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6th Biennial

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2002

Margaret (Peg) Van Patten, editor

Proceedings of the Long Island Sound Research Conference, held jointly with the New England Estuarine Research Society (NEERS) meeting on October 26-28, 2002 at the University of Connecticut at Avery Point.

Acknowledgments

Since 1992, the Long Island Sound Foundation has sponsored six Long Island Sound Research Conferences, in cooperation with the Connecticut and New York Sea Grant programs. We have worked together to provide financial support for each of the biennial conferences and for the publication costs of the Proceedings volumes from the conferences.

In 2002, the Conference was combined with the annual NEERS (New England Estuarine Research Society) meeting, and held as a joint conference at the University of Connecticut's Avery Point campus. NEERS participants were invited to include their presentations in this volume alongside the Long Island Sound presentations.

The Board of Directors of the Foundation would like to thank Susan McNamara, the Executive Director of the Foundation for facilitating and coordinating the conferences, Joyce Wood-Martin and Barbara Mahoney of the University of Connecticut for their assistance and Peg Van Patten, Communications Director of the Connecticut Sea Grant for compiling and editing the 2002 Long Island Sound Research Conference Proceedings. Special thanks to Dr. Carmela Cuomo for assisting with the Introduction and to Dr. W. Frank Bohlen for reviewing the manuscripts and making many valuable suggestions.

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Introduction

The State of Long Island Sound in 2002

Carmela Cuomo[1] and Margaret Van Patten[2]
[1]Yale University and Univ. of New Haven; [2]Connecticut Sea Grant, University of Connecticut

Numerous exciting scientific investigations into the physical, chemical, biological and ecological functioning of Long Island Sound are underway. The results of many of these investigations were presented at the joint Long Island Sound Research–New England Estuarine Research Society (NEERS) Conference held in at the University of Connecticut's Avery Point campus in Groton, Connecticut, in October, 2002. The backdrop for these studies is the Long Island Sound (LIS) and the environmental conditions that occurred within it during the years 2000-2002. Compiled in one place, by means of this proceedings volume, these studies attempt to provide an overview of these conditions as well as a summary of the conference presentations. This introduction provides some background about the Sound, and presents information such as locations of monitoring stations used by the environmental agencies in Connecticut and New York, which would be repetitive if included in several papers.

Long Island Sound, like other urban coastal areas at the start of the 21st Century, faces increasing environmental pressure from a variety of sources, not the least of which is expanding urbanization of its watershed. Increased urbanization of the Sound's watersheds results in more surface runoff, greater non-point source pollutant loading, and higher volumes of treated sewage effluent–all running into LIS, the ultimate sink. The LIS watershed includes six states and a portion of Canada, and holds roughly 16 million people. To put the population into perspective, it is roughly equivalent to the entire population of the nation of Chile. In addition, LIS is, like most of the world's water bodies, presently being affected by a warming of global temperatures. The exact nature of the responses of the various components of LIS to these increasing pressures is, at present, unknown. Whether the lobster mortality event of 1999 is one response to the environmental stressors presently at work in the Sound or is simply a unique biological catastrophe is not known with certainty as of this writing. What is known, however, is the fact that the LIS ecosystem is one of the most heavily utilized and densely populated estuaries in the United States. As such, it warrants a thorough understanding of its various biological, chemical, physical, and geological components.

The location of Long Island Sound, adjacent to one of the largest metropolitan areas in the world, makes it subject to all the coastal problems associated with large human populations, including over-fishing, point source pollution (e.g. sewage disposal, power plant effluents), non-point source pollution (e.g. pesticide and fertilizer run-off, road residue run-off), oil tanker spills, the introduction of non-indigenous species, loss of wetlands to development and sea level rise, and the development of moderate-to-severe hypoxia during certain times of the year.

Long Island Sound occupies a basin that is approximately 150 km long and 30 km wide. Its average water depth is 24 m, although a central E-W-running axis extends from 30-60 m depth through the Western and Central basins of LIS and depths can reach over 100 m near the Race, a narrowing at the eastern end.

Geologically, LIS began when its basin began to form about 3 million years ago, from eroding river banks. This basin was then modified and remodified by glacial action. During the last glacial retreat 20,000 years before present, a recessional moraine formed at the southern end of the basin.

As the glacial retreat continued northward, the meltwaters flowed into the basin and accumulated behind this morainal dam, forming freshwater Lake Connecticut. Glacial clays, carried by the waters, began to accumulate at the base of the glacial lake. As the glaciers retreated, sea level began to rise and the land began to rebound from its glacially-depressed state. These two factors combined to cause a breach in the moraine that dammed Glacial Lake Connecticut at its eastern edge. Seawater flowed into the basin sometime between 15,000 and 13,500 years ago. Rebound and sea level rise continued until approximately 9,000 years ago, after which most rebound ceased, although sea level rise continued until 3,000-5,000 years ago, forming the present day Long Island Sound estuary. There are three raised features, or sills, that cross the Sound's bottom, dividing LIS into three basins, the Western, Central, and Eastern Basins.

The Sound's east-west orientation and distribution of its freshwater inputs is unusual for an estuary. Because of these factors and the interesting bottom topography, the circulation is more complex than can be thoroughly iterated here. Marine waters enter from the Atlantic Ocean at the eastern end, through the Race, where it narrows and deepens due to gouging by the glacier. It flows westward along the Connecticut shore, or northern boundary, and is joined by freshwater entering from three major rivers located in Connecticut–the Connecticut, Thames, and Housatonic Rivers. An additional significant source of brackish water is the East River, a tidal strait located at the western end of LIS. The denser marine water sinks as it flows westward in a generally counterclockwise motion.

General weak vertical salinity stratification occurs basin-wide throughout the year, although strong, local salinity gradients occur during times of high freshwater discharge. An additional longitudinal salinity gradient of 5-6 $^{0}/_{oo}$ (ppt) exists throughout the year. Superimposed on the general estuarine circulation of LIS are various spatially and temporally dispersed density-driven currents. Tidal currents are dominant throughout the majority of LIS, with the flood tide shorter than the ebb tide.

Sediment distribution in LIS is linked to the tidal and wind-driven currents. Nearest to the Race is a region of active erosion or non-deposition. As one moves slightly westward, sediments become finer and consist primarily of silt. The Central and Western basins are dominated by soft-bottom, mud and clay.

The central and western portions of LIS experience strong thermal stratification during the summer months whereas strong tidal currents keep thermal stratification relatively weak in the eastern portion of the Sound. Breakdown of thermal stratification generally occurs in the late summer-early fall when major winds begin to shift from the southwest to the northwest. The strongest winds usually occur during the winter months.

Summer thermal stratification is associated with the onset of hypoxic conditions in the western end of LIS. Hypoxia - the presence of waters containing $\leq$ 3.25 mg/l dissolved oxygen - was first observed in the Sound in the 1970's and has been a persistent summer condition in the westernmost part of the Sound since the 1980's. The extent and severity of hypoxic events in the Sound has received much study over the past thirty years. The causes of hypoxia in LIS are complex although one of the primary ones appears to be nitrogen enrichment associated with increasing urbanization of the watershed. Nitrogen enrichment results in increased phytoplankton production in the water column, which translates into increased organic matter deposition to bottom sediments. Bacterial decomposition of organic matter - both in the water column and in the sediments - act as an additional oxygen sink via aerobic bacterial decomposition of organic matter. Additionally, anaerobic decomposition of organic matter can result in the production of hydrogen sulfide, the oxidation of which results in further consumption of water column and pore water oxygen. Sediment oxygen demand can become quite large if bottom water oxygen renewal is limited by either water temperature or the existence of a stratified water column which prevents vertical mixing, ultimately setting the stage for the onset of hypoxic and perhaps even anoxic bottom waters. The CT DEP, under the auspices of the US EPA, has conducted a hypoxia monitoring program in Long Island Sound since the early 1990's. These studies result in a yearly series of maps that depict the extent and duration of hypoxia in the Sound (see Figure 1).

From historical times, the Sound has been used as a source of food, recreation, and commerce. Indian middens attest to the fact that the Sound was an important source of food for the first peoples of the region. Sailing ships carried goods to various ports within the Sound, and whaling ships sailed into and out of Mystic. The diverse and abundant species present in LIS were used by both recreational and commercial fishermen. The Sound was a known commercially-important source of oysters, scallops, hard clams, lobsters, and numerous finfish. The Long Island Sound lobster fishery, alone, was worth over $30 million in 1998.

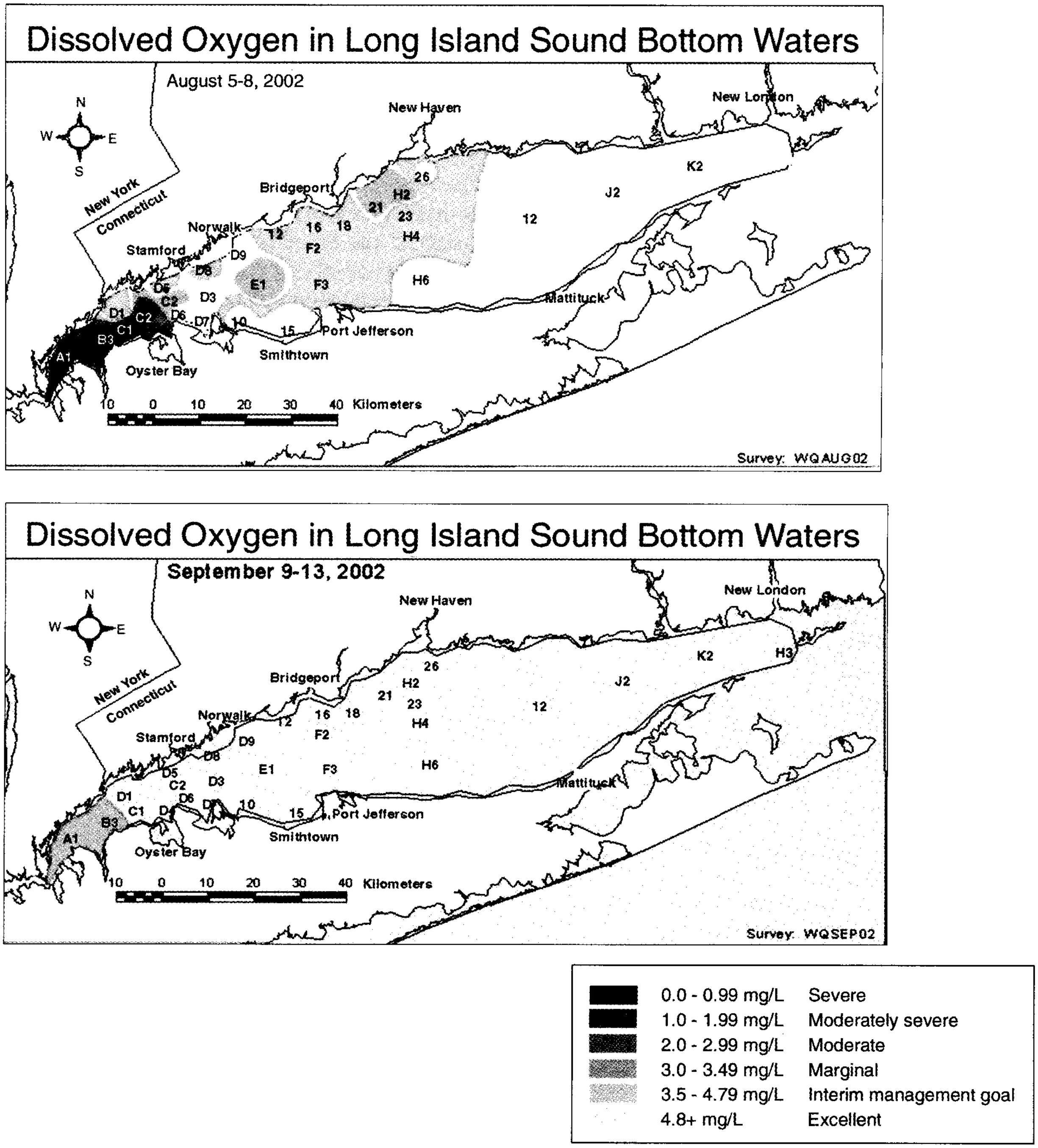

Figure 1. Two hypoxia maps for Long Island Sound in 2002 (Redrawn from Connecticut DEP). Maps for various years and months are available online. To compare them, go to: <http://www.dep.state.ct.us/wtr/lis/monitoring/lis_page.htm>. This example shows the hypoxic situation in August, 2002, and the more normal situation in September as hypoxic conditions subside.

Oyster harvesting, traditionally a very large LIS industry, declined precipitously in 1997 and 1998, following outbreaks of the diseases MSX and Dermo. While these diseases did not affect human consumers, they did cause massive oyster mortality and dramatically reduced harvests Sound-wide. Harvest values dropped from nearly $50 million in 1992 to about $8 million in 1997. On the bright side, hard clams continue to be a valuable LIS industry, as its clam beds are among the most productive in the nation.

However, perhaps no other event, or resource, has served to focus attention on Long Island Sound more than the recent massive lobster die-off that began in Western Long Island Sound in 1998-1999 and continues to this day. This die-off effectively shut down the LIS lobster fishery, especially in the Western Sound. The event may have resulted from a deadly synergy of environmental and biological stressors, including but not limited to, warmer than normal water temperatures, hypoxia, and a parasitic neoparamoeba. Many lobstermen also believe that pesticides and other anthropogenic contaminants played an important role in the lobster mortality. Research into the cause(s) is ongoing, and results will be reported elsewhere in addition to several papers in this volume.

One cannot help but ask, was this die-off a result of a one-time confluence of events ongoing in the Sound, or is it a harbinger of a "new" environmental state of the Sound ? If so, what will that state be, and how will living resources adapt (or fail to)? How much of the changes taking place in the dynamic Sound system are natural, and what portion are anthropogenic? These are the questions many scientists are attempting to answer.

Are hypoxic events in LIS only a phenomenon of the latter 20th Century or do they have a geo-historical basis? Several papers in this volume (Lugoboli, *et al.*, Thomas, *et al.*, and Varekamp, *et al.*) explore the historical record of hypoxia in LIS and its relationship to temperature change, organic carbon storage in WLIS sediments, benthic foraminiferal changes, and increasing eutrophication. The results of these studies indicate that increased carbon storage in the sediments of the Sound can be linked to increasing human habitation of the region; these studies, however, also point to the potential influence of global temperature patterns on local faunal changes and organic carbon storage within LIS sediments.

The development of summertime hypoxic conditions in LIS is believed to be directly related to excess nitrogen loading. Did nitrogen loads change in some important way in recent times? In a

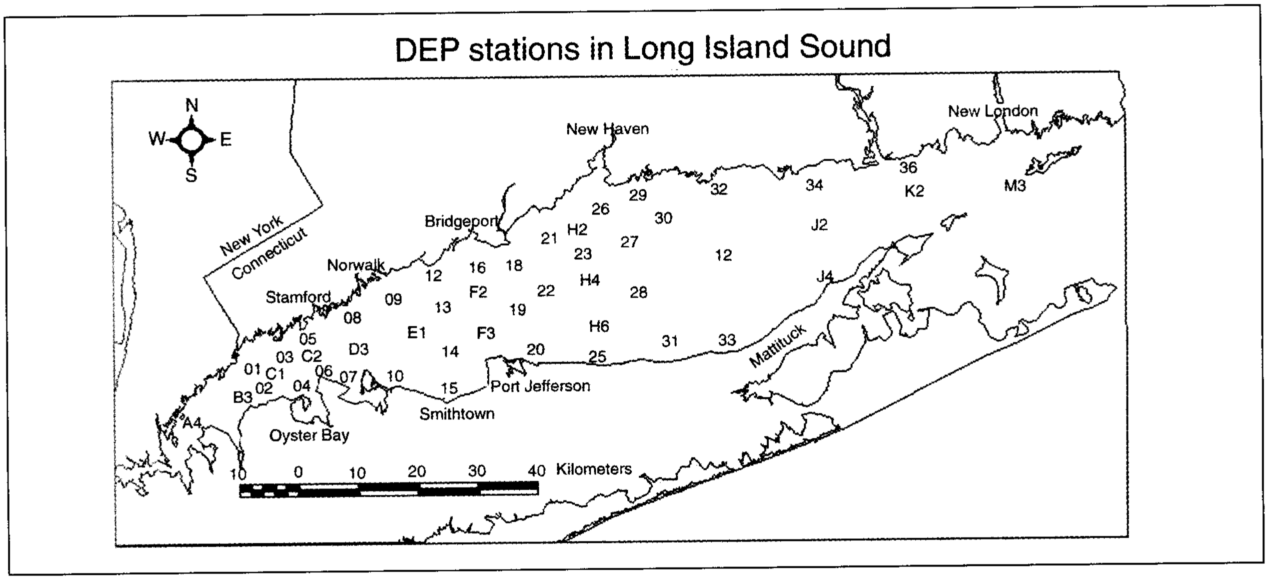

Figure 2. Connecticut Department of Environmental Protection water quality sampling stations in Long Island Sound. Stations B3, D3, F3, H6,12 and M3 were initiated in 1991. The rest of the 12 stations were established in 1994. (Redrawn from CTDEP original).

paper in this volume, Li, *et al.* examine the contribution of atmospheric nitrogen deposition to overall nitrogen loading in LIS during the time period 1991-1999 whereas Dettman and Mason (this volume) explore the effects of warm winter water temperatures on chlorophyll-a and total nitrogen content of surface waters and their implications for summer plankton production using data obtained by the CT DEP monitoring program for the years 1995-2001. Many investigators used the CTDEP's established monitoring stations, shown in Figure 2.

What is the relationship between water column hypoxia and the benthic environment in WLIS? How might water column hypoxia have affected organisms, such as lobsters, living on and within the sediments of LIS ? Immediately following the lobster mortality event in 1999, a series of sediment profile images (SPI) were taken at 32 of the CT DEP's stations in WLIS. These revealed reduced anoxic sediments at a majority of the stations sampled (Figure 3). These sediment images and geochemical changes associated with sediment hypoxic and anoxic conditions and their relationship to bottom water geochemistry are discussed in Fitzgerald, *et al.* (this volume) and Cuomo, *et al.* (this volume).

The lobster mortality event of 1999 was preceded by higher than normal water temperatures and dry weather for much of the summer. The remains of Hurricane Floyd arrived in New England in September of 1999 and resulted in a sudden, dramatic input of freshwater to Long Island Sound over the period of two days. The effects of such meteorological forcing on the responses of the water column in Central and Western LIS during the Fall of 1999 are examined in a paper by Wilson (this volume) whereas Codiga, *et al.* (this volume) explore the relationship between stratification and hypoxia development in WLIS.

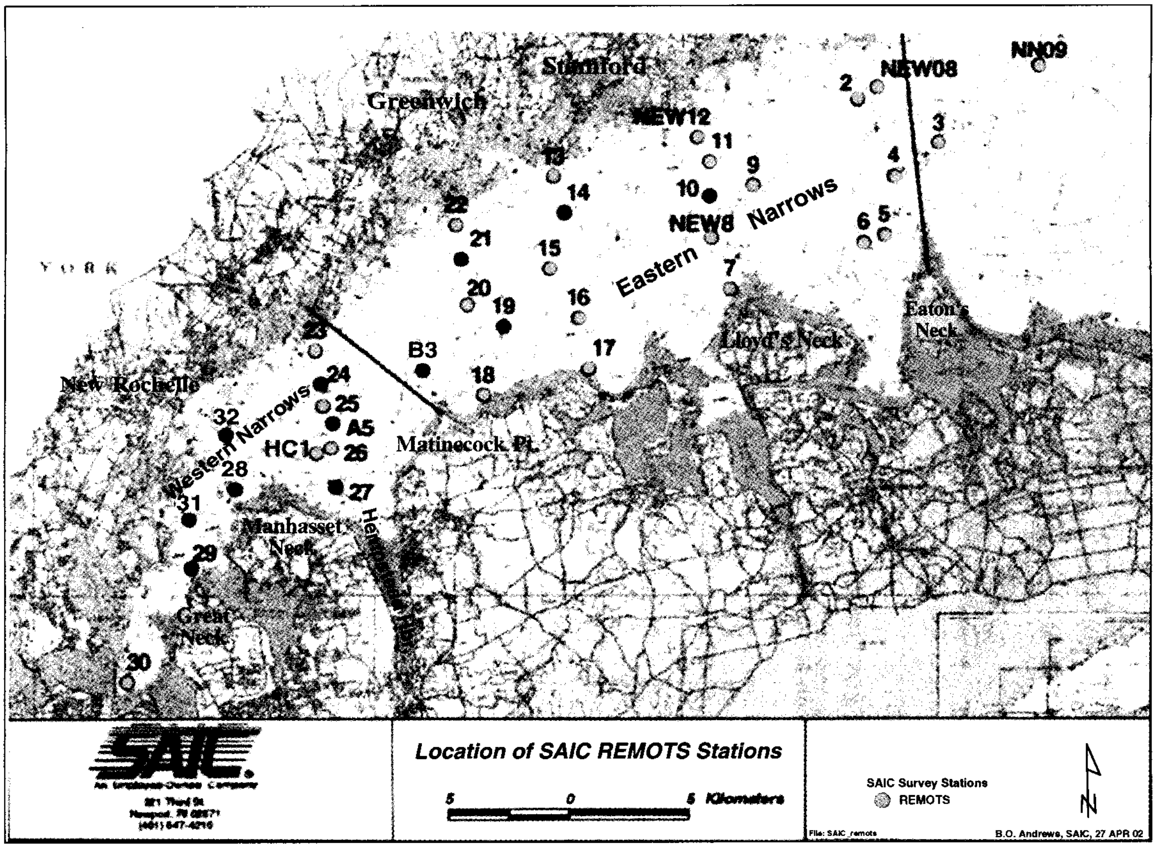

Figure 3. Monitoring stations: DEP monitoring stations are dark numbered circles; SAIC REMOTS sampling stations are lighter circles, sampled by Cuomo and Valente. Redrawn from original.

Whether or not increasing water temperatures alone might be a driving force behind changes in LIS fisheries is explored for the years 1975-2001 in a paper by Tomichek and Roseman.

New in this Proceedings is an index by topic, which will facilitate readers in looking for the material they most wish to peruse.

Together, the papers contained in this volume will, at the very least, provide a historical record and a framework for further scientific inquiry into the factors responsible for large-scale changes in the ecosystems that comprise Long Island Sound.

Save the Date!

The next Long Island Sound Research Conference, "Long Island Sound in Transition", will be held on November 4-5, 2004 at the Charles B. Wang Center, Stony Brook University, Stony Brook NY.

Congratulations to Connecticut Science Fair 2002 Award Winners

The Long Island Sound Foundation would like to congratulate those who received awards at the Connecticut Science Fair 2002. Below are their names and titles of the winning projects.

Alexandra M. Bourque, Greenwich High School, Grade 11
"Sediment's Effects on the Filtration Rate of Cyanobacteria in Mussels"

Cara L. DeCresente and Rebecca A. Henshaw, St Rita School, Grade 7
"'What is the Matter' with Beach Sand?"

Yannique R. Salabie, Harding High & Health Magnet School, Grade 11
The Spectrophotometric Analysis of the Effects of Salinity on the Chemoluminescense of Dinoflagellates"

Valerie A. Schoepfer,* Fields Memorial School, Grade 8
"Nitrate and Phosphate Concentrations in Salt and Fresh Water"
* *see paper in this volume*

Observed Evolution of Vertical Profiles of Stratification and Dissolved Oxygen in Long Island Sound

Dan Codiga[1], Duane Waliser and Bob Wilson[2]
[1]University of Connecticut, Department of Marine Sciences
[2]State University of New York, Stony Brook

Motivation

Levels of dissolved oxygen below the 3 mg/L threshold for hypoxia have been identified as an important threat to the ecological health of Long Island Sound (LIS). Associated management plans incorporate costly regulatory actions based on total maximum daily load analysis for nutrients. Hypoxic conditions are generally understood to be the combined result of a number of biological and physical processes in the marine environment. The conventional wisdom is that the onset of density stratification due to solar heating in the spring and summer facilitates isolation of deep water from atmospheric contact, in which depletion of oxygen by biological consumption proceeds at an accelerated pace; increases in phytoplankton biomass that accompany nutrient enrichment exacerbate the problem. Motivated in part by the need to assess the extent to which this broadly accepted mechanism applies, the present work targets improved understanding of density stratification and the role it plays. A goal is to test the hypothesis that inter-annual variation in spatial extent and severity of hypoxia is tied to inter-annual changes in the seasonal cycle of stratification.

Approach

Results presented here are part of an effort to improve understanding of mechanisms controlling development and breakdown of density stratification, and changes in dissolved oxygen concentrations, based on direct observations. The suite of measurements needed to properly investigate such processes is somewhat demanding. Important factors potentially include advection by sub-tidal residual currents arising from exchange between the estuary and adjacent shelf waters, as well as riverine input; stirring of water column properties in the frictional bottom boundary layer due to tidal currents; and fluxes of momentum, heat, and precipitation across the air-sea interface. Because property exchanges between shallow and deep layers are expected to be important, improved diagnosis of the responsible mechanisms will be limited without measurements that resolve vertical profiles through the water column.

A ferry-based observing system is being developed to help address these needs. The system includes the use of ferries as sampling platforms, together with oceanographic moorings for in situ sampling of vertical profiles of water properties. Two ferry transects are involved, New London to Orient Point and Bridgeport to Port Jefferson. Each will be outfitted with an acoustic Doppler current profiler (ADCP) to measure current profiles beneath the vessel as it transits, and with sensors to measure surface water temperature, salinity, dissolved oxygen, and chlorophyll concentrations. The Bridgeport - Port Jefferson ferry will also collect atmospheric parameters. Because ferry crossings occur multiple times daily, data collection over periods longer than a few months will enable tidal and sub-tidal signals to be isolated from each other. Such temporal sampling achieved with spatial coverage along shore-to-shore transects, is critical to improve understanding of sub-tidal variability including the residual circulation, and would be prohibitively costly without use of ferries. Future reports will include results based on ferry sampling; here, we present preliminary findings from the moored profiler component of the project.

There are three mooring sites, one each in Western, Central, and Eastern LIS. The sites have been selected in order to span the length of the Sound and to sample its deeper sections, while avoiding locations where vessel traffic is unacceptably high or tidal currents too strong for profilers to function. The Western site (41° 13.58' 72° 14.32') is in water 31 m deep, roughly mid-way between Long Neck Point and Eaton's Neck Point. The Central site (41° 00.96' 73° 05.16') is in water 37 m deep, just south and east of Stratford Shoal. The Eastern site (41° 13.58' 72° 14.32') is in water 46 m deep, just east of a point roughly midway between Plum Gut and Black Point, Niantic. It is understood that oxygen depletion at the Eastern site, and to some extent at the Stratford site, is not of as much concern as at the Western site. All three sites are sampled to help diagnose the mechanisms driving the evolution of stratification and assess the extent to which they operate basin-wide. Given the known seasonal onset and retreat of hypoxia, mooring deployments are scheduled for spring and summer/fall periods, nominally for two months during the most concentrated development and breakdown of stratification.

Methods

As mentioned above, the moorings need to measure temperature, salinity, and dissolved oxygen; sample vertically throughout the water column; and operate unattended for two months or more, collecting at least several profiles each day. Technology to achieve these aims, particularly the sampling of entire vertical profiles in the presence of very strong tidal currents, and unattended collection of dissolved oxygen values for more than a few weeks has not been available until recently.

Two types of moored profiling instrument are used. The first type is the buoyancy-driven Ocean Sensors APV 500 model, outfitted with oversized piston and custom roller brackets, and equipped with the Aanderaa Optode oxygen sensor. This profiler moves upward (downward) along the mooring wire based on the buoyancy excess (deficit) associated with a piston held in an outward (inward) position. The second type is a wave-driven Seahorse model by Brooke-Ocean Technology, with SeaBird 911-Plus CTD and model 43 oxygen sensor. It collects data while driven upward along the mooring wire by its buoyancy excess, and is then driven downward by its own inertia due to surface wave motions, harnessed by a one-way ratchet attachment to the wire. The buoyancy-driven profilers are programmed to collect data on an upward profile, then again ~15 minutes later on a downward profile, repeated each ~4 hours. The wave-driven profilers are subject to less severe battery constraints, given their conversion of wave energy to mechanical movement, so are programmed to profile once per hour.

Vertical movement of profilers is impeded by drag associated with strong currents. The buoyancy-driven profiler is quite vulnerable to drag, despite use of an oversized piston to gain additional buoyant force and custom roller brackets to reduce friction between the wire and the profiling package. This makes it unsuitable for deployment at the Eastern site where tidal currents are strong. The wave-driven profiler can function in stronger currents, but will not profile in very calm sea states when wave energy is low. For these reasons, not all scheduled profiles return data from top to bottom, and the records are punctuated by occasional partial or missing profiles. Vertical resolution is nominally several cm; values are binned and averaged in 1-m depth intervals, and averaged across upward and downward profiles in the case of the buoyancy-driven profiler.

Time series vertical profiles of salinity, temperature, density anomaly (sigma-t), and dissolved oxygen from the mooring deployments are presented together with tidal heights and wind speeds. The tidal heights are hourly values from the Bridgeport station acquired from the National Oceanic and Atmospheric Administration (NOAA) website <http://www.co-ops.nos.noaa.gov>. Wind speeds are from Sikorsky airport, Bridgeport, acquired from the climate section of the NOAA website <http://www.ncdc.noaa.gov>.

To date, there have been two deployment periods: Spring 2002 and Summer/Fall 2002. The Spring 2002 deployment was from April 18 to June 3 and involved one wave-driven profiler at the

Central site near Stratford Shoal, which returned a good record. The Summer/Fall 2002 deployment was from August 12 to October 18 (one week prior to the LISRC/NEERS conference, hence the preliminary nature of this report). It included moorings set at all three sites; the instruments at the Eastern and Central sites returned good records. A few weeks following its deployment, the mooring initially located at the Western site was reported to be in a navigation channel off Hart Island, near the Bronx in New York. The mooring hardware was recovered in its entirety, with the exception of the profiling sensor package itself and the upper mast and light on its surface buoy, which had been sheared off. What little evidence there is in hand indicates that a hawser line between a tug and barge had ensnared the buoy and dragged it to Hart Island, in the process stripping off the profiling sensor package causing it to be lost. Such an outcome, while a significant setback to the project, is perhaps not entirely surprising, given the level of commercial traffic in Western LIS. This is true despite that, as with all project buoys, this unit had a lighted tower and was licensed by the U.S. Coast Guard. In addition, detailed announcements of buoy positions and descriptions are always made well ahead of deployments both through the USCG Notice to Mariners and via a network of mariners including pilot boat captains and tug and barge operators.

Preliminary results

Results from three mooring deployments are described: Spring 2002 at the Central site (Figure 1), Summer/Fall 2002 at the Central site (Figure 2), and Summer/Fall 2002 at the Eastern site (Figure 3).

The Spring 2002 Stratford Shoal deployment used a wave-driven profiler (Figure 1). Tidal fluctuations are apparent in the vertical profiles and are consistent with the presence of an ambient horizontal gradient in properties being advected past the mooring by tidal currents. The spring-neap cycles during this interval included one 4-5 day period of distinctly weaker tides (yearday ~123-130); overall, the link between spring/neap tides and variations in vertical profiles is not strongly apparent. The profile of density anomaly (sigma-t) began nearly vertically uniform at ~23 kg/m^3 except for the upper few meters, which were ~0.8 kg/m^3 less dense. At the end of the record, a more nearly two-layer structure was in place, with a difference of ~1.5 kg/m^3 across an interval roughly at mid-depth, separating an upper (~20 kg/m^3) layer from a lower layer (~21.5 kg/m^3). The development of this expected overall springtime reduction in densities and vertical stratification occurred as a result of changes to both the temperature and salinity structure during the record. The temperature was nearly vertically uniform (~8°) until about yearday 125, when it steadily increased at all depths. This increase was more rapid and intense near the surface during the final weeks. (Surface temperature ~15°; bottom temperature ~12°). Salinity was also nearly uniform vertically (~29 PSS) at the beginning of the record, but, in contrast to temperature, showed substantial variations starting at about yearday 120, when relatively fresh values (~27.8 PSS) appeared in the upper ~5 m. Ending values were ~27 in the upper water column and more than ~28 at depth. Winds generally stayed in the range of 5-15 knots, with several events at higher speeds. A wind event at yearday ~123 rapidly modified the entire water column, apparently destroying much of the stratification; restratification occurred within a few days. Strong winds on yeardays 133-135 are also associated with substantial profile modification. It remains to be determined whether these events are due to local vertical processes or advection by wind-driven currents; similarly strong winds on yearday ~118, though not of the same duration, did not cause such modifications. Dissolved oxygen showed high levels (~11-12 mg/l) in the upper 5-10 m at the start of the record, then weak vertical variations superimposed on a gradual decrease to ~8 mg/l over the remainder of the deployment.

The Summer/Fall 2002 Stratford shoal deployment (Figure 2) used a buoyancy-driven profiler. This record reveals a number of relatively abrupt changes as summer transitions to fall, in contrast to the spring data from this site (Figure 1) during which the overall transition from spring to summer profiles generally occurred gradually throughout the deployment. One neap period of tides was sampled fully. Winds were generally weaker than in Spring, as expected, but reached their peak sustained values during the neap tides. Prior to yearday ~240, strong density stratification is seen at shallow, then middle, depths, due almost entirely to temperature. Vertical structure subsequently breaks down abruptly in association with strong winds; this occurs during neap tides and is not attributable to tidally-driven mixing. In the remainder of the record, temperature decreases and maintains weak vertical differences. Quite independently, there is a distinct shift, at all depths, to lower salinities at yearday ~252-257, which reduces densities by up to 2 kg/m^3. The oxygen signal parallels temperature most closely throughout the record, with structure dominated by deeply penetrating high values (~6.5-8 mg/L) from yearday ~248-260, and varies only weakly with the swings in salinity and density at yearday ~252-257. Vertical oxygen contrasts (~8 mg/L shallow vs ~5 mg/L at depth) persist at the end of the record despite the weak vertical density differences.

The Summer/Fall 2002 Eastern LIS deployment (Figure 3) used a wave-driven profiler. Periods of missing profiles correspond to spring tides, indicative that drag associated with tidal currents degrades profiler performance. On comparison with the simultaneous Stratford Shoal record (Figure 2) described above, it is clear that the evolution of all measured fields occurs quite distinctly at the two sites. The Eastern LIS site shows density profiles that begin with stratified structure and slowly homogenize over the first few weeks of the deployment. This includes a response to the wind-related event of yearday ~240 seen at Stratford Shoal, which is followed by much more rapid restratification at the Eastern site. Starting about yearday 255 a high-salinity deep layer arrives and builds in strong

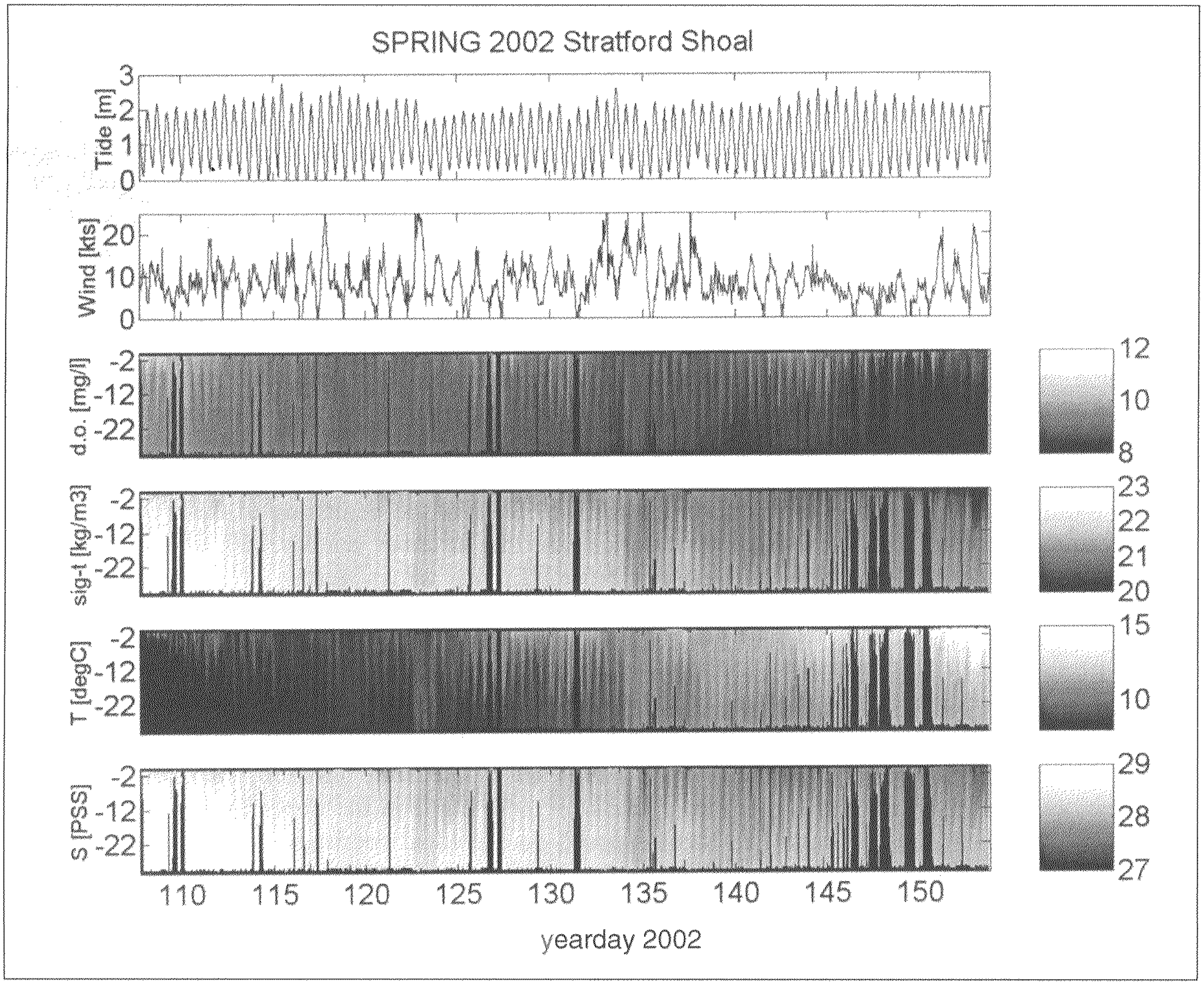

Figure 1. Results from wave-driven moored CTD profiler deployed near Stratford Shoal in water 37 m deep from April 18 to June 3, 2002. The upward axis labeled at left is the vertical coordinate in meters, positive upward, relative to zero at the sea surface. Missing data are shown as black and appear identically in all four of the lower frames. Wind speeds from Sikorsky Airport, Bridgeport. Tidal heights from Bridgeport Harbor.

stratification. Vertical temperature differences are nearly absent from yearday ~260 forward, while in contrast vertical salinity differences persist until about yearday ~275. In the final days of the record, sharp vertically uniform tidal fluctuations are prominent and suggest the presence of a frontal boundary being advected past the mooring. Overall the oxygen signal is quite uniform vertically, with values diminishing from ~6.5 to 6 mg/L over the duration of the record, despite the density stratification early on. High near-surface values build in from yearday ~258-275; these are not concurrent (later by ~weeks) with similar high shallow values at the Stratford site.

Conclusions

Based on the preliminary description of the moored profiler data we draw the following conclusions regarding the Central and Eastern sites. Both temperature and salinity contribute substantially to the density stratification, and these two variables fluctuate independently from one another on timescales of days to weeks. Wind events can nearly homogenize the water column in a matter of hours, and restratification can occur about as quickly; following an event in Summer/Fall 2002, restratification occurred within about a day at the Eastern site and after a few days at the Central site. Fluctuations in profile structure that follow the spring-neap pattern in tidal heights are not strongly apparent, suggesting that, while tidal mixing may be important in setting the basic basin-

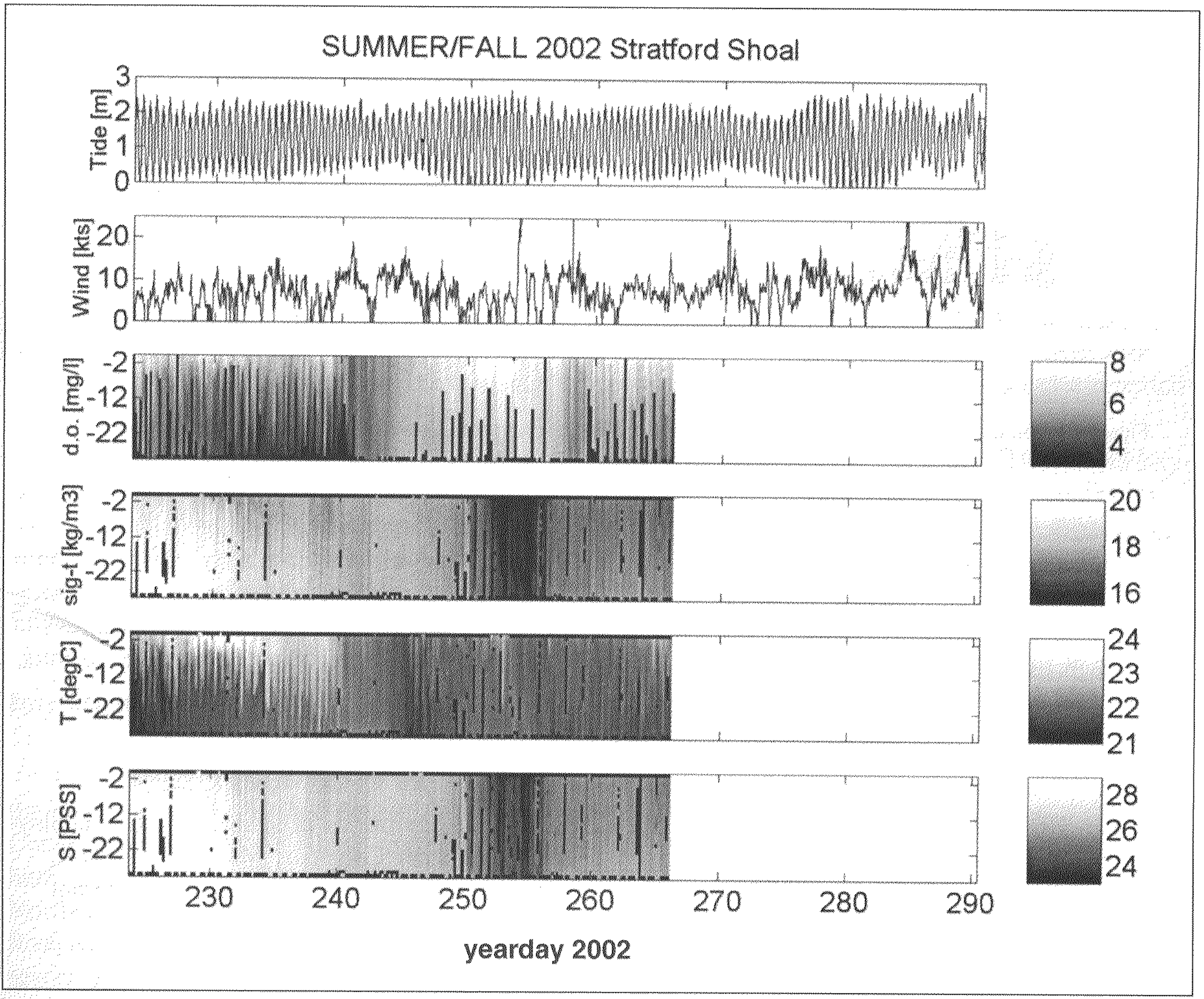

Figure 2. Results from buoyancy-driven moored CTD profiler deployed near Stratford Shoal in water 37 m deep from August 12 to October 18, 2002, presented as in Figure 1. For ease of comparison, time axis spans full length of simultaneous deployment shown in Figure 3.

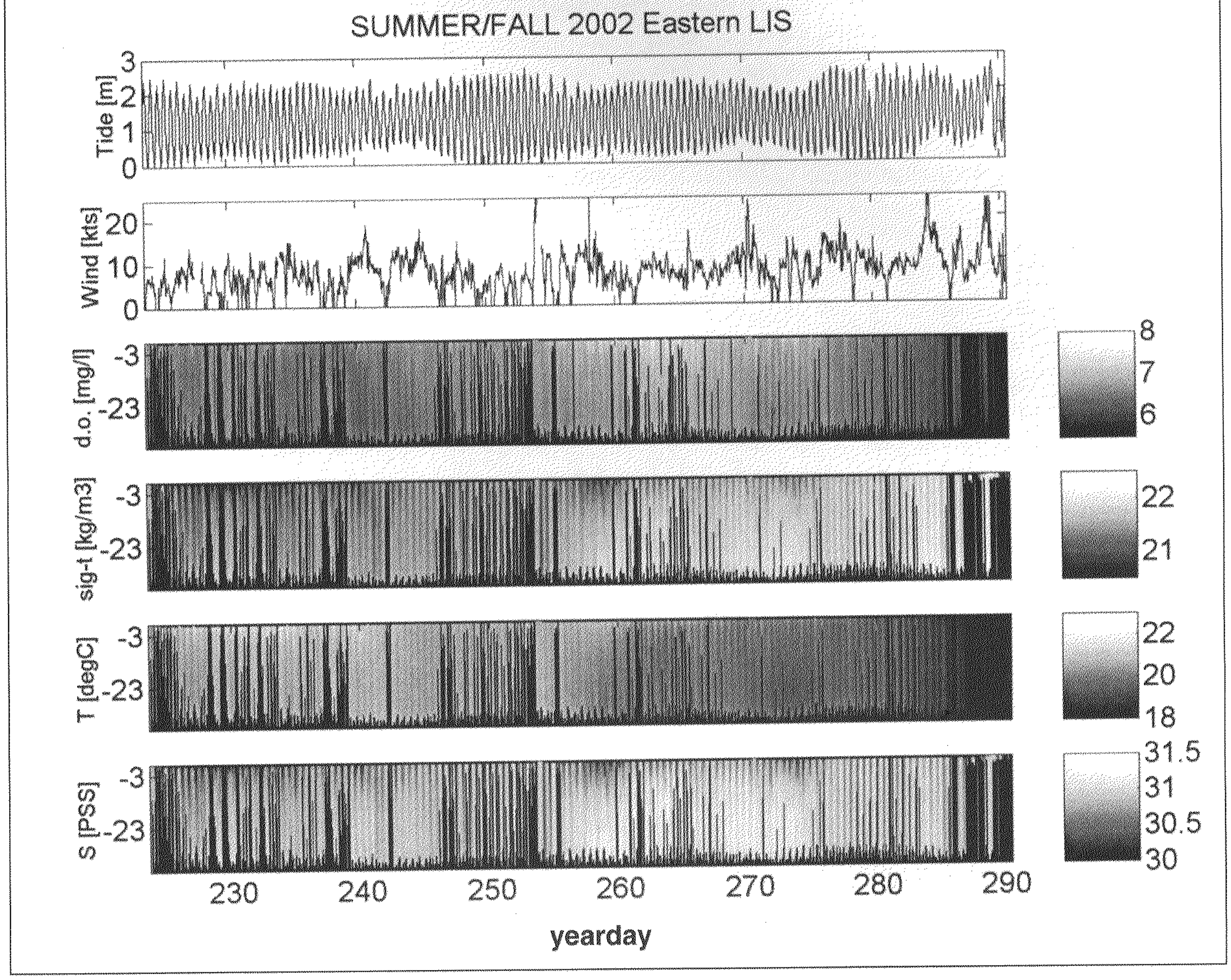

Figure 3. Results from wave-driven moored CTD profiler deployed in Eastern LIS in water 46 m deep from August 12 to October 18, 2002, presented as in Figure 1.

wide stratification it is of secondary importance to density profile evolution over days and weeks. There are indications that events in the residual circulation advect salt and have a strong impact on the density profile, for example as the onset of high salinities at depth that strengthen stratification. Evolution of profiles occurs independently at the Eastern and Central sites. Oxygen concentrations fluctuate at times independently from, and at times in parallel with, changes in temperature, salinity, and density. Additional analysis is underway to expand on these preliminary findings.

Acknowledgments

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Monitoring of Bottom Water and Sediment Conditions at Critical Stations in Western Long Island Sound

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Abstract

The overall objective of this study is to obtain an understanding of bottom water and sediment chemistry dynamics in western Long Island Sound (WLIS), and the potential influence of near-bottom chemistry on benthic community structure and lobster health. Beginning in May 2002, water samples were collected at heights of $\leq$ 5 cm and 1.0 m above the bottom at 12 selected stations in WLIS and analyzed for NH_4, H_2S, and O_2. Grab samples for analysis of benthic community structure and cross-section photographs of the sediment-water interface (REMOTS sediment-profile images) were simultaneously obtained along with the water samples at six of the 12 stations. The 12 stations selected for sampling in 2002 were chosen from among 36 stations for which historical bottom water chemical (NH_4 and H_2S) data and REMOTS sediment-profile images already exist (see Figure 3 in Cuomo & Van Patten, this volume). The results of the benthic sampling are not reported on in this paper.

The data collected so far provide an accurate field record of the apparent levels of dissolved oxygen in the sediments of western Long Island Sound, as reflected both in apparent redox depths (as recorded in sediment-profile images) and the amount of ammonia, hydrogen sulfide, and oxygen present in the bottom waters, over a time period critical to the LIS lobster fishery. The results of this study are discussed with regard to the data obtained by the Connecticut Department of Environmental Protection (DEP)'s Long Island Sound (LIS) Monitoring Program.

Introduction

Long Island Sound is one of the most well-known and commercially important estuaries along the east coast of the United States. Its location–bounded by Long Island, New York City, Connecticut, and Rhode Island–makes it susceptible to all of the problems associated with large human coastal populations. In addition to these problems, LIS, like other environments in the Northeast, has been experiencing warmer than average temperatures (NOAA, 2002; Tomichek & Roseman, this volume; Wilson, this volume), which may further compromise its quality. Prior to severe die-offs in 1999 and 2002, LIS was known to be an important source of commercially harvested lobsters and shellfish. The 1999 die-off effectively closed down much of the commercial lobster fishery in Western LIS and the 2002 die-off did the same for the central and eastern portions of the Sound. Although a neoparamoeba (S. Frasca, personal communication) has been implicated in the death of the lobsters in 1999, most experts agree that the lobsters were weakened and stressed by an assortment of other factors prior to the sudden mass mortality. Lobster mortality events have occurred occasionally in LIS – just not to the extent of the 1999 and 2002 events (CT DEP Bureau of Natural Resources, Marine Fisheries Office, 2000). One environmental factor that appears associated with many of the lobster die-offs in WLIS is hypoxia – the presence of extremely low amounts of dissolved oxygen in waters one meter or more above the bottom – as measured by the CT DEP's ongoing hypoxia monitoring program in LIS.

Background

Immediately following the September 1999 mortality event, SAIC was contracted to collect sediment-profile images (SPI) at 30 stations in WLIS to determine if there was anything unusual occurring at the sediment-water-interface that might explain the lobster mortality. The SPI images revealed a very shallow (< 1 cm) apparent redox potential discontinuity (aRPD - a measure of the apparent depth of oxygen penetration into the sediment) at most stations. The sediment surface at most stations consisted of black and/or dark-colored sediment, indicative of hypoxic to anoxic conditions. It was considered highly likely that lobsters had experienced low levels of dissolved oxygen and/or had been exposed to low levels of sediment-derived anoxic metabolites, such as ammonia and hydrogen sulfide in the time preceding the die-off.

A follow-up survey was conducted during August–November of 2000 (Valente & Cuomo, 2000). This survey entailed using SPI images and bottom water chemical samples in an attempt to evaluate the role of near-bottom hypoxia/anoxia and associated reduced end-products of anaerobic organic matter decomposition (e.g. hydrogen sulfide and ammonia) as potential structuring influences on the benthic environment and benthic communities, including lobsters, of WLIS. During this survey dissolved oxygen levels were recorded 10 cm above the bottom by a YSI DO Meter and discrete water samples were taken for ammonia and hydrogen sulfide analysis within 2-4 cm of the seafloor.

The SPI images from the second survey revealed very shallow (< 2cm) aRPD depths at a majority of stations sampled in the Western Narrows, with more variability present in stations sampled in the Eastern Narrows. No clear temporal trends of either increasing or decreasing sediment oxygenation were seen in these images across the sampling period. The SPI images at many stations throughout the survey revealed very black sediment either at or just below the sediment-water-interface. Such sediments are inferred to have a high inventory of sulfides, resulting from the anaerobic bacterial breakdown of organic matter in the sediments. The SPI images revealed a fauna dominated by surface-dwelling opportunistic marine organisms. Such a fauna is characteristic of hypoxic events in WLIS and elsewhere.

Dissolved oxygen measurements revealed a steady increase from 2.97 mg/l in August up to 8.52 mg/l in November; they also revealed a general west-to-east gradient of increasing bottom water DO during August. Elevated sulfide and ammonia levels were observed at several stations in the Western Narrows in August, at all stations in September, and at all stations sampled in November, despite the fact that dissolved oxygen levels were higher at this time than they had been in August. These measurements suggested an apparent decoupling between the chemical conditions measured within 2-4 cm of the sediment-water-interface and those measured at 10 cm or more above the bottom. Sediment-related anaerobic decomposition processes appeared to dominate over aerobic processes on the bottom of WLIS during the late summer-early fall, even as water column stratification broke down and the overlying water column became re-aerated. It appeared that organic-rich sediments in WLIS might play a key role, along with temperature, in structuring the chemical environment at the sediment-water-interface where lobsters and other benthic organisms live. Questions remained, however, regarding the persistence of this apparent decoupling of water column hypoxia from bottom water chemical conditions and its development over the course of a year.

The objectives of the present study were: to obtain sediment profile images and bottom water ($\leq$ 2 cm and 1 m above the seafloor) chemical data (DO, H_2S, NH_4) from 12 selected critical stations in WLIS between the months of April and November, 2002; to obtain benthic samples from a subset of stations in order to provide detailed information on the organisms identified in the SPI images; and to investigate how key sediment quality parameters and associated bottom DO, H_2S, and NH_4 levels change over the course of the sampling period as near-bottom (1 m above) water conditions change from oxic to hypoxic in WLIS.

This paper presents a summary of the SPI and water quality data obtained to date, as this study is ongoing. Benthic samples obtained at the same time as the SPI and water samples are still in the process of being analyzed and are not reported on here.

Methods

SPI images were taken at 12 selected sites in WLIS (see Figure 3 in Cuomo & Van Patten, this volume) during the months of May, August, September, and October 2002. During each survey, a REMOTS® sediment profile camera owned and operated by SAIC was lowered to the seafloor multiple times at each of the sampling stations to obtain three replicate images suitable for subsequent analysis and interpretation. Following each field survey, the REMOTS® images were analyzed using SAIC's computer-based image analysis system. Standard measurements obtained during analysis include grain size, depth of the aRPD, infaunal successional stage, and the presence/absence of methane bubbles or surficial bacterial mats (hypoxia indicators). These parameters are all assigned numerical values that are then used to calculate an overall indicator of the condition of the site, known as the Organism-Sediment Index.

Bottom and near-bottom water samples were obtained at these same 12 sites using a Cuomo-Dobkowski (patent pending) water sampler on these same three cruises. Additional water samples from WLIS were obtained twice in August, twice in September, and once in October. The C-D sampler was slowly lowered to the seafloor where it was allowed to sit for a full minute prior to samples being taken, permitting dispersion of any pore waters and/or particles accidentally released from the sediments during the landing of the sampler. Thus, "clean" bottom water samples were taken. All water samples taken were preserved in the field and analyzed in the laboratory (within 24 hours) for hydrogen sulfide, ammonia, and dissolved oxygen according to standard EPA protocols. Sulfide samples were measured using modifications of both the Cline technique (Strickland & Parsons, 1977; Parsons, *et al.* 1984) and the Fonselius technique (Strickland & Parsons, 1977; Parsons, *et al.* 1984). The latter technique is appropriate for use with samples containing $\leq$ 300 µmol/l whereas Cline's technique is useful for samples containing $\leq$ 1000 µmol/l. Three near-bottom and three bottom water samples were taken per station, for a total of six samples per station; three replicates were run on each sample. Dissolved ammonium was done according to a modification of the Parsons, *et al.* (1984) technique, which measures the sum of NH_4^+ and NH_3. Two to three samples were taken per level per station for a total of between four and six samples per site. Three replicates were run on each sample. Dissolved oxygen was determined colorimetrically using a modification of the standard Winkler procedure (Parsons, *et al.*, 1984). Once again, three near-bottom and three bottom water samples were taken per station, for a total of six samples per station; three replicates were run on each sample.

A standard 0.04 m Van Veen grab sampler was used to collect surface sediments at several selected stations during the months of May and September in WLIS. The objective of this sampling was to ground-truth the benthic communities identified in the sediment-profile images. Each sample was preserved in 10% formalin, stained with rose bengal, and sieved down to 250 µm. All samples are in the process of being sorted and will not be reported on in this paper.

As of this writing, REMOTS® images and benthic samples have been collected on two cruises in LIS and water samples have been collected on a total of seven cruises. Had another ship been available, at least three more water-sampling cruises would have taken place–one in April and two during

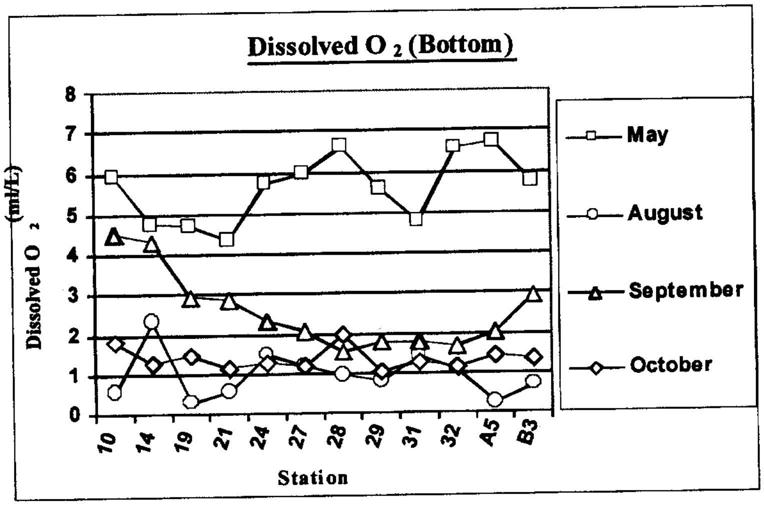


Figure 1a.

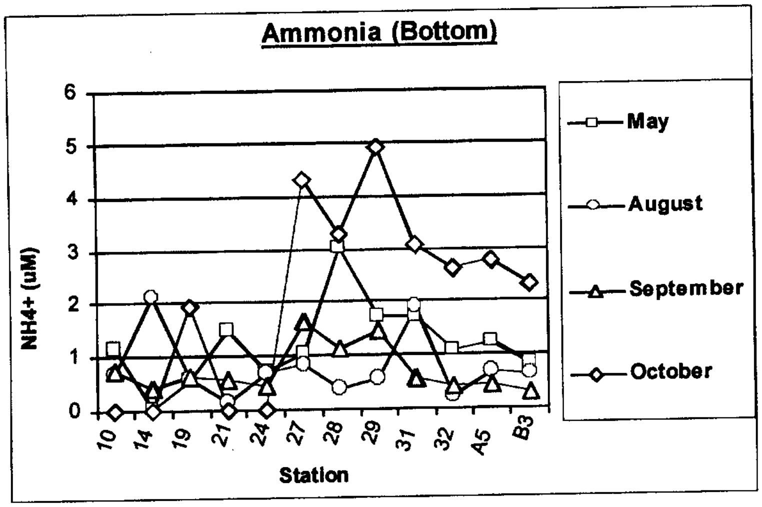


Figure 1b.

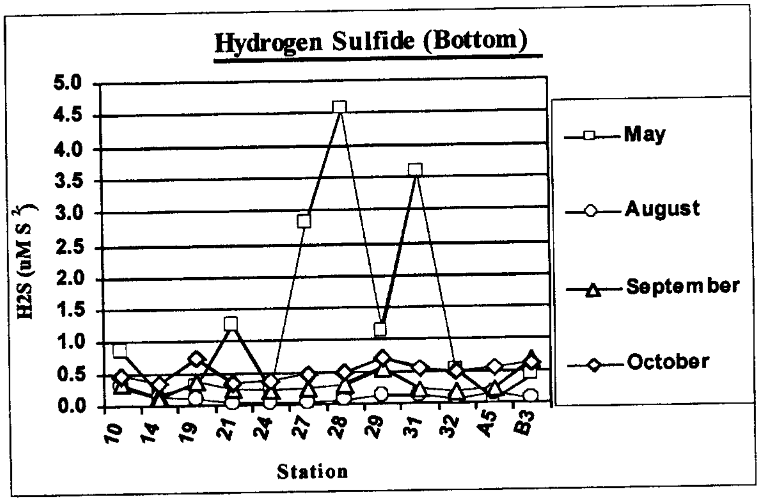


Figure 1c.

Figure 1. Bottom Water Levels of Dissolved Oxygen (a), Ammonia (b), and Sulfide (c) in 2002.

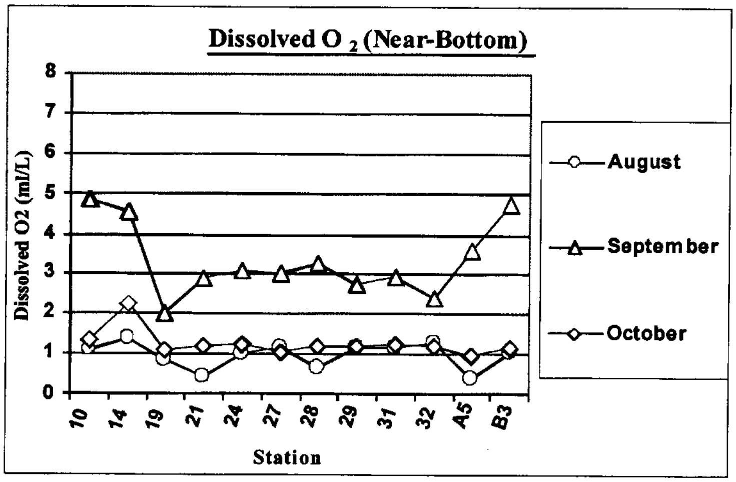


Figure 2a.

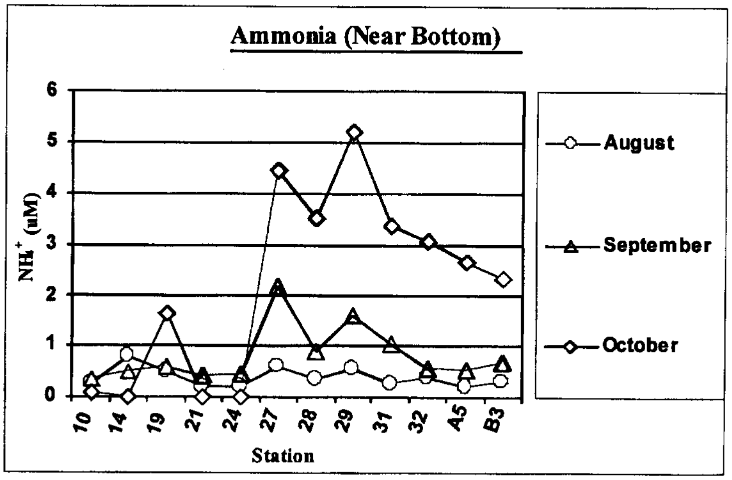


Figure 2b.

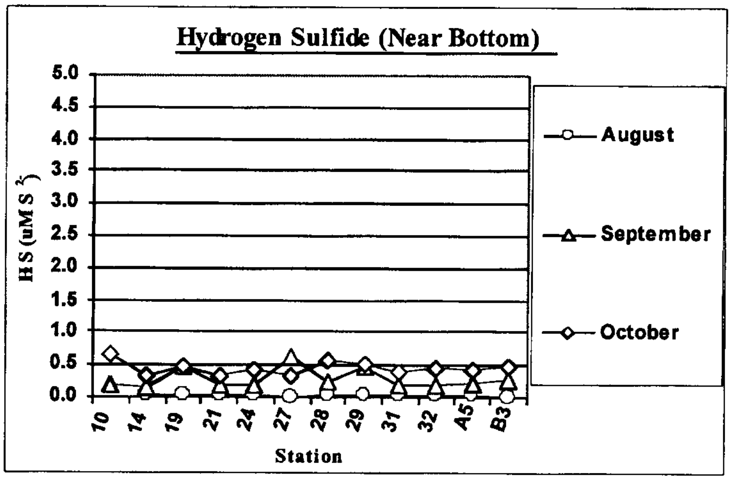


Figure 2c.

Figure 2. Near-Bottom Water Levels of Dissolved Oxygen (a), Ammonia (b) and Sulfide (c) in 2002.

the critical June-July window. Additional REMOTS® images, benthic samples, and water samples will be taken throughout the fall and early winter in order to complete this study.

Results

The results are shown in Figures 1-3. Bottom water dissolved oxygen levels (Figure 1a) were higher in May, ranging from a low of 4.2 ml/l at station 21 to a high of 6.8 mg/l at station A5, than in August, September, or October. The lowest levels recorded thus far in this study occurred during August when 11 of the 12 stations had bottom water dissolved oxygen levels below 2.0 ml/l. Except for two stations in September, bottom water dissolved oxygen levels at all stations in August and September were below 3.0 ml/l. The near bottom water dissolved oxygen levels (Figure 2a), in general, displayed the same trend as the bottom water dissolved oxygen levels, although the actual levels of dissolved oxygen were, in general, slightly higher in the near-bottom samples.

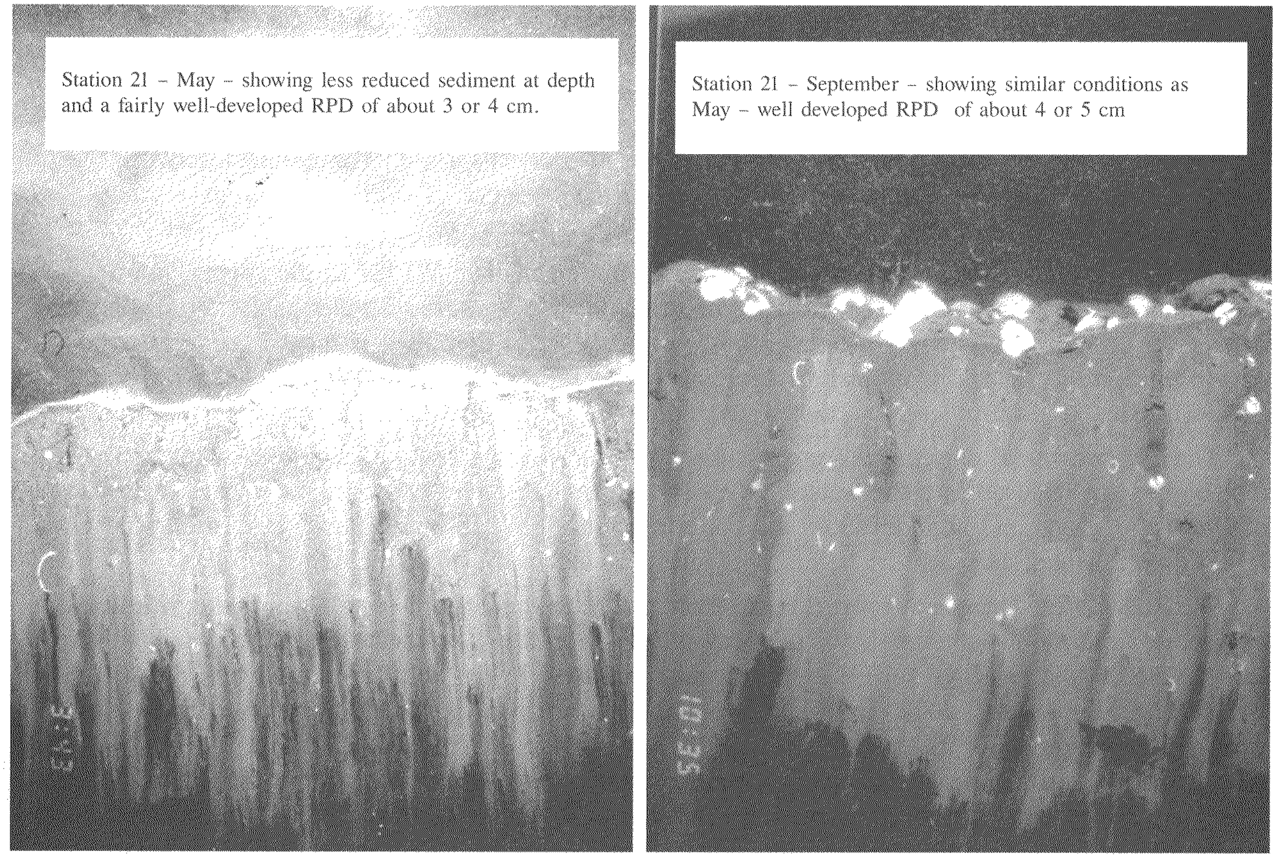
Ammonia was present in the bottom waters at all stations sampled in WLIS in May, August, and September at levels < 3.0 uM/l (Figure 1b). Near-bottom waters follow the same trends, although levels tend to be marginally lower (Figure 2b).

Sulfides were measured in higher concentrations in the bottom waters in May and were absent, for all intents and purposes, in August. In September and October, they were present in bottom waters at extremely low levels (Figure 1c). Near-bottom waters, once again, parallel the trends seen in the bottom waters; that is, there were no measurable sulfides present in the near-bottom waters in August and very little present in September and October (Figure 1c).

SPI for two stations are shown in Figure 3. Images taken in May reveal reduced sediments at depths of > 3 cm overlain by a fairly well-developed apparent oxygenated layer of surface sediments at all stations, including those in the westernmost part of the study area. Images obtained in September, however, reveal distinct differences among stations. Images taken at Station #21 (Figure 3), located in the eastern of the study area, contained similar sediment oxygenation levels (as indicated by aRPD), to those taken in May. Figure 3 also shows images obtained from Station 29, located in the westernmost portion of the study area. These images reveal mostly black, reduced sediments at the sediment surface and the almost total absence of any oxygenated sediment (measured as aRPD). Some images even reveal methane bubbles at depth in these sediments.

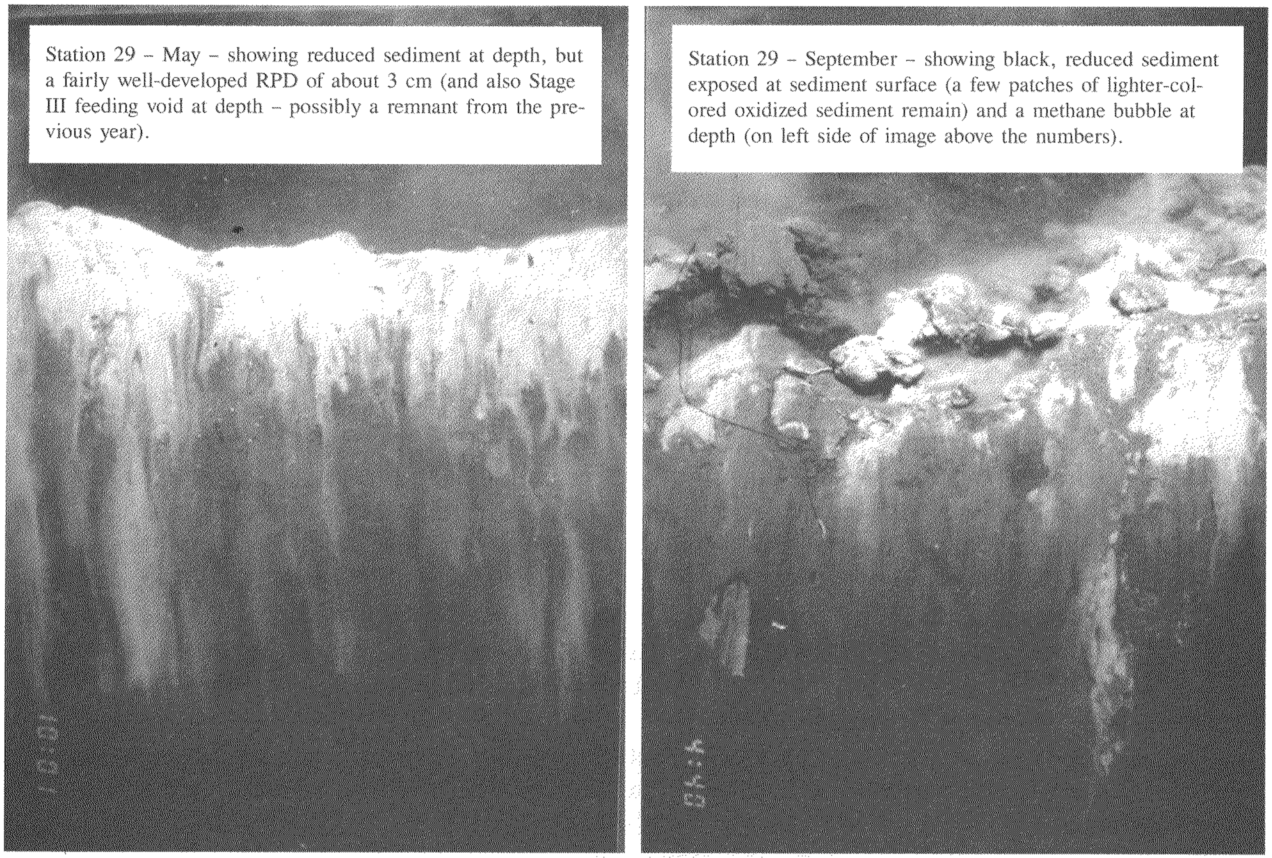
Discussion

The relationships among dissolved oxygen, hydrogen sulfide, ammonia, water temperature, water stratification, and sediment organic matter are extremely complex in Western Long Island Sound. The data collected for this study are consistent with that collected during the 2000 survey, although the fall increase in sulfides and ammonia appears to occur later in 2002 than it did in 2000. This may be a result of the differences in the extent and duration of the summer hypoxic event in WLIS between 2000 and 2002 (see Figure 1 on page 3, this volume, and maps on the CTDEP website by year at <http://www.dep.state.ct.us/wtr/lis/monitoring/lis_page.htm>.) The disconnect between measurements taken at the sediment-water interface and measurements taken more than 1 meter or so above the bottom can clearly be seen when the data collected here is compared to that collected by the state of CT DEP as part of their ongoing monitoring program of summertime hypoxia in WLIS. The DEP data show the presence of a hypoxic event in WLIS beginning in early July 2002. This event continued through August, when it reached its maximum severity. This was followed by oxygenated conditions in September. During the time of hypoxic conditions above the bottom during the summer, the bottom water oxygen levels measured in this study were consistent with the DEP results, albeit lower. Such consistency is to be expected when hypoxic conditions are being controlled by stratification.



Station 21 – May – showing less reduced sediment at depth and a fairly well-developed RPD of about 3 or 4 cm.

Station 21 – September – showing similar conditions as May – well developed RPD of about 4 or 5 cm



Station 29 – May – showing reduced sediment at depth, but a fairly well-developed RPD of about 3 cm (and also Stage III feeding void at depth – possibly a remnant from the previous year).

Station 29 – September – showing black, reduced sediment exposed at sediment surface (a few patches of lighter-colored oxidized sediment remain) and a methane bubble at depth (on left side of image above the numbers).

Figure 3. Sediment Profile Images from Two Stations in WLIS during May and September 2002.

The DEP data collected in September, however, reveal oxygenated conditions (above 3 ml/l DO) at all stations in the Sound, whereas this study found measurable levels of ammonia present in bottom waters during this same time period supporting the disconnect first observed in 2000 (Valente & Cuomo, 2000). Interestingly, the magnitude of the disconnect appears to fluctuate from year to year. Data from May, collected in this present study, also reveals a strong disconnect between the amount of oxygen present in near-bottom waters and the presence of sulfide in bottom waters.

Bottom water chemistry appears much more closely tied to sediment processes and less directly influenced by surface-down processes during the spring and fall in WLIS. In the spring, bottom water chemistry is most likely driven by the increase, as temperatures rise, of bacterial decomposition of organic materials remaining from the previous year; in the fall it is most likely related to decomposition of excess organic matter associated with fall plankton blooms, coupled with simple rapid diffusion of anaerobically-produced metabolites into the re-oxygenated overlying water column.

The May chemical data further suggest that there are, in fact, two time periods when anaerobic metabolites are released into the waters of WLIS. Such a persistent twice-yearly release of low levels of hydrogen sulfide and ammonia into the bottom waters of WLIS may contribute to a chronic low-level environmental stress on the benthic organisms living in the area, including lobsters.

The REMOTS® images support the existence of reduced sediment conditions at the sediment-surface for the month of September, yet the May images indicate a somewhat oxygenated sediment surface. However, the levels of sulfide and ammonia measured in the bottom waters were higher in May, 2002 than in September, 2002. In order to interpret this correctly, it must be remembered that sediment-color changes (aRPD) are time-averaged records of the sediment geochemical environment. Thus, sediment-profile photographs are revealing the average chemical conditions that an area has seen over several months, not the immediate. Undoubtedly, sediments become oxygenated over the winter as temperatures decrease and bacterial reaction rates drop to a minimum. This results in the development of an oxygenated zone in the sediment. In the winter months in WLIS, the depth of this zone will be a function of many things, including sediment temperature, overlying water temperature, and amount of sediment organic load. May is a time when bacterial reaction rates are readily increasing as the temperature increases; it is also a time when the spring plankton bloom brings fresh organic matter to the sediment surface. Both of these contribute to the likelihood of anaerobic bacterial processes occurring at or near the sediment surface, resulting in the release of sulfides and ammonia into the bottom waters. As the summer progresses and the sediments become dominated by anaerobic decomposition processes, the sediment "color" becomes dominated by iron monosulfides, giving the SPI an apparent RPD that is located on the surface.

Conclusions

It must be remembered that data are still being gathered even as this paper is being written, so the conclusions presented here are a summary of information found so far. A disconnect appears to exist between the chemistry of near-bottom waters, which are the ones most commonly sampled for dissolved oxygen, and that of the waters close to the sediment-water interface. The magnitude of this disconnect appears to vary from year to year. If one is to understand the factors affecting the health of benthic organisms then attention must be paid to the geochemistry of the waters that the animals are actually exposed to, which may, if fact, be different than that of waters 1 m above the sediment surface. REMOTS® images do not necessarily catch this disconnect but rather record time-averaged (months) conditions within the sediments.

The disconnect that exists in WLIS implies that bottom-dwelling organisms, such as lobsters, are subject to low levels of potentially toxic metabolites (H_2S & NH^{3+}, NH_4) during at least two time periods a year in WLIS. Exposure to low-levels of chemical stressors, combined with other environmental stressors, such as high temperatures, may be enough to impair the lobsters' immune systems and leave them vulnerable to pathogens. Such a scenario, in and of itself, might help to explain the lobster die-off of 1999.

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Relationships Between Total Nitrogen and Planktonic Chlorophyll *a* in Long Island Sound

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Introduction

Water quality in a large fraction of U.S. estuaries and other marine embayments has declined, in many cases because of overenrichment by nutrients. In particular, nitrogen has been identified as the cause of excess primary production in many marine systems, including Long Island Sound (LIS). In this paper, we present the results of a preliminary examination of the relationship between concentrations of total nitrogen and of planktonic chlorophyll a in LIS.

Data

The analyses described here utilize data collected and made available by the Connecticut Department of Environmental Protection's (CT DEP) Long Island Sound Water Quality Monitoring Program. Beginning in January 1991, monthly cruises have been conducted to collect water samples and to perform *in situ* observations. Water samples were routinely collected two meters below the surface, and five meters off the bottom (State of Connecticut, 2001). Data have been collected since January 1991 at seven stations that span most of the length of LIS from the boundary with the East River in New York City to the western end of Fisher's Island (stations A2, B3, D3, F3, H6, I2, and M3 in the map of CT DEP water quality sampling stations, page 4). Beginning in January 1995, these stations have been supplemented with an additional 12 stations that span the entire length of the Sound to the seaward boundary north of Montauk Point, New York (see map, page 4). Chemical analyses have been conducted by the Environmental Research Institute at the University of Connecticut, using strict quality assurance and quality control procedures (Environmental Research Institute, 2001).

Methods

All samples were filtered through a 0.7 mm GF/F filter. Pigments on filter pads were extracted with an aqueous acetone magnesium carbonate solution, and chlorophyll *a* concentration measured with a Turner 450® Fluorometer. Total dissolved nitrogen was measured by autoanalyzer following persulfate digestion. Particulate nitrogen was measured by high temperature combustion using a CHN analyzer. Total nitrogen is calculated as the sum of total dissolved and particulate nitrogen. Further details concerning these methods may be found in the standard operating procedures manual for the Long Island Sound Study (Environmental Research Institute, 2001).

Since the primary focus of this work is on the relationship between chlorophyll *a* and total nitrogen, only data collected near the surface were used. The analyses described here are based on data for 18 stations. Since data were available for only seven stations for 1991–1994, our analyses are confined to the period 1995-2001, to make use of the full complement of stations in a consistent manner. Early analyses showed that data for Station A2 did not fit the general pattern of data for other stations. Since Station A2 is located at the boundary between the East River and LIS, it was decided that data from this location were probably not representative of LIS as a whole, and this station was dropped from further analyses.

Results

Early analyses showed that both chlorophyll *a* and total nitrogen (TN) displayed similar spatial gradients, with concentrations increasing with westward distance from the seaward boundary of LIS, north of Montauk Point, New York. The similarity of these two spatial gradients suggested that a relationship could be found between chlorophyll *a* and TN.

Seasonal averages of chlorophyll *a* concentrations are plotted vs. seasonally averaged TN concentrations in Figure 1. Each point for a given season represents a single station. Seasons were defined as described in the legend for Figure 1. Concentrations for each season were averaged over the period 1995-2001. For each season, the points with lowest concentrations are for stations in the outer (eastern) Sound, those with highest concentrations are for stations in the innermost (western) Sound. The lines and equations are for linear regressions to these data, with $R^2 \geq 0.78$ for each season. The implication of these results is that, at least for long-term averages, there is a strong seasonal correlation between chlorophyll *a* and TN. Slopes for the regressions for summer and spring data exceed those for fall and winter. Much of the focus for the rest of this paper will be on summer data.

Interannual changes in the relationship between chlorophyll *a* and TN were explored by plotting seasonally-averaged concentrations for individual years. The results for summer concentrations (Figure 2 and Table 1) show substantial year-to-year changes in the slope of this relationship. The slope progressively declines from a value of 47.7 in 1995 to 11.0 in 1999, followed by a sudden increase to values of 60.6 and 53.0 in 2000 and 2001 respectively.

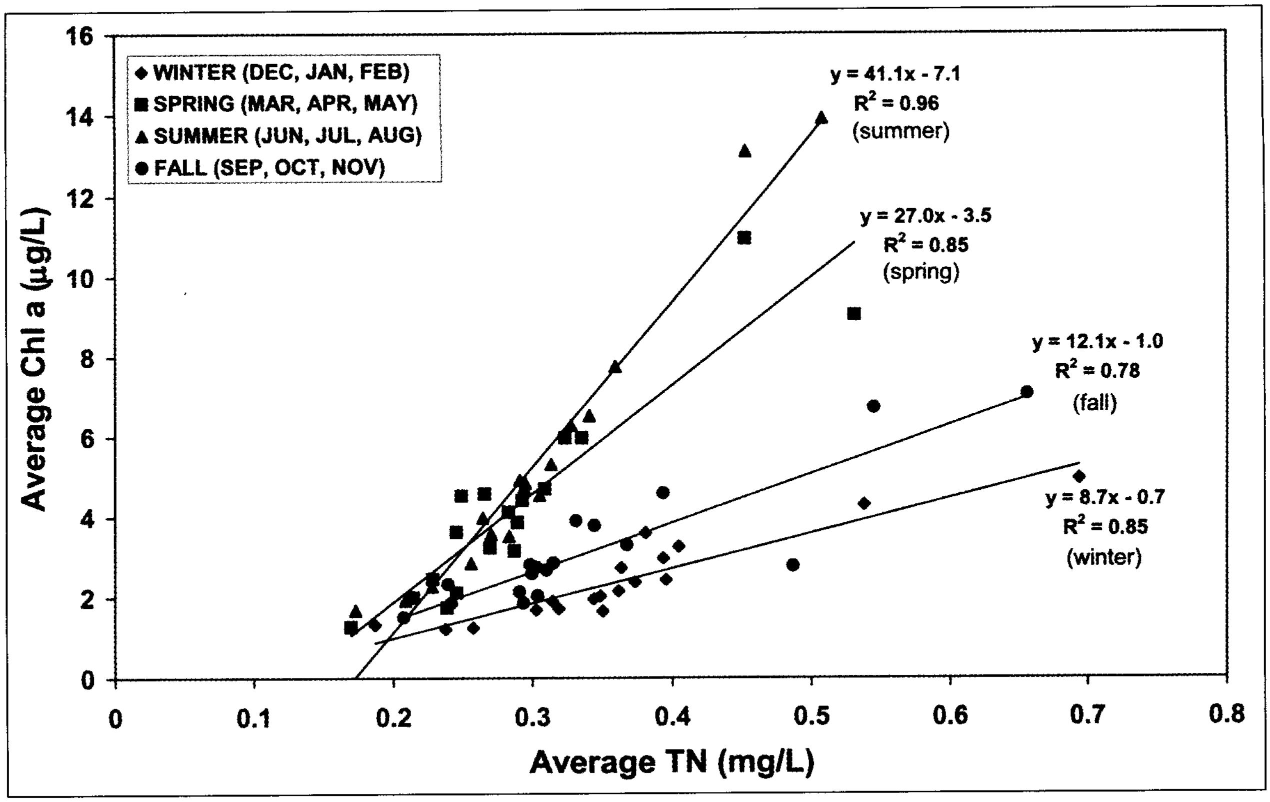

Figure 1. Relationship of seasonally averaged chlorophyll a to seasonally averaged total nitrogen in surface water. The averages are over the years 1995–2001. Each point for a given season is for a single station. The lines show results of linear regression for each season.

Table 1. Regression parameters for the chl *a*:TN relationships in Figure 2.

Year	Slope	Y-Intercept	R^2
1995	47.7	-8.3	0.79
1996	31.8	-7.7	0.88
1997	17.3	-2.5	0.84
1998	12.5	-1.7	0.84
1999	11.0	-1.9	0.70
2000	60.6	-4.0	0.78
2001	53.0	-5.4	0.56

Chlorophyll *a* concentrations in the upper water column declined significantly from summer 1995 and 1996 values during the period 1997–1999, and then increased sharply in 2000 and 2001 (Figure 3). Concentrations of TN (Figure 3) showed considerable short-term variability. There was a large increase in TN between the summers of 1995 and 1996, but a much smaller percentage decrease in summertime concentrations in 1996-1999 than for chlorophyll *a*. Thus, the decline in slope of the chl:TN relationship between 1995 and 1996 in Figure 2 is driven by a large change in summertime TN, while the steady subsequent decline in slope through 1999, followed by a sudden increase in 2000, is driven mainly by changes in summer chlorophyll *a*.

We examined a number of factors that could account for the observed trend in chlorophyll *a*, including flow rates of two major tributary rivers to the Sound, solar irradiance, concentrations of total suspended solids, wind speed, and water temperature.

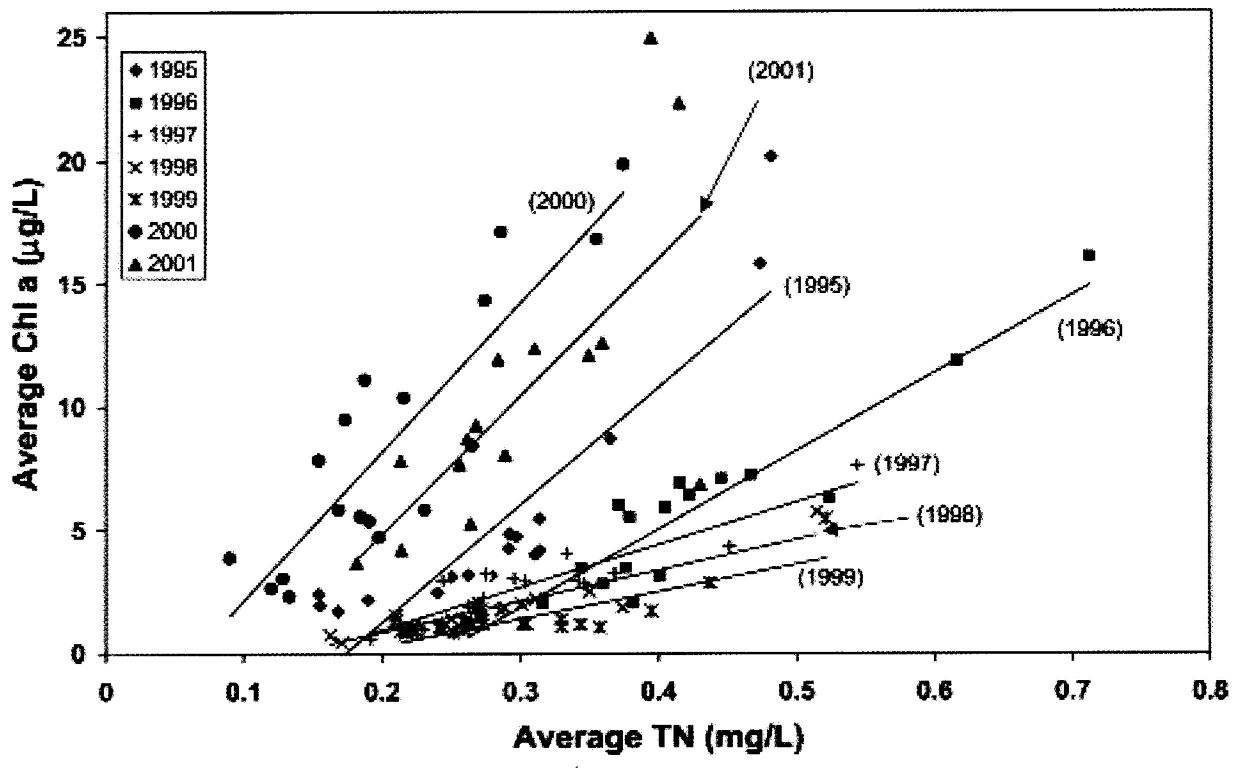

Figure 2. Relationships between summer average chlorophyll a and summer average total nitrogen in surface water for individual years. Each point for a given year is for a single station. The lines show results of linear regression for each year.

Changes in flow rates of tributaries could indicate variations in the rates of nutrient delivery to the Sound and in the flushing rate of the Sound, or be correlated with weather patterns that could affect primary productivity. Time series discharge data for 1995–2000 obtained from the U.S. Geological Survey for the Connecticut and Housatonic Rivers, two major tributaries to the Sound, showed no pattern that could explain the observed interannual changes in summer chlorophyll *a* concentrations. The only conclusion resulting from this analysis was that 1996 was significantly wetter than the other years, and therefore may have had more cloudy days and larger nutrient inputs. This may explain the high TN concentrations observed in 1996 and early 1997.

Variations in solar irradiance could indicate changes of photosynthetically active radiation due to changes in cloud cover. Preliminary analysis made use of measurements of total solar radiation (total incident irradiance) for 1995–1999 collected by The Eppley Laboratory, Inc. in a near-coastal location at Newport, Rhode Island, approximately 70 km east-northeast of the eastern boundary of the Sound, using an Eppley Precision Spectral Pyranometer. This analysis showed that mean monthly irradiance was smaller in the summer months of 1996 than in 1995 and 1997–1999. This is consistent with the increased river flows in 1996, but data for subsequent years showed no overall trends that could explain the observed changes in chlorophyll in 1997–1999. Analysis of data collected adjacent to the Sound for the full period 1995-2001 is planned.

Comparison of average summertime concentrations of total suspended solids at four representative stations that span the inner and middle Sound showed what appear to be random changes that do not explain the temporal trends observed for chlorophyll *a* concentrations.

Wind could affect vertical mixing of nutrients and phytoplankton, but preliminary analysis of monthly average wind speed at Bridgeport and at Groton, adjacent to the western and eastern Sound, respectively, show no patterns that correlate with observed changes in chlorophyll *a* concentrations. We recognize that wind speed and direction are highly variable, and that analysis of monthly data could obscure significant short-term mixing events, and plan to refine this analysis.

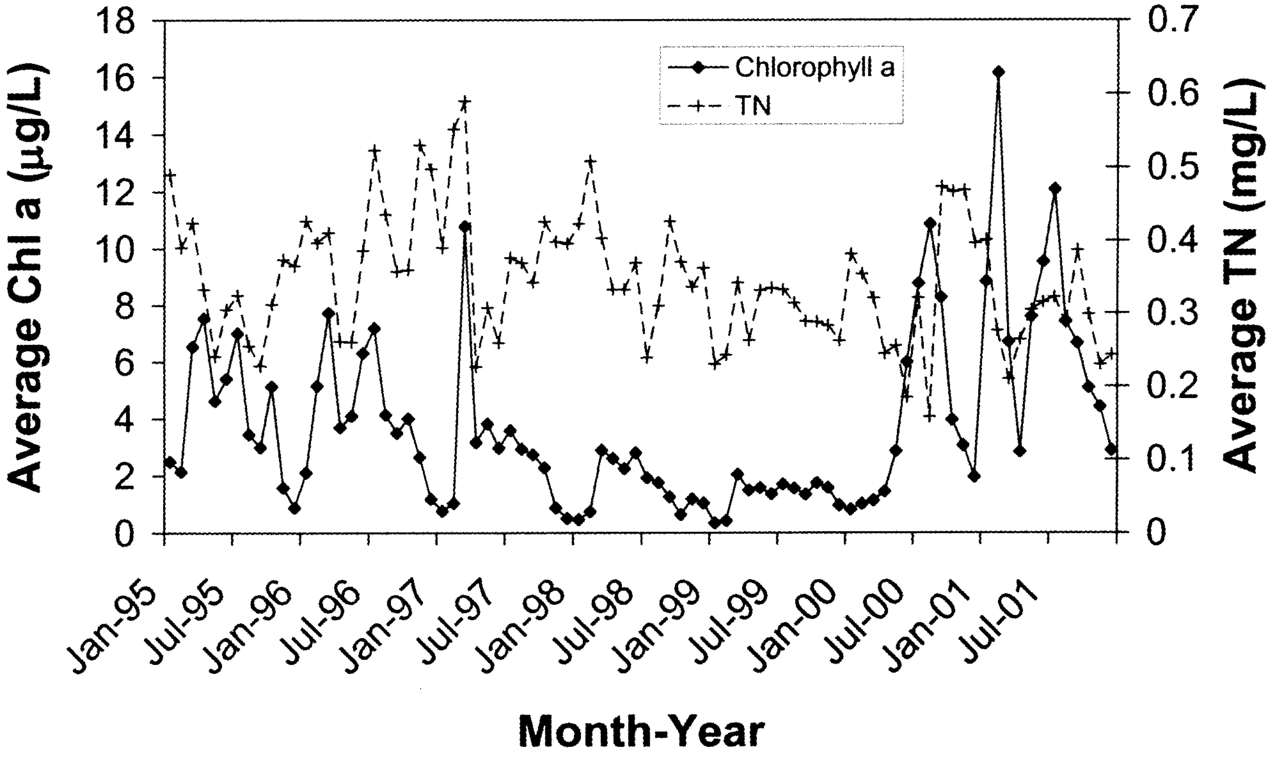

Figure 3. Monthly average chlorophyll a (solid line) and total nitrogen (dashed line) concentrations in surface water, 1995–2001. Each value is an average of concentrations at all 18 stations included in this study.

We also examined temporal trends in water temperature. While we did not find a consistent relationship between summer water temperatures and chlorophyll *a*, both surface and bottom winter water temperatures in 1997–1999 were higher than in previous or subsequent years (Table 2). Winter water temperature was calculated as the average for January, February and March, the three months with coldest water.

The linear regressions shown in Figures 1 and 2 all have negative intercepts on the chlorophyll axis, and examination of the residuals for the linear regressions suggests that fitting a nonlinear function to the data may be appropriate. Nonlinear regression with a power law function ($[\mathrm{chl}] = a\ [\mathrm{TN}]^b$, with a and b the regression coefficients) provided fits that pass through the origin with $0.75 \leq R^2 \leq 0.97$ for the data shown in Figure 1, and $0.56 \leq R^2 \leq 0.84$ for the data shown in Figure 2, similar to the range of values for linear regressions.

Table 2. Average winter (January–March) surface and bottom water temperatures, averaged over all stations.

Year	**Surface Temperature (°C)**	**Bottom Temperature (°C)**
1995	2.9	3.1
1996	1.1	1.2
1997	3.6	3.8
1998	4.4	4.3
1999	3.7	3.8
2000	3.2	3.3
2001	2.1	2.2

Summary

This work shows that it is possible to derive reasonably good long-term chl:TN relationships for Long Island Sound. These relationships vary seasonally, with higher chlorophyll *a* concentrations for a given TN concentration in spring and summer than in fall and winter. The slope of the summertime chl:TN relationship shows substantial interannual variation. In particular, the slope of this relationship progressively declines between 1995 and 1999, and then increases for 2000 and 2001.

Numerous environmental factors, i.e. river inflows, solar irradiance, total suspended solids, wind speed, and water temperature, were examined for patterns that could explain this trend. Only winter water temperature showed a pattern that appeared related to the reduced chlorophyll *a* concentrations observed in the summers of 1997–1999. These summers with low chlorophyll concentrations and a small slope for the chl:TN relationship were preceded by winters with elevated water temperatures. The differences between these wintertime temperatures and those in other years are small, but winter water temperature is the only parameter we have examined that shows a pattern that corresponds with the three years having minimum summer chlorophyll concentrations. Water was cold the winter of 1996, and mean chlorophyll concentrations did not change much between the summers of 1995 and 1996. The change in the slope of the chl:TN relationship between these two years is explained by an increase in TN concentration, perhaps because of increased riverine nitrogen loading in 1996.

Others (Oviatt *et al.*, 2002; Keller *et al.*, 1999 & 2001) have noted a relationship between a depressed or missing winter-spring phytoplankton bloom and concurrent water temperature, and have attributed these effects to higher abundance and activity of zooplankton during warmer winters. The effects noted in this study appear to be more complicated, since they occur months after the warmer winter water temperatures and appear unrelated to summer water temperature. Stachowicz *et al.* (2002) have observed decreased recruitment of native ascidians (sea squirts or tunicates) and increased recruitment of nonnative species adapted to warmer water in years with higher winter water temperatures. They also found in a laboratory study that growth rates of nonnative species exceeded those of a native species at water temperatures characteristic of LIS in summer. While our study did not have data appropriate to investigate the effects of such indirect food-web effects, the apparent relationship between summer chlorophyll and winter water temperature suggests that study of this link may be valuable, especially since water temperature in LIS and other estuaries in the region has been increasing in recent decades (Keller *et al.*, 1999; Stachowicz *et al.*, 2002).

Acknowledgments

We thank Paul Stacey, Christine Olsen, Judy Li, and Matthew Lyman of the Connecticut Department of Environmental Protection for making these data available to us, for useful discussions, and for useful background information concerning CT DEP's Long Island Sound Water Quality Monitoring Program and data quality assurance and control procedures. Solar irradiance data measured by The Eppley Laboratory, Inc. were available for 1995-1998 from data sheets maintained at the Pell Marine Science Library at the University of Rhode Island. This is contribution number AED-03-005 of the U.S. Environmental Protection Agency's Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division.

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Methylmercury Cycling and Production in the Coastal/Estuarine Waters of Long Island Sound and its River-Seawater Mixing Zones

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Introduction and Objectives

Consumption of marine fish and seafood products is the principal pathway by which humans are exposed to the very toxic organo-mercurial, monomethylmercury (MMHg). Consequently, there is an urgent need for increased knowledge and understanding of the marine biogeochemical cycling of mercury (Hg), especially MMHg, and the impact of anthropogenically related Hg inputs. Biologically productive, nutrient-rich near shore regions, which support major commercial and recreational fisheries, are of special interest. Accordingly, our EPA-STAR Hg research is focused on Long Island Sound (LIS), its watershed, and river-seawater mixing zones. This major natural resource can provide a valuable analog for other near shore/urban marine ecosystems. Our reaction-speciation focused investigations are designed to allow the results to be applied in other marine regions. Such an approach is essential given the complexity and variability of fertile estuaries and adjacent coastal waters, which are major repositories for natural and pollutant riverborne/watershed derived substances such as Hg. Our specific objectives are concerned with several major features of the aquatic - biogeochemistry of Hg, particularly sedimentary MMHg production, elemental mercury (Hg^0) cycling and emissions, interactions between terrestrial watersheds, rivers and near shore marine waters, and the role of organic matter in governing the availability of Hg for competing methylation/reduction reactions.

Methylmercury Production in Sediments

We are investigating the relationships between microbial activity and sediment geochemistry in affecting Hg methylation in Long Island Sound, focusing on complex linkages between sulfur chemistry, Hg speciation, and organic matter. A preliminary mass balance of MMHg in Long Island Sound (Langer *et al.*, 2001; Vandal *et al.*, 2002) suggests *in situ* production as the main source, and we hypothesize that most is synthesized microbially in sediment. Our hypothesis is that Hg^0 is the predominant Hg cycling product of bacterial activity in the oxic zone (i.e., surface mixed layer of water column), while net *in situ* synthesis of MMHg is most significant in redox transition zones (i.e., shallow sedimentary regimes). Additionally, we hypothesize that *in situ* sedimentary production (sulfate-reducing bacteria) and mobilization is the primary source of MMHg to the waters and biota of LIS, and may account for up to 75% of MMHg in fish (Hammerschmidt *et al.*, 2002b).

We examined factors influencing MMHg in sediment at three representative sites spanning the benthic trophic gradient in LIS, ranging from fine-grain, organic-rich substrate in the west to sandy, low organic material in the east. Surveys were conducted in August 2001 and March 2002 aboard the Connecticut Department of Environmental Protection (CT DEP) vessel R/V *John Dempsey*, and in November 1999 and June 2002 aboard the UCONN vessel R/V *Connecticut*. Surface sediment total Hg and MMHg increase from east to west in LIS. Sediment MMHg concentrations were dependent on the availability of Hg and were directly related to the organic matter content of the sediment (Hammerschmidt & Fitzgerald, 2001; Hammerschmidt *et al.*, 2002a; 2002b). However, rates of ^{200}Hg (stable isotope) methylation in August 2001 were greatest in sandy, low-organic LIS sediment.

We hypothesize that availability of dissolved ionic (labile) Hg substrate controls the methylation rate in LIS sediments. Organic matter and AVS affect Hg methylation in LIS by influencing partitioning of inorganic Hg between dissolved and particulate phases, thereby regulating the availability of dissolved Hg to methylating bacteria (Hammerschmidt *et al.*, 2002b). In low-organic sediments, where (sulfate-reducing) bacterial activity is also low, more of the inorganic Hg is methylated because HgS^0 (bioavailable Hg) is the dominant dissolved Hg-sulfide complex as a result of minimal sulfide accumulation. Sediments collected in March and June of this year are expected to support this hypothesis and show the seasonal effects of temperature and autochthonous organic matter inputs on Hg methylation in LIS sediments.

Elemental Hg in Long Island Sound

Cycling of elemental Hg (Hg^0) plays a governing role in the aquatic biogeochemistry of Hg and bioavailable Hg species (MMHg). Aqueous production of Hg^0 competes for reactant (i.e., labile/ionic Hg) with the *in situ* biological synthesis (methylation) of MMHg, and high Hg^0 production may result in less MMHg in biota and less total Hg in sediment. It has been shown that *in situ* Hg^0 production in the waters of LIS and emissions to the local/regional atmosphere are major processes (Rolfhus & Fitzgerald, 2001), and extensive investigations of the air-sea partitioning of Hg are being conducted in LIS. We are testing the hypothesis that the Hg^0 distribution in LIS is spatially/temporally variable, related to the distribution of labile Hg (labile inorganic and organically associated Hg species), and the *in situ* supply of reducing agents (bacterial activity and solar radiation).

Our investigations of LIS have added to our understanding of the seasonally varying DGM (dissolved gaseous mercury; approximately 99% Hg^0 in seawater) distribution in LIS. We are using an Automated aqueous Gaseous Elemental Mercury sampling and analysis System (AGEMS), designed by our laboratory for shipboard use (Balcom *et al.*, 2000; 2002), which allows for direct analysis of Hg^0 in surface waters. Field measurements are supported by laboratory analyses using a semi-automatic Dissolved Elemental Mercury Analyzer (DEMA; Tseng *et al.*, in press-2003). In addition to the earlier (1995-1997) seasonal surveys of Rolfhus & Fitzgerald (2001), AGEMS surveys of DGM in LIS were conducted on-board the CT DEP vessel R/V *John Dempsey* during the spring and summer of 1999, 2000, 2001 and 2002. Samples for Hg speciation measurements were collected at selected surface sites during the surveys, and the CT DEP Water Quality Monitoring Program provided nutrient and other water quality data.

Our DGM surveys have elucidated spatial/seasonal patterns in the distribution of Hg^0 in LIS and their correlation with hydrographic conditions. Rolfhus & Fitzgerald (2001) revealed DGM maxima in central LIS, and elevated concentrations during the warm months. DGM often decreases with decreasing salinity west of the Connecticut River (salinity decreases from east to west in LIS), particularly during the spring. These spring distributions are attributable to enhanced Hg^0 production at a labile Hg source (e.g., Connecticut River; Rolfhus *et al.*, submitted), with the distribution maintained by biological production and simple mixing of DGM in LIS (Rolfhus & Fitzgerald, 2001). Hg^0 surface maxima have been observed along the central axis of LIS west of the Housatonic River in May and September, in the region between two bottom sills that traverse LIS (Balcom *et al.*, 2002). Since river inputs in the region do not support this Hg^0 production, we hypothesize that there is a hydrographically-generated supply of labile Hg from deep waters in this region. Inputs of labile Hg from deep water mixing (e.g., degradation of organic matter at depth), as indicated by surface peaks in nitrate, appear to supply Hg reactant for DGM production at the surface following breakdown of thermal stratification in August/September.

Supersaturation of Hg^0 in all seasons indicates that Hg is lost from the Sound to the atmosphere via evasion. The average calculated Hg^0 flux (Wanninkhof, 1992) for the earlier 1995-1997 surveys was calculated as 334 p moles m^{-2} d^{-1} (Rolfhus & Fitzgerald, 2001). AGEMS surveys conducted from 1999 through 2002 have produced a similar average calculated flux of 300 p moles m^{-2} d^{-1}. General trends include increased Hg^0 flux with increased Hg^0 concentrations during warmer months and at higher wind velocities. Annual emissions from LIS are estimated at 85-90 kg (Rolfhus & Fitzgerald, 2001; Balcom *et al.*, 2000), which indicates remobilization of approximately 35% of the Hg inputs (230 kg/y) to LIS (Fitzgerald *et al.*, 2000; Vandal *et al.*, 2002).

Hg-Organic Interactions

Organic matter-Hg interactions are the major control over the behavior and fate of Hg in aquatic systems. Therefore, organic matter governs the availability of Hg for competing methylation/reduction (Hg^0 production) reactions. Complexation of inorganic mercury cations (Hg^{2+}) by natural organic compounds has been posited as an influential and often controlling feature of the aquatic biogeochemical cycling of this toxic metal, and is a working hypothesis for the present study. While many studies suggest that the majority of Hg present in natural waters is complexed with natural dissolved organic matter (called "ligand"), little quantitative information currently exists regarding the abundance and strength of such Hg-complexing agents in natural waters (Lamborg *et al.*, submitted). This is due in large measure to the lack of a suitably sensitive and reliable technique with which to probe Hg speciation. A new method has been developed in our laboratory (Lamborg *et al.*, submitted) for the determination of activities and conditional stability constants of dissolved organic matter towards Hg using an *in vitro* reducible-Hg titration approach. This is a wet chemical analog to the electrochemical titrations now in use for ligand studies of many other trace transition metals in natural waters. Results indicate that complexing agents are present in the dissolved-phase (<0.2 µm) at ca. <1-60 n*N* concentrations and with log conditional stability constants (logK') in the range of 20-25 (Lamborg *et al.*, submitted). Such ligand characteristics indicate that the majority of Hg^{2+} dissolved in fresh ($>99\%$) and coastal salt waters ($>50\%$) is associated with organic complexes.

The major flux of Hg delivered to the Sound from the Connecticut River, its largest source of fresh water, is the result of watershed leaching. Spring runoff contributes large amounts of Hg to the Connecticut River that is tightly bound to dissolved and colloidal organic ligands and particulate matter, which are largely unreactive (not reducible with Sn(II); Rolfhus *et al.*, submitted; Lamborg *et al.*, submitted). Based on analysis of the ligand activity through the salinity gradient of the Connecticut River (conservative type distribution with activity decreasing at higher salinity), it is clear that ligands in Long Island Sound are of terrestrial origin. Additionally, an inverse relationship between water flow and ligand activity was observed in the Connecticut River. Though data are few, they suggest that the watershed releases a relatively fixed amount of ligand that is diluted to a greater or lesser extent in the river depending on the hydrographic conditions (Lamborg *et al.*, submitted). We have found that estuarine reactions (i.e., mixing of river borne Hg species with seawater high in Cl^- and major cations) increase the labile Hg fraction available for reduction or methylation. This physicochemical transformation results from shifts in speciation associated with the presence of inorganic complexing ligands (i.e., Cl^- ions) in coastal seawater, and displacement of sequestered Hg in river water by the increased activity and competition from cations such as Ca^{++} and Mg^{++} (Rolfhus *et al.*, submitted). Therefore, the river-coastal water mixing regime is one of the most critical parts of the watershed. Similar trends are also observed in LIS proper.

Implications

The estimated sediment-water flux of dissolved MMHg (~ 10 kg y^{-1}) is consistent with the preliminary mass balance for LIS, pointing to sedimentary production and mobilization of MMHg as the principal source (Hammerschmidt *et al.*, 2002a, 2002b). Additionally, bioaccumulation estimates suggest that most of the MMHg in LIS plankton is attributable to sedimentary synthesis and mobilization. Assuming that all of the sediment-derived MMHg is accumulated by phytoplankton in LIS, which has 200-400 g C m^2 y^1 primary production, we predict that they would have 0.7-1.4 ng MMHg g^{-1} wet weight. Our average measured level of MMHg in suspended particulate matter of

LIS, most of which in autochthonous, is 0.9 ng g^{-1} (Hammerschmidt *et al.*, 2002b). It is likely that most of the MMHg in higher trophic levels of LIS, which ultimately derive organic matter and MMHg from primary producers, also can be attributed to Hg methylation in the sediments. We project that 40-75% of the MMHg in LIS fish is from methylation of inorganic Hg deposited to sediments, both recently and historically.

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Stratigraphic Control of Salt Marsh Bank Retreat in the Housatonic Estuary

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Abstract

The wetland complex in the lower-most Housatonic estuary is part of the Charles E. Wheeler Wildlife Management Area and the Stewart B. McKinney Wildlife Refuge and is significant both in terms of its ecology and its size. Portions of the salt marsh complex are undergoing bank retreat at rates locally as high as 1 m/yr. The high marsh habitat is particularly threatened. Bank failure in many areas is driven by erosion of low cohesion, sand-rich layers by a combination of wave and current action, and probably by groundwater sapping and tidally-induced groundwater fluxes. When the weight of the undercut block exceeds the tensile strength of the overlying material, a block fall occurs with blocks being on the order of 2 m long, 1 m high, and 0.5 m deep. Subsequent wave and current action disaggregates the blocks. The low cohesion, sand-rich strata appear to be storm deposits and have been correlated with historical hurricanes. Age-depth relationships were estimated via comparison with ^{14}C and ^{210}Pb data obtained by researchers from Wesleyan University for a core taken nearby. Significant erosional strata correlate well with the hurricanes of 1783, 1788, and 1815. Another erosion-prone horizon consists of glacial outwash sands that underlie the marsh. Besides the presence of weak horizons in the marsh stratigraphy, other controls on bank erosion appear to be exposure to high-energy wave action, e.g., via boat wakes, and potentially the interaction of tides and sea level change in determining the vertical distribution of energy against the bank.

Introduction

The Charles E. Wheeler Wildlife Management Area is a salt marsh complex in the estuary of the Housatonic River (Figures 1). The 840-acre marsh is protect by the Milford Point barrier beach. It is a relatively young marsh complex (most of the expansion occurred in the early 1900's), and is an important breeding and feeding area for birds, such as clapper rails, black ducks, ospreys, egrets, and herons, as well as a number of species that merit special concern. The surrounding waters serve as a nursery ground for striped bass, bluefish, and summer flounder (Jonas, 2002). Over the last century, the Wheeler salt marsh complex has escaped many of the anthropogenic alterations, such as mosquito ditching, infilling, and artificial flow restriction, that have plagued other Connecticut salt marshes.

In conjunction with the adjacent Great Meadow wetlands in the Stratford-Bridgeport area, the Wheeler marsh complex is one of Connecticut's largest tidal wetlands. The core of the wetland complex is Knell's (Nell's) Island (Figure 1). In the late 1800's, the core of Knell's Island comprised all of the tidal marshland in the lower Housatonic estuary. Most of the area that is now Wheeler salt marsh was a shallow, sandy oystering bay (Figure 1). Based on a review of historical maps and aerial photographs, the growth of the majority of the salt marsh occurred between the 1890 and 1940. Knell's Island also expanded during that period to approximately three times its 1890 size (Figure 1). Today much of Knell's Island is high marsh and provides the principal high marsh habitat in the

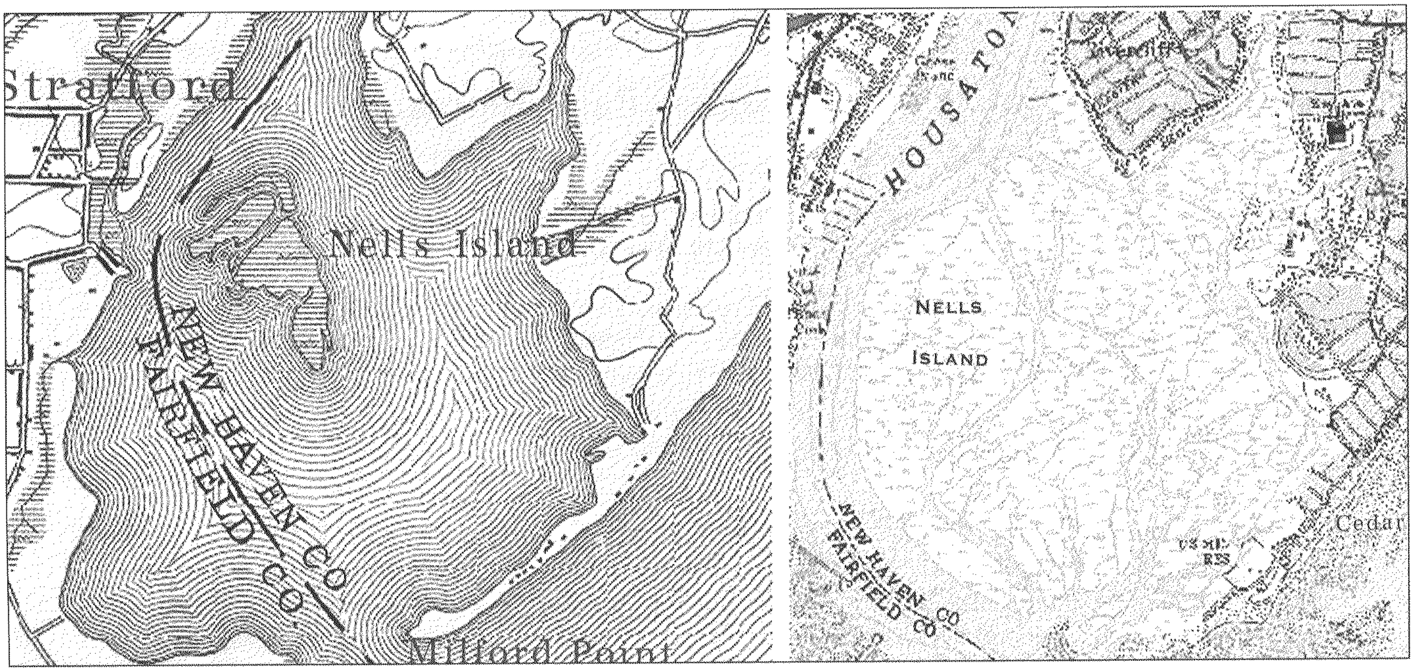

Figure 1: Historical maps showing marsh expansion and growth of Knell's Island for the period of 1893 (left map) to 1984 (right map). (Map source: 1893 15-min and 1984 7- min (USGS) quadrangles). The maximum north-south length of the island in the 1984 map is approximately 1340 m.

Wheeler salt marsh complex.

Field observations at Knell's Island in 1998 revealed numerous areas of bank erosion, especially along the island's northern and western boundaries. This represents a clear reversal of the historical trend of rapid marsh expansion. The most severe erosion is occurring along the boundaries of Knell's historical core, the area that comprises the high marsh habitat for the Wheeler marsh complex. A system for monitoring bank erosion rates was established and data collected during the period from 1998 to 2001. In order to understand the causes of bank retreat, studies of the geomorphic mechanisms at work were initiated.

Data Collection

Short-Term Erosion Rates

Benchmarks were established at 50-m intervals during the summer of 1998 around the western and northern perimeter of Knell's Island creating a baseline for subsequent measurements. Distances from this baseline to the edge of the island were measured at 20-m intervals and at each benchmark. During the summer of 2000 and 2001 (Jonas, 2002), subsequent measurements were made. Due to difficulties in precisely defining the marsh edge, cumulative measurements of change in position (over the period of 1998 to 2001) of the marsh edge <0.5 m were defined as unreliable and changes >0.5 m were considered reliable. Thus, erosion and accretion rates <0.17 m/yr (0.5m/3yrs) were termed uncertain. An average rate of accretion or erosion was calculated for each 50-m segment over the three-year period.

Bank Statigraphy

In 2001, an approximately 1-m long core was collected near the northernmost benchmark, where the most severe erosion was occurring. The core was continuously logged and refrigerated in the laboratory for preservation. Each of the distinct layers was visual described for composition, color,

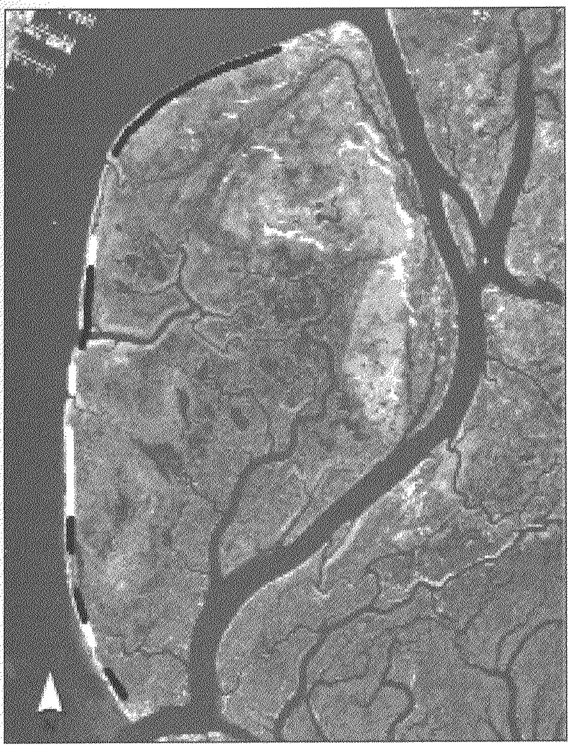

Figure 2: Distribution of erosion and accretion along the western shore of Knell's Island. Areas of erosion (0.17 to 0.8 m/yr) are in black, areas of accretion (0.17 to 1.8 m/yr) are in white, and areas of uncertainty (<0.17 m/yr) are left blank (Aerial photograph source: Connecticut Dept. of Environmental Protection).

and the coarseness of the grains. Each distinct layer was assigned a letter. Grain size analysis was then performed on samples from each layer through dry sieving, and in addition wet sieving for samples with abundant aggregates. The marsh bank stratigraphy was then compared to the depth of each layer observed in the core.

Storm and Flood Correlations to Sand-Rich layers

Correlating the sand-rich layers with storm and flood events was a multi-step process. First, records regarding storm and flood for Southern New England were compiled in order to determine the date of occurrence, severity of event and proximity to Knell's Island. Next, age-depth relationships were estimated via comparison with ^{14}C and ^{210}Pb data obtained by researchers from Wesleyan University (Varekamp *et al.*, unpublished data) from a core taken nearby. Finally, the storm events were correlated with the approximate age of each sand-rich layer.

Results and Conclusions To Date

Although there are portions of the western side of Knell's Island where accretion is observed, the predominant geomorphic change is bank retreat. Retreat rates are especially high along the northern side of the island with average retreat rates ranging from 0.5 m/yr to 0.8 m/yr (Figure 2). Along the western edge of the marsh, retreat rates range from 0.2 m/yr to 0.6 m/yr.

Bank failure in many areas is driven by erosion of low cohesion, sand-rich layers (Figure 3) by the interaction of waves and currents, and probably by groundwater sapping and tidal pumping. When the weight of the undercut block exceeds the tensile strength of the material, a block fall occurs. Measured fallen blocks are approximately 2 m long, 1 m high, and 0.5 m deep. Subsequently wave and current action disaggregates the fallen blocks.

The low cohesion, sand-rich strata appear to be storm deposits and have been correlated with historical hurricanes (Figure 3).

Significant erosional strata correlate reasonably well with the hurricanes of 1783, 1788, and 1821. Another erosion-prone horizon consists of glacial outwash sands that underlie the marsh. Besides the presence of weak horizons in the marsh stratigraphy, other controls on bank erosion appear to be exposure to high-energy wave action, e.g., via boat wakes, and potentially the interaction of tides and sea level change in determining the vertical distribution of energy against the bank.

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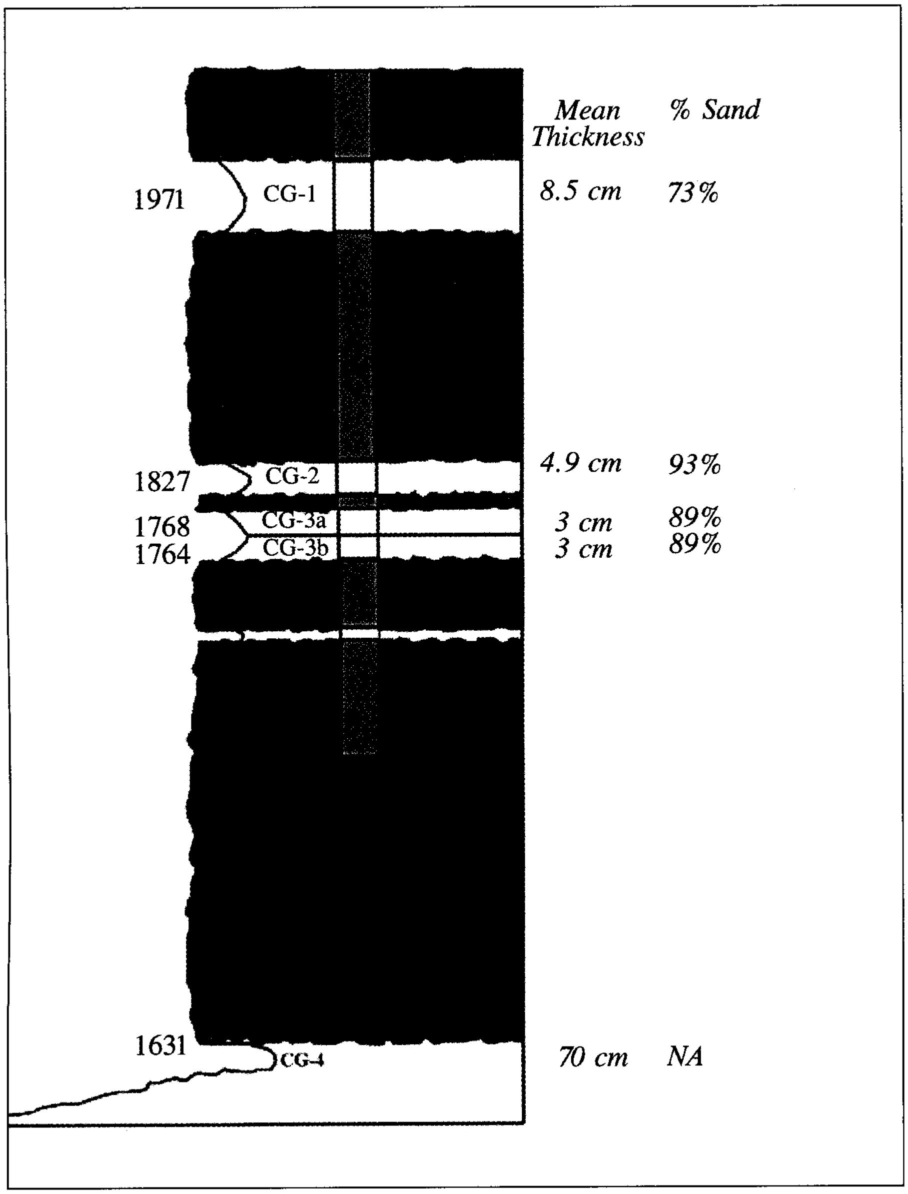

Figure 3: Generalized stratigraphic profile of Knell's Island. The dates listed on the left represent the inferred storm event date.

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Mandibular Organ Inhibiting Activity Identified from Lobster Crustacean Hyperglycemic Hormones (CHHs)

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Introduction

Lobster crustacean hyperglycemic hormones (CHHs) are a family of neuropeptides that regulate many physiological processes and are also secreted in response to stresses. These stresses include heat shock, anoxia, (Chang *et al.*, 1998, 1999) and possibly insecticide exposure. The CHHs are synthesized mostly in the sinus gland X-organ complex (SG) of the eyestalk. The CHH hormones, between 72 and 81 amino acids in length, are multifunctional. That is, while most of them are involved in mobilizing sugars into the hemolymph (CHHa and CHHb), particular ones have additional functions such as molt inhibiting hormone activity (MIH), while others have gonad inhibiting hormone activity (GIH). GIH has also been identified as vitellogenin inhibiting hormone (VIH). Other activities such as osmoregulation capacity (OC) have been attributed to CHHs (Charmantier-Daures *et al.*, 1994). CHHs from several species of crustaceans including lobsters have been identified, sequenced, and cloned (Tensen *et al.*, 1991; de Kleijn *et al.*, 1995; reviewed by Böcking, Dirksen and Keller, 2001). While mandibular organ inhibiting hormones (MOIHs) have been identified as being members of the CHH family (Liu & Laufer, 1996) and subsequently sequenced and cloned from the spider crab *Libinia emarginata* (Liu *et* al, 1997), no MOIH has been identified in the lobster *Homarus americanus.*

MOIHs inhibit the synthesis of methyl farnesoate (MF), which is a close homologue of insect juvenile hormone (JH). The mandibular organ (MO) is homologous to the insect corpus allatum. Both of the secretions from these structures enhance reproduction of the adult and control morphogenesis and metamorphosis in earlier stages of the growing organism (Laufer & Biggers, 2001). The close structural similarity of MF to JH makes the use of larvicidal substances that interfere with metamorphosis of insects such as methoprene, candidates for interfering with the same processes of metamorphosis and adult reproduction in crustaceans.

In the present study we used several methods and techniques to investigate the MOIH activity of *H. americanus* CHHs in *in vitro* bioassays.

Methods and Materials

We devised a technique of MO cell dissociation (Laufer *et al.*, 2002), to compare the ability of specific CHH antibodies to reverse the inhibitory effects of SG extracts on MF synthesis by dispersed MO cells in vitro. In other dissociated MO cell assays we tested whether a recombinant CHHb peptide expressed by Pichia yeast cell cultures possessed MOIH activity.

Animals–American lobsters, *Homarus americanus*, from Long Island Sound–were destalked 7-10 days before MF synthesis assays were conducted. Preparation of MO cells was accomplished by incubating the glands in 0.1% collagenase (Worthington CLS 3) for 20 minutes with occasional vortexing as previously described (Laufer *et al*, 2002). Sinus glands were dissected from frozen eyestalks in cold Pantin saline and homogenized in a glass micro-homogenizer, transferred to a 1.5mL microfuge tube in which they were centrifuged for 3–5 minutes at 5000 rpm. One fifth and 2/5 sinus gland equivalents were applied to cultures of dissociated cells. Other similar MO cell cultures received peptides from yeast cloned CHHb cultures. Other MO cell cultures received SG extracts previously absorbed with CHH antibody. Each culture received 4 µL of H3-Methyl methionine with a specific activity of 71 Ci/µmol. The synthesis of MF was compared in MO cultures with and without experimental treatment.

Results

Antibody absorption assays performed with SG extracts to determine if MIH, CHHa and VIH antibodies can reverse MOIH inhibition by SG extracts showed that MIH and CHHa antibodies did not significantly affect the ability of SG extracts to inhibit MF synthesis by MO cells. Antibodies to VIH however, reversed the ability of SG extracts to inhibit MF synthesis by MO cells by 50%. Media containing CHHb peptides expressed by Pichia cell cultures assayed for MF synthetic activity of MO cells in culture also showed a marked response. Fifty mL of Pichia media was equivalent to 0.4 SG in terms of MOIH activity. That is, 50mL of CHHb media contained similar amounts of MOIH activity as 0.4 SG extract.

Conclusions

We conclude that both CHHb and VIH or GIH from lobster exhibit significant amounts of MOIH activity in in vitro bioassays. Furthermore, they may account for most of the MOIH activity present in SG-X organ complexes and thus appear to be lobster MOIHs. There may be additional MOIHs in SG-X organ preparations, which we have not been able to detect with our present procedures. It should be pointed out that the levels of MOIH detected are significant and appear to be in a physiological range. That is, activity equivalent to 0.4 SGs was detected by the CHHb protein, and by the VIH (GIH) specific antibody.

Most significant is the finding that lobster eyestalks contain MOIHs. These members of the CHH family respond by increasing in response to environmental stressors such as temperature shock and anoxia, according to Chang *et al.* (1999). Other stressors such as insecticides, for example, methoprene if in sufficient amounts, should also increase these CHHs. Since MOIHs are among the CHHs as a class, it indicates that environmental stressors will shut down or drastically curtail or interfere with such activities as reproduction, growth, morphogenesis and metamorphosis of the lobster through enhanced MOIH production. Such physiological alterations in metabolism provide the lobster with valuable adaptations for dealing with environmental stressors for it allows them to shunt their energies from activities such as growth and reproduction to carbohydrate mobilization for maximum swimming ability and escape from harmful environments.

Acknowledgments

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Temporal and Spatial Variability of Chlorophyll in Long Island Sound

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Introduction

Phytoplankton, as well as other trophic components of marine ecosystems, exhibit temporal and spatial variabiltiy (Reid *et al.* 1987, Cushing 1982). Phytoplankton variability occurs in various patterns such as cycles, trends, fluctuations, events and pulses. The variability occurs at various scales from hours or less, daily, seasonally and interannually (Smayda 1998). Causes of variability are complicated. Variability could be climatologically induced, such as temperature, precipitation, wind speed and direction, and circulation patterns of the water. In fact, natural variability is a characteristic of phytoplankton (Colijn 1998, Smayda 1998). Large-scale atmospheric phenomena, such as the North Atlantic Oscillation (NAO), have been documented to influence regional precipitation and snowfall and are related to the variability to sea surface temperature (Bates 2001, Vinje 2001). NAO has also been related to the change in phytoplankton biomass and other components in the marine ecosystem (Edwards *et al.* 2001, Planque and Taylor 1998). However, the mechanisms through which NAO affects marine ecosystem are often not known (Irigoien *et al.* 2000, Conversi *et al.* 2001). Anthropogenic processes such as nutrient enrichment can also cause phytoplankton variability. Thus it is important to characterize long term variability of chlorophyll in different regions, not only as a feature of natural variation, but as an essential step in detecting any anthropogenic effects on phytoplankton dynamics in coastal waters (Li and Smayda 1998, Smayda 1984). In this article, we describe the inter-annual variability of phytoplankton in Long Island Sound based on monthly measurements made since 1991. This is particularly meaningful because the input of nitrogen to the Sound has been reduced in recent years, in order to reduce the hypoxia in the Sound (Stacey *et al.* 2000, Stacey 2002). The reduction in nitrogen loading is expected to continue with a goal of 58.5% total reduction by 2014. The long-term variability of phytoplankton in the past eleven years can provide baseline information against which any change due to the reduction of nitrogen may be detected.

Materials and methods

Monthly water samples were collected from 18 stations throughout Long Island Sound (CT DEP monthly water quality monitoring stations on page 4). Samples were started in 1991 at 7 axial stations, and 11 stations were added in 1994. Water samples were taken from both surface (2 m from surface) and bottom (approximately 5 m off the bottom). Chlorophyll was measured fluorometrically. In addition to chlorophyll, nutrients were also measured including ammonia (NH_4^+), nitrate plus nitrite ($NO_3^- + NO_2^-$), total dissolved nitrogen (TDN), particulate nitrogen (PN), dissolved inorganic phosphorus (= orthophosphate, DIP), particulate phosphorus (PP), dissolved organic carbon (DOC), particulate carbon (PC), dissolved silica (SiO_2), particulate (biogenic) silica (BioSi), total suspended solids (TSS) and biological oxygen demand (BOD). CTD profiles including temperature, salinity, depth and irradiance were taken at each station.

Data were tested for normality prior to the trend analysis. The Student Range test was used to test normality using EPA's software Dquest. The commonly used Kolmgorov-Smirnov test was not chosen because the test tends to be conservative with large data sets (Sokal and Rohlf 1981). For normally distributed data, simple linear regression analysis was used for trend analysis. For data that were not normally distributed, non-parametric statistical procedure (seasonal Kendall's tau) was used for trend analysis (Hirsch *et al.* 1982, Hirsch and Slack 1984).

Results and discussion

Chlorophyll in Long Island Sound showed a distinct west to east gradient, high in the west (8 $\mu g\ l^{-1}$) and low in the east (about 2 $\mu g\ l^{-1}$). Figure 1 showed the mean chlorophyll from 1995 to 2001. This spatial distribution of chlorophyll in the Sound generally followed that of nitrogen.

The analysis of 11 years of chlorophyll from multiple stations showed that the timing of major phytoplankton blooms varied from year to year and were not restricted to the winter-spring period. The long term mean (1991 to 2001) annual cycle showed that chlorophyll levels were somewhat constant throughout the year (Figure 2). However, large inter-annual variability was associated with each month as indicated by the standard deviation.

Early work by Riley (1959) in Long Island Sound showed a distinct annual cycle of phytoplankton with one large bloom in winter-spring (February or March) and another small bloom in the fall (September/October). Peterson and Ballantoni (1987) showed a similar cycle based on their 1985 sampling. This annual pattern has been considered the classical cycle of phytoplankton in temperate estuaries including Long Island Sound.

Our results show that in some of the years a bloom did not occur in winter-spring. This may be an artifact of our sampling design. Our monthly sampling may haveindeed missed some of the winter spring blooms. However, it was interesting that summer blooms were observed in our long-term sampling which indicate that blooms were indeed not restricted to the winter/spring period as described before. Li and Smayda (1998) also reported phytoplankton blooms at various months from year to year based on long term (1973 to 1996) weekly observation in Narragansett Bay, Rhode Island.

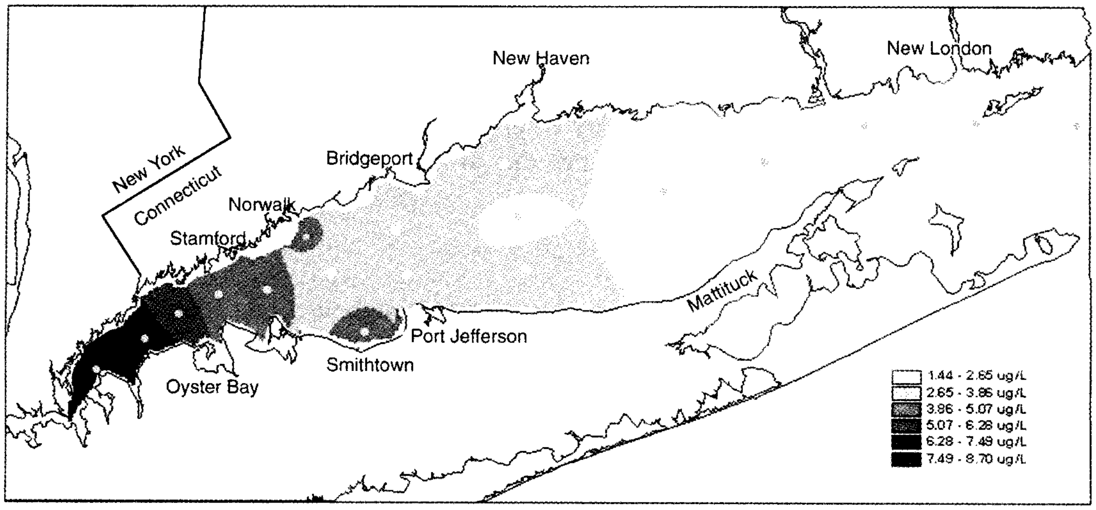

Figure 1. Spatial distribution of chlorophyll in Long Island Sound. Values are mean from 1995 to 2001. Dots indicate the sampling stations.

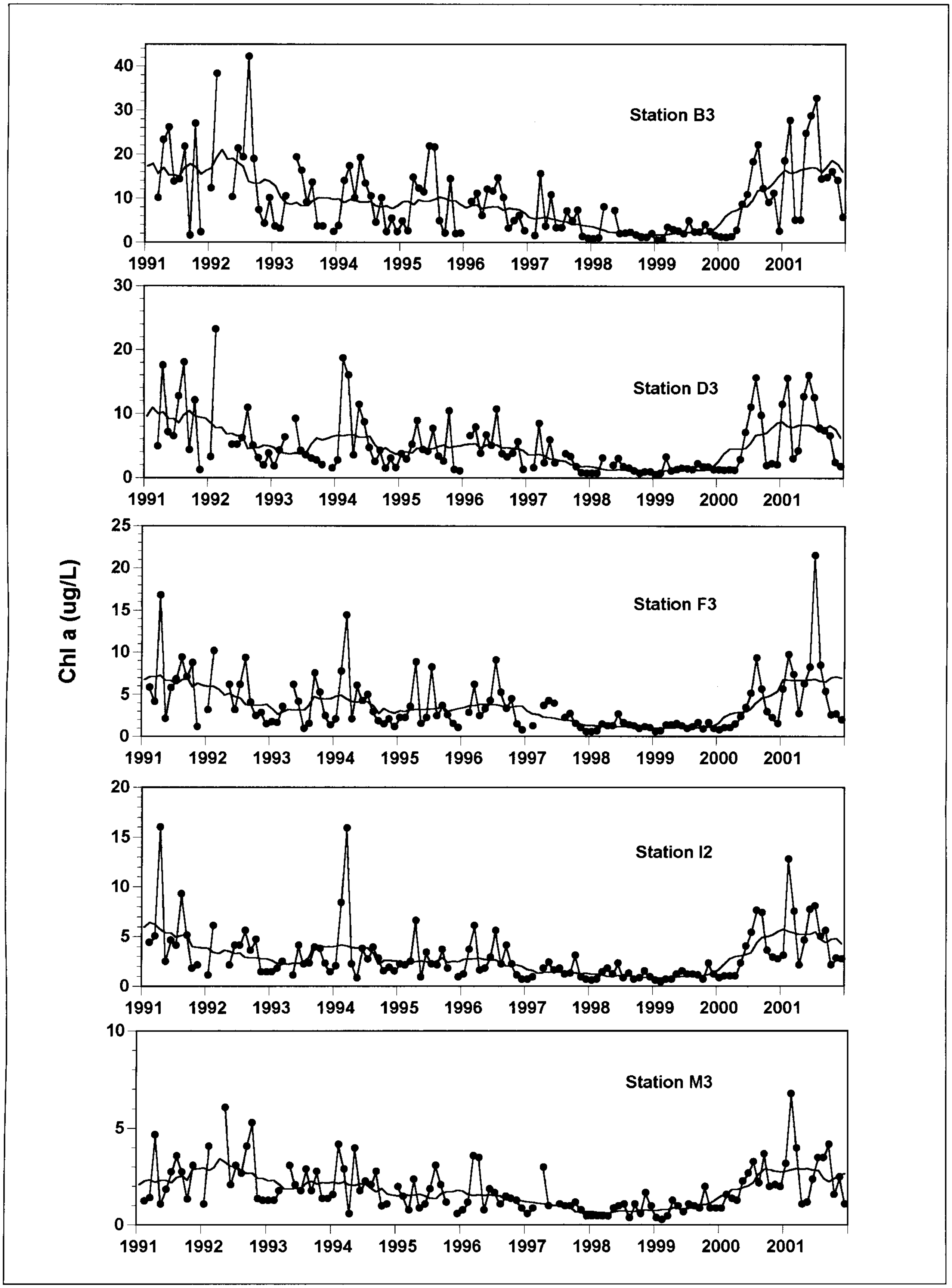

Figure 2. Chlorophyll time series for 5 stations. The solid lines are 12-month moving average to show the trend without seasonal effect, if there is any.

The initiation of winter-spring blooms is mainly driven by physical forces. Riley (1942) suggested that the surface layer production increases as the depth of the surface layer is reduced due to the seasonal warming. Sverdrup (1953)'s critical depth model formalized the mechanisms of the winter-spring blooms. The model predicts that the winter-spring bloom begins when the depth of the surface mixed layer becomes less than the critical depth (above which total phytoplankton production is balanced by the loss through grazing and respiration). However, the critical depth model was originally developed for open water. In coastal waters, other mechanisms of natural variability, such as salinity stratification by fresh water and tidal currents, may overwhelm the seasonal thermal stratification (Lucas *et al.* 1998). In addition, increased human activity (increased nutrient inputs) may have modified the phytoplankton's habitat and consequently, the bloom patterns. For example, the warmer winter temperature can promote the growth of zooplankton, whose grazing may be responsible for some of the missing winter-spring phytoplankton blooms (Oviatt *et al.*1999). Increased nutrient input from rivers and point sources may temporarily release the summer nitrogen limitation in the upper water column and promote a bloom.

There was a significant long-term decreasing trend in chlorophyll in the Sound from 1991 to 2001. All six stations sampled since 1991 showed a significant decrease (3 stations at 95% confidence level, another 3 stations at 90% confidence level). The linear decreasing rate ranged from 0.057 µg l^{-1} year^{-1} in the east to 0.40 µg l^{-1} year^{-1} in the west. These decreasing rates were about 4 to 7% of the median per year. However, within the overall decreasing trend over this 11-year time series, the true decrease occurred from 1991 to 1999, while chlorophyll concentration

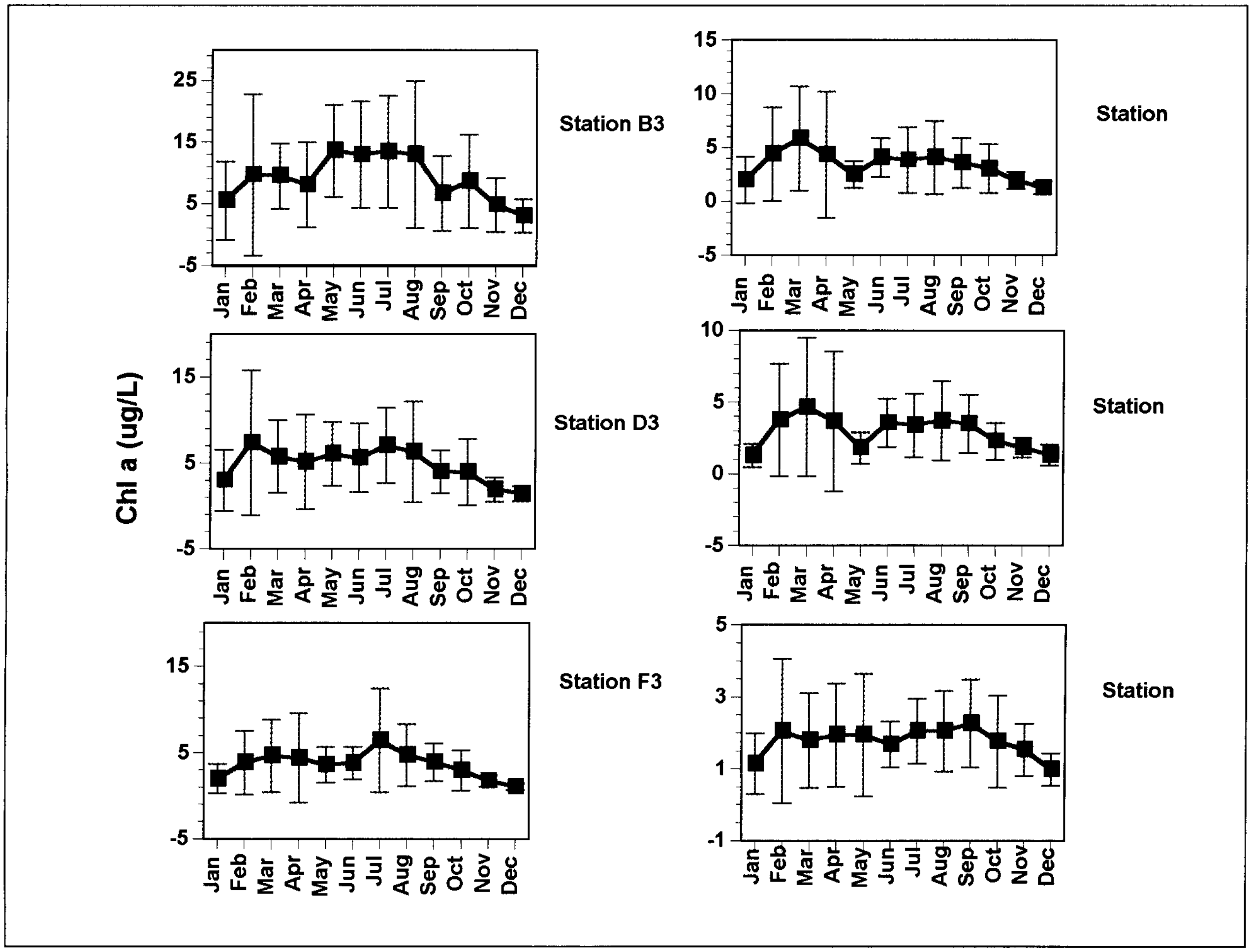

Figure 3. Long-term mean (1991 – 2001) annual cycles of chlorophyll in 6 axial stations. Vertical bars represent one standard deviation.

increased in 2000 and 2001. Due to this trend change in 2000, the 12 stations sampled since 1994 did not show an overall long-term decrease in chlorophyll because the increase in 2000 and 2001 offset the decrease in previous years.

Chlorophyll decreases from 1991 to 1999 ranged from 2.35 to 0.18 µg l^{-1} $year^{-1}$, much higher than the rate for 1991 to 2001. These decreasing rates were about 3 to 8% of the median per year. All 18 stations showed a significant decrease in chlorophyll when time series include only 1991 to 1999.

To find out if the long-term trends were associated with any particular season(s), the chlorophyll data were divided into four seasons, January to March, April to June, July to September and October to December. The observed trend appeared to be present at all of the four seasons.

Along with the long-term trends observed in chlorophyll, there was a significant long-term decrease in inorganic nitrogen (mainly nitrate plus nitrite), a significant increase in inorganic silicate and a significant increase in inorganic phosphate. A slight increase in surface water temperature was also observed.

No correlation was found between chlorophyll concentration and inorganic nitrogen, inorganic silicate, inorganic phosphate, river discharge, or sea surface temperature. However, it was not surprising that no such direct correlation existed. The growth and standing stock of phytoplankton (as indicated by chlorophyll) are subject to variability of habitat parameters that vary themselves with time and may interact with each other. Multivariate analysis will be performed and results will be reported elsewhere.

Acknowledgments

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The Use of Stable Carbon Isotopes in Foraminiferal Calcite to Trace Changes in Biological Oxygen Demand in Long Island Sound

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Introduction

Low oxygen conditions (hypoxia) were first observed in Long Island Sound (LIS) in the early 1970's (Parker and O'Reilly, 1991). Bottom waters in LIS and many coastal bodies of water around the U.S. suffer hypoxia during the summer, because the high temperatures result in thermal stratification, which hinders mixing of the oxygenated surface water with bottom waters (e.g., Welsh and Eller, 1991). If there are large amounts of organic carbon present in the water column, their oxidation will use oxygen (biological oxygen demand, BOD), and the bottom waters may become depleted in oxygen.

Organic carbon in Long Island Sound is mainly derived from two sources: 1) Land sources, including effluent from waste water treatment plants; 2) Biological productivity in LIS, which has become enhanced by the influx of nutrients (nitrogen, phosphate) from anthropogenic sources (e.g., waste water treatment plants, fertilizer). Because of the importance of anthropogenic inputs in the production of organic matter, scientists working on LIS have wondered whether hypoxia also occurred in the past. We define "past" as the time before human activities started to strongly influence the water chemistry of LIS, probably the 1600's (European settlement) followed by the Industrial Revolution and the increasing human population of the 19th century. Whether hypoxia occurred in the past is an important question, because the duration, areal extent and severity of hypoxia in LIS may be caused by biological productivity and/or may be associated with changes in climatic conditions which affect the strength of stratification. The question then is whether the present hypoxia in LIS is caused by climatic change, or by anthropogenic nutrient input. If climatic effects are important, human activities causing global warming might increase the frequency of very hot summers and high influxes of fresh-water, thus increasing the strength of stratification and possibly hypoxia (e.g. O'Shea and Brosnan, 2000).

Our research aims at a reconstruction of organic productivity and evidence for the oxidation of organic matter in bottom waters (possibly with resulting low oxygen conditions) over the last several hundreds of years, using samples from cores taken in LIS (Buchholtz ten Brink *et al.*, 2000; Thomas *et al.*, this volume).

Working Hypothesis

Most organic matter has a specific carbon isotopic signature, because the lighter isotope, ^{12}C, is taken up preferentially by the organism during photosynthesis. Terrestrial organic matter is isotopically somewhat lighter than marine-produced organic matter. If a large amount of phytoplankton-produced organic matter sinks to the bottom, and is locally oxidized, the concentration of ^{12}C in dissolved inorganic carbon (DIC) in the bottom waters will increase (e.g., Voss *et al.*, 2000).

Benthic foraminifera such as *Elphidium excavatum* use DIC to construct their calcium carbonate shells (tests), which take on the DIC carbon isotopic signature. The $\delta^{13}C$ values of the foraminiferal tests depend on the amount and type of organic carbon that was oxidized to form DIC, and thus on the degree of BOD (e.g. Thomas *et al.*, 2000).

Many species of benthic foraminifera precipitate their tests out of isotopic equilibrium with the bottom waters, with offsets generally consistent within a species (e.g., McCorkle *et al.,* 1990). Benthic foraminifera commonly have $\delta^{13}C_{cc}$ values that are more negative than those in bottom water DIC, because infaunal foraminifera live within the sediment and calcify in contact with sediment pore waters (e.g., Grossman, 1984; Zahn *et al.*, 1986; McCorkle *et al.*, 1990). The species *E. excavatum* is an epifaunal species, however, that lives at the sediment-water interface in LIS (Buzas, 1965). A related subspecies, however, is shallow infaunal and has $\delta^{13}C_{cc}$ values slightly more negative than DIC values in bottom waters (Polyak *et al.,* 2003).

Methods

About 51 water samples, 90 surface samples and 9 cores were collected between 1996 and 2001 (Buchholtz ten Brink *et al.*, 2000; Thomas *et al.,* this volume) in addition to 28 water samples from the Connecticut and Housatonic Rivers. All samples were analyzed for $\delta^{13}C$ and $\delta^{18}O$. The LIS waters are mixtures of river and sea water, and each end member has a distinct oxygen and carbon isotopic composition. The mixing proportions (salinity) are therefore reflected in the isotopic composition of both the water ($\delta^{18}O_w$) and the DIC ($\delta^{13}C_{DIC}$). In order to eliminate the effect of salinity on the carbon isotopic values of the foraminiferal tests ($\delta^{13}Ccc$), we estimated the <u>excess</u> $\delta^{13}C_{cc}$ values ($\delta^{13}C*$), which are equal to $\delta^{13}C_{cc}$ (observed) <u>minus</u> $\delta^{13}C_{DIC}$ (predicted). The predicted $\delta^{13}C_{cc}$ values are derived from the measured $\delta^{18}O_{cc}$ values in the tests, which are recalculated to $\delta^{18}O_w$ assuming a constant mean annual bottom water temperature of 12.5 ^{O}C, using the fractionation factors of Kim and O'Neil (1997)) and a 1.1%*o* off-set between water and carbonate test for this species. The salinity is derived from the calculated $\delta^{18}O_w$ and then the $\delta^{13}C_{DIC}$ is calculated with the LIS mixing model. The use of an annually averaged temperature implies that the species *E. excavatum* lives for about a year (Murray, 1991), and averages the seasonal temperature variability. The $d^{13}C*$ is a measure of the amount of organic carbon that was oxidized in the bottom waters and thus indirectly the BOD.

Results

Based on salinity and our LIS mixing model, $\delta^{13}C_{DIC}$ in LIS waters values should range between -0.2‰ and -1.0‰, but the observed values are between -2.4‰ and -6.0‰ with the

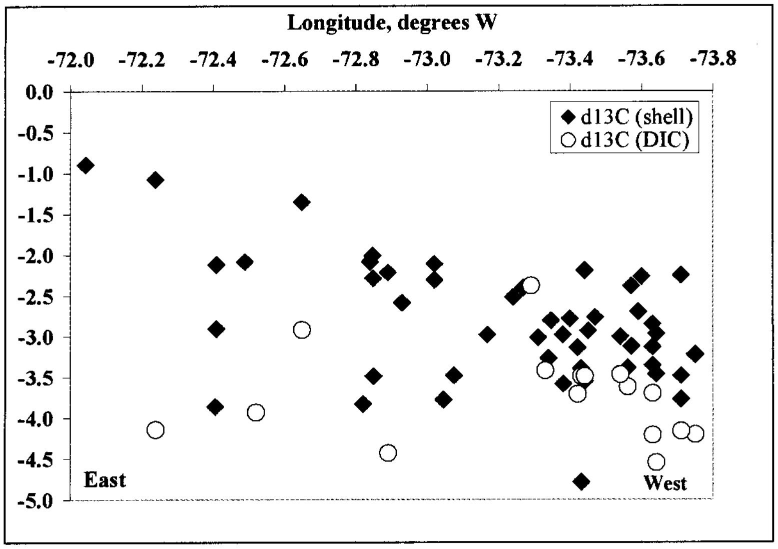

Figure 1. Variation of stable carbon isotopic compositions of foraminiferal shells from surface samples and $\delta^{13}C$ of dissolved inorganic carbon (DIC) across the Sound. Note the East-West decrease in the maximal shell parameters.

lightest values in areas that experience seasonal hypoxia (Figure 1) (western LIS). The $\delta^{13}C_{cc}$ values of surface samples similarly decrease from ~ -1.0‰ in the East to ~-4.5‰ in western LIS. The $\delta^{13}C_{DIC}$ values of LIS waters are thus lighter than expected from their salinity, indicating that oxidation of organic matter occurred, especially in western LIS (e.g. Keough *et al.*, 1998). The off-set of about 1‰ between the $\delta^{13}C_{cc}$ in surface sample tests and $\delta^{13}C_{DIC}$ in LIS waters may be analytical in nature or systematic between DIC in water and calcite. The heaviest measured $\delta^{13}C^*$ values are about 1‰ lighter than DIC, indicating either asystematic, species-specific off-set (as expected from observations by Polyak *et al.*, 2003), or oxidation of organic matter throughout the Sound.

The $\delta^{13}C_{cc}$ values of all the core samples become lighter from core bottom towards the top, i.e., from the past towards the present. The $\delta^{13}C$ values were around -2.0‰ prior to 1740, then shifted to significantly lower values of up to -5‰ in the middle through late part of the 20th century. The lightest values occur in cores from western LIS (WLIS75, A1C1) and near the main river mouths (B1C1, G1C1). The $\delta^{13}C^*$ values of surface samples show the same East-West trend as the raw data. The $\delta^{13}C^*$ values of the core samples (Figures 2A, 2B) show values around -1 to -2 ‰ pre-1850, and become much lighter over the last 150 years (values -2 to -4.5 ‰).

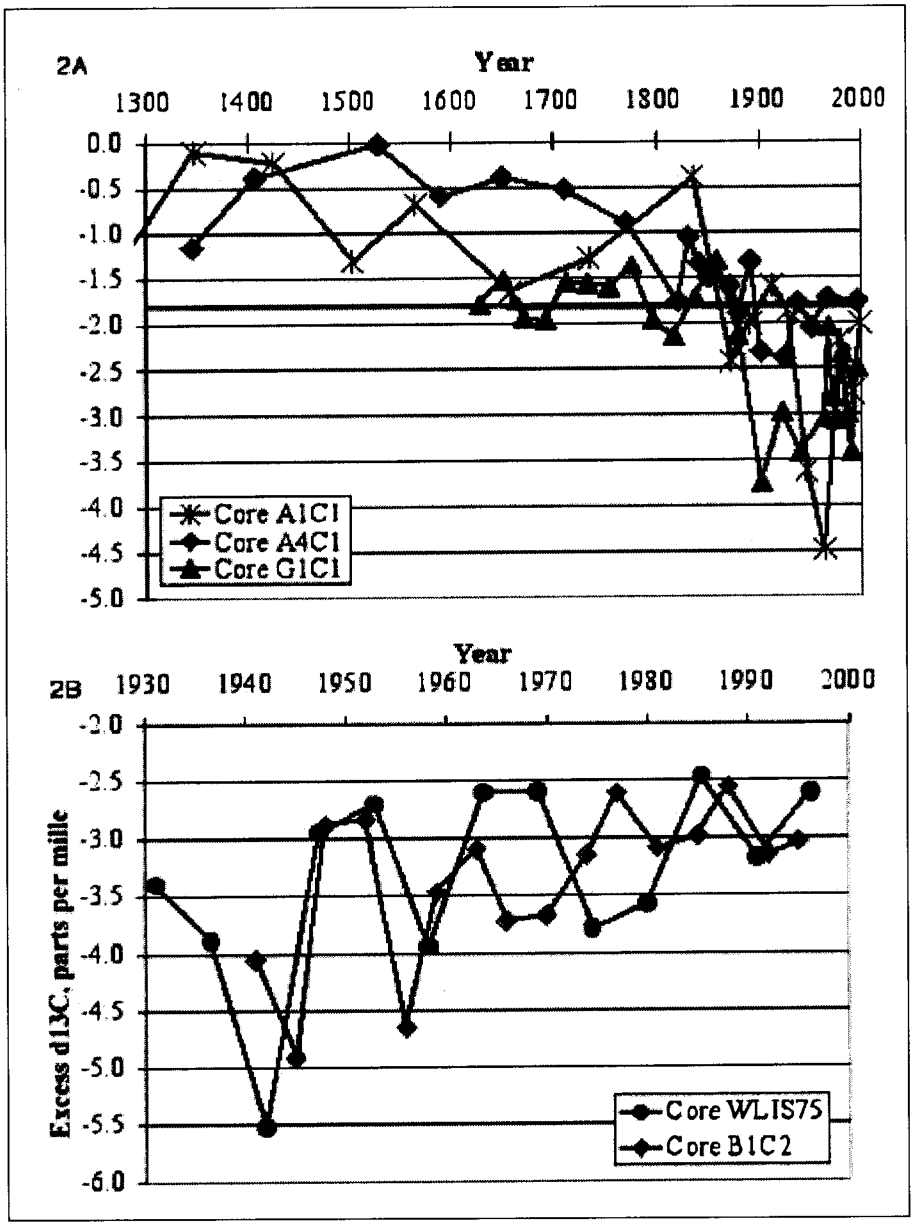

Figure 2. Changes in d^{13}C* values of core samples with time. Note that the $\delta^{13}C^*$ values of the long records A1C1, A4C1 and G1C1 (Figure 2A) start to decrease around the late 1700's and early 1800's, and, similar to the short records WLIS75 and B1C2 (Figure 2B), start to increase in the mid-twentieth century.

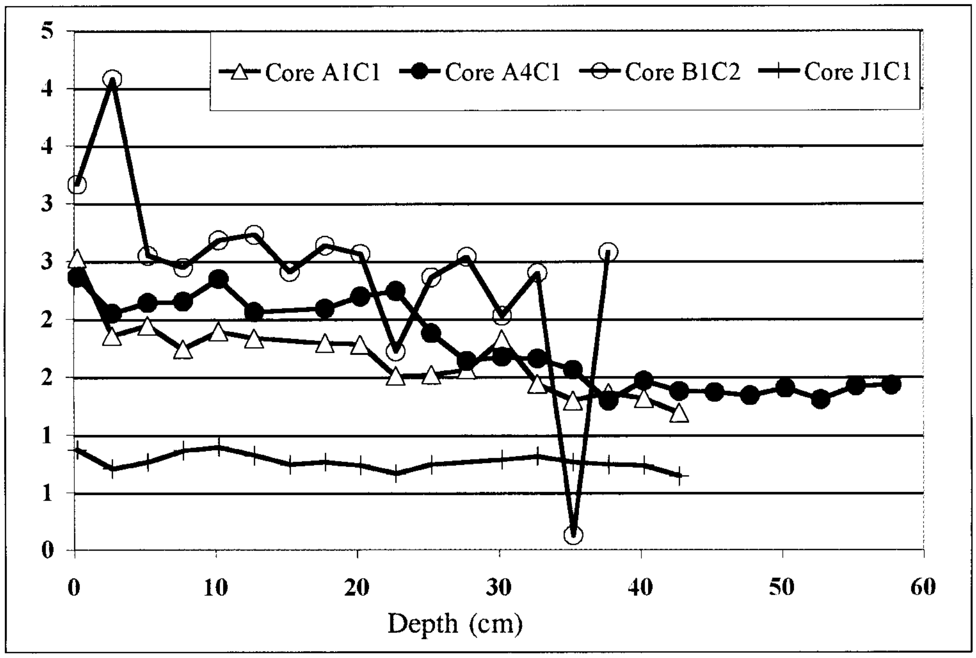

Figure 3. Changes in sediment organic carbon with depth. Note the low levels in cores taken from eastern LIS (J1C1) and increasing levels in cores taken from the WLIS (A1C1, A4C1 and B1C2).

Discussion

The temporal trends of $\delta^{13}C_{cc}$ and $\delta^{13}C^*$ in cores A1C1, A4C1 and G1C1 suggest that only small amounts of organic carbon were oxidized at the bottom of LIS prior to 1800 AD. At around 1850 AD, the carbon isotopic values started to decline, indicating the increased oxidation of organic carbon. At the same time, the burial rate of organic carbon increased, as shown by increasing concentrations of organic carbon (Figure 3 and Varekamp *et al.*, this volume). These trends correlate well with the changes in human population and trends in deforestation and industrialization over this time period (e.g., Koppelman *et al.*, 1976).

The $\delta^{13}C$ values were possibly at their lightest around the 1960-1970's in cores A1C1 and G1C1, and became slightly heavier towards the end of the 20th century. Trends in the concentrations of *Clostridium perfringens* (Buchholtz *et al.*, 2000), a tracer for sewage inputs, support the notion of increased eutrophication that was driving the increased organic productivity. The extremely light $\delta^{13}C$ values in carbonate samples from core WLIS75 in the Narrows near New York City may also be impacted by the inputs of light particulate organic carbon in effluent from local sewage treatment plants.

Conclusions

Our data indicate that the amount of organic carbon in LIS sediment increased since the late 19th century, either indirectly, as a result of increased primary productivity caused by anthropogenic water column fertilization and/or directly, through antrhopogenic addition of terrestrial organic carbon, especially in western LIS and the Narrows. The carbon isotope data indicate that more and more organic carbon was oxidized in the bottom waters since the 1850's, probably leading to a large BOD. The common occurrence of hypoxia over the last 40 years may thus very well be a direct result of the strong eutrophication of the Sound over that period, although we can not exclude that the effects of modern global warming over the last 100 years also impacted the rate of organic carbon mineralization at the bottom of LIS, in addition to enhancing water column stratification (Varekamp *et al.*, this volume).

Acknowledgments

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Temporal and Spatial Variations of Atmospheric Nitrogen Deposition in Connecticut

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Introduction

The Connecticut Department of Environmental Protection (CTDEP), in conjunction with the United States Environmental Protection Agency (USEPA), has been evaluating the effect of nitrogen loading on dissolved oxygen levels in Long Island Sound (LIS). Studies show that excessive loading of nutrients, especially nitrogen, is one of the major contributors to the hypoxia problems that LIS has been experiencing (USEPA Long Island Sound Study, 1997). Previous studies by the University of Connecticut show that nitrogen from the atmosphere is an important component of the nitrogen loading to LIS. Emissions from combustion processes and agriculture are primary sources of the atmospheric nitrogen, which can be transported into the Sound and its estuaries via both direct and indirect deposition.

The National Atmospheric Deposition Program (NADP) and the Clean Air Status and Trends Network (CASTNet) include only a single rural site at Abington, Connecticut since 1994 (Clark *et al.*, 1997). To study the atmospheric concentrations and deposition fluxes of nitrogen in Connecticut, a network of eight additional monitoring stations was established throughout the State since late 1996. Based on the measurements from this monitoring network for the period from February 7, 1997 through December 30, 1999, we have examined the temporal and spatial variations of atmospheric nitrogen deposition in Connecticut. Results of this study will be useful in understanding the trend and variability of atmospheric nitrogen deposition in the study area, in comparing deposition measurements with different networks, and in evaluating and improving air quality control and management.

Materials and Methods

A network of eight monitoring stations was established to study the atmospheric concentration and deposition of nitrogen in the State of Connecticut (Figure 1). The sites were classified as urban, rural, coastal, or inland (Table 1), based on their geographical location and representative surrounding land use. Nitrogen species including nitrate (NO_3^-), ammonium (NH_4^+), nitric acid vapor (HNO_3) and total dissolved nitrogen (TDN) in air and in precipitation were collected and used to infer nitrogen concentrations and dry and wet deposition flux densities, using independently collected meteorological data. In the monitoring network, the methods of dry and wet deposition collections are similar with those used in the Clean Air Status and Trends Network (CASTNet) and the National Atmospheric Deposition Program (NADP, Nadim *et al.*, 2001).

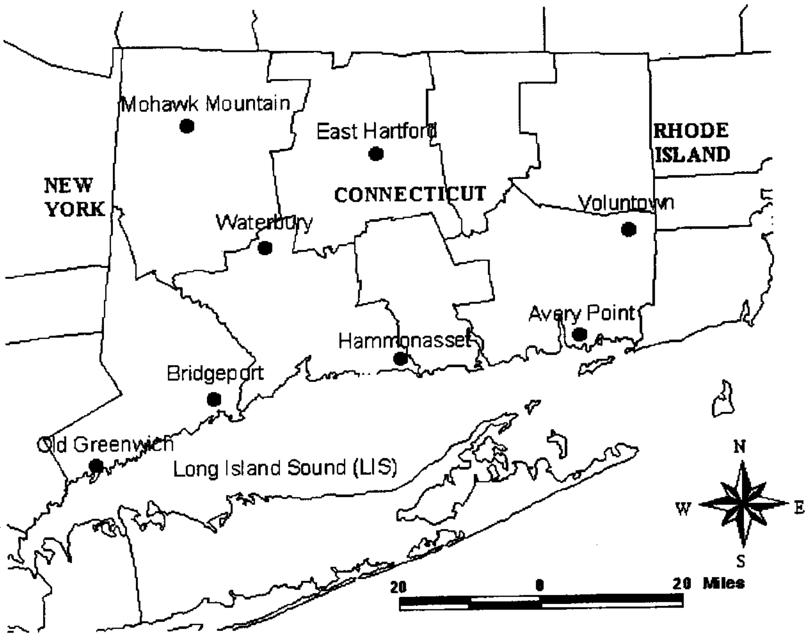


Figure 1. Location of the monitoring stations in the State of Connecticut used studying the atmospheric concentration and deposition of nitrogen. The stations were classified as urban, rural, coastal and interior, as shown in Table 1.

Weekly samples from dry and wet collectors were delivered to the analytical laboratory of the Environmental Research Institute, University of Connecticut, for chemical analyses of each nitrogen species and total value. In this study, the dissolved organic nitrogen (DON) concentration was defined as the difference between total dissolved nitrogen and the dissolved inorganic nitrogen (= $NO_3^- + NH_4^+$, Scudlark *et al.*, 1998). Because the detection limit for TDN was only 0.005 mg l^{-1}, the method allowed accurate estimation of DON concentrations. The reported mean atmospheric concentrations of nitrogen species (C_a, μgN m^{-3}) were determined from the collected mass of each specific species in the dry samples divided by the total volume of the air flowing through the filter-pack during the sampling week. Similarly, the reported mean rainwater concentration of nitrogen species (C_w, mgN l^{-1}) were computed from the collected mass of each specific species in the wet samples divided by the total volume of the precipitation collected during the corresponding week. The laboratory procedures and quality control plan can be found in Carley and Perkins (2001).

Table 1. Classification of the monitoring stations in the State of Connecticut

Site Name	**Abbreviation**	**Classification**	
		Location	Land use
Avery Point	AP	Coastal	Rural
Bridgeport	BR	Coastal	Urban
East Hartford	EH	Inland	Urban
Old Greenwich	OG	Coastal	Urban
Hammonasset	HA	Coastal	Rural
Mohawk Mountain	MM	Inland	Rural
Voluntown	VO	Inland	Rural
Waterbury	WA	Inland	Urban

The weekly dry deposition flux density for each nitrogen species (F_d, mgN m^{-2} wk^{-1}) was estimated by, with proper units conversion,

$$F_d = V_d * C_a \quad (1)$$

where V_d is the corresponding weekly mean dry deposition velocity in cm s^{-1}. The weekly mean dry deposition velocity was a simple average of the hourly dry deposition velocity calculated from the meteorological parameters, using the dry deposition inference model (DDIM) adopted by the National Dry Deposition Network. Details of the DDIM can be found in Hicks et al. (1987), Yang *et al.* (1996), and Luo *et al.* (2002).

Based on the weekly precipitation (P, mm wk^{-1}), the weekly wet deposition flux density (F_w, mgN m^{-2} wk^{-1}) was calculated by, with appropriate units conversion,

$$F_w = P * C_w \quad (2)$$

Results and Discussion

Seasonal dynamics

The dependence of the quarterly means on the factors of season, geographic location and surrounding landuse of the monitoring stations, and their interactions were evaluated via the general linear model (GLM), with year as a replicate measure (Table 2). Most of the collected nitrogen quantities peaked during warm months, especially from June through August (Table 3). For nitrogen deposition in dry or wet form, more than one-third of the atmospheric nitrogen loadings to the coastline occurred in summer (34.0% and 37.3%, for dry and wet depositions, respectively). The prevailing wind direction is an important factor for determining atmospheric pollutant levels from long-range transport.

The surface winds along the Connecticut coastline of LIS are most frequently from northwest in cold season and southwest in warm season (Xu *et al.*, 1997, Yang *et al.*, 1996). Air quality is poor with southwest winds due to the intense emissions from the New York-New Jersey industrial area.

Table 2. Results of the analysis of variances on the quarterly means of the pertinent quantities of nitrogen deposition using GLM, at the eight monitoring sites in Connecticut during 1997-1999.

Factor	F Statistic on the quarterly means for							
	C_a[a]	V_d^g	V_d^p	F_d	C_w	P	F_w	F
Season	31.37	1.44	18.74	8.64	11.63	2.44	16.28	20.57
	***	NS	***	***	***	NS	***	***
Landuse	117.58	45.06	23.43	28.42	0.23	0.56	2.89	7.19
	***	***	***	***	NS	NS	NS	**
Location	10.29	0.58	39.41	0.15	1.45	1.61	1.76	1.54
	**	NS	***	NS	NS	NS	NS	NS
Season*Landuse	2.14	0.50	0.11	6.24	0.11	0.51	0.10	0.24
	NS	NS	NS	***	NS	NS	NS	NS
Season*Location	1.19	0.48	1.22	0.13	0.89	0.63	0.28	0.22
	NS	NS	NS	NS	NS	NS	NS	NS
Landuse*Location	0.16	7.48	43.84	0.10	0.01	0.00	0.56	0.47
	NS	**	***	NS	NS	NS	NS	NS

a. C_a = Ambient concentration; F_d = Dry deposition flux density; V_d^g = HNO_3 deposition velocity; V_d^p = Particulate deposition velocity; C_w = Rainwater concentration; F_w =Wet deposition flux density; F =Total (dry+wet) deposition flux density;

b. Statistical significance, where **: $p < 0.01$, ***: $p < 0.001$, NS: not significant ($p > 0.05$).

Table 3. Quarterly means of concentration and deposition flux density of total nitrogen over the eight monitoring sites in Connecticut during 1997-1999

Season	C_a[a] _gN m^{-3}	F_d mgN m^{-2} wk^{-1}	C_w mgN l^{-1}	F_w mgN m^{-2} wk^{-1}	F mgN m^{-2} wk^{-1}
Spring	1.41 A[b]	4.26 A	1.19 A	24.50 A	28.76 A
Summer	2.09 B	6.06 B	1.90 B	34.46 B	40.53 B
Fall	1.56 A	4.13 A	1.25 A	17.80 C	21.94 C
Winter	1.73 A	3.36 A	0.98 A	15.70 C	19.06 C

a. C_a = Ambient concentration; F_d = Dry deposition flux density; C_w = Rainwater concentration; F_w =Wet deposition flux density; F =Total (dry+wet) deposition flux density;

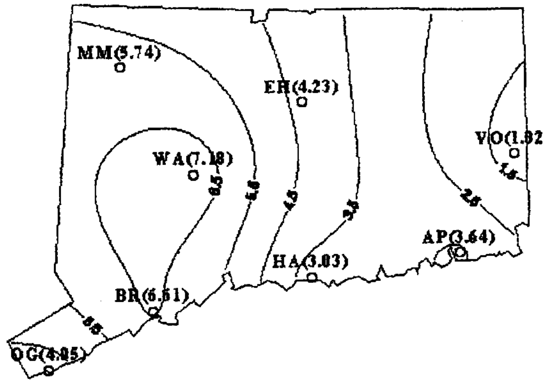
b. Indices of Fisher's least significant difference (LSD) among means. Means with the same letter are not significantly different.

Spatial variations

Descriptive statistics were employed to characterize the weekly nitrogen data collected in the monitoring network. The overall mean weekly dry and wet deposition fluxes of total nitrogen were 4.4 and 24.0 mgN m^{-2} wk^{-1} during 1997-1999, respectively. The 3-year mean values show a general decreasing trend in nitrogen deposition fluxes from west to east in Connecticut (Figure 2). HNO_3 was the major contributor to dry deposition at these sites, while NO_3^- was predominant in wet deposition.

This study also allowed the evaluation of the effects of geographic location and land use on the atmospheric concentration and deposition of nitrogen species in Connecticut (Table 2). The southwestern urban region had higher ambient concentrations and dry deposition of nitrogen compared to the rural areas (Figure 3a). Ambient nitrogen concentration over the four urban sites (Old Greenwich, Bridgeport, Waterbury, and East Hartford) averaged about 38.9% higher than that over the rural sites (Mohawk Mountain, Hammonasset, Avery Point, and Voluntown), resulting a 58.0% higher dry deposition flux over these sites compared to their rural counterpart. The maximum 3-year mean dry deposition of total nitrogen was measured at Waterbury (7.18 mgN m^{-2} wk^{-1}). Bridgeport and Mohawk Mountain sites followed Waterbury with three-year mean values of 6.61

(a) Dry deposition flux



(b) Wet deposition flux

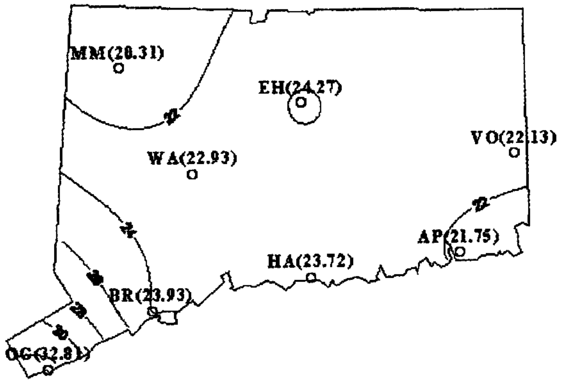
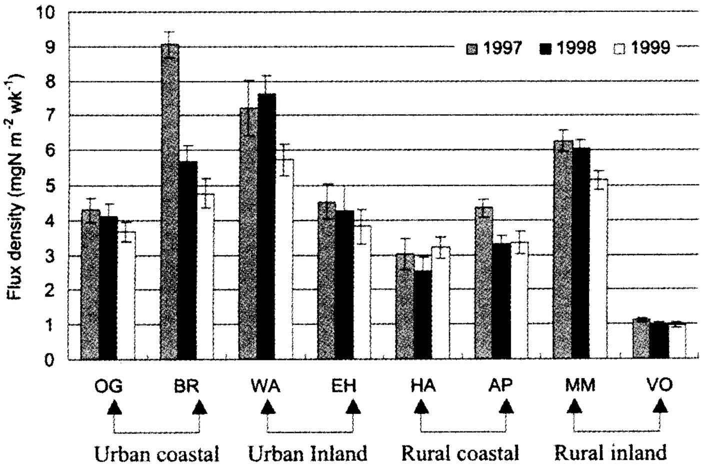


Figure 2. (a) Dry and (b) wet deposition flux of total nitrogen (mgN m^{-2} wk^{-1}) in Connecticut, during 1997-1999)

and 5.74 mgN m^{-2} wk^{-1}, respectively (Figure 2). Besides the ambient nitrogen concentration, the deposition velocities of nitrogen species were also important contributors to the spatial pattern of dry nitrogen deposition. Both deposition velocities of HNO_3 and nitrogen particles were significantly higher over the urban sites than the rural sites ($F = 435.9$ and $p < 0.001$ for HNO_3 deposition velocity; $F = 91.53$ and $p < 0.001$ for particulate velocity). Local industrial activities and traffic emissions of nitrogen at urban areas had significant effects on the spatial distribution of atmospheric nitrogen concentration and dry deposition flux in Connecticut.

Wet deposition rates were nearly invariant over the monitoring network. The only exception was the Old Greenwich site, located in the southwestern corner of the State (Figure 2). Wet deposition of total nitrogen at Old Greenwich (32.8 mgN m^{-2} wk^{-1}), was quite high compared to the average wet deposition flux of 22.7 mgN m^{-2} wk^{-1} over the other seven sites during 1997-1999 (Figure 3b). Because intense wet deposition happens in near-source locations (Geigert, 1991 and Hu, 1997), the high quantities of wet deposition at Old Greenwich might be attributed to high nitrogen emissions in the industrial areas around New York and New Jersey demonstrated in previous studies (NADP, 1994; Xu, 1995; Yang *et al.*, 1996; Carley and Perkins, 2001).

(a) Dry deposition



(b) Wet deposition

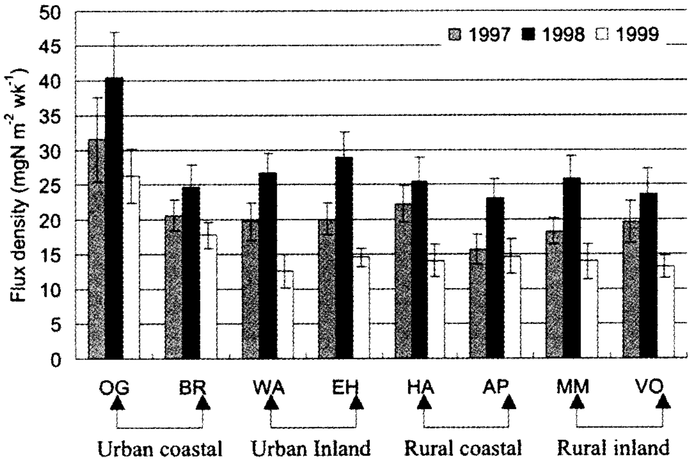


Figure 3. Annual means of weekly (a) dry and (b) wet deposition of total nitrogen in Connecticut during 1997-1999. The stations were classified as urban, rural, coastal and interior, as shown in Table 1.

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Sediment Metal Contamination at the West River Memorial Park, New Haven, Connecticut

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Introduction

The West River, a tidal tributary of Long Island Sound, once a salt marsh, was eradicated as such to create upland recreational area following the release of the *Civic Improvement Plan* by Cass Gilbert and Frederick Law Olmsted (Casagrande, 1997). Tide gates were installed at the southern end of what would become the West River Memorial Park, to reduce the mosquito population. A long straight channel was dredged through the length of the park to create a reflective pool of West Rock, reminiscent of that in front of the Washington Monument. These changes resulted in an ecologically disturbed wetland, dominated by common reed (*Phragmites australis*) forming visual and physical barriers to the water (Casagrande, 1997; Figure 1). The possibility of sediment contamination within the marsh by heavy metals and excess organic matter was hypothesized due to the proximity of multiple contaminant sources.

This study is designed to test the following hypothesis: the West River Memorial Park sediment is contaminated with heavy metals and that sediment metal contents will vary with different sediment types. The specific research objectives are to: (1) Conduct a high spatial resolution sampling of the sediment of the West River Memorial Park; (2) Measure the chemical (lead, zinc, copper, iron, nickel and organic content) and physical (texture and grain size) properties of the sediment and (3) Examine the environmental factors affecting sediment metal distribution within the marsh sediment.

Methods

Sampling of sediments began on March 20, 2001 (sites 1-12c) and were completed on May 12, 2001 (sites 13-17). Sediment samples at each site were collected from a canoe using a ponar grab to the depth of approximately 6 cm. Sediment organic content was determined from weight loss on ignition (LOI) after sediment samples dried (70°C) and ashed (550°C). Dried sediment (65°C) was wet digested using the HNO_3-H_2O_2 digest method according to U.S. EPA method 3050a. Digests were analyzed for iron (Fe), copper (Cu), nickel (Ni), lead (Pb), and zinc (Zn) using air-acetylene flame atomic absorption spectrophotometry (Perkin-Elmer 2380).

Results

Sediment Metal Contents

The metal content of the sediment ranged from 12 to 216 mg kg^{-1} for Cu, 24 to 433 mg kg^{-1} for Pb, 27 to 597 mg kg^{-1} for Zn and 6 to 46 mg kg^{-1} for Ni. The enrichment of metals in the sediment within the marsh relative to thier respective crustal abundances most likely reflects the degree of urbanization of the surrounding terrain where specific anthropogenic sources contribute metals. Results also show that the sediment metal contents at a particular location within the marsh are correlated with the sediment type at that location. Where fine-grained sediments containing high

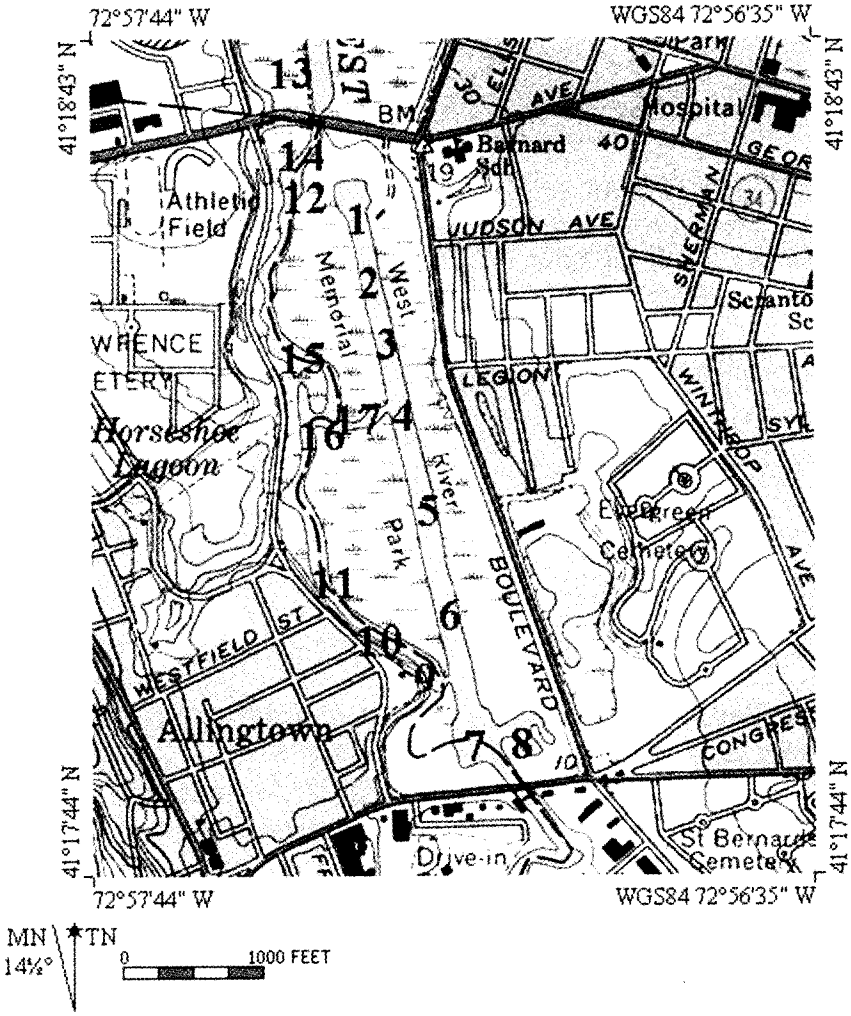

Figure 1. Topographic map of the West River Memorial Park. The sampling sites are identified on the map by number.

organic matter are present, the metal content of the sediment is high. In contrast, where coarse-grained (sandy) sediments containing low organic matter are present, the corresponding metal content of the sediment is low.

Geographic Distribution of Metals in the Marsh

Results of the characterization of the physical properties of the sediment within the West River Memorial Park show differences in the sediment types based on the general geographic locations within the marsh. For example, the northwestern portion of the marsh is characterized by coarser sandy sediment containing low organic matter. In contrast, the southeastern portion of the marsh is characterized by *Phragmites austrialis* along the shoreline and fine-grained sediment containing high organic carbon (Figure 1). The differences in location and sediment properties allowed for the grouping of sampling locations to form four different areas of the marsh for comparison and discussion.

The seventeen sampling sites were grouped into four areas according to their geographic location within the marsh, the sediment LOI and the sediment grain size. Area 1 was subdivided to compensate for differences in sediment type. Area 1a was comprised primarily of sandy sediment. With the absence of *P. austrialis* stalks in this location, the mean sediment LOI was very low (1.6%). Also, the metal contents within this area were comparatively low (34.5 mg kg^{-1} for Pb and 25.1 mg kg^{-1} for Cu) (Figure 2).

Area	Location	Sampling Sites
1a	Northwest	12a, 13 - 15
1b	Northwest	12b, 12c
2	Southwest	9-11,16, 17
3	Southwest	1-8

Table 1. Geographic areas and respective sampling sites within the West River Memorial Park

In contrast, the sediment sites in Area 1b were comprised of sand and silt-sized particles and had a much higher mean metal content compared to Area 1a. Sediments within Area 1b had a mean LOI of 5.1%. Mean lead, zinc and copper contents were 155 mg kg^{-1}, 203 mg kg^{-1} and 66.5 mg kg^{-1}, respectively (Figure 2). The differences in the average sediment metal contents between Area 1a and 1b support the fact that sediment type was influential in determining the degree of metal enrichment.

Area 2 (Table 1) represents a transition area where the natural stream meets the main, dredged channel and terminates at the tide gates. The sediments were finer than that of Area 1a and Area 1b, consisting primarily of silt-sized particles at most stations. Area 2 had a significantly higher mean LOI value of 18.3% compared to LOI values for Areas 1a and 1b. Mean sediment metal contents within this area were 236 mg kg^{-1} for Pb, 114 for Cu, 310 for Zn and 34.9 for Ni (Figure 2).

The sediment within Area 3 was primarily comprised of silt-sized particles. The sediments within this area were characterized by high LOI values, with a mean of 18.4%. The high organic content of the sediment within this area was most likely derived from leaf litter and *Phragmites* stalks that covered the upper 1-2 cm sediment layer of the streambed. Area 3, discernibly the region most highly impacted by anthropogenic influences, showed a significant increase in sediment metal contents compared to the other areas (Figure 2).

This area of the main dredged streambed was beset with storm drains and was within close proximity to high traffic roadways (Figure 1). The highest mean metal contents within the park were measured in this area (348 mg kg^{-1}, 175 mg kg^{-1}, 462 mg kg^{-1} and 41.2 mg kg^{-1} for Pb, Cu, Zn and Ni, respectively).

Sources of Metals to the West River Memorial Park (WRMP)

Contaminant metals are introduced to Long Island Sound from a variety of sources including sewage effluent, disposal of dredged material, industrial discharges, urban and agricultural runoff and atmospheric deposition (Mecray and Buchholtz ten Brink, 2000). Within the WRMP, the sources of Pb and Zn are apparent; however, direct sources of Cu and Ni may lie external to the park.

The roadways surrounding the WRMP are likely the primary sources of Pb and Zn to the river sediment. The WRMP is bound on three sides by heavily trafficked roadways including Ella Grasso

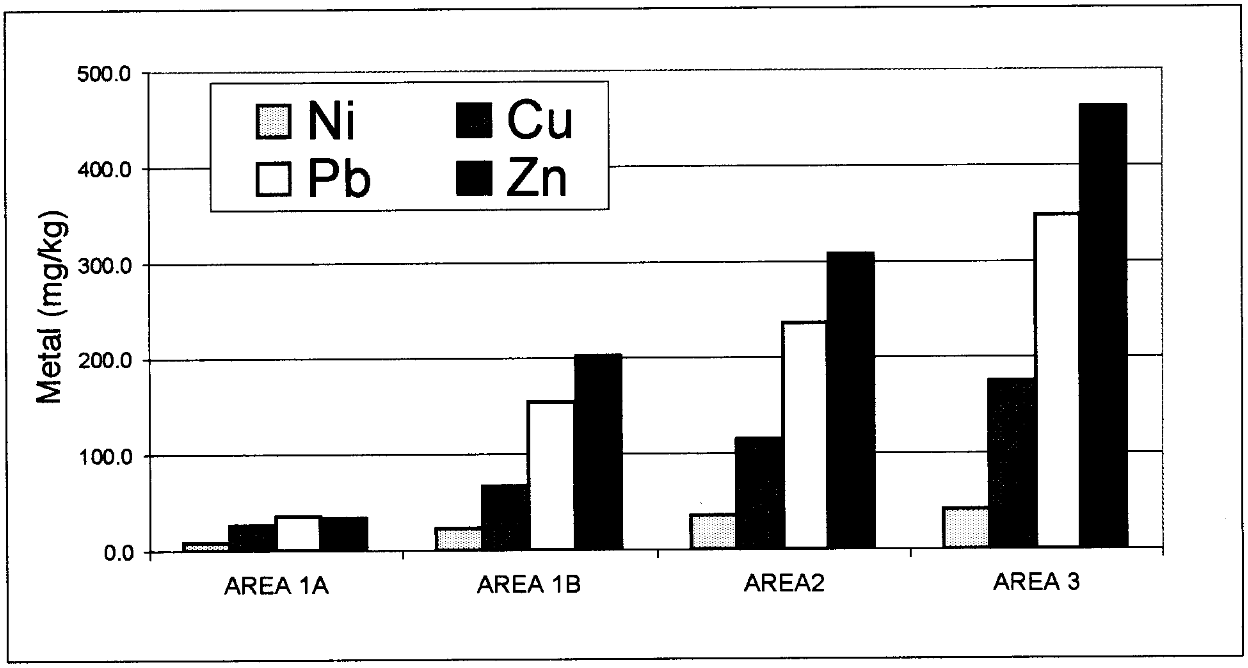

Figure 2. Mean sediment metal content for each area within the West River Memorial Park.

Blvd. to the east, Rte. 34 to the north and Rte. 1 to the south (Figure 1). Zinc is used in the vulcanization of tires, which wear on the high traffic roadways (Nriagu, 1980). Historically, lead has come mostly from atmospheric deposition of anti-knock compounds in the exhaust gases from vehicles that use gasoline fuel (Murozumi *et al.* 1969; Ng and Patterson, 1981). Both Zn and Pb from these sources can become trapped in the soils along the roadways and are susceptible to erosion during heavy rains and enter the WRMP via storm drains.

Zinc and Copper may be derived from natural sources in the soil and sediment and sewage effluent discharges (Benoit *et al.*, 1999). Copper may be enriched in soils in the West River watershed due to the historical use of copper-based fungicides (copper sulfate and calcium hydroxide) on agricultural land. Nickel, although not enriched in the WRMP sediment, is a constituent of industrial oil and may have been deposited atmospherically due to the combustion of oil at oil-fired power plants (Kowalczyk, 1977). The sources of these contaminants appear to have no direct point of entry within the WRMP. However, these and other metals may be transported to the WRMP on suspended sediment particles during periods of high water flow from areas north of the marsh. These suspended sediments are then deposited in the marsh sediment.

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Geologic Framework Data for Long Island Sound, 1981-1990: a Digital Data Release

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Since 1982, the U.S. Geological Survey (USGS) and the Connecticut Department of Environmental Protection (DEP) have maintained a cooperative agreement to investigate the geological framework and surficial processes of Long Island Sound. As part of this agreement during the 1980's, a regional set of high-resolution seismic-reflection profiles and vibratory cores were collected to interpret the stratigraphy of the Long Island Sound basin. These unique seismic records and coring logs, which are still under great demand (e.g. from energy companies who need sub-bottom information when siting offshore transfer platforms and routes for pipelines and cables), were irreplaceable, fragile, existed only in analog form, and were becoming ragged from frequent use. To preserve these data, they were converted into digital form, and the files were organized into a Geographic Information System (GIS) product on DVD-ROM media that would allow the data to be more readily accessed and disseminated.

The seismic records are provided in two resolutions: reduced GIF images for rapid browsing to provide an overview of the data, and 300-dpi, full-resolution TIF images for full-scale plotter reproduction. Images of these records can be accessed through links with both tables and maps. Information on the vibratory cores includes visual descriptions, core photographs, and maps showing station locations. Associated seismic navigation and core location data have been converted to GIS compatible formats and are listed in the data catalog with FGDC compliant metadata for each data layer. All available ancillary reports (cruise, summary, and scientific) from the seismic and coring programs have been included to supply additional background and to broaden the perspective and understanding of the user. A gallery of photographs showing the equipment and procedures from the geophysical and coring programs and the personnel who conducted these operations is also provided for general interest.

Scientific questions and policy issues related to the Sound often require data of historical, regional, and interdisciplinary scope. Because acquisition of new data is expensive and may duplicate previous efforts, the body of existing data needs to be utilized to its maximum so that it can serve as a foundation and baseline for further work. This accessible and documented compilation of seismic profiles and core descriptions comprises the largest single body of existing data on the geological framework of Long Island Sound and, is valuable to environmental managers, policy-makers, scientific researchers, industry, and interested members of the public.

Web sites of interest:

<http://www.usgs.gov>

<http://marine.usgs.gov>

<http://woodshole.er.usgs.gov/project-pages/longislandsound>

Maine Horseshoe Crab (*Limulus polyphemus*) Spawning Surveys, 2001-2002

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Abstract

Spawning surveys of horseshoe crabs (*Limulus polyphemus*) were conducted in 2002, continuing a volunteer monitoring project begun in 2001. Periods of horseshoe crab spawning activity generally follow the lunar phases of late May and June, but the specific days of peak activity have proven difficult to predict. Seven sites were surveyed, with the help of seventy volunteers. Counts were conducted by walking shoreline transects (100m x 1m minimum) on the daily high tide for two five-day periods following the late May new moon and the early June full moon. An index of relative abundance was constructed, based on the daily average of the highest five consecutive days of activity following the late May lunar phase—the more active of the two data periods. The most abundant populations among the sites surveyed during 2001 and 2002 were found at Thomas Point Beach (Brunswick), Middle Bay (Brunswick), and the Damariscotta Mills site (Nobleboro) on Salt Bay. Other spawning sites persist in the Damariscotta River at Days Cove (formerly referred to as Hospital Cove; Damariscotta), Bailey Cove (Wiscasset), the Bagaduce River (Sedgwick) and Taunton Bay (Franklin).

A concurrent tagging study was conducted on Taunton Bay (Hancock County), which is currently the northernmost documented spawning site for this species. During the 2002 season at the tagging site, there were 1,119 observations of 741 individuals, of which 269 were females and 472 were males (380m of shoreline). These numbers were approximately fifty per cent lower than the 2001 data, which recorded 2,273 observations of 1,292 individuals, of which 337 were females and 955 were males. Although there were differences in the number of days on which tagging was conducted, both seasons captured spawning activity in the period subsequent to three lunar phases.

Introduction

American horseshoe crabs (*Limulus polyphemus*) are widely distributed on the coasts of the western Atlantic; spawning is documented from Frenchman Bay (Hancock County), Maine (Schaller *et al.*, 2002), to Mexico's Yucatan Peninsula (Shuster 1982). Harvest pressure and declining horseshoe crab stocks in the mid-Atlantic states generated several iterations of regulations during the 1990's, followed by geographic expansion of the harvest to adjacent unregulated states. Maine's horseshoe crab fishery was unregulated until November 2000, when the Maine Department of Marine Resources (DMR) implemented catch restrictions limiting possession in Maine to 25 per person, per day. Anecdotal reports indicate that Maine's horseshoe crab populations have been in decline for a decade or more, in spite of the harvest pressure having been relatively recent and localized (so far as is known) to the Damariscotta and Bagaduce Rivers. Prior to initiation of this project in 2001, the status of the species in Maine was last documented in 1977 by Born (1977) in his report entitled "Significant Breeding Sites of the Horseshoe Crab (*Limulus polyphemus*) in Maine and Their Relevance to the Critical Areas Program of the State Planning Office".

Horseshoe crabs are sparsely distributed in Maine, and spawning counts probably offer the best opportunity to observe large numbers of adults when they migrate to protected shorelines to spawn at high tide. Males and females move to spawning sites with the rising tide. Males either clasp onto a female and ride along piggy-back fashion, or congregate around a spawning pair. A female, accompanied by one or more males, burrows into the bottom in shallow water and deposits a mass of small grey-green or buff colored eggs. The egg masses vary from clusters of hundreds to thousands of eggs (Rudloe 1980) and the male (or males) shed milt that fertilizes the eggs externally. As the tide begins to drop, the spawners retreat to subtidal refugia, and wet sand slumps into the hole covering the eggs and closing the simple nest.

Horseshoe crab eggs develop and hatch in a period of two weeks or longer, depending on the microclimate of the nest (Shuster 1990). The grain size of the sediments, variations in tide heights, local weather, and depth of the nest and cross-beach elevation of the nest will determine the temperature, salinity, oxygen and moisture regime available to the developing eggs, and fostering or slowing the rate of development (Jegla and Costlow 1982, Penn and Brockman 1994). Juvenile horseshoe crabs grow by molting up to five times the first year, two or three times the second year, a couple of times the third year, and once yearly thereafter. They reach maturity in 15 or 16 molts and 9 to 10 years, respectively, for males and females. (Shuster 1990)

The Maine Horseshoe Crab Surveys were initiated in 2001 in a joint venture between the Maine Department of Marine Resources and Bar Mills Ecological, by means of a grant from the Maine Outdoor Heritage Fund. The goal of the project was to determine whether the species was still present at breeding sites reported by Born in 1977, and to provide baseline date that could be used to monitor species status. Three staff and seventy volunteers collected data during 2001, establishing the first quantitative baseline population data for the state. Data collection for the 2001 counts was limited to periods following the late May new moon and the early June full moon. Analysis of the 2001 data suggested that the timing of the surveys did not necessarily document peak activity for all sites, as would have been shown by counts that increased, peaked and declined. In an effort to improve the data in 2002, within the limitations of reliance on volunteer staffing, the surveys were repeated at fewer sites for two five-day periods. The resulting data are more representative, although it appears that peak activity lagged at two sites. For those two sites, increasing daily counts were documented, but the data series ended before documenting a decline in the daily counts.

In addition to the spawning counts, a tagging study was conducted on Taunton Bay (Hancock County). Sparser populations at the edge of the species' range make it possible to tag every animal encountered. Tagging with individually numbered tags enables distinction between the number of observations and the number of individuals observed throughout the season, as well as clarifying sex ratios and the number of repeat visits by individuals during the late May to June spawning season.

Methods

Spawning counts were conducted only during daylight hours for reasons that included safety, limitations on visibility, and the potential for disrupting the animals. Counts started approximately 30 minutes before high tide, so that the work would be completed close to high tide. The tagging crew started as much as 45 minutes before high tide to accommodate handling time and a longer transect—380m vs. 100m. Completing the transect soon after high tide is important because nearly all the animals retreat to deeper waters as the tide drops. Transects were located at historic spawning sites, based on Born's work which designated sites as having occasional occurrence, juveniles, or being breeding sites, sometimes with a plus or minus to denote greater or lesser abundance, or that crabs were absent.

In conducting the counts, observers walked designated shoreline transects, and counted the number of horseshoe crabs they observed. Transects were a minimum of 100m in length, divided into contiguous 10m sections. To minimize observation bias, data were recorded according to the section, differentiating between the number of horseshoe crabs observed within 1m seaward of the water's edge, and the number of horseshoe crabs observed in the section that were beyond the 10m by 1m swathe. Horseshoe crabs encountered in the 10m by 1m section were recorded as "IN" (within) the transect. Animals encountered more than 1m from the water's edge were recorded as "OUT" (outside) of the transect, but still within the 10m long segment. The "IN" data provided the basis for the index of relative abundance. For other aspects of the study, the data from all observations are included, such as the number of visits by an individual, the ratio of males to females, and the number of animals within the spawning group—often two, but sometimes as many as five or six.

Tagging was conducted using standard fish marking tags, FD-94 (nylon tags anchored with a double T-bar) by Floy Tag and Manufacturing, Seattle, Washington. Tags were attached by drilling a 7/64" hole through the genal angle—at either of the lateral posterior points of the prosoma. The drill bit was sanitized in 5% Povidone-Iodine solution between animals to avoid cross contamination. Although molting occurs infrequently if at all in mature horseshoe crabs, the tag is located so that it would allow molting, and would be shed with the molted shell. In the event of entanglement, a small piece of the edge of the shell would break away resulting in tag loss, but saving the horseshoe crab. The animals appear to be insensitive to the tag and the tagging process, and usually resume mate searching within minutes of being released. The tags are individually numbered, stating "Report number XXXX" on one side (where XXXX represents the tag number), and having an email address, "HorseshoeCrabs@aol.com", on the other side. Gender and prosomal width are documented at the time of tagging. Males are differentiated by claspers on their first (anterior) pair of legs. Claspers are modified pincers that resemble a boxing glove, or a miniature version of a lobster's crusher claw on their anterior-most pair of legs and enable males to clasp onto the shell of a female in amplexus.

Tagging Results

Tagging was conducted for 49 days in 2002, from May 19 to June 29. There were 1,119 observations of 741 individuals, of which 269 were females and 472 were males. These numbers represent a little less than half of the 2,273 observations in 2001 of 1,292 individuals, of which 337 were females and 955 were males. The 2001 tagging covered a shorter period of 25 (of 27) days between May 22 and June 17. Both seasons of data included three lunar phases and the additional days in 2002 were during periods of minimal spawning activity in some cases. Animals tagged in 2001 accounted for 16% of the individuals observed in 2002: 12% of the females and 18% of the males. In the context of the number of animals tagged in 2001, 9% were documented as returning in 2002; representing 10% of the 2001 females and 9% of the 2001 males. The ratio of tagged males to females at this site declined to 1.7:1 in 2002, from 2.8:1 in 2001.

Data from the tagging site for 2001 and 2002 are shown in Table 1, and graphed in Charts 1 and 2 with environmental data for water temperature, tide height, and lunar phase. Bottom temperature from a DMR gauge in Boothbay Harbor was used as a surrogate for regional bottom temperatures due to project limitations. The pattern of two peaks of activity was repeated in 2002, although synchronization with a 4-5 day delay following lunar phase did not occur for the second lunar phase. The first period each year spanned a full week, and was followed by five days of low counts, followed by another peak. The second peak in each year was lower than the first peak of activity, and followed by a low count day, then some intermittent periods of spawning activity. These intermittent periods of low activity involved fewer than half the number of animals observed during either of the prior peaks. Tidal amplitude does not appear to influence spawning activity based on the data.

Although spawning activity occurs on the overnight tide as well as the daytime tide (Barlow *et al.* 1986), only daylight survey data was collected. Analysis of the number of spawning visits documented 274 tagged females making 1.5 spawning visits on average (range 6-1), with 30% observed more than once. Of 467 tagged males, spawning site visits averaged 3.2 (range 9-1), with 63% observed more than once (range 9-1; n=467). In 2001 337 tagged females were observed making 1.7

Table 1. Summary of 2001-2002 tagging data from Shipyard Point on Taunton Bay, Franklin (Hancock County), Maine

2001 Data	*Lunar Phase*	*Transect 1m x 380m*	*Total Obs.*	*Water Temp °C*	*Tide 1 Hgt. Ft.*	*Tide 2 Hgt. Ft.*	2002 Data	*Lunar Phase*	*Transect 1m x 380m*	*Total Obs.*	*Water Temp °C*	*Tide 1 Hgt. Ft.*	*Tide 2 Hgt. Ft.*
							5/19		1	1	11.0	11.1	10.2
							5/20		0	0	9.0	11.0	10.6
5/21		6	6	16.5	11.7	10.7	5/21		0	0	10.5	11.0	11.2
5/22	New	32	37	16.0	12.1	10.9	5/22		0	0	10.0	11.2	11.8
5/23		59	87	16.2	12.4	11.1	5/23		0	0	10.5	11.5	12.4
5/24		94	105	17.5	11.1		5/24		2	3	14.0	11.7	12.9
5/25		149	162	17.5	12.6	11.1	5/25		2	4	14.0	11.8	13.1
5/26		190	206	17.5	12.5	11.0	5/26	Full	5	5	12.0	11.8	13.1
5/27		191	217	15.0	12.3	10.9	5/27		11	19	15.0		11.6
5/28		138	174	15.0	11.9	10.8	5/28		45	49	17.0	12.8	11.3
5/29		169	181	17.0	11.5	10.8	5/29		63	66	16.0	12.4	10.9
5/30			no data		11.2	11.0	5/30		34	47	15.0	11.8	10.4
5/31		12	13	13.5	11.3	11.0	5/31		86	106	18.0	11.2	10.0
6/1		4	5	13.5	11.7	11.0	6/1		128	128	17.0	10.6	9.8
6/2			no data		12.0	11.1	6/2		81	88	15.0	10.1	9.7
6/3		0	0	13.0	12.2	11.2	6/3		56	69	14.0	9.8	9.7
6/4		3	3	13.0	12.3	11.2	6/4		1	3	12.5	9.7	9.9
6/5	Full	14	31	15.0	12.2	11.1	6/5		7	12	14.5	9.7	10.1
6/6		26	40	16.0	10.9		6/6		7	11	13.0	9.8	10.4
6/7		124	150	17.5	12.0	10.7	6/7		9	22	13.0	9.9	10.7
6/8		127	136	16.0	11.6	10.4	6/8		19	22	14.5	10.0	11.0
6/9		154	194	19.0	11.3	10.1	6/9		49	59	16.0	10.1	11.3
6/10		173	193	17.0	10.9	9.8	6/10	New	83	95	17.0	10.2	11.5
6/11		89	96	17.5	10.5	9.6	6/11		18	36	14.0	10.3	11.7
6/12		9	17	16.0	10.1	9.5	6/12		3	5	13.5	*10.4*	10.4
6/13		45	55	17.0	9.8	9.5	6/13		10	13	15.5	11.9	10.4
6/14		73	74	21.0	9.7	9.7	6/14		16	33	15.0	11.9	10.5
6/15		70	74	23.5	10.0	9.6	6/15		0	1	13.0	11.8	10.6
6/16		31	38	19.0	10.4	9.7	6/16		0	1	12.5	11.6	10.8
6/17		23	29	19.5	10.9	9.9	6/17		2	3	14.0	11.4	11.0
6/18		26	33	20.5	11.5	10.2	6/18		4	19	14.0	11.1	11.3
6/19		25	27	19.5	12.0	10.5	6/19		22	39	15.0	11.0	11.6
6/20		25	26	21.0	12.5	10.8	6/20		0	2	15.0	10.9	12.0
6/21	New	26	27	18.0	12.8	11.1	6/21		10	15	17.5	11.0	12.3
6/22		9	9	15.0	11.3		6/22		16	19	18.0	11.1	12.6
6/23		6	6	14.5	12.9	11.4	6/23		15	16	18.0	11.1	12.6
							6/24	Full	12	14	20.0	11.1	12.5
							6/25		8	17	17.5	11.0	*11.0*
							6/26		3	8	17.5	12.3	10.8
							6/27		6	6	19.5	11.9	10.6
							6/28		4	7	19.0	11.5	10.3
							6/29		4	5	19.5	11.0	10.1
Total:		1078	2451				Total:		842	1068			

Table 2. Index of Relative Abundance (100x1m transect at high tide)

2002 Survey Sites	*Town*	*2002 5 Day Mean **	*2001 Highest Count*	*2002 Highest Count*
Middle Bay	Brunswick	**84**	56	103
Thomas Pt. Beach	Brunswick	**373**	103	703
Bailey Cove	Wiscasset	**8**	n/a	15
Damariscotta Mills	Nobleboro	**81**	192	99
Days Cove	Damariscotta	**36**	50	102
Bagaduce River	Sedgwick	**10**	16	16
Taunton Bay	Franklin	**21**	46	34

* Based on the 5 highest consecutive counts from 5/27-6/2/02.

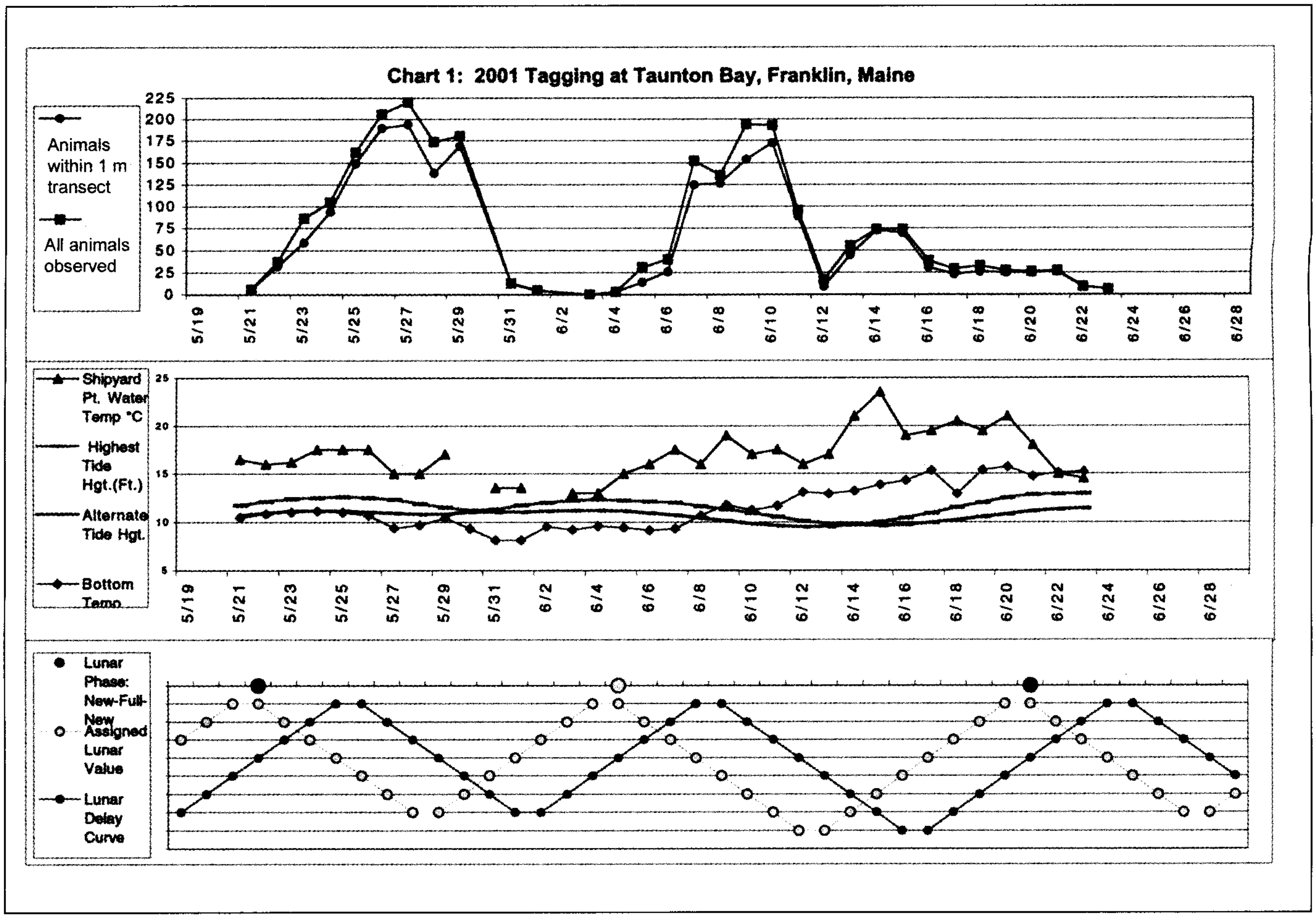

Chart 1: 2001 Tagging at Taunton Bay, Franklin, Maine

visits on average (range 8-1), with 42% observed more than once; 955 tagged males averaged 1.8 visits (range 8-1) and 69% seen more than once.

Survey Results

Spawning counts were conducted at seven sites during 2002, dropping several sites with little or no activity in 2001 (Stover Cove, Merepoint Neck, Pleasant Cove, Salt Pond), and combining the effort from multiple sites on the Damariscotta River in 2001 to two sites there in 2002. Most surveys were conducted along a 100m transect, although the Taunton Bay transect was 380m. Surveys at Middle Bay and Bailey Cove each included longer stretches of shoreline, but in both cases only the first 100m section of the transect was used in the analysis. The Taunton Bay tagging site data have been normalized to 100m. Data are provided in Appendix A for the counts and water temperature readings.

An index of relative abundance for the survey sites was constructed using the average of the highest five consecutive days for each site. This period included the first and larger of the two seasonal peaks of spawning activity at all sites. The index is shown in Table 2. Thomas Point Beach (Brunswick), Middle Bay (Brunswick), and the Damariscotta Mills site (Nobleboro) on Salt Bay host the most abundant populations among the sites surveyed during 2001 and 2002. Other important spawning sites for this species persist in the Damariscotta River at Days Cove (formerly referred to as Hospital Cove; Damariscotta), Bailey Cove (Wiscasset), the Bagaduce River (Sedgwick) and Taunton Bay (Franklin).

Temperature appears to function as a limiting factor in spawning activity, since most activity this season was documented as occurring at temperatures of 14 °C or higher; only three animals were observed in 2002 at water temperatures of 12 °C or lower. In 2001, periods of spawning activity occurred at 14 °C or warmer, except for one event at Damariscotta Mills. That particular event occurred at 12 °C, bracketed by days when water temperature was 14 °C or higher. The temperature

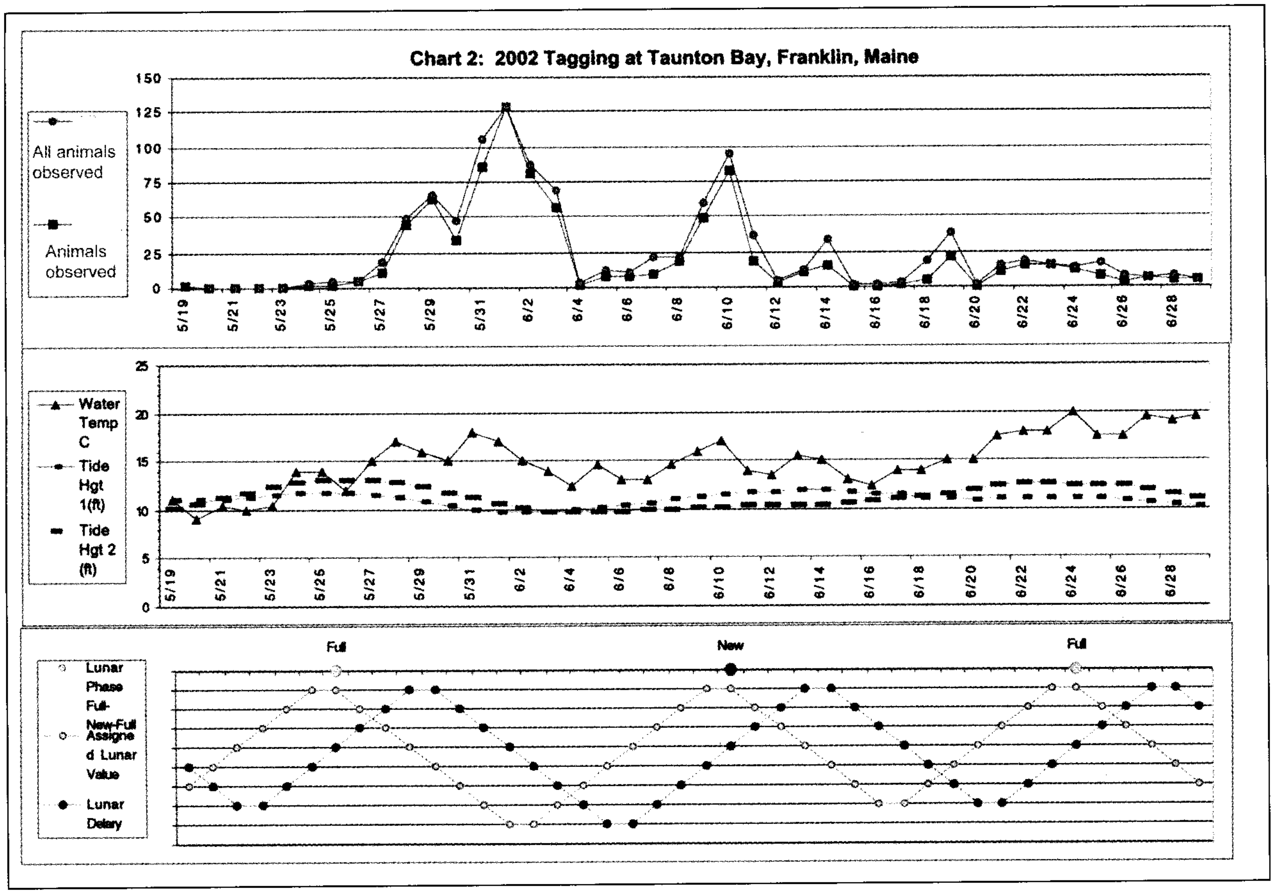
Chart 2: 2002 Tagging at Taunton Bay, Franklin, Maine

drop may have resulted from freshwater inflows from the Darmariscotta River into Salt Bay, close to the survey site (also known as the Mills Sewer Filter for the wastewater treatment discharge pipe 500' offshore).

Charts 3a - 3f graph water temperatures and counts for the period of May 24–June 3. The count data has been normalized to represent each day's count as a percent of the peak day for the period, for the site. There was a full moon on May 26, and activity for all sites peaked during the period of May 29 to June 1.

Discussion

The survey data from 2002 offers a longer data series for each site than the 2001 data, and is believed to have captured data for the day of highest spawning activity for most sites, as demonstrated by counts that recorded low numbers, increased over two or more days, reached a peak level, and subsequently declined for one or more days. The trend was incomplete at two sites, the Bagaduce River and Days Cove, which lagged behind the other sites. These sites documented increasing counts, but reached the end of the data collection period before documenting decreasing counts. It appears that one or two more days of data collection would have shown the expected post-peak decline. It cannot be known for certain if the peak was recorded, nonetheless the data are presented.

Assuming that counts for Days Cove and the Bagaduce River are accepted as having reached their maximum, the pattern of activity graphed in chart 9, Activity by Percent of Peak for All Sites, demonstrates that activity for all sites peaked between May 29th and June 1. This means that peak activity spanned a range of only 3-6 days following the full moon on the May 26th.

The horseshoe crab population in Middle Bay is probably under-represented by the high tide counts, because fine silt renders water opaque at the shoreline. The 2001 high tide counts reported a maximum of 57 animals, but a volunteer reported counting over 800 on the flats at low tide one day. In 2002, low tide counts were conducted along a creek that flows into the northeast corner of Middle Bay. The highest number counted at low tide was 1,389 animals over a length of 140m (992/100m) on June 10, 2002. Most of the creek is 1m in width; parts of the last 10m at the mouth widening to approximately 2m.

Conclusion

Continuation of counts is recommended to monitor the status of this species. Efforts are underway to continue the tagging study for 8-10 more years in hopes of documenting adult longevity in Maine, and the dynamics of this population. Additional horseshoe crab populations in Maine could be identified by conducting shoreline surveys for molted shells in late summer. Volunteer participation is integral to the continuation and success of this project. Even with its limited scope, this project provides important baseline data and has identified factors that influence horseshoe crab spawning behavior in northern waters.

Acknowledgments

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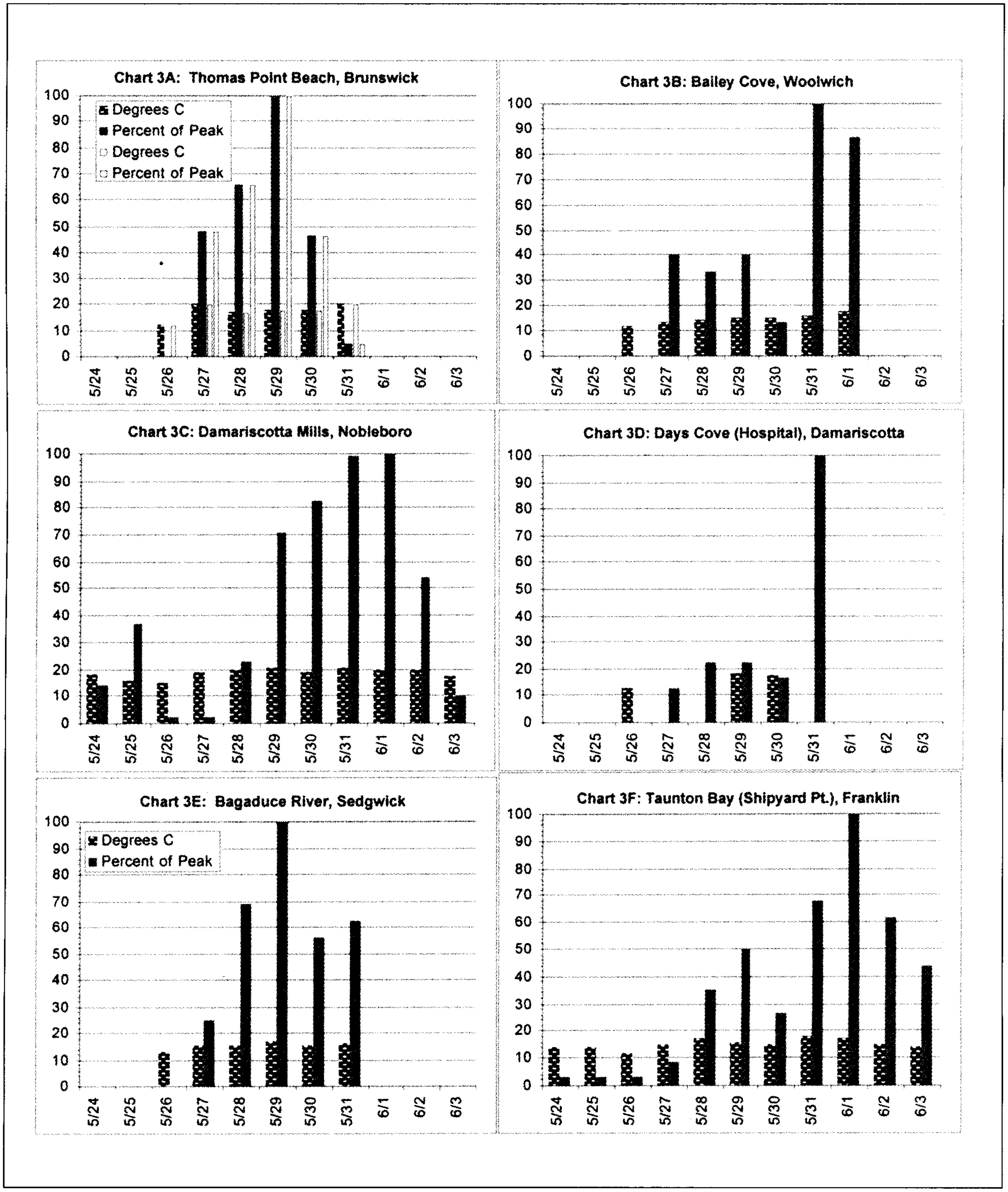

Charts 3a - 3f. Water temperatures and counts for May 24 - June 3.

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Appendix A: Survey Data by Site, showing counts and water temperature (100 x 1m).

Date	Taunton Bay*		Middle Bay		Thomas Pt. Be.		Bailey Cove		Damar. Mills		Days Cove		Bagaduce R.	
	Temp	Count	Temp	Count	Temp	Count	Temp	Count	Temp	Count	Temp	Count	Temp	Count
5/15/02									12.0	0				
5/16/02									14.0	0				
5/17/02									14.0	2				
5/18/02										0				
5/19/02		0							13.0	4				
5/20/02		0							7.0	0				
5/21/02		0							12.0	0				
5/22/02		0							14.0	0				
5/23/02		0		0					16.0	0				
5/24/02	14.0	1		32					18.0	14				
5/25/02	14.0	1		38					16.0	37				
5/26/02	12.0	1		46	12.0	2	12.0	0	15.0	2	13.0	0	12.5	0
5/27/02	15.0	3		**99**	16.0	**335**	13.4	6	19.0	2		**13**	15.0	**4**
5/28/02	17.0	12		**80**	17.0	**465**	14.0	**5**	20.0	23		**23**	15.0	**11**
5/29/02	16.0	**17**		**80**	18.0	***703***	15.3	**6**	20.5	**70**	18.0	**23**	16.5	***16***
5/30/02	15.0	**9**		***103***	18.0	**327**	14.9	**2**	19.0	**82**	17.5	**17**	15.0	**9**
5/31/02	18.0	**23**		**60**	20.0	**35**	16.2	***15***	20.5	**98**		***102***	16.1	**10**
6/1/02	17.0	***34***		10			17.3	**13**	20.0	***99***				
6/2/02	15.0	**21**		16					20.0	**54**				
6/3/02	14.0	15		0					17.0	10				
6/4/02		0							20.5	12				
6/5/02		2							17.0	18				
6/6/02		2							17.0	23				
6/7/02		2							16.5	26				
6/8/02		5							21.0	42				
6/9/02		13					16.0	0	20.0	43				
6/10/02		22			17.0	8	16.0	2	21.0	143	18.0	35	15.0	16
6/11/02		5			15.0	6		0	17.0	14	16.0	11	14.0	10
6/12/02		1		34	13.0	3			15.0	3	15.0	3	12.0	1
6/13/02		3		0	16.0	12			19.0	19	15.5	26	14.0	8
6/14/02		4		42	16.0	5			17.0	1	15.5	5	14.0	5
6/15/02		0		2					14.0	3	14.0	7		
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* Taunton Bay data normalized from 380m transect.

Nitrate and Phosphate Concentrations in Salt and Fresh Water

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Nitrate and phosphate are limiting nutrients needed by salt and fresh water organisms. My experiment showed that several factors influence concentrations of nutrients in salt and fresh water. I hypothesized that both nutrients would have higher concentrations in salt water than in fresh and there would be more nutrients present at ebb than flood tide.

To do this experiment, I collected water samples and tested with the Hach and LaMotte testing kits. I tested the nitrate and phosphate concentrations in salt and fresh water at ten different locations. Salt water stations were in the Poquonock River and Baker's Cove at Project Oceanology in Groton, Connecticut. I tested at eight fresh water stations along the Yantic River and seven stations in lakes, tributaries, and streams in Bozrah. I compared the results with water quality standards for fresh and salt water. This study indicated that the conditions that might affect the concentrations of nitrate and phosphate include precipitation, waste disposal, ice melt, and the salt concentration.

My hypothesis was partially correct. In salt water, the concentrations of both nitrate and phosphate were higher than that in fresh water. There were more nutrients at ebb tide than flood tide and more in salt water than fresh water. In fresh water, the nitrate concentration was higher after precipitation.

Originally, 52 graphs were made to show the data. Figure 1 shows the compiled graphs of nitrate concentrations in salt water. It is possible that a combination of precipitation, runoff, and leaching during ebb tide was responsible for the higher concentrations seen at all stations following these events. Five stations on August 13 were greater that normal. Three of these were in the enriched range. The runoff from higher elevations and the increased fresh water from all the contributing streams may have added to the nitrate concentration. After this tide, the same day, most values were back within the normal range, except for station 10, which was still high. Nitrate values remained normal for the rest of the study.

Figure 2 shows the compiled data of nitrate concentrations in fresh water. All days and stations were within the normal range. The highest of these was January 25th, at Station 4. This could have been high because of possible waste disposal from a nearby chicken farm. Over the course of this study, nitrate readings never exceeded the normal limit of 2.5 mg/L. However, readings did range from 0 - 1.76 mg/L.

Figure 3 shows all phosphate data gathered in both salt and fresh water. All salt water concentrations were in the normal range for the entire study. However, the highest within these was Station 9 on August 13, flood tide, with a concentration of .65 mg/L. This sample was taken during flood tide, the day after it rained. Tides and precipitation did not affect the phosphate concentrations. The fresh water data showed that phosphate was scarce or not able to be tested in fresh water. There was only one reading of phosphate in fresh water, on November 12. The concentration was 0.1 mg/L. This reading was before a rain event. After that testing date, the values went back to 0 mg/L for the rest of the study.

A suspected reason for the low values was the lack of precipitation. In each month, there a lower than average precipitation. From September 1 to November 21 there was a -5.88 inch decrease in rainfall. We received only 42% of the normal amount of precipitation that this part of the country

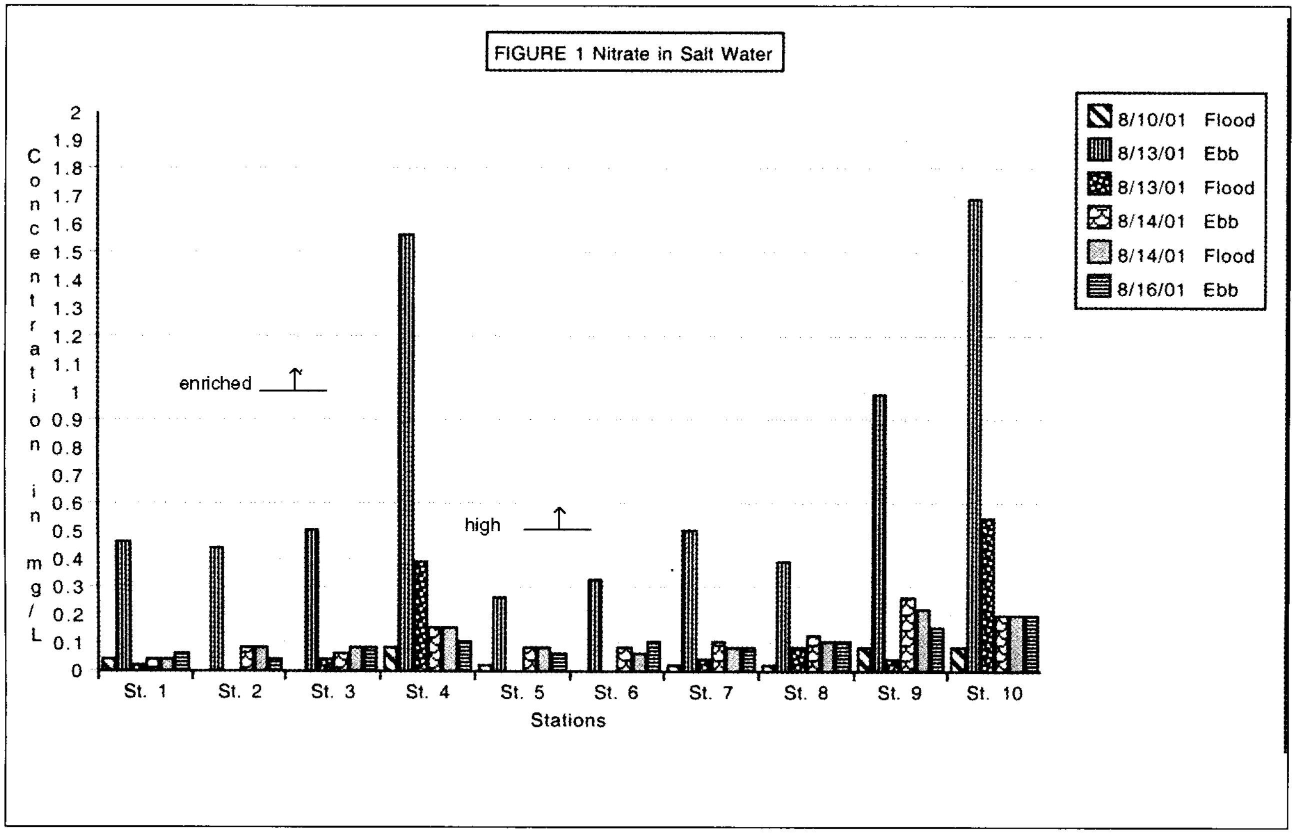

Figure 1. Nitrate in salt water.

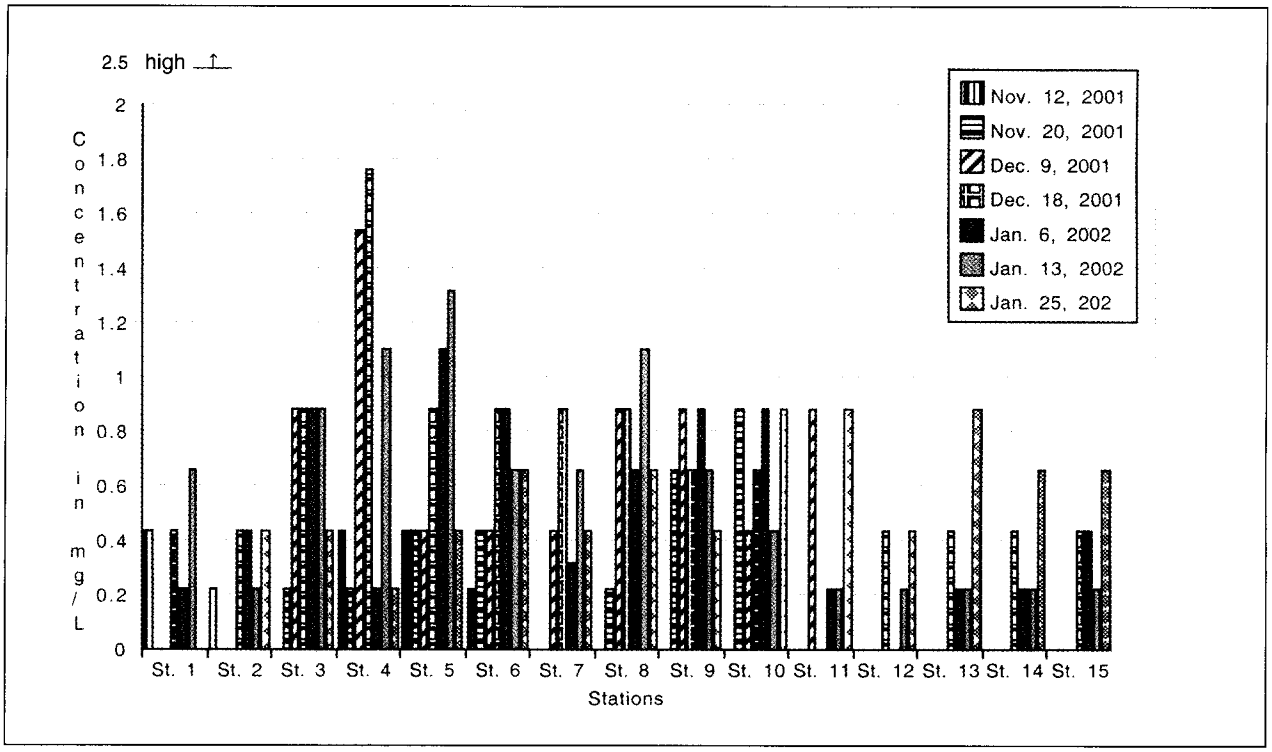

Figure 2. Nitrate in fresh water.

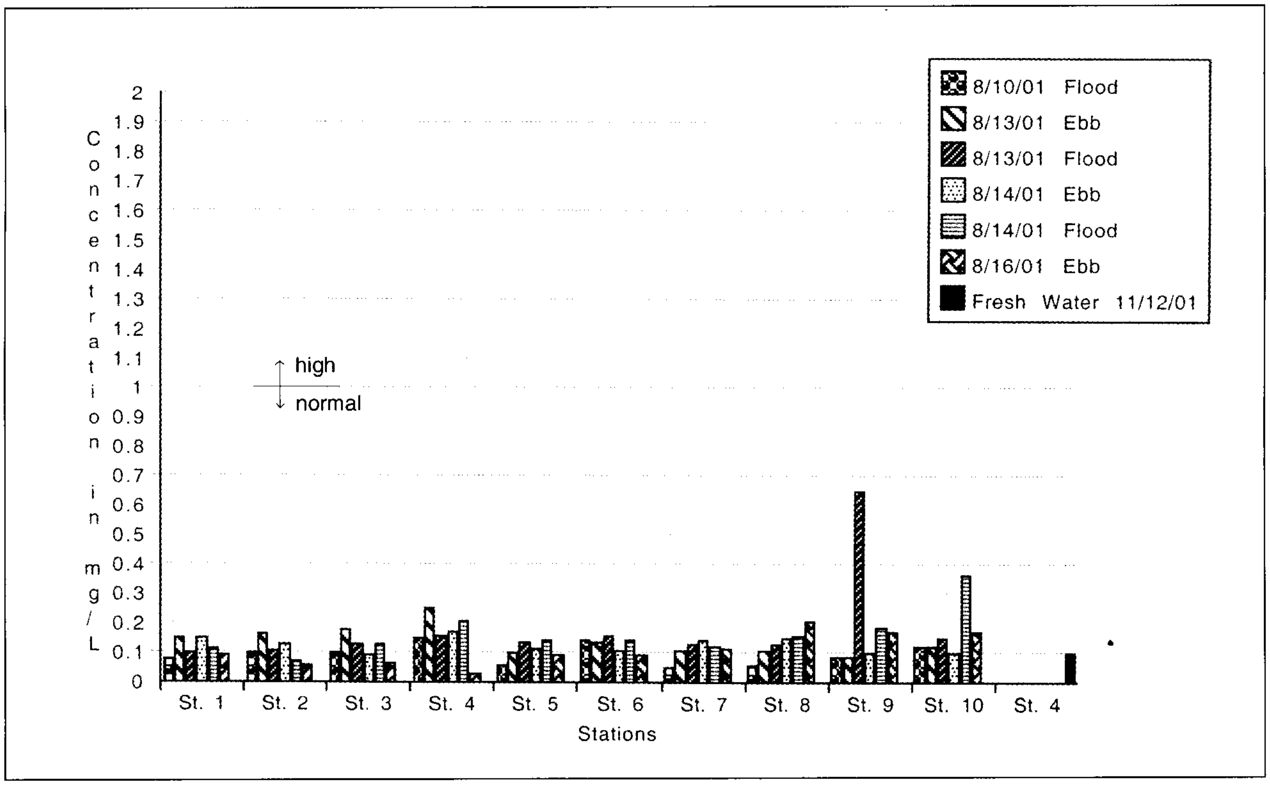

Figure 3. phosphate in salt and fresh water.

usually receives. This could be why there wasn't as much nutrient in the water during fall and winter seasons as there was in the summer. There was insufficient data to make a conclusion about the impact for rainfall on fresh water nutrient load.

To eliminate the testing system as a variable between salt and fresh water, all fresh and salt water was tested with a Hach Colorimeter. Generally, the results were slightly higher for Hach than LaMotte when compared. The Hach had greater precision. It showed the results to the hundredths rather that the tenths as the LaMotte did. However, the testing system used did not change the conclusions.

A recent sediment survey of Fitchville Pond, conducted by Baystate Environmental Consultants, tested for levels of nitrate and phosphate among other things. Their study found high levels of both nitrate and phosphate at three of six sites in the pond. No actual number values were given. More tests would be needed to determine what affect the pond nutrients would have on the overall watershed nutrient load.

There were some variables in this experiment. The time of day I collected the water could have impacted the nutrient levels tested if agricultural waste was disposed of into tributaries feeding into the fresh water pond. The fact that I tested in a salt environment versus fresh water also could have made a difference. The salt water stations were located at the end of a river and had many contributing streams. This gave the watershed more sources to provide nutrients. The fresh water was at the beginning of the watershed. Here, the water might not get as many nutrients built up from traveling through as many places. Another variable was that in salt water I used an electronic Hach DR/700 Colorimeter, which gave precise reading. For fresh water, I used the LaMotte methods, which included looking at an axial reader and comparing colors to get a less precise reading. This variable was eliminated through additional testing with the same Colorimeter that was used for the salt water tests. The amount of time between collecting and testing could also make a difference. For salt water, I tested samples immediately after I collected them. For fresh water, I tested up to a week after I collected the water samples.

If I were to continue this experiment, I could test water at different times of the day to see if morning or afternoon affected the concentrations of nutrients. I could also test different parts of the same body of water. For example, I could test at various locations in Fitchville Pond. Testing the same samples over time would help determine if bacteria and other microscopic organisms change the nutrient levels. I could also test to see if an increased nutrient level causes an algae problem. Another test could be to test the soil in different locations for levels of nutrients. Although over 330 tests were conducted over the course of this study, it became apparent that many variables played a role in the nutrient levels in a watershed. Many more studies would be needed to pinpoint the actual events producing the observed nutrient level.

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Connecticut's Nitrogen Control Program to Manage Long Island Sound Hypoxia

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Introduction

The Long Island Sound Study (LISS) was initiated in 1985, when Congress directed the U.S. EPA, in cooperation with the states of Connecticut and New York to sponsor the LISS. A Management Conference was established, including federal, state, interstate, and local agencies, universities, environmental groups, industry, and the public, and was charged with developing a Comprehensive Conservation and Management Plan (CCMP) that addressed Long Island Sound's most pressing environmental problems. The problem of hypoxia drew most of the characterization and planning attention and funding because of its severity and also because it is a problem whose significance had not been previously recognized. Low oxygen levels impact large areas of the Sound's bottom waters during the late summer hypoxic period. LISS research and computer modeling efforts have linked the problem to excess loading of a nutrient–nitrogen–to the system. Nitrogen stimulates the growth of microscopic algae that ultimately die and decay on the bottom of Long Island Sound. During decay, oxygen is consumed.

The recently adopted Total Maximum Daily Load (TMDL) for Long Island Sound requires Connecticut and New York to reduce total nitrogen loadings to Long Island Sound (LIS) by 58.5% by 2014. This represents the most ambitious nitrogen management program formalized under EPA's TMDL program in the U.S. To meet the 58.5% reduction, Connecticut must reduce sewage treatment plant (STP) nitrogen loads 64% and urban and agricultural runoff by 10%. Connecticut's nitrogen-trading program brings 79 individual STPs under one general permit (GP) and into a Nitrogen Credit Exchange (NCE). The NCE is an unprecedented and innovative approach to meet the TMDL requirements, taking advantage of natural attenuation processes to determine each STP's relative impact on LIS. The NCE is expected to save $200 million in statewide capital cost savings for upgrades. While only in its first year of operation, aggregate nitrogen reductions appear to be meeting planned targets established for 2005.

Management History

In the CCMP, the LISS proposed a phased approach to control nitrogen loading to Long Island Sound. Phase I capped nitrogen loads from western Long Island Sound sewage treatment plants at 1990 base levels. Phase II, also aimed at the largest source - sewage treatment plants - committed the states to reduce nitrogen loads from key, western Sound treatment plants by 25%. These actions yielded a reduction in the overall human-generated load of nitrogen from Connecticut and New York of more than 18%. Phase III, formalized in the TMDL approved by EPA in 2001 requires Connecticut and New York to reduce their collective load of nitrogen to LIS by 58.5% by the year 2014. Additional phases of management call upon the federal EPA to coordinate nitrogen reductions from neighboring states (MA, NH, VT) in the watershed and plan for meaningful reductions in atmospheric nitrogen deposition.

The Connecticut Department of Environmental Protection (CTDEP) has a strong commitment to implement the plan. The plan builds upon the programs and longstanding commitments Connecticut has already established to clean up and protect Long Island Sound. In support of Phases I and II, Connecticut earmarked $18 million from its Clean Water Fund to allow municipalities to begin controlling nitrogen discharges from southwestern Connecticut coastal STPs. Grants were awarded to 11 communities to «retrofit» their treatment plants to remove nitrogen and attain the Phase II 25% target reduction in the amount of human-produced nitrogen entering Long Island Sound from those plants. Through 2001, CTDEP awarded more than $55 million to 22 municipalities for nitrogen control out of an overall $280 million in STP upgrades since 1993. Full implementation of Phase III nitrogen control projects for STPs required by the TMDL is expected to cost Connecticut taxpayers on the order of $200 million by 2014 of an overall $700 million bill for STP construction projects.

The General Permit and the Nitrogen Credit Exchange

In June of 2001, just months after EPA approved Connecticut and New York's TMDL, the Connecticut General Assembly passed Public Act (PA) 01-180 and Governor Rowland signed it into law. An Act Concerning Nitrogen Reduction in Long Island Sound provided the foundation for a Nitrogen General Permit (GP) to put all 79 municipal STPs under one umbrella authority for nitrogen control, and established both a Nitrogen Credit Exchange (NCE) to allow trading of nitrogen and a Nitrogen Credit Advisory Board (NCAB) to oversee pricing and the actual trading. In accordance with due process, including ample opportunity for public input through public hearings and meetings, the GP was issued in December 2001, setting the wheels in motion for the first annual trade based on nitrogen loads monitored during calendar year 2002.

The GP establishes annual end-of-pipe nitrogen limits in pounds per day for each of 79 STPs for 2002 through 2006. Each year's limit has an equal percent step down from the baseline for each facility, i.e., the permit limit is lowered each year, and a final limit to be met in 2014 is also specified. The permit will be reissued at the end of 2006 to include the next five-year cycle of limits, also with equal incremental step-downs for each facility, for the years 2007 through 2011.

The annual step-down in nitrogen permit limits is based on anticipated, aggregate statewide reductions from state Clean Water Fund projects scheduled for that respective year. Because CTDEP and affected municipalities have been proactive, and aware that nitrogen control was looming on the horizon, many plant retrofits and full upgrades had been implemented prior to issuance of the permit (Figure 1). As a result, the first year, 2002 permit limits are based on more than a 25% reduction from the baseline established in the TMDL. Subsequent years are in the range of 7 to 17 percent incremental reduction, depending on anticipated upgrade project completion schedule. At this rate of progress, Connecticut's statewide nitrogen reductions from point sources will be met well in advance of the 2014 TMDL requirement. The permit also sets conditions for effluent monitoring, reporting, compliance, and allows for the sale of excess credits and the purchase of deficit credits by permittees within the NCE.

What makes nitrogen trading so attractive in Connecticut is the anticipated cost savings realized when market forces help determine which facilities remove more nitrogen and which find economic advantage to become buyers of credits. What makes nitrogen trading feasible is the geographic distribution of STPs throughout the state. Geographic distribution positions some STPs very close to the hypoxic zone in western Long Island Sound (Figure 1). Their nitrogen is very efficiently delivered and, pound for pound, the nitrogen discharged at the end of the pipe has a greater impact on oxygen levels than more distant STPs. Nitrogen from those more distant STPs is attenuated as it travels down rivers and also as it travels from east to west through the Sound. Pound for pound, nitrogen discharged at the end of the pipe from those STPs has less impact on oxygen levels in western Long Island Sound than the more proximate plants.

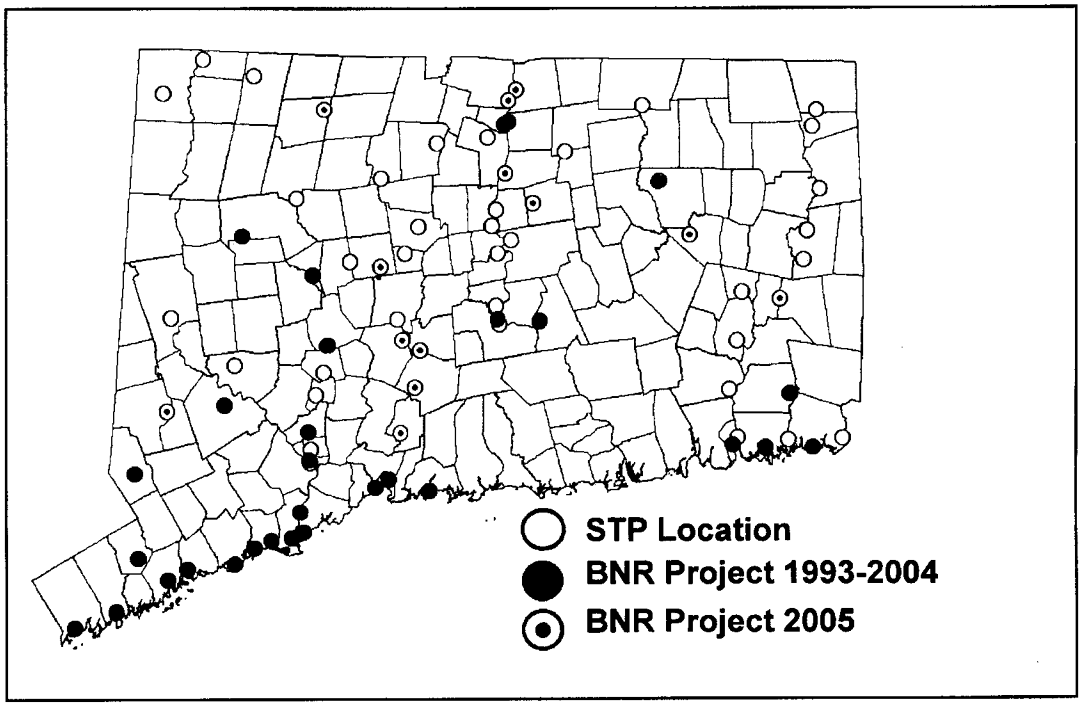

Figure 1. Upgrade status of 79 municipal STPs involved in the Nitrogen Cred Exchange in Connecticut.

Using empirical evaluations of nitrogen transport losses, relative attenuation factors were calculated for each STP based on location. They range from 86% nitrogen loss from plants in northwestern Connecticut such as Putnam and Plainfield, to 51% in the upper Quinnipiac River, to 0% in the Norwalk to Greenwich portion of Fairfield County. Attenuation factors are specified in the TMDL and the GP. These relative impacts of nitrogen because of attenuation make nitrogen removal much more cost-effective in Norwalk than Putnam, for example, and provide an incentive for Norwalk to remove more nitrogen than its permit requires and sell those credits to the NCE. Similarly, Putnam may find the cost of credits at a very favorable exchange ratio financially attractive compared to the high cost of a plant upgrade and choose not to remove nitrogen and purchase their credits on the NCE.

Using the attenuation factors, end-of-pipe nitrogen loads can be converted to "equivalent" pounds, or E-pounds, that equalize each STP's nitrogen load based on relative impact on oxygen in LIS. E-pounds facilitate nitrogen trading and pricing by incorporating relative values of nitrogen from each STP, much as a currencies might be converted to a common currency (e.g., U.S. dollars) to describe relative pecuniary values and facilitate trading. In the case of Norwalk, one end-of-pipe pound would equal one E-pound. In the case of Putnam, one end-of-pipe pound would equal 0.14 E-pound. If Norwalk removed 100 more end-of-pipe pounds of nitrogen than their permit required, they would receive payment from the NCE at the going price for 100 E-pounds. If Putnam exceeded their end-of-pipe permit limit by 100 pounds of nitrogen, they would only need to buy 14 E-pounds from the NCE at the going price.

The conversion of end-of-pipe pounds to E-pounds does not affect the end result of removing 64% of the baseline STP load that reaches the LIS hypoxic zone. By applying a percent reduction, end-of-pipe and E-pound targets are in direct proportion and the reduction of nitrogen delivered to LIS is the same. Only the distribution of source reductions among the 79 STPs is flexible while the aggregate, statewide reduction still would comply with the TMDL schedule under any distribution scenario. Use of E-pounds simplifies the pricing and trading of nitrogen on the NCE without compromising the TMDL reduction goal and the anticipated water quality benefits to LIS.

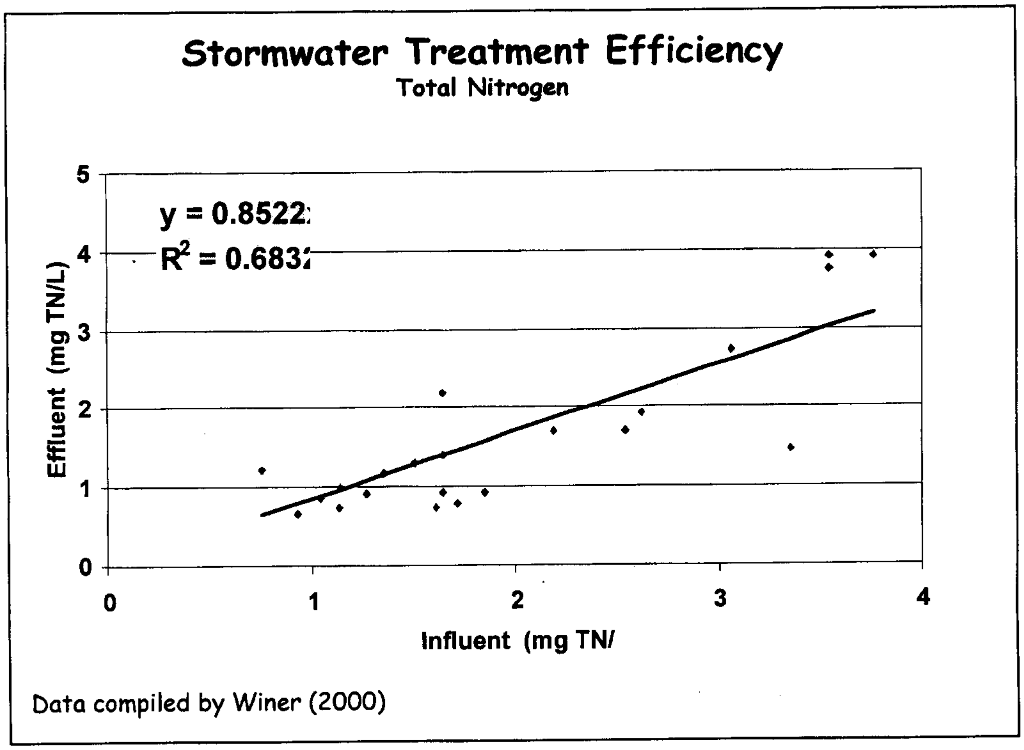

Figure 2. Connecticut STP performance for end-of-pipe (EOP) and equalized (E-Pound) nitrogen loads to Long Island Sound in 2002.

The First Year's Operation of the Nitrogen Credit Exchange

The GP was issued in December 2001, marking calendar year 2002 as the first operational year of the NCE. The 79 STPs involved in the NCE are required to monitor their effluents throughout the year and report the results to CTDEP on a monthly basis. The results of the first year's monitoring will be published by March 2003 and will identify those facilities that have credits to sell to the NCE and those that need to buy credits, and the respective numbers. The NCAB has been meeting on a monthly basis to track progress and to estimate what the cost of an equalized credit will be. The cost is based on the non-loan, amortized portion of the BNR projects that generated nitrogen reduction credits plus the annual operation and maintenance costs. The price will be set by March so municipalities will know how much they will receive or pay to the NCE. The current estimate of the cost of an E-pound (credit) of nitrogen for 2002 is about $1.50. Payments will be due in July of 2003 for those that need to buy credits and disbursements will be made to those who sold credits to the NCE by August.

The nitrogen credit program has already met many management expectations and benefits of a trading program. Management efforts are economically pushed towards the southwestern portion of Connecticut, where the impacts to LIS are more direct and the equalization factors larger. The 2002 data demonstrate the added efficiency of trading to meet the LIS TMDL. Although during the first five months of 2002 end-of-pipe nitrogen loads exceeded the aggregate permit limit for 2002, the equalized loads were consistently below the aggregate permit limit (Figure 2). In fact, 2002 performance is approaching the equalized nitrogen discharge permit limit set for 2004-2005.

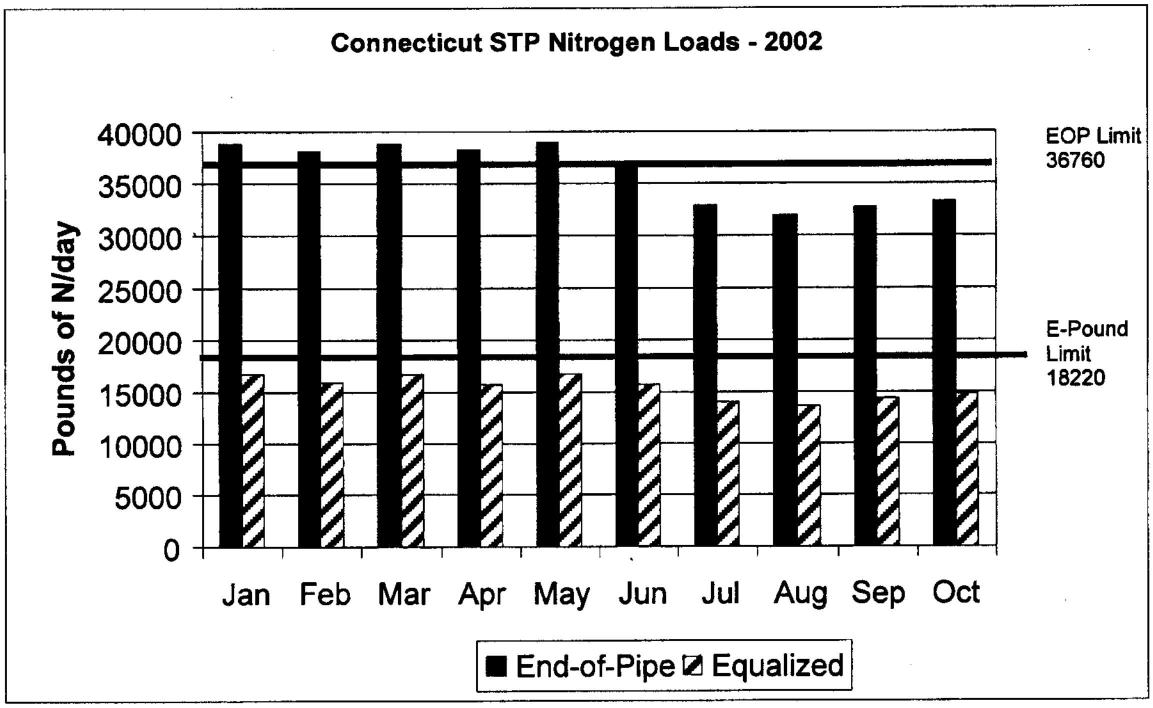

Figure 3. Nitrogen removal efficiency of stormwater best management practices.

Nonpoint Source Management

Management of nonpoint sources presents a more difficult challenge. Although Connecticut's goal is to reduce only 10% of the nonpoint source baseline from urban and agricultural land covers, standard best management practices (BMP) do not very efficiently remove nitrogen. Using literature values, for example, stormwater BMPs that are recognized as the most efficient nitrogen removers capture or denitrify only about 15% of the influent total nitrogen (Figure 3). Treating existing loads is difficult but, despite application of the most efficient management practices and smart growth development techniques, new development will still produce a net increase in nitrogen loading over natural land cover. Coupled with anticipated growth, a town like Wallingford, CT, for example, could only expect to lower its urban loading of nitrogen by 6% even with aggressive application of BMPs and smart growth techniques for new growth. Statewide, only about a 5% reduction in NPS nitrogen from urban areas could be expected at a cost of over $2 billion. For nitrogen management, pollution prevention techniques are likely to yield the most benefits as, once in the soil and groundwater, there are limits as to how much nitrogen can be removed through management.

Conclusions

By 2014 Connecticut is required to remove nearly 50% of its baseline nitrogen load to LIS to alleviate hypoxia in western LIS and meet water quality standards. Most (64%) will come from point source reductions, primarily STPs. Through application of a nitrogen credit exchange program, Connecticut municipalities have made excellent progress towards meeting nitrogen reduction goals and are already at a level not expected until 2005. Nonpoint source reductions of nitrogen are more of a challenge and probably will require a strong pollution prevention ethic coupled with aggressive BMP application.

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Eutrophication of Long Island Sound as Traced by Benthic Foraminifera

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Introduction

Benthic foraminifera are marine, unicellular eukaryotes which secrete a shell (test) of calcium carbonate or agglutinate mineral grains in an organic matrix. Benthic foraminiferal assemblages in grab samples taken from Long Island Sound (LIS) in the early 1960s were described extensively (Buzas, 1965). These faunas were low-diversity and dominated by 1-3 species, as expected for marginally marine assemblages. At water depths of less than 10-15 m the dominant species was *Elphidium excavatum*, which consumes living diatoms (Murray, 1991; Bernhard and Bowser, 1999). In deeper areas *Buccella frigida* and *Eggerella advena*, which use more refractory organic carbon (Murray, 1991), were common. In the mid-1990's assemblages in grab samples had changed considerably, and in western LIS, which has suffered summer hypoxia since the early 1970's (Parker and O'Reilly, 1991), the species *Ammonia beccarii* had become common to dominant (Thomas *et al.*, 2000).

In order to document ecosystem changes since the European settlement in the mid-1600's and to constrain the timing of the change from *Elphidium*- to *Ammonia*-dominated faunas, we collected benthic foraminiferal assemblage data in samples from 9 cores on depth transects in Long Island Sound (LIS) (Figure 1; Buchholtz ten Brink *et al.*, 2000). Preliminary ages of core samples were determined using ^{137}Cs, ^{210}Pb and metal pollution data (Varekamp *et al.*, 2000, 2003).

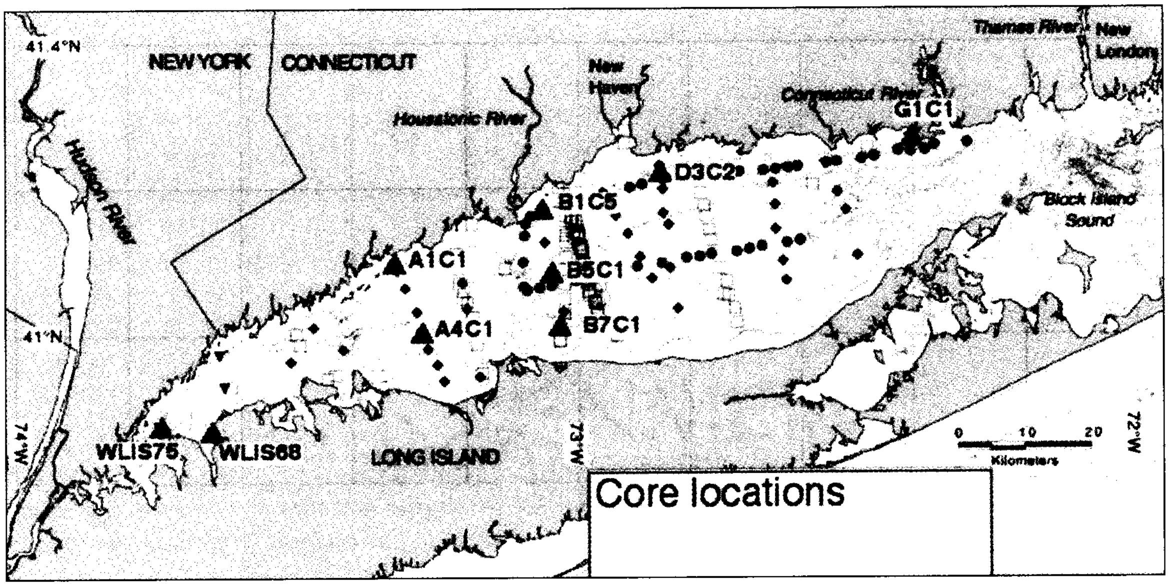

Figure 1. Location of cores studied in Long Island Sound. WLIS 68 and WLIS 75: Narrows; A1C1, A4C1, western Long Island Sound; B1C5, B5C1, B7C1, D3C2: central Long Island Sound; G1C1: eastern Long Island Sound, at the mouth of the Connecticut River.

Results

In most cores, foraminifera increased in absolute abundance (number of foraminifera per gram of dry bulk sediment) in the early through middle 1800's, coeval with an increase in relative abundance of *E. excavatum* (Figures 2a, b). These faunal changes coincided with an increase in metal contamination in the sediment (Varekamp *et al*, 2000, 2003), an increase in the concentration of the bacterial spore and sewage indicator *Clostridium perfringens* (Buchholtz ten Brink *et al.*, 2000), and in the concentration of organic carbon in the sediments (Lugolobi *et al.*, this volume) as well as in the accumulation rate of organic carbon (Varekamp *et al.*, this volume).

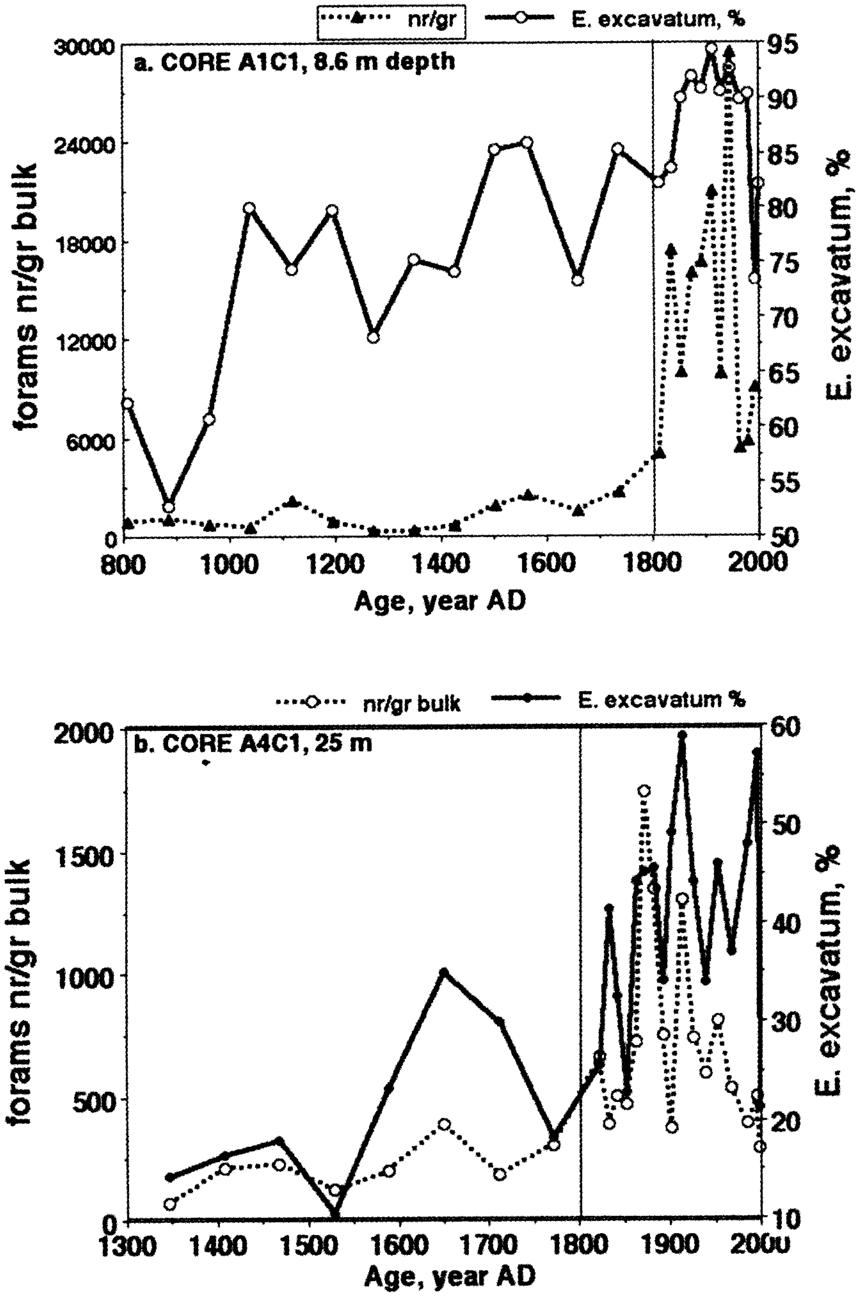

Figure 2. Relative abundance of *Elphidium excavatum* and absolute abundance of foraminifera (number per gram dry bulk sediment) for cores A1C1 (a) and A4C1 (b).

Additional and more profound changes in LIS benthic foraminiferal assemblages occurred after the late 1960's. In cores in western LIS and close to the mouth of the Connecticut River, overall foraminiferal abundance decreased (Figure 2), while the species *Ammonia beccarii* became common or dominant (Figure 3). The increase in *A. beccarii* was most extreme in western LIS where episodes of summer hypoxia/anoxia are most common and most severe (Parker and O'Reilly, 1991). This profound faunal change started in the 1960's-1970's in western LIS and outside the mouth of the Connecticut River, then extended further into the Central Basin in the 1990's.

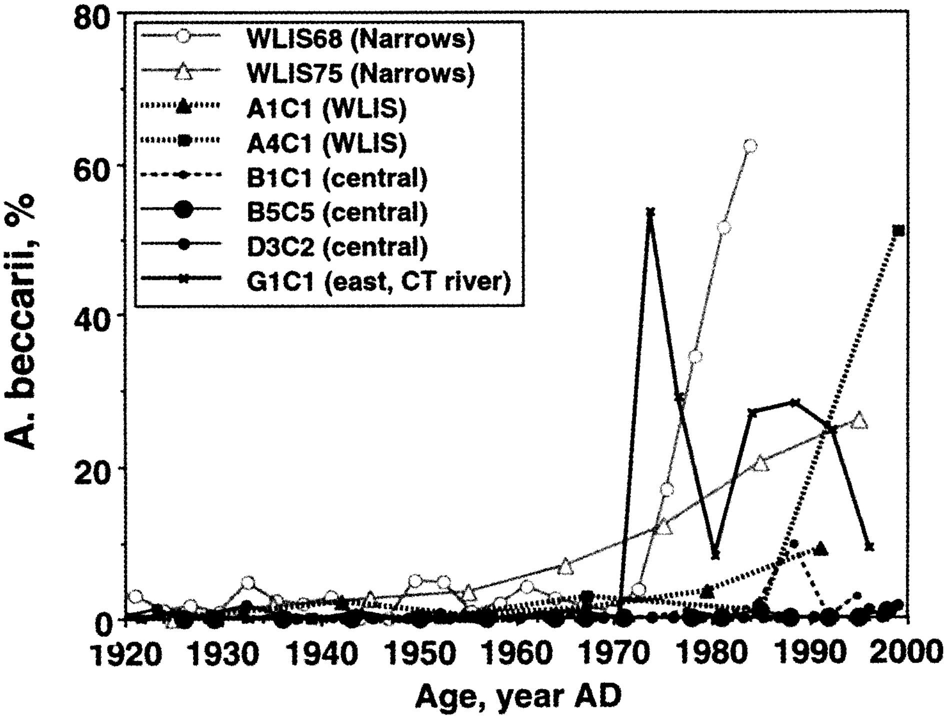

Figure 3. Relative abundance of *A. beccarii* in cores in different parts of LIS.

Discussion

Increased nutrient influx in the middle of the 19th century may have led to increasing biological productivity, as documented by increasing concentrations of organic carbon and nitrogen in the sediments and increasing accumulation rates of organic carbon, especially in western Long Island Sound and the Narrows (Lugolobi *et al*, this volume; Varekamp *et al* this volume). We suggest that this increasing productivity included an increase in productivity of planktonic diatoms, the main food source of *E. excavatum*, as was also documented in Chesapeake Bay (Cooper and Brush, 1993). An increase in productivity of diatoms in response to increased anthropogenic nutrient input may well have caused the changes in benthic foraminiferal faunas in the mid-19th century: increasing percentages of the diatom-consuming *E. excavatum* and increasing overall foraminiferal abundance.

Less clear is the cause of the increase in relative abundance of *A. beccarii* and the coeval decrease in overall abundance of foraminifera. Both *Elphidium* and *Ammonia* species are adapted to marginally marine conditions and survive in highly polluted waters (e.g., Alve, 1995). A replacement of *Elphidium*-dominated by *Ammonia*-dominated assemblages in the Gulf of Mexico (Sen Gupta *et al*, 1996; Platon and Sen Gupta, 2001) and Chesapeake Bay (Karlsen *et al*, 2000) has been argued to have been caused by declining oxygen levels. This explanation, however, is problematic because both taxa easily survive low-oxygen conditions in both natural (e.g. Alve, 1995) and laboratory settings (Moodley and Hess, 1992).

A factor that could have caused the increase in relative abundance of *A. beccarii* is the recent increase in water temperature in LIS, which could have given the competitive advantage to *Ammonia* over *Elphidium. A. beccarii* reproduces successfully in the laboratory only if temperatures exceed 20°C for at least several weeks (Bradshaw, 1957; Schnitker, 1974), whereas *Elphidium excavatum* is abundant in assemblages as far North as the Arctic Ocean (e.g., Polyak *et al*, 2002). One can thus argue that increasing temperatures may have played a role in the faunal changes in LIS, but this was probably not so in the Gulf of Mexico or even in Chesapeake Bay, where temperatures have not been in the limiting range for *A. beccarii* in the past.

We speculate that the main cause for the replacement of *Elphidium*-dominated assemblages by *Ammonia*-dominated assemblages in LIS may have been the influx of N-rich effluent from waste water treatment plants, which has led to an increase in N/Si values. High N/Si values strongly influence phytoplankton composition (Rabalais and Turner, 2001; Dortch *et al* 2001), because they give the competitive advantage to organic-walled primary producers, including cyanobacteria and dinoflagellates, over diatoms which form a siliceous frustule (Escaravage and Prins, 2002). Many organisms that are higher in the food chain, however, strongly prefer or require diatoms rather than organic-walled phytoplankton as their food source, and changes in phytoplankton assemblages thus reverberate through the food chain (e.g., Rabalais and Turner, 2001).

Phytoplankton in western LIS shows a recent shift from being diatom-dominated to being dominated by organic-walled phytoplankton (Capriulo *et al* 2002). *E. excavatum* preferentially consumes diatoms, and such a change in primary producers would thus give the competitive advantage to other species of foraminifera. The total foraminiferal abundance is expected to decline when primary producers are dominated by flagellates because many foraminiferal species do not use these as food (Murray, 1991). The changes in composition of the phytoplankton resulting from high N-concentrations would be expected to be most severe in the regions with most severe eutrophication, in which areas we also see the most severe hypoxia, explaining the correlation between the increasing relative abundance of *Ammonia* and low-oxygen conditions.

Conclusions

Benthic foraminifera are very common in LIS sediment cores and we used their tests to reconstruct historical changes in the LIS ecosystem. The foraminiferal assemblages indicate that the biota started to react to eutrophication in the middle of the 19th century. Profound ecosystem changes occurred from the late 1960s-early 1970s on, particularly in western LIS. These changes may have been caused by increasing N/Si ratios, which would cause major changes in the primary producers in LIS by giving the advantage to organic walled photosynthesizing algae over diatoms. This change then reverberated throughout the entire ecosystem, including the benthic microfauna.

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Trends in Abundance of Eggs, Larvae, Juvenile and Adult Fish Collected From 1976 Through 2001 in Eastern Long Island Sound

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Abstract

Long Island Sound supports a diverse assemblage of fish species, including year-round residents, seasonally migratory fishes, and transient species. Several of these species support important commercial and sport fisheries including winter flounder, scup, and tautog. At Millstone Power Station located on Long Island Sound in Waterford, Connecticut, a total of 121 fish taxa was recorded as eggs, larvae, juveniles, or adults from collections in the trawl, seine, and ichthyoplankton programs. This total includes 111 taxa taken by trawl, 49 by seine, and 58 enumerated in ichthyoplankton samples. Anchovies accounted for almost half of the larvae collected, winter flounder comprised another 15% and thirteen other taxa made up most of the remainder. Cunner, tautog, and anchovies accounted for nearly 90% of the eggs collected. Silversides dominated (>80%) the seine catch. Eight taxa accounted for about 80% of the total catch at the trawl stations. These were the winter flounder, scup, silversides, windowpane, grubby, skates, anchovies and cunner. Temporal changes in the composition of fish collected over the past 26 years in these monitoring programs are examined.

Introduction

Long Island Sound supports a diverse assemblage of fish species, including year-round residents, seasonally migratory fishes, and transient species. Several of these species support important commercial and sport fisheries including winter flounder (*Pseudopleuronectes americanus*), scup (*Stenotomus chrysops*), and tautog (*Tautoga onitis*). Monitoring of fish taxa has been conducted at the Millstone Power Station (MPS) located on Long Island Sound in Waterford, CT, since 1976 to determine impact of the power facility on the local fish assemblages. Fish taxa have been recorded as eggs, larvae, juveniles, or adults from collections in the trawl, seine, and ichthyoplankton programs. Sampling methods for all three monitoring programs have been consistent at fixed sites over the entire period, providing a continuous 26-year time series of comparable abundance data for all life stages of fish.

Materials and Methods

Ichthyoplankton Program

Sampling frequency of ichthyoplankton entrained through the MPS cooling-water system varies seasonally. Both day and night samples are collected twice a week during June through August, once a week in September and February, and three times a week during March through May. One daytime sample per week is collected during October through January. Samples are collected at the plant discharge, using a 1.0 x 3.6-m conical plankton net with 335-µm mesh deployed with a gantry system. Sample volume is determined from the average readings of the four General Oceanic flowmeters. Under normal plant operation, the net is usually deployed for 3 to 4 minutes. All ichthyoplankton collections are preserved in 10% formalin.

Ichthyoplankton samples are subdivided in the laboratory using a splitter. Fish eggs and larvae are removed from the subsamples with the aid of a dissecting microscope. Successive splits are completely sorted until at least 50 larvae (and 50 eggs for samples processed for eggs) are found, or until one-half of the sample has been examined. Larvae are identified to the lowest practical taxon and enumerated in all samples. All ichthyo-plankton densities are reported as number per 500 m^3 of water filtered.

Trawl Program

Triplicate bottom tows are made to collect demersal fish using a 9.1-m otter trawl with a 0.6-cm codend liner every other week throughout the year at three stations: Niantic River (NR), Jordan Cove (JC), and Intake (IN) (Figure 1). A typical standard tow is 0.69 km in length. Catch is expressed as the number of fish per standardized tow (CPUE).

Seine Program

Shore-zone fish are sampled using a 9.1 x 1.2-m knotless nylon seine net of 0.6-cm mesh. Triplicate shore-zone hauls (standard distance of 30 m) are made parallel to the shoreline at Jordan Cove (JC) biweekly from May through November (Figure 1). Collections are made during a period 2 hours before and 1 hour after high tide. Fish from each haul are identified to the lowest possible taxon and counted. Catch is expressed as number of fish per haul.

Data Analysis

A Δ-mean is used as an index of abundance for fish eggs and larvae in the ichthyoplankton program. The Δ-mean is the best estimator of the mean for abundance data that approximates the lognormal distribution and contains numerous zeros (Pennington 1983, 1986). Because of varying

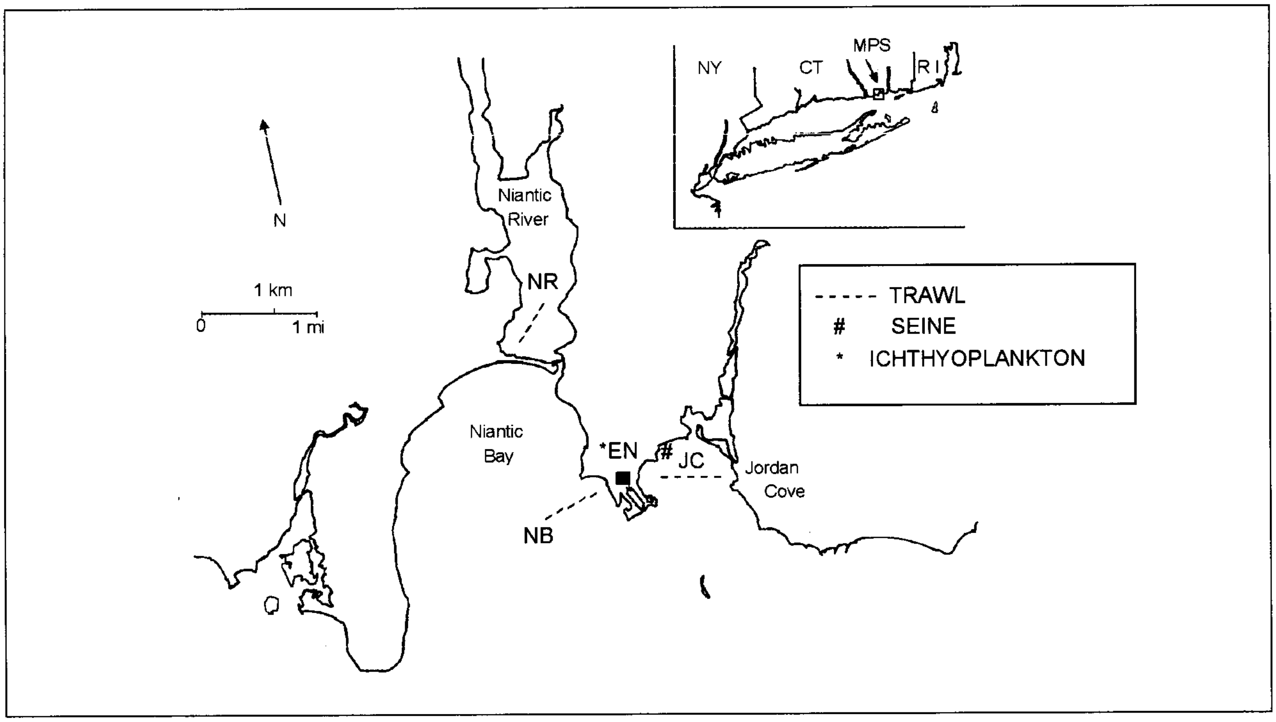

Figure 1. Location of current trawl, seine and ichthyoplankton sampling stations.

sampling frequencies, the Δ-mean indices of ichthyoplankton taxa were weighted by the largest number of samples collected in a week to standardize data across weeks and years. Since ichthyoplankton taxa occur seasonally, data to calculate each Δ-mean are restricted by month to reduce the number of zero values in the distribution tails, which extend beyond the occurrence seasonal boundaries. Temporal changes of juvenile and adult fish taxa collected in the trawl and seine programs during the 25-year period are compared using annual catch per unit effort (CPUE). Catches in both programs are equally distributed through the years.

Results

A total of 121 fish taxa (eggs, larvae, juveniles, or adults) were recorded from collections made in the trawl, seine, and ichthyoplankton programs as part of the monitoring studies at MPS from January 1976 through December 2001. This total includes 111 taxa taken by trawl, 49 by seine, and 58 enumerated in ichthyoplankton samples (Table 1).

Cunner eggs (*Tautogolabrus adspersus*) were always the most abundant of the fish eggs collected and tautog (*Tautoga onitis*) were the second most abundant eggs (Table 2). A large decline in anchovy, mostly bay anchovy (*Anchoa mitchilli*) egg abundances occurred after 1984 with the 2000 _-mean density (< 1.0) being the lowest in this time-series. Annual abundance of most of the 15 dominant larval taxa was determined (Table 2). Historically, anchovies were the most abundant larvae, but the past two years winter flounder, Atlantic menhaden (*Brevoortia tyrannus*) and grubby had much higher larval _-mean densities. On occasion, larvae of other species, particularly those of cunner, tautog, rock gunnel (*Pholis gunnellus*), fourbeard rockling (*Enchelyopus cimbrius*), and radiated shanny (*Ulvaria subbifurcata*) were relatively abundant.

Silversides (Atlantic silverside, *M. menidia*, and inland silverside, *M. beryllina*) dominated (80%) the seine catch at station JC (Table 3); another 10% were killifishes (striped killifish, *Fundulus majalis*, and mummichog, *F. heteroclitus*), with fourspine stickleback (*Apeltes quadracus*) and Atlantic menhaden each making up 4% of the catch. There was a dramatic increase in the numbers of Atlantic menhaden in the seine catch recently, with the catches in 1998, 1999 and 2000 accounting for over 70% of the 26-year total.

Eight taxa accounted for over 80% of the total catch at the three trawl stations (Table 4-6). These were the winter flounder (41%), scup (*Stenotomus chrysops*; 17%), silversides (mostly Atlantic silverside; 7%), windowpane (*Scophthalmus aquosus*; 5%), grubby (4%), skates (mostly little skate, *Raja erinacea*; also the winter skate, *R. ocellata*, and clearnose skate, *R. eglanteria*; 3%), cunner (3%), and anchovies (mostly bay anchovy; 3%).

Eight species had higher abundance during the past 2 or 3 years than during any other time of the 25-year period. These were tautog, scup and hakes (*Urophycis* spp.) at all three stations; summer flounder (*Paralichthys dentatus*) at NB and NR, black sea bass (*Centropristis striata*) at JC and NB, smallmouth flounder (*Etropus microstomus*) at NB; and rock gunnel (*Pholis gunnellus*) and cunner at JC (Tables 4-6). Two species, winter flounder and windowpane, had lower abundance at all three stations in 2000 and 2001 than at any other time in the past (Tables 4-6). Anchovies also exhibited low abundance over the past 2 years, but they are schooling species and catches of this species have always been patchy. The total catch of fish over the 26-year period was relatively similar between IN (132,203) and NR (120,131), with the catch at each of these stations more than twice the total of 75,158 fish at JC.

Discussion

Fish abundance and assemblage composition can be highly variable in the marine environment. While seasonal changes in distribution and abundance of fish are somewhat consistent and predictable (Potter *et al.* 1986), variability in population and community dynamics can be caused by various interconnected physical and biological attributes. Many studies of fish communities are relatively short in duration and therefore lack temporal resolution to identify complex trends such as changes in stability and variability due to climate changes and habitat decline. Examination of the Millstone data reveals changes in species abundance and assemblage composition across the 26-year period that monitoring has occurred.

Variability in environmental factors, diel and seasonal movements of resident species, migrations of non-resident species, species interactions, and fishing pressure can influence the abundance, distribution, and catchability of a species (Hilborn and Walters 1992. In this study, methods and sampling stations remained unchanged throughout the study period and equal effort was applied among each sampling year. Therefore, the variability observed in our catches should reflect natural fluctuations in population structure and catchability of taxa at our sampling stations and not artifacts of the methods employed.

Many of the factors affecting fish assemblages listed above of have been documented over the past two decades in eastern Long Island Sound. Studies of eelgrass (*Zostera marina*), an ecologically important marine vascular plant, conducted over the last 16 years indicate that this once dominant habitat component (McGill 1974; Marshall 1994) has all but disappeared (DNC 2002; Vozarik *et al.* 2000). Heavy sets of blue mussel (*Mytilus edulis*) and macroalgal mats (*Agardhiella subulata, Ulva lactuca, Desmarestia viridis*) now cover large areas of the bottom of Niantic River (DNC 2002), precluding settlement and growth of other species dependent on sandy/silt substrates. Long-term temperature monitoring indicates a significant warming trend in January-June seawater temperatures over the last 21 years in eastern Long Island Sound (Foertch 2000). High fishing mortality is suspected as a significant contributor to the causes of the winter flounder stock decline (Sinclair and Murawski 1997). The precipitous decline in catch of winter flounder seen in our data is not unique to the immediate area but was common to the entire northeast (Simpson *et al.* 1996; Jeffries 2000).

Long-term monitoring studies can reveal shifts in species assemblages, relative abundance, and environmental factors such as seawater temperature. Comparable to our results are trends revealed in the nearly 40-year data set from the University of Rhode Island (URI) Narragansett Marine Laboratory and Graduate School of Oceanography bottom trawl program (Jeffries 2000). The URI data also show increases in relative abundance of skates, while winter flounder, silver hake, and northern searobin declined. Jeffries (2000) also reports that mean annual seawater temperature in Narragansett Bay has increased by approximately 1.4°C between 1966 and 2000, similar to the temperature rise in eastern Long Island Sound (Foertch 2000).

Results of our study revealed changes in fish assemblages occurring across a 26-year period and emphasized the usefulness of long-term data sets to detect these changes. We surmise that many of the observed changes are linked to increased water temperature, and declines in eelgrass populations. Trends observed in our study have been detected in other areas sampled along the northeast coast of North America.

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Table 1. List of fishes collected in the Fish Ecology sampling programs in the vicinity of the Millstone Power Station, Waterford, CT from January 1976 through December 2001.

Scientific name	Common name	Trawl[a]	Seine[a]	Ichthyoplankton
Acipenser oxyrhynchus	Atlantic sturgeon	*		
Alosa aestivalis	blueback herring	*	*	
Alosa mediocris	hickory shad	*		
Alosa pseudoharengus	alewife	*	*	*
Alosa sapidissima	American shad	*	*	
Alosa spp.	river herring	*	*	*
Aluterus schoepfi	orange filefish	*		
Ammodytes americanus	American sand lance	*	*	*
Anchoa hepsetus	striped anchovy			*
Anchoa mitchilli	bay anchovy	*	*	*
Anguilla rostrata	American eel	*	*	*
Apeltes quadracus	fourspine stickleback	*	*	*
Bairdiella chrysoura	silver perch	*		
Bothus ocellatus	eyed flounder	*		
Brevoortia tyrannus	Atlantic menhaden	*	*	*
Brosme brosme	cusk	*		
Caranx crysos	blue runner	*	*	
Caranx hippos	crevalle jack	*	*	
Centropristis striata	black sea bass	*		*
Chaetondon ocellatus	spotfin butterflyfish	*		*
Chilomycterus schoepfi	striped burrfish	*		
Clupeidae	herrings	*		*
Clupea harengus	Atlantic herring	*	*	*
Conger oceanicus	conger eel	*		*
Cyclopterus lumpus	lumpfish	*		*
Cynoscion regalis	weakfish	*	*	*
Cyprinodon variegatus	sheepshead minnow		*	*
Dactylopterus volitans	flying gurnard	*		
Dasyatis centroura	roughtail stingray	*		
Decapterus macarellus	mackerel scad	*		
Decapterus punctatus	round scad	*		
Enchelyopus cimbrius	fourbeard rockling	*		*
Etropus microstomus	smallmouth flounder	*		*
Eucinostomus lefroyi	mottled mojarra		*	
Fistularia tabacaria	bluespotted cornetfish	*		
Fundulus diaphanus	banded killifish		*	
Fundulus heteroclitus	mummichog	*	*	
Fundulus luciae	spotfin killifish		*	
Fundulus majalis	striped killifish		*	
Gadidae	codfishes	*		*
Gadus morhua	Atlantic cod	*		*
Gasterosteus aculeatus	threespine stickleback	*	*	*
Gasterosteus wheatlandi	blackspotted stickleback	*	*	*
Gobiidae	gobies	*		*
Gobiosoma ginsburgi	seaboard goby	*		
Hemitripterus americanus	sea raven	*		*
Hippocampus erectus	lined seahorse	*		*
Labridae	wrasses			*
Lactophrys spp.	boxfish	*		
Leiostomus xanthurus	spot	*		
Limanda ferruginea	yellowtail flounder	*		*
Liparis spp.	seasnail	*		*
Lophius americanus	goosefish	*		*
Lucania parva	rainwater killifish	*	*	
Lutjanus apodus	schoolmaster		*	
Macrozoarces americanus	ocean pout	*		
Melanogrammus aeglefinus	haddock	*		
Menticirrhus saxatilis	northern kingfish	*	*	*
Menidia beryllina	inland silverside	*	*	
Menidia menidia	Atlantic silverside	*	*	*
Merluccius bilinearis	silver hake	*	*	*
Microgadus tomcod	Atlantic tomcod	*		*

Table 1. (continued).

Scientific name	Common name	Trawl	Seine	Ichthyoplankton
Monacanthus hispidus	planehead filefish	*		
Monocanthus spp.	filefish	*		
Morone americana	white perch	*		*
Morone saxatilis	striped bass	*	*	
Mugil cephalus	striped mullet	*	*	*
Mugil curema	white mullet		*	
Mullus auratus	red goatfish	*		
Mustelis canis	smooth dogfish	*		
Myliobatis freminvillei	bullnose ray	*		
Myoxocephalus aenaeus	grubby	*	*	*
Myoxocephalus octodecemspinosus	longhorn sculpin	*		*
Myoxocephalus spp.	sculpin	*		
Ophidiidae	cusk-eels	*		
Ophidion marginatum	striped cusk-eel	*	*	*
Ophidion welshi	crested cusk-eel	*		
Opsanus tau	oyster toadfish	*		
Osmerus mordax	rainbow smelt	*	*	*
Paralichthys dentatus	summer flounder	*		*
Paralichthys oblongus	fourspot flounder	*		*
Peprilus triacanthus	butterfish	*	*	*
Petromyzon marinus	sea lamprey	*		
Pholis gunnellus	rock gunnel	*	*	*
Pollachius virens	pollock	*		*
Pomatomus saltatrix	bluefish	*	*	
Priacanthus arenatus	bigeye	*		
Priacanthus cruentatus	glasseye snapper	*		
Pristigenys alta	short bigeye	*		
Prionotus carolinus	northern searobin	*	*	*
Prionotus evolans	striped searobin	*	*	*
Pseudopleuronectes americanus	winter flounder	*	*	*
Pungitius pungitius	ninespine stickleback	*	*	*
Raja eglanteria	clearnose skate	*		
Raja erinacea	little skate	*		
Raja ocellata	winter skate	*		
Salmo trutta	brown trout	*		
Sciaenidae	drums			*
Scophthalmus aquosus	windowpane	*	*	*
Scomber scombrus	Atlantic mackerel	*		*
Scyliorhinus retifer	chain dogfish	*		
Selar crumenopthalmus	bigeye scad	*		
Selene setapinnis	Atlantic moonfish	*		
Selene vomer	lookdown	*	*	
Synodus foetens	inshore lizardfish	*		
Sphyraena borealis	northern sennet	*		
Sphoeroides maculatus	northern puffer	*	*	*
Squalus acanthias	spiny dogfish	*		
Stenotomus chrysops	scup	*		*
Strongylura marina	Atlantic needlefish		*	
Syngnathus fuscus	northern pipefish	*	*	*
Tautogolabrus adspersus	cunner	*	*	*
Tautoga onitis	tautog	*	*	*
Trachinotus falcatus	permit	*	*	
Trachurus lathami	rough scad	*		
Trachinocephalus myops	snakefish	*		
Trinectes maculatus	hogchoker	*		
Ulvaria subbifurcata	radiated shanny	*		*
Upeneus parvus	dwarf goatfish	*		
Urophycis chuss	red hake	*		
Urophycis regia	spotted hake	*		
Urophycis tenuis	white hake	*		
Urophycis spp.	hake	*	*	*

[a] Includes species collected at other stations near Millstone Power Station that are no longer sampled.

TABLE 2. The annual Δ-mean density (no./500 m^3) of the most abundant fish eggs and larvae collected at EN for each year from January 1976 through May 2001.

Taxon	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988
EGGS[a]													
T. adspersus	-	-	-	8,348	8,379	7,326	7,874	7,580	6,707	12,842	2,579	5,017	5,388
T. onitis	-	-	-	1,648	3,741	2,501	3,561	2,372	1,817	4,027	2,833	2,972	2,211
Anchoa spp.	-	-	-	1,558	999	769	499	2,415	3,631	118	586	64	32
LARVAE[b]													
Anchoa spp.	1,152	931	483	2,168	2,430	5,768	816	1,421	302	1,102	1,244	126	359
P. americanus	-	106	143	114	285	129	233	297	210	180	87	109	116
A. americanus[c]	-	94	318	119	111	136	21	27	18	9	3	13	41
B. tyrannus	5	3	3	1	2	1	9	18	2	38	2	2	5
M. aenaeus	-	41	38	36	38	107	72	68	50	68	34	29	95
P. gunnellus	-	13	13	16	58	58	27	13	14	14	22	4	26
T. adspersus	29	58	1	13	58	78	31	49	4	12	4	5	9
T. onitis	37	36	1	11	46	83	44	33	3	15	3	7	17
E. cimbrius	2	8	6	8	6	1	6	13	5	8	8	12	45
Liparis spp.	27	30	10	16	22	5	13	8	36	1	4	42	18
U. subbifurcata	-	5	9	14	14	16	17	6	4	60	7	9	23
S. fuscus	3	7	4	9	8	13	7	9	9	5	4	6	7
S. aquosus	9	11	1	5	5	5	2	13	3	1	4	3	5
P. triacanthus	14	3	1	2	11	17	9	9	1	2	3	<1	9
C. harengus	-	1	1	1	<1	6	1	0	1	<1	2	1	14

Taxon	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
EGGS[a]													
T. adspersus	6,255	7,269	6,987	2,776	4,535	8,722	4,266	8,801	3,610	1,459	3,530	8,245	-
T. onitis	3,373	1,942	2,040	1,189	1,394	1,350	1,807	2,323	587	897	1,37	2,322	-
Anchoa spp.	32	89	317	62	329	234	118	36	5	114		<1	-
LARVAE[b]													
Anchoa spp.	619	1,122	799	178	203	475	181	175	131	106	129	65	-
P. americanus	203	106	99	381	21	142	224	81	255	137	182	328	296
A. americanus[c]	31	24	7	18	28	43	63	18	11	28	13	53	12
B. tyrannus	47	16	81	37	8	44	56	145	23	28	58	293	-
M. aenaeus	63	30	24	58	34	48	43	85	140	55	39	45	188
P. gunnellus	9	6	3	15	8	28	17	41	22	42	12	22	21
T. adspersus	14	68	209	8	10	25	12	12	7	21	15	17	-
T. onitis	15	33	99	13	6	12	8	18	2	14	59	12	-
E. cimbrius	31	37	98	5	18	9	8	2	10	9	4	2	-
Liparis spp.	12	3	23	14	12	5	<1	46	2	2	0	4	-
U. subbifurcata	41	51	34	28	2	18	8	<1	18	1	10	1	1
S. fuscus	5	3	5	3	6	4	6	2	2	2	4	3	-
S. aquosus	3	4	12	2	2	3	1	4	2	1	2	3	-
P. triacanthus	5	29	10	2	2	5	3	1	2	1	2	4	-
C. harengus	1	1	2	9	7	4	4	33	11	7	13	0.5	1

[a] Fish eggs not identified prior to 1979. Data were seasonally restricted to May-August for *T. adspersus*, May-August for *T. onitis*, and June-August for *Anchoa* spp.

[b] Data seasonally restricted to July-September for *Anchoa* spp., March-June for *P. americanus*, December-May for *A. americanus*, June-December for *B. tyrannus*, February-May for *M. aenaeus*, January-May for *P. gunnellus*, June-August for *T. adspersus*, June-August for *T. onitis*, March-May for *Liparis* spp., April-September for *S. fuscus*, April-June for *U. subbifurcata*, April-July for *E. cimbrius*, May- October for *S. aquosus*, June-September for *P. triacanthus*, and February-May for *C. harengus*.

[c] Period of occurrence (December-May) extends across calendar year; year given for Δ-means is that of the latter 5 months.

Table 3. Total number of fish caught by seine at station JC from 1976 through 2001.

Taxon[a]	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	Total
Menidia spp.	37,265	15,956	411	542	5,610	1,812	2,005	7,057	1,658	467	3,087	3,375	4,075	2,515	5,668	3,686	4,513	2,227	2,931	4,459	6,193	7,991	3,795	4,189	4,090	4,312	139,889
Fundulus spp.	1,684	713	706	472	507	307	646	479	1,477	756	106	351	2,269	697	812	1,170	312	299	1,635	587	140	114	59	139	68	490	16,995
B. tyrannus	0	0	0	0	0	0	1	1	0	0	0	3	2	7	528	171	2	47	14	881	23	381	2,341	1,797	1,194	24	7,417
A. quadracus	463	591	253	267	35	107	78	1,767	230	105	297	96	135	312	122	1,068	65	37	29	9	48	13	23	7	27	15	6,199
C. variegatus	41	285	35	16	7	33	133	23	29	23	1	1	11	2	14	1,169	8	0	55	11	2	3	0	4	1	0	1,907
P. saltatrix	1	1	1	1	0	1	49	86	2	0	0	1	1	3	797	2	0	10	4	3	1	2	0	10	6	5	987
P. pungitius	3	0	2	5	3	2	3	296	9	2	9	2	11	6	3	4	0	0	0	0	0	0	0	0	0	0	360
G. aculeatus	8	142	3	13	2	2	2	47	7	1	14	2	34	0	0	2	0	0	1	0	0	1	0	1	0	0	282
S. fuscus	1	1	6	1	1	2	12	8	10	3	3	5	15	1	12	8	5	2	12	12	7	40	32	4	33	19	255
Gadidae	0	0	9	0	0	20	21	2	6	0	4	1	0	1	1	0	0	1	0	1	2	17	2	0	12	11	111
M. cephalus	0	0	3	0	42	1	4	0	5	0	0	1	42	0	0	0	1	0	0	0	0	0	0	0	1	0	100
M. curema	0	0	0	0	0	0	0	1	9	0	0	0	42	1	9	1	0	1	30	0	1	1	1	0	0	1	98
L. parva	1	0	0	0	0	0	0	2	0	1	0	15	9	3	0	32	3	0	0	1	0	1	1	0	1	1	71
C. harengus	0	0	0	0	0	0	2	0	0	0	0	30	0	6	1	0	0	0	0	0	0	5	0	22	0	3	69
P. americanus	1	1	0	0	2	4	0	0	5	6	3	3	0	1	11	4	2	0	3	0	3	2	10	1	3	2	67
T. onitis	0	0	0	0	0	0	4	0	0	0	0	0	0	0	7	1	0	0	4	4	0	1	2	11	11	15	60
G. wheatlandi	0	0	0	0	0	0	3	1	2	6	11	2	6	2	2	0	1	2	0	0	0	3	0	0	0	0	41
A. rostrata	10	2	13	2	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	2	35
T. falcatus	0	0	1	0	1	0	0	0	0	0	0	0	22	6	0	0	0	0	0	2	0	0	1	0	0	0	33
M. aenaeus	0	0	1	0	0	0	1	0	0	2	2	0	1	0	0	0	1	2	3	1	2	2	1	0	10	1	30
C. hippos	0	0	1	0	0	1	0	0	0	0	0	0	3	1	3	0	1	1	2	0	0	9	0	1	0	0	23
Anchoa spp.	0	0	0	0	2	0	7	2	0	0	0	0	0	0	3	2	0	1	0	2	0	0	0	1	0	0	20
O. mordax	0	0	0	0	0	0	0	0	0	0	0	0	18	0	0	0	0	0	0	0	2	0	0	0	0	0	20
T. adspersus	0	0	2	0	0	0	3	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	5	0	3	0	14
A. pseudoharengus	0	0	0	0	0	0	0	0	5	0	0	0	0	3	0	0	0	0	0	2	1	0	0	1	0	0	12
L. apodus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	7
A. americanus	0	0	0	1	0	0	1	0	0	0	0	0	1	0	0	1	0	1	0	0	0	0	0	0	1	0	6
S. marina	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	2	0	1	0	0	0	0	5
A. aestivalis	0	1	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0	0	0	0	0	0	0	0	4
S. vomer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	0	0	1	0	0	0	0	4
Men. saxatilis	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	3
C. regalis	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	2
C. ocellatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1
L. xanthurus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1
P. gunnellus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Prionotus spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1
TOTAL	39,478	17,693	1,448	1,320	6,213	2,294	2,977	9,773	3,455	1,373	3,537	3,888	6,697	3,568	7,996	7,321	4,915	2,632	4,732	5,977	6,427	8,591	6,274	6,189	5,461	4,901	175,130

Table 4. Total number of fish caught by trawl in Niantic Bay from January 1976 through December 2001.

Year	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	Total
S. chrysops	742	1,465	388	425	903	800	1,417	1,861	1,265	391	1,528	1,314	996	739	1,378	7,903	3,749	271	1,853	283	64	516	2,043	749	1,627	6,250	40,920
P. americanus	1263	991	1,308	1,791	2,036	2,273	1,470	2,754	1,825	1,314	1,702	1,999	1,893	1,730	1,235	1,490	969	2,293	1,109	2,025	863	912	827	722	745	278	37,817
S. aquosus	285	253	133	96	208	131	176	303	257	225	360	788	512	710	394	362	634	988	841	561	501	196	432	424	273	121	10,164
Raja spp.	185	52	86	60	126	114	121	258	305	183	370	400	345	482	463	472	498	554	340	510	438	262	326	403	274	193	7,820
Menidia spp.	278	660	538	984	266	82	92	117	61	127	158	87	79	32	12	233	1,740	286	63	141	70	103	38	743	223	65	7,278
T. adspersus	710	822	214	992	645	341	174	107	64	39	9	8	13	36	30	17	88	10	23	30	8	20	48	16	45	70	4,579
Anchoa spp.	166	58	805	1	44	354	1	9	24	1,799	95	41	11	1	2	17	2	2	3	12	0	0	187	2	6	4	3,646
C. striata	9	2	0	3	5	39	13	24	23	39	247	8	32	45	47	38	47	4	278	33	8	22	465	253	329	878	2,891
M. aenaeus	33	43	86	61	76	211	125	215	226	84	129	53	90	121	75	60	57	144	63	136	45	120	121	61	74	108	2,617
Gadidae	1	22	86	80	35	481	275	146	182	91	116	35	387	17	21	9	52	26	3	27	59	15	24	24	19	8	2,241
P. dentatus	90	39	17	9	23	49	30	55	80	27	103	124	77	10	57	81	116	85	100	69	131	125	116	193	199	198	2,203
Prionotus spp.	45	30	28	48	66	60	42	67	29	40	105	22	40	214	76	23	23	41	96	137	11	12	46	110	15	51	1,477
P. triacanthus	5	2	11	4	4	9	4	7	0	3	10	5	928	11	328	60	24	16	1	7	0	0	6	1	0	0	1,446
E. microstromus	7	0	0	0	1	16	2	14	18	20	114	36	66	12	58	81	105	42	66	53	39	57	170	183	163	89	1,412
Urophycis spp.	6	3	10	1	19	26	57	168	22	22	20	17	52	27	40	30	39	63	21	90	54	36	57	186	250	26	1,342
M. bilinearis	100	15	3	36	108	48	30	57	27	41	41	7	23	49	47	39	81	7	19	29	13	16	64	29	36	34	999
T. onitis	76	70	83	70	46	28	52	40	46	47	25	13	37	25	17	13	8	26	13	28	22	24	12	27	45	106	999
H. americanus	4	10	5	15	29	85	84	112	46	8	4	2	0	1	2	1	3	0	0	0	3	4	2	7	2	2	431
O. mordax	7	5	9	6	7	4	23	25	0	6	14	11	3	3	3	19	99	38	1	16	3	21	1	35	2	1	362
S. fuscus	0	7	15	11	14	26	9	12	25	13	10	13	8	14	5	7	9	14	3	11	8	7	15	6	4	11	277
A. pseudoharengus	0	2	222	2	2	3	1	0	1	0	6	2	1	0	0	0	9	17	3	1	0	0	2	0	0	0	274
B. tyrannus	1	0	0	1	1	0	0	0	0	0	0	0	0	0	0	5	24	0	1	2	1	1	1	1	120	2	161
S. maculatus	2	1	0	0	1	0	0	2	0	0	1	0	4	3	8	22	16	1	7	5	4	1	9	19	10	5	121
Clupeidae	1	0	0	0	0	0	0	0	0	110	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	113
C. regalis	1	8	0	1	0	3	0	1	0	1	4	2	10	0	1	0	2	0	0	3	1	0	8	2	13	9	70
C. lumpus	0	8	2	5	16	4	1	3	0	4	0	0	9	0	4	0	0	1	0	0	0	7	1	0	0	2	67
P. oblongus	1	0	1	0	3	1	1	0	6	3	8	1	2	0	0	0	1	2	1	0	4	5	2	0	0	0	42
H. erectus	0	0	0	0	0	0	0	0	0	2	1	0	1	0	2	6	0	0	0	4	0	4	7	5	8	0	40
M. americana	1	3	5	1	6	0	1	0	0	0	0	0	1	2	0	0	2	1	1	1	0	2	1	0	3	8	39
Liparis spp.	2	2	1	5	1	3	3	3	1	3	0	0	2	2	0	0	0	0	0	0	0	2	0	1	0	0	31
A. sapidissima	15	2	0	0	4	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	24
Alosa spp.	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	3	5	0	0	0	3	5	0	2	2	0	22
C. harengus	0	0	1	0	0	0	0	1	2	0	0	0	0	0	0	0	1	1	0	0	0	0	7	2	3	2	20
M. hispidus	2	0	2	0	0	0	1	0	3	1	0	0	1	2	2	0	0	3	1	0	0	1	0	0	1	0	20
D. volitans	1	0	0	0	0	0	0	1	0	0	1	4	1	2	3	0	1	0	0	2	0	0	0	1	0	1	18
G. aculeatus	0	1	0	1	1	0	2	2	0	1	0	0	0	0	0	1	1	1	1	0	1	3	1	0	1	0	18
M. octodecemspinosus	0	0	2	0	2	6	3	2	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	16
A. aestivalis	0	0	0	1	0	0	0	6	3	0	2	1	0	0	0	0	1	0	0	0	0	1	0	0	0	0	15
M. saxatilis	0	0	0	0	0	1	0	0	0	0	1	0	1	0	0	0	0	5	1	2	1	3	0	0	0	0	15
O. tau	2	0	0	1	4	1	1	0	2	0	0	2	0	0	0	0	0	1	0	0	0	0	0	0	1	0	15
M. aeglefinus	0	0	0	0	0	0	2	0	0	0	1	0	3	0	0	0	0	5	0	0	0	0	0	0	3	0	14
F. tabacaria	1	1	0	0	1	0	0	0	0	0	0	0	0	1	3	1	0	1	0	0	0	1	0	3	0	0	13
A. schoepfi	0	0	1	0	0	0	0	0	1	1	0	0	0	2	1	0	1	0	0	1	0	2	0	1	0	0	11
A. americanus	0	0	2	0	1	1	1	1	0	0	2	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	10
A. rostrata	0	1	0	1	0	0	0	1	2	1	0	1	0	0	0	1	0	0	1	0	0	0	0	0	0	0	9

Table 4. (cont.)

Year	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	Total
M. canis	0	0	1	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	8
P. cruentatus	0	0	0	0	0	0	0	0	3	0	0	0	2	2	1	0	0	0	0	0	0	0	0	0	0	0	8
C. hippos	0	0	0	0	0	0	1	0	0	1	0	0	0	2	0	0	0	0	0	0	0	0	0	2	0	0	6
L. xanthurus	0	0	0	0	0	0	0	1	0	2	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	6
S. setapinnis	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	6
P. arenatus	0	0	0	0	0	0	0	0	2	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	1	6
A. quadracus	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	3
C. ocellatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	1	0	0	3
C. schoepfi	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	3
Gobiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	1	0	3
O. marginatum	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	2	0	0	4
P. alta	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	3
S. foetens	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	1	0	0	1	4
C. crysos	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
E. cimbrius	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	1	0	3
L. americanus	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	1	3
M. auratus	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
S. vomer	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	2
C. oceanicus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1
Lactophrys spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	2
P. pungitius	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
P. saltatrix	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. scombrus	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
T. lathami	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1
T. maculatus	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
U. subbifurcata	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
TOTAL	4,042	4,580	4,064	4,712	4,704	5,202	4,214	6,377	4,548	4,652	5,189	5,001	5,640	4,295	4,321	10,995	8,409	4,950	4,915	4,221	2,357	2,508	5,041	4,216	4,501	8,527	132,181

Table 5. Total number of fish caught by trawl in Jordan Cove from January 1976 through December 2001.

Year	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	Total
P. americanus	1,525	878	1,069	1,362	695	622	1,072	1,710	915	938	930	683	625	499	655	559	642	786	331	770	393	451	414	328	491	167	19,510
S. chrysops	444	250	162	214	65	12	713	374	26	332	426	196	77	203	610	4301	628	17	384	132	11	323	455	598	1205	2037	14,195
Menidia spp.	484	146	77	663	209	17	69	256	106	159	1,450	303	48	66	16	226	177	131	48	88	32	31	15	190	804	50	5,861
T. adspersus	134	79	88	228	191	262	196	131	73	30	32	14	145	36	50	93	162	52	58	48	6	35	712	642	407	237	4,141
P. gunnellus	10	58	30	44	119	214	99	133	49	95	77	117	231	100	37	85	55	96	174	213	60	110	527	399	404	307	3,843
Anchoa spp.	285	9	32	4	6	5	5	18	31	323	286	164	256	492	15	1,407	4	1	137	16	0	1	62	0	22	28	3,609
Raja spp.	98	55	30	37	40	34	27	181	137	50	114	103	122	88	280	225	246	270	80	217	136	142	118	73	86	77	3,066
Gadidae	18	43	112	97	48	427	404	132	155	108	212	62	63	12	30	45	101	350	34	19	77	38	31	40	134	60	2,852
S. aquosus	164	89	54	119	99	32	48	176	180	103	154	74	66	76	93	82	117	158	62	141	150	70	64	33	33	7	2,444
M. aenaeus	34	53	81	109	51	128	66	160	81	83	42	91	175	41	31	51	79	126	86	111	42	66	146	96	120	127	2,276
G. aculeatus	10	0	0	52	16	2	3	208	934	29	342	384	35	0	1	30	29	110	4	0	4	4	8	4	3	5	2,217
P. dentatus	87	36	21	16	9	9	61	59	151	44	165	85	36	9	68	76	110	32	33	40	97	65	51	94	134	56	1,644
T. onitis	73	113	59	56	20	24	35	19	16	27	58	33	31	23	34	44	90	51	19	73	47	26	71	138	261	123	1,564
O. mordax	47	141	58	16	72	11	2	21	215	263	227	235	90	5	6	2	10	1	0	6	1	3	0	9	0	0	1,441
S. fuscus	8	11	10	28	36	38	32	111	127	49	63	59	146	19	16	21	65	76	25	100	21	21	59	79	69	17	1,306
Urophycis spp.	8	21	19	18	12	20	44	79	41	39	6	16	21	20	99	13	54	37	18	91	31	19	21	115	113	52	1,027
C. striata	3	0	0	0	0	6	1	6	0	4	46	0	1	3	3	11	10	1	65	4	0	17	441	135	122	145	1,024
B. tyrannus	0	0	0	0	0	0	0	0	0	0	0	1	0	520	0	6	5	4	0	0	2	0	2	4	5	0	549
E. microstromus	11	0	0	1	1	2	3	10	7	23	34	12	6	6	44	14	69	28	26	42	7	34	17	27	50	9	483
H. americanus	3	7	2	24	40	73	86	141	52	15	0	1	1	0	1	0	0	0	0	0	0	0	1	0	1	0	448
A. quadracus	0	1	3	1	0	3	1	5	17	17	84	19	11	0	0	0	19	86	2	2	0	1	0	0	1	0	273
M. bilinearis	64	3	1	3	42	4	9	3	5	5	3	0	0	3	10	42	1	1	3	1	2	4	5	1	10	1	226
C. lumpus	2	9	11	17	37	4	0	7	0	18	1	2	30	4	3	0	3	13	7	1	4	37	5	0	1	6	222
Prionotus spp.	29	6	6	5	4	3	7	1	6	5	16	0	4	7	5	8	12	13	1	4	2	3	8	9	0	3	167
H. erectus	0	0	0	0	0	0	0	0	3	1	8	8	3	0	7	20	3	0	0	5	0	1	1	8	17	0	85
P. triacanthus	1	1	2	0	4	0	0	11	0	0	5	1	26	4	14	3	1	0	0	0	0	0	0	1	3	1	78
C. harengus	0	0	0	0	0	0	0	0	0	63	0	2	0	0	1	1	2	1	0	0	0	0	0	2	2	0	74
F. tabacaria	1	1	0	0	2	0	1	0	8	1	2	0	0	1	11	5	5	2	1	0	0	0	0	8	19	1	69
A. pseudoharengus	1	2	1	0	0	0	0	3	0	0	0	0	0	40	1	0	2	0	0	0	0	0	1	0	0	0	51
A. rostrata	2	3	0	1	1	7	9	4	4	4	1	0	2	1	0	5	0	0	0	0	0	0	0	0	1	0	45
S. maculatus	4	2	1	0	0	0	0	1	3	1	0	0	0	2	3	3	0	0	0	5	1	2	2	1	9	1	41
C. regalis	1	11	1	0	0	5	0	0	0	0	0	2	1	1	0	0	0	0	1	0	0	0	0	7	5	6	41
A. americanus	0	3	2	6	0	6	1	0	0	1	0	0	0	0	0	0	1	0	0	0	1	0	0	0	14	0	35
Liparis spp.	2	0	1	1	3	1	5	0	1	5	0	1	2	3	2	0	0	0	1	0	0	5	0	1	0	0	34
O. marginatum	0	0	0	0	0	0	0	0	0	0	0	0	3	10	1	0	1	0	0	0	0	0	0	0	10	0	25
M. hispidus	0	1	2	0	0	0	6	0	2	4	1	0	0	1	2	1	0	1	0	0	0	0	0	0	1	0	22
S. setapinnis	0	0	0	0	0	0	0	0	0	0	0	0	16	0	0	0	0	0	0	0	0	0	0	0	1	0	17
M. canis	2	1	1	0	0	0	0	0	0	0	0	0	2	0	4	0	0	0	0	0	0	0	1	0	0	1	12
O. tau	2	0	0	0	0	0	0	2	0	1	2	0	2	1	0	0	0	0	0	0	0	0	0	0	1	0	11

Table 5 (cont.)

Year	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	Total
M. americana	0	1	4	0	0	0	0	1	0	0	0	0	2	0	0	0	0	1	1	0	0	0	0	0	0	0	10
P. pungitius	0	0	0	0	0	0	0	0	5	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10
Lactophrys spp.	0	0	0	0	0	0	0	0	2	0	0	0	0	0	1	3	0	1	0	0	0	0	1	1	0	0	9
A. schoepfi	0	0	0	1	1	0	1	0	1	0	1	0	2	1	1	0	0	0	0	0	0	0	0	0	0	0	9
A. sapidissima	1	0	0	0	0	1	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8
Alosa spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	4	0	0	1	0	0	0	0	0	0	0	8
G. wheatlandi	0	0	0	0	0	0	1	1	1	0	1	2	0	0	0	0	1	0	0	0	0	1	0	0	0	0	8
S. borealis	0	0	0	0	0	0	0	1	1	2	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	6
T. maculatus	3	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5
Gobiidae	0	0	0	0	0	0	0	1	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	4
M. saxatilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	1	1	0	0	4
P. saltatrix	1	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	4
T. lathami	0	0	0	1	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	4
U. subbifurcata	0	0	1	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	4
L. xanthurus	0	1	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	3
M. octodecemspinosus	0	0	0	2	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
P. arenatus	0	0	0	0	0	1	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	3
Monacanthus spp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	3
D. centroura	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	2
E. cimbrius	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	2
Gasterosteidae	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
M. auratus	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	2
P. cruentatus	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	2
S. vomer	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	2
A. aestivalis	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
A. mediocris	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
A. maculatus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
C. oceanicus	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
C. ocellatus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
D. macarellus	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
D. volitans	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Fundulus spp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
L. americanus	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
M. aeglefinus	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. marina	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1
T. myops	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
U. parvus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
TOTAL	5,541	4,013	3,921	5,107	3,804	3,957	4,990	5,957	5,344	4,831	6,783	4,661	4,275	4,288	4,149	9,378	4,696	4,439	3,598	4,124	3,123	3,507	5,239	5,046	6,562	5,526	75,158

Table 6. Total number of fish caught by trawl in Niantic River from January 1976 through December 2001.

Species	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	Total
P.americanus	2,983	1,311	1,157	2,168	3,674	4,465	5,840	4,861	4,613	2,827	4,088	3,452	6,746	3,862	3,679	5,142	5,871	1,675	1,860	1,210	1,438	559	992	1,291	1,394	345	77,503
Menidia spp.	1,203	304	61	93	105	161	144	189	39	81	1,460	416	604	187	217	542	456	144	108	43	268	213	84	600	897	196	8,815
M. aenaeus	53	32	34	65	230	336	390	418	236	281	365	282	439	590	407	354	411	249	224	235	402	284	368	149	125	54	7,013
S. aquosus	98	41	24	130	158	177	208	228	243	114	295	272	173	333	377	236	228	289	95	105	76	104	72	46	43	11	4,176
P. dentatus	43	29	16	14	21	100	82	60	165	58	240	208	200	20	143	171	257	102	167	66	98	232	76	161	254	121	3,104
S. fuscus	23	9	14	22	62	177	100	97	63	101	102	205	148	63	84	105	73	195	32	141	62	27	49	31	45	17	2,047
Prionotus spp.	119	23	1	7	4	10	253	2	2	3	44	0	19	147	98	265	40	12	258	57	6	40	20	463	71	3	1,967
T. onitis	46	15	27	47	25	126	80	31	5	25	100	26	50	36	89	67	22	15	12	129	40	36	125	238	324	130	1,866
G. aculeatus	4	17	13	59	21	185	110	45	5	164	84	12	16	63	11	11	433	111	20	60	3	116	53	80	3	7	1,706
A. quadracus	19	5	2	22	32	192	763	72	5	102	115	29	29	13	9	26	107	30	3	6	2	13	1	47	7	8	1,659
Anchoa spp.	10	195	0	0	0	0	11	2	5	15	9	11	168	734	2	12	1	0	12	2	14	3	21	139	0	0	1,366
B. tyrannus	0	4	10	0	1	1	0	0	1	31	10	3	0	788	0	179	26	19	15	26	2	1	2	106	101	18	1,344
Gadidae	3	7	9	14	101	205	75	59	18	47	174	7	34	27	31	5	13	18	15	17	111	64	128	14	13	15	1,224
S. chysops	62	10	11	5	6	38	27	57	2	3	22	15	3	5	10	175	119	0	83	38	2	12	4	94	263	166	1,232
P. gunnellus	3	0	1	11	3	17	32	77	27	16	14	45	33	48	93	47	22	56	30	35	39	34	132	52	23	19	909
O. tau	98	22	6	16	28	34	20	26	16	33	53	49	58	29	52	23	4	10	14	15	6	7	7	6	7	6	645
T. adspersus	32	3	4	15	4	64	70	62	30	13	34	20	9	14	11	4	13	11	9	12	2	18	105	35	24	65	683
Clupeidae	5	31	24	6	12	3	3	10	3	2	2	2	2	3	17	0	21	17	111	9	299	0	0	7	0	0	589
C. striata	18	0	0	1	2	3	2	0	1	8	112	0	3	1	16	44	1	0	7	8	0	3	5	40	65	4	344
Urophycis spp.	0	0	0	0	0	3	3	11	0	8	6	17	14	11	7	4	7	6	0	28	11	3	13	43	130	19	344
O. mordax	55	86	2	0	6	13	14	17	3	31	8	7	34	4	1	6	13	1	1	3	1	4	0	3	1	0	314
A. rostrata	17	11	7	4	8	23	14	21	16	25	24	18	20	4	14	3	1	4	9	9	8	2	5	7	3	6	283
S. maculatus	11	5	0	0	7	11	12	7	1	2	1	0	2	4	12	23	3	2	16	16	1	3	1	28	8	1	177
Gobiidae	3	0	0	0	4	0	0	2	2	12	3	3	12	2	15	11	9	25	0	4	0	2	6	6	3	9	133
H. errectus	0	0	0	0	0	0	0	1	1	1	9	4	2	2	8	36	1	0	0	0	0	0	6	12	25	0	108
A. americanus	1	0	1	74	1	0	2	14	1	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	96
H. americanus	0	0	0	2	5	9	6	32	24	3	1	0	0	0	0	0	0	0	1	1	1	0	1	0	0	0	86
Raja spp.	2	0	0	0	2	1	1	3	0	1	1	0	4	1	3	2	20	2	6	2	9	16	4	4	1	0	85
E. microstomus	0	0	0	0	2	0	0	1	2	0	4	2	4	3	2	6	9	2	2	4	1	6	2	10	10	2	74
D. volitans	2	0	0	0	0	1	3	0	0	1	1	0	0	0	0	3	2	2	1	1	4	1	0	3	3	0	28
M. americana	2	1	0	1	5	2	0	0	0	0	0	0	2	8	1	0	1	2	0	0	0	0	0	0	1	0	26
F. tabacaria	0	1	0	0	0	0	0	0	0	0	0	0	1	1	7	3	0	0	1	0	0	0	0	3	4	0	21
M. saxatilis	0	0	2	1	0	1	1	0	0	1	0	0	0	4	1	0	0	2	2	0	3	1	0	1	0	0	20
P. triacanthus	1	0	0	0	0	0	0	2	0	0	0	0	0	8	0	0	2	0	0	0	0	0	1	0	6	0	20
Liparis spp.	0	0	0	0	0	1	0	0	0	0	0	0	0	9	0	1	0	0	1	1	0	0	1	0	0	2	16

Table 6. Total number of fish caught by trawl in Niantic River from January 1976 through December 2001.

Table 6 (cont.)

	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	Total
P. oblongus	0	0	0	2	0	0	0	0	0	0	0	0	2	7	0	1	0	0	0	0	0	0	0	2	0	0	14
F. majalis	0	0	0	0	0	4	3	0	0	0	2	0	0	1	1	0	0	1	0	0	0	0	0	1	0	0	13
C. lumpus	0	0	1	1	0	0	1	0	1	0	0	2	0	0	0	2	1	1	0	0	1	1	0	0	0	2	14
O. marginatum	0	0	0	0	0	0	0	0	0	0	0	0	3	0	3	0	0	0	0	0	0	0	0	4	0	0	10
P. saltatrix	0	1	0	0	0	0	2	0	0	0	0	1	0	0	1	0	0	0	1	0	0	0	0	0	0	0	6
S. borealis	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	6
Mer. saxatilis	0	1	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	2	0	0	1	0	0	0	6
C. oceanicus	0	0	0	0	0	0	0	0	1	0	0	0	1	1	0	1	0	0	0	0	0	0	0	0	0	0	4
S. foetens	0	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	4
M. hispidus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	3
P. pungitus	0	0	0	0	1	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
C. ocellatus	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	3
G. wheatlandi	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	1	3
Lactophrys spp.	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	2
L. ferruginea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2
M. curema	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	3
T. maculatus	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
B. ocellatus	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
C. regalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1
C. variegatus	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Myoxocephalus spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1
M. canis	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
P. alta	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
P. arenatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1
s. trutta	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. retifer	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. setapinnis	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1
Total	6,894	4,142	3,407	4,760	6,511	8,345	10,256	8,391	7,516	5,997	9,371	7,097	10,831	9,024	7,415	9,505	10,181	4,996	5,112	4,281	4,906	3,802	4,283	5,726	5,854	3,229	120,131

The Paleo-environmental History of Long Island Sound as Traced by Organic Carbon, Biogenic Silica and Stable Isotope/Trace Element Studies in Sediment Cores

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Introduction

The bottom waters of western and central Long Island Sound (LIS) have suffered seasonal hypoxia and anoxia over the last 30 years (Parker and O'Reilly, 1991). Oxygen-depletion in bottom waters is an expression of a perturbed ecosystem and deserves a thorough study regarding its potential causes (Koppelman *et al.*, 1976; O'Shea and Brosnan, 2000). The eutrophication of the Sound over the last 150 years (Lugolobi *et al*, this volume; Thomas *et al.*, this volume) is commonly seen as the cause of the hypoxia in LIS and other coastal waters, but other parameters such as water temperature may have an impact as well. The severity of hypoxia is loosely correlated with climatic conditions such as El Niño events and ambient summer temperatures (Wilson, pers. comm.).

The effect of high summer temperatures on LIS can be summarized (e.g., Welsh and Eller, 1991) as follows: 1. Enhanced water column stratification cuts off the O_2 supply from the atmosphere to the bottom waters, 2. Water at higher temperatures can hold less oxygen at saturation, leading to a smaller available aqueous O_2 reservoir, 3. High bottom water and sediment temperatures promote rapid bacterial decay of labile organic matter, both at the sediment-water interface and in the upper sediment column. If organic matter is available for decomposition, the ambient temperature has a significant influence on the rate of oxygen consumption in the bottom water column. The Sound may have suffered from hypoxia in the past, prior to the fertilization of the Sound waters by anthropogenic nutrient releases, as a result of high water temperatures.

We present data that can be used as proxies for organic productivity in the Sound: C_{org} and biogenic silica accumulation rates. In addition, we reconstruct the paleo-temperature and paleo-salinity history of the Sound for the last 1000 years using stable isotope and trace element data from carbonates in sediment cores. We thus provide a historic reference framework of paleo-environmental conditions in which we can place the environmental problems in LIS waters of the last 50 years (similar to the work of Cooper and Brush, 1993, for Chesapeake Bay). An extension of this work is the study of carbon isotopes on the same carbonate samples (Lugolobi *et al.*, this volume).

Methods

We determined C_{org} concentrations with a carbon analyzer and biogenic silica concentrations through extractions with hot alkaline solutions. Cores were dated using radiogenic isotopes and age-matching with mercury (Hg) pollution profiles that have been dated extensively in the coastal marshes of Connecticut (Varekamp *et al.*, 2000; 2003). Bulk dry densities of the sediment samples were derived from measured water contents and assumed rock densities of 2.6 gr/cm^3. The stable isotope and trace element studies were carried out on the tests of the foraminiferal species *Elphidium excavatum* (Thomas *et al.*, this volume; 2000), which were picked from both surface and core sediment samples. Living specimens in surface samples were detected through Rose Bengal staining. Cores were taken along transects through LIS (Buchholtz ten Brink *et al.*, 2000; Thomas *et al.*, this volume).

We developed a simple mixing model for LIS waters based on mixing of Atlantic sea water and river water, with salinity as the mixing parameter. From the properties of the two end-members we derived a relationship between salinity and the oxygen isotopic composition of water. We use the Ca/Mg values in calcite as a proxy for temperature (e.g., Lea, 1999), as preliminarily calibrated on the core tops using a mean annual LIS water temperature of 12.5 °C. The Mg/Ca in calcite does not depend on salinity in the range of modern LIS salinities. The Mg/Ca paleo-temperatures were used to calculate $\delta^{18}O_w$ from $\delta^{18}O_{cc}$ using the fractionation factors of Kim and O'Neil (1997) and a species specific offset of 1.1‰ (as determined from surface sample and water column studies). The obtained $\delta^{18}O_w$ values were then translated into paleosalinities using the modern LIS mixing model.

Results

We studied cores A1C1 (WLIS), D3C2 (Central LIS) and WLIS75 (Narrows; Thomas *et al.*, this volume) for C_{org} and biogenic silica (BSi) concentrations, which were recalculated into BSi and C_{org} accumulation rates (mg/cm^{2}yr) for cores D3C2 and A1C1 (Figures 1a, 2). The results show an increase in C_{org} and BSi accumulation rates since about 1800-1850, with a 5-6 times increase in the midst of the 20th century compared to pre-colonial times, and very high values in the most recent sediments. Both BSi and C_{org} show an exponential increase in accumulation rate over time in core A1C1. Core D3C2 shows steadily increasing C_{org} accumulation rates over the last 150 years, whereas BSi accumulation rates show a strong increase around the 1970's. The BSi contents of core WLIS75 (Figure 3) are overall high (~3.5%) relative to pre-colonial values (~0.5 %), but show a strong drop in the 1980's-early 1990's. That drop correlates with changes in the benthic foraminiferal faunas in the core (Thomas *et al.*, this volume).

The paleo-temperature data from Ca/Mg values (Figure 1b) were determined for core A1C1, which shows a period of high temperatures about 1000 years ago, followed by a cooler interval (with two peaks between 1500 and 1900) and a period of warming over the last century. This temperature pattern shows a strong resemblance to an oxygen isotope record from the GISP2 (Greenlan) ice core (GISP2 CD), which we translated into an approximate temperature scale for the northern hemisphere. We recognize the Mediaeval Warm Period (MWP), Little Ice Age (LIA) and Modern Global Warming

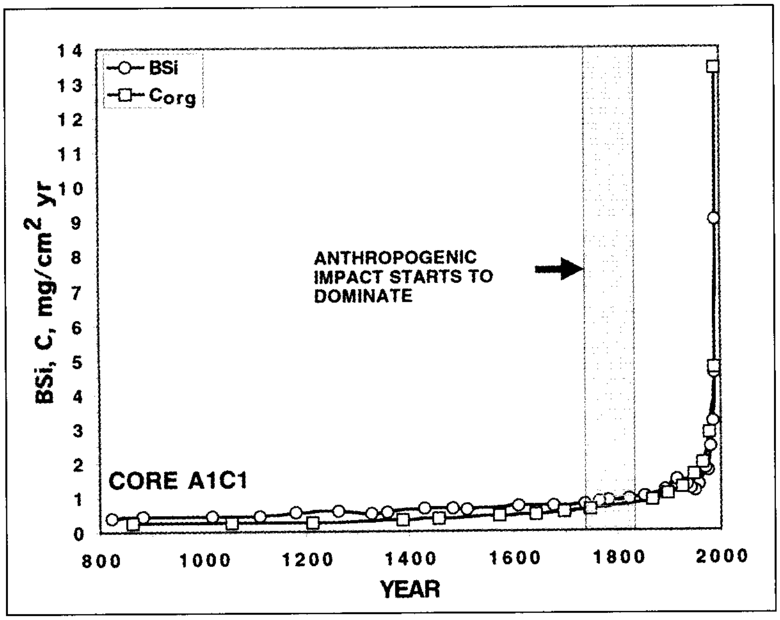

Figure 1. Paleo-environmental records of Core A1C1.

Figure 1a. Organic carbon and biogenic silica accumulation rates in core A1C1. The rates start to increase around 1800-1850 AD and increase exponentially over the last 50 years.

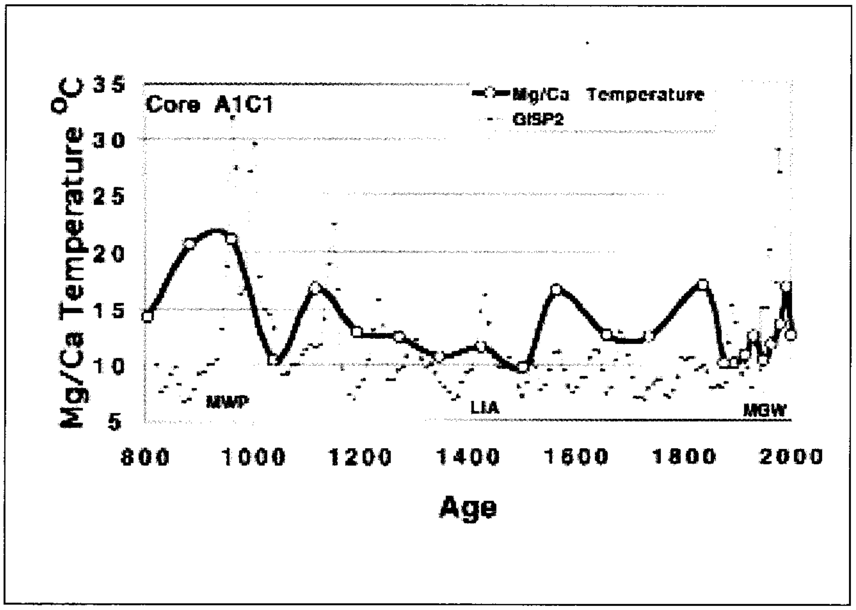

Figure 1b. Paleo-temperature record of core A1C1. The Mediaeval Warm Period (MWP), Little Ice Age (LIA) and Modern Global Warming (WGW) intervals are clearly recognizable. For comparison, the GISP2 record is plotted with a non-calibrated temperature scale.

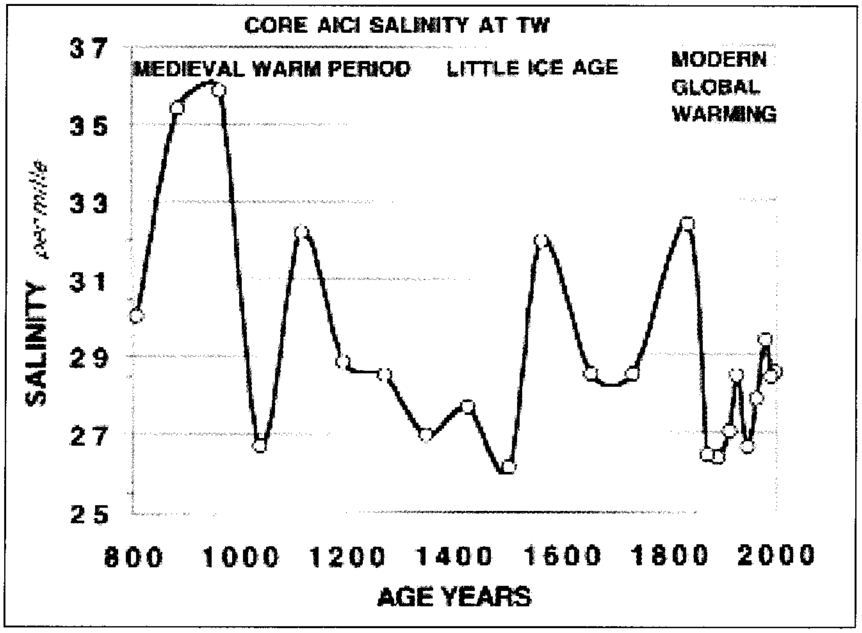

Figure 1c. Paleo-salinity record of core A1C1, with salinities calculated based on the Ca/Mg water temperatures (TW). The data indicate a high salinity at the MWP (hot and dry) and a lower salinity during the cooler LIA period.

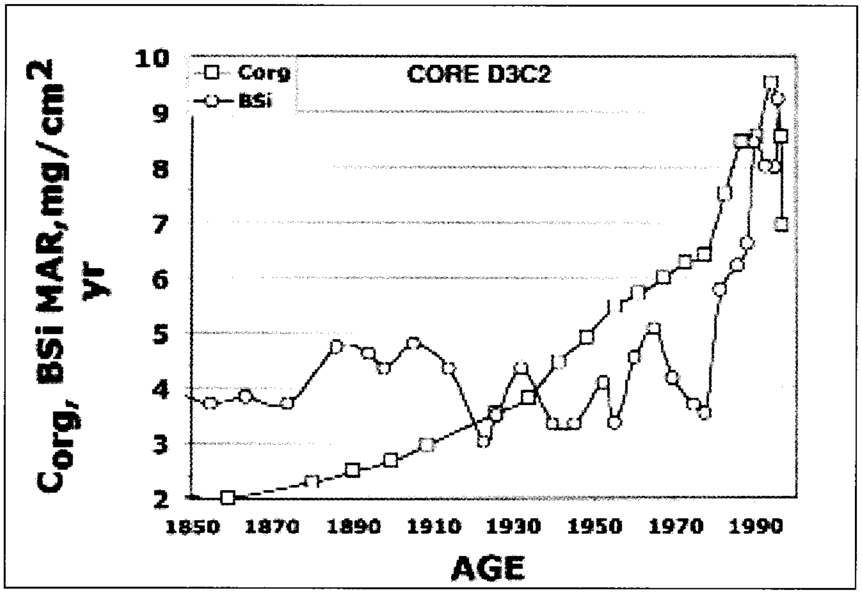

Figure 2. Organic carbon and biogenic silica accumulation rates in core D3C2. The carbon accumulation rate increases steadily since 1850 AD, whereas BSi increases strongly in the 1980's.

(MGW) trends in both records. The two peaks in the 1500-1900 time interval in the A1C1 record are much more pronounced than in the ice core record, which may be the result of poor temperature calibration or a true regional signal as compared to the much larger-scale record in the ice core.

The paleo-salinity record of core A1C1 (Figure 1c) shows a maximum during the MWP and lower values during the LIA (wet and cool). There is no secular trend apart from the high salinity period about 1000 years ago (a warm and dry MWP).

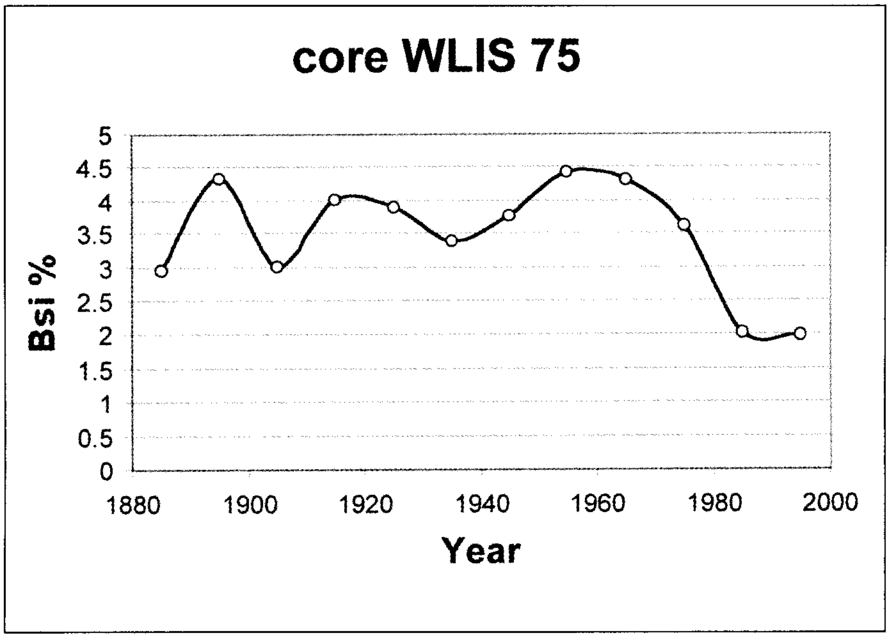

Figure 3. Biogenic silica concentrations in core WLIS75. The concentrations are the highest found among LIS samples, but drop in the 1980's.

Conclusions

The BSi and C_{org} data in western and central LIS show strong eutrophication, which started around 1850 AD. The carbon loading apparently increased unabatedly over the last 100 years. Core WLIS75 shows evidence for decreasing amounts of BSi in the last 20 years of the 20th century, indicating reduced productivity of diatoms. Ecological studies suggest that a switch from diatoms to dinoflagellates may have occurred in recent times (Thomas *et al.*, this volume; Capriulo *et al.*, 2002).

The paleo-temperature record shows a clear pattern of paleo-climate and the paleo-salinity pattern suggests alternating 'dry and hot' and 'wet and cool' climate periods. Our data suggest that the carbon loading in western LIS exerts a strong control on the occurrence of hypoxia: the hypoxic periods started when the amount of organic carbon in the sediment cores went up dramatically (see also Lugolobi *et al.*, this volume). Nonetheless, the rising temperature trend also correlates with the occurrence of common summer hypoxia, and temperature may have played a subsidiary role in its development.

Acknowledgments

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Macroinfaunal Composition at Three Sandy Beaches in Eastern Long Island Sound From 1980-1992

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Abstract

Macroinvertebrate communities at three eastern Long Island Sound (LIS) sandy beaches of varying wave exposure were sampled seasonally from 1980 through 1992. Beaches within LIS are protected from direct ocean swell. However, wind-generated wave stress and *Zostera* detritus incorporated in sediments were a major factor in structuring macroinvertebrate communities in this area. A total of 213 taxa were identified during the 13-year period. Among the organisms identified to species, polychaetes were dominant (85 species), followed by arthropods (80 species) and mollusks, (38 species). Oligochaeta, Nermertea, Echinodermata and Archiannelida were also counted but were not identified to species during the study. Numbers of species and population densities were highest in the sheltered area Jordan Cove (JC). The sheltered assemblage included higher numbers of oligochaetes and surface feeding polychaetes. In contrast, exposed communities, Giants Neck (GN) and White Point (WP) included high numbers of ryhnchocoels and burrowing polychaetes.

Introduction

The data in this study were collected as part of the environmental monitoring program conducted in the vicinity of the Millstone Power Station from 1980 to 1992. This study is the first long-term study of sandy beach infauna within Long Island Sound (LIS). The few long-term studies that exist (Croker *et al* 1975; Croker 1977; Croker and Hatfield 1980) were performed in open coastal environments, unlike the more sheltered embayments of LIS.

Several studies have described the composition of intertidal sand communities along the eastern coast of the United States (Dexter 1969; Howard and Dejores 1972; Croker *et al.* 1975; Dauer and Simon 1975; Croker 1977; Holland and Polgar 1976; Whitlatch 1977; Mauer and Aprill 1979); most have been short-term.). Croker (1967) published a list of fauna typical of the ocean exposed beaches along the south shore of Long Island. Thus, the long-term spatial and temporal trends in the composition and abundance of intertidal sandy beach communities have not been fully addressed. The major emphasis of this paper will be to describe the composition of LIS sandy beach infaunal communities from a sheltered area and areas exposed to wind-driven waves.

Methods and Materials

The study area is located in southeastern Connecticut, USA (41°18' N and 10' W) on the north shore of LIS (Figure 1). Three sandy beaches, subjected to differing degrees of wave exposure, were sampled quarterly (March, June, September and December) from March 1980 through June 1992. The Jordan Cove station (JCI) is a sheltered, southeasterly facing beach. This site is exposed to wave scour primarily during periods of strong southeast winds (primarily during fall and winter storms). Eelgrass (*Zostera marina*) and algae often cover this beach and are incorporated within the sediments as they decay. Sediment deposited onto the beach during storms covers this organic

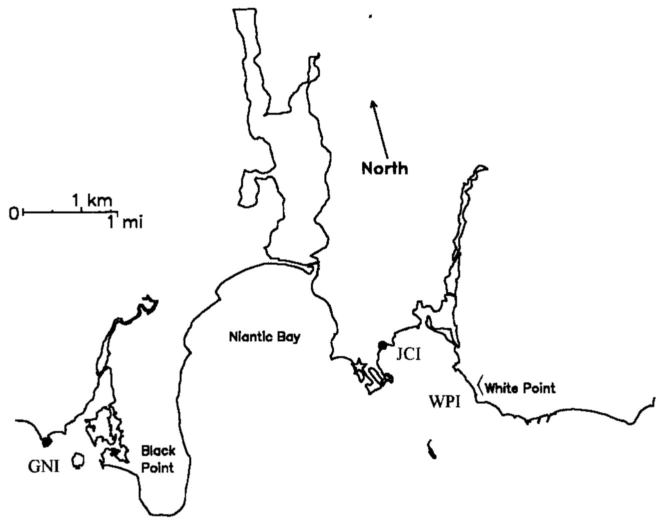

Figure 1. Map of the Millstone Point area showing the location of intertidal infaunal sampling stations (WPI=White Point, GNI=Giants Neck, JCI=Jordan Cove).

material, resulting in a surface layer of sand often (3-5 cm deep) overlying a distinct layer of detritus. Giants Neck (GNI) faces south and is exposed to the more constant wave scour produced by the prevalent southerly winds. This constant wave action and wave-driven longshore transport produces a clean sandy beach where little organic material accumulates. White Point (WPI) is the most exposed of the three sites and is influenced by prevailing southwesterly, as well as southerly and northerly winds. Similar to the GNI beach, WPI is a highly dynamic, clean sandy beach.

At each station, ten replicate samples were collected using a hand-held coring device 10cm in diameter X 5cm deep. Samples were collected at approximately 0.5 m intervals along the water line at mean low water. At the laboratory, samples were fixed with a 10% buffered formalin/Rose Bengal solution and, after a minimum of 48 h, organisms were floated from the sediments onto a 0.5 mm mesh sieve and preserved in 70% ethyl alcohol. Samples were examined using dissecting microscopes (10x) and organisms were sorted into major groups (annelids, arthropods, molluscs and others) for identification to the lowest practical taxon and counted. Oligochaetes and rhynchocoels were treated as taxonomic units because of the difficulties associated with identifying these organisms to a lower taxonomic level. Organisms that were too small to be quantitatively sampled by our methods (e.g., nematodes, ostracods, copepods, and foraminifera) were not sorted from samples. At each station, comparisons of annual collections were made by calculating the Bray-Curtis similarity index between each pair of years, using the formula (Clifford and Stephenson 1975):

$$S_{jk} = \frac{\sum_{i=1}^{n} 2\min(X_{ij}, X_{ik})}{\sum_{i=1}^{n} (X_{ij} + X_{ik})}$$

where S_{jk} is the similarity index between year j and year k; X_{ij} is the log transformed (ln+1) abundance of taxon i in year j; X_{ik} is the abundance in year k; and n is the number of taxa in common, for which, on average, at least two individuals were found per year. A group-average-sorting, clustering algorithm was applied to the resulting similarity matrix (Lance and Williams 1967).

Grain size and the silt/clay fraction were determined on a 3.5cm diameter x 5 cm core, taken at the time of infaunal sampling. Sediment samples were analyzed using the dry sieving method described by Folk (1974).

Results

Sedimentary Environment

Sediments at the three intertidal stations were comprised of medium to coarse sands with a low silt/clay component. Average grain size at Giants Neck (GN) and White Point (WP) typically fell between 0.3 and 0.6 mm during the study (Figure 2), although values in the 0.7 to 0.9 mm range were recorded at these stations. The similarity of the sediment at these stations was also evidenced by the consistently low silt/clay component within the sediments. For example, quarterly values for silt/clay at GN typically ranged between 0.1% and 0.5%, over the study, while quarterly values at White Point (WP) were below 0.1%.

Sediments at Jordon Cove (JC) were generally coarser and had higher silt/clay and organic content and much more pronounced seasonal and year to year fluctuations in all sediment characteristics than sediments at the GN and WP stations. Quarterly mean grain size ranged from 0.3 mm (medium sand) to >1.3 mm (very coarse sand), although grain size most frequently ranged from 0.5 to 0.8 mm (medium to coarse; Figure 2). Silt/clay content of JC sediments over the entire study period also exhibited greater seasonal and annual fluctuation than at GN or WP. Percent silt/clay ranged from <0.1% to 5.7% over the thirteen-year period.

General Community Composition

Intertidal infauna communities were numerically dominated by polychaetes, oligochaetes and rhynchocoels. Polychaetes were the dominant infaunal group at GN and WP and, on average, represented ~50% of the species identified and >50% of the total individuals collected (Table 1). Rhynchocoels were second in abundance at the GN (16%) and WP (28%) beaches, while arthropods and molluscs were consistently less important contributors both in terms of species and numbers of individuals. At JC, oligochaetes (78%), were numerically dominant and were nearly four times as abundant as polychaetes (21%), the next most numerous group at this station. Arthropods and polychaetes comprised 47% and 31% of all species identified at JC, and accounted for only 2% and 21% of the total number of individuals, respectively.

Annual total numbers of species ranged from 13-40 at GN, 26-66 at JC and 17-28 at WP (Figure 3). Total accumulated numbers of species over the entire study was highest at JC throughout the study followed by GN and WP (Table 1). Year-to-year variation in total numbers of species at GN and WP was similar and generally lower than that observed at JC. The JC infaunal community exhibited wide fluctuations in species number during this study. Annual numbers of polychaete species was generally similar at all stations while numbers of arthropod and mollusc species was higher at the sheltered site, JC, than the more exposed sites GN and WP, which were more similar to each other throughout the study. Additionally, a list of taxa collected at each site is provided, indicating seasonal occurrence during the entire study (Appendix 1).

Community Dominance

Oligochaetes, rhynchocoels and five polychaetous annelids (*Marenzelleria virdis, Leitoscoloplos fragilis, Paraonis fulgens, Hediste diversicolor* and *Streptosyllis arenae*) accounted for 85.6%, 94.7% and 91.6% of the individuals collected at the GN, JC, and WP, respectively, (Table 2). Other taxa were highly variable over the years and generally accounted for <6% of the total individuals collected during the study period.

	(S)	RANGE	(N)	RANGE
Giants Neck				
Polychaeta	55	9-30	13592	291-1836
Oligochaeta	-	-	3076	49-839
Mollusca	17	1-6	60	1-17
Arthropoda	50	2-14	599	3-261
Rhynchocoela	-	-	3148	63-686
Total	122	13-40	19475	638-2428
Jordan Cove				
Polychaeta	52	11-25	23746	133-3671
Oligochaeta	-	-	88459	1827-12769
Mollusca	36	5-14	1103	12-389
Arthropoda	78	7-29	2240	10-724
Rhynchocoela	-	-	917	2-353
Total	166	27-54	113386	2576-15040
White Point				
Polychaeta	56	12-21	9763	356-1476
Oligochaeta	-		2511	9-591
Mollusca	16	1-6	97	1-25
Arthropoda	33	1-10	85	1-13
Rhynchocoela	-		4822	35-809
Total	105	17-29	17276	445-2510

Table 1. Total number of species (S), number of individuals (N) and ranges for each major taxon collected at intertidal sand stations 1980-1992.

The GN and WP infaunal communities were each dominated by rhynchocoels (15.3% and 27.8%, respectively), the polychaetes *Marenzelleria viridis* (GN; 19.5%), (WP; 21.5%), and *Leitoscoloplos fragilis* (17.4% and 16.9%, at GN and WP respectively; Table 2). Oligochaetes were also numerically dominant, accounting for 15.0% and 14.5% of the individuals at GN and WP, respectively.

The JC intertidal community was distinct from the other stations because oligochaetes were the overwhelmingly dominant taxon (78.0%). Other taxa accounting for more than 2% of the individuals were the polychaetes, *Marenzelleria viridis* (8.9%) and (7.0%). The remaining seven taxa accounted for no more than 1.4% each of the total individuals collected at JC (Table 2).

Cluster Analysis

Cluster analysis of intertidal communities from 1980-1992 resulted in three major groups (I-III) of station/years at similarity level of 38% (Figure 4.). Annual station collections (1980-1992), exhibited higher within than between station similarity. Station specific clusters reflected differences in composition and abundance of infaunal communities particular to exposed (GN and WP) and sheltered (JC) intertidal sites. Segregation of annual collections within stations, which clustered at

Station/Taxa	Feeding Type	Percent (%)	Range (%)
Giants Neck			
Marenzelleria virdis	SDF	19.5	1.0 -78.6
Leitoscoloplos fragilis	BDF	17.4	2.3 - 45.5
Rhynchocoela	C	15.3	4.7 -36.7
Oligochaeata	SDF	15.0	4.4 - 57.5
Paraonis fulgens	SDF	14.1	0.7 -35.7
Capitella spp.	BDF	5.3	0.5 -12.8
Leitoscoloplos spp.	BDF	2.4	0.0 - 17.5
Neohaustorius biarticulatus	SF	2.0	0.0 - 1.3
Hediste diversicolor	O	1.9	0.0 - 7.5
Polydora cornuta	SDF	1.4	0.0 - 6.7
Jordan Cove			
Oligochaeta	SDF	78.0	65.3 -92.1
Marenzelleria virdis	SDF	8.9	1.1-21.8
Hediste diversicolor	O	7.0	1.0-16.4
Capitella spp.	BDF	1.4	0.0 - 4.6
Polydora cornuta	SDF	1.0	0.1 - 3.7
Gammarus lawrencianus	SDF	1.0	0.0 - 7.3
Rhynchocoela	C	0.8	0.0 -6.4
Gemma gemma	SF	0.7	0.0-14.6
Microphthalmus sczselkowii	SDF	0.6	0.0 - 4.0
Fabricia sabella	SF	0.4	0.0 - 3.3
White Point			
Rhynchocoela	C	27.8	7.9 - 62.3
Paraonis fulgens	SDF	21.5	6.3 - 60.2
Leitoscoloplos fragilis	BDF	16.9	4.9 - 39.0
Oligochaeta	SDF	14.5	1.0- -29.8
Streptosyllis arenae	C	6.3	0.6 - 28.0
Leitoscolplos spp.	BDF	2.4	0.0 - 10.8
Marenzelleria viridis	SDF	2.2	0.0 - 9.5
Capitella spp.	BDF	1.3	0.0 - 3.8
Parapionosyllis longicirrata	C	1.1	0.0 - 4.4
Polydora cornuta	SDF	0.8	0.0 - 3.6

Table 2. Ten dominant taxa (%), range (%) and feeding type at intertidal sand stations from 1980-1992.

greater 50%, were caused by changes in the abundance of dominant taxa. For example, high abundance of the amphipod, *Neohaustorius biarticulatus*, and low abundance of the polychaete, *Paraonis fulgens* were obvious at GN. White Point collections consisted of two main groups and three outliers. Lower abundances of dominant taxa such as *Rhynchocoela* and the polychaete, *Leitoscoloplos fragilis* were evident in 1983, 1985 and 1992, resulting in the outlier years. The main groups were linked at the 60% level and had higher abundances of taxa such as *Rhynchocoela, Paraonis fulgens*, Oligochaeta and Streptosyllis arenae. Jordan Cove showed lower similarity (~50%) among years compared to other intertidal stations. Low abundances of oligochaetes, *Marenzelleria viridis,* and *Capitella* spp. and high abundances of rhynchocoels and *Gemma gemma* in 1981, 1989 and 1991 resulted in their separation from the majority of JC collection years. Two other single linkage years, (1980 and 1991), showed high densities of oligochaetes, the polychaetes *M. viridis, Microphthalmus*

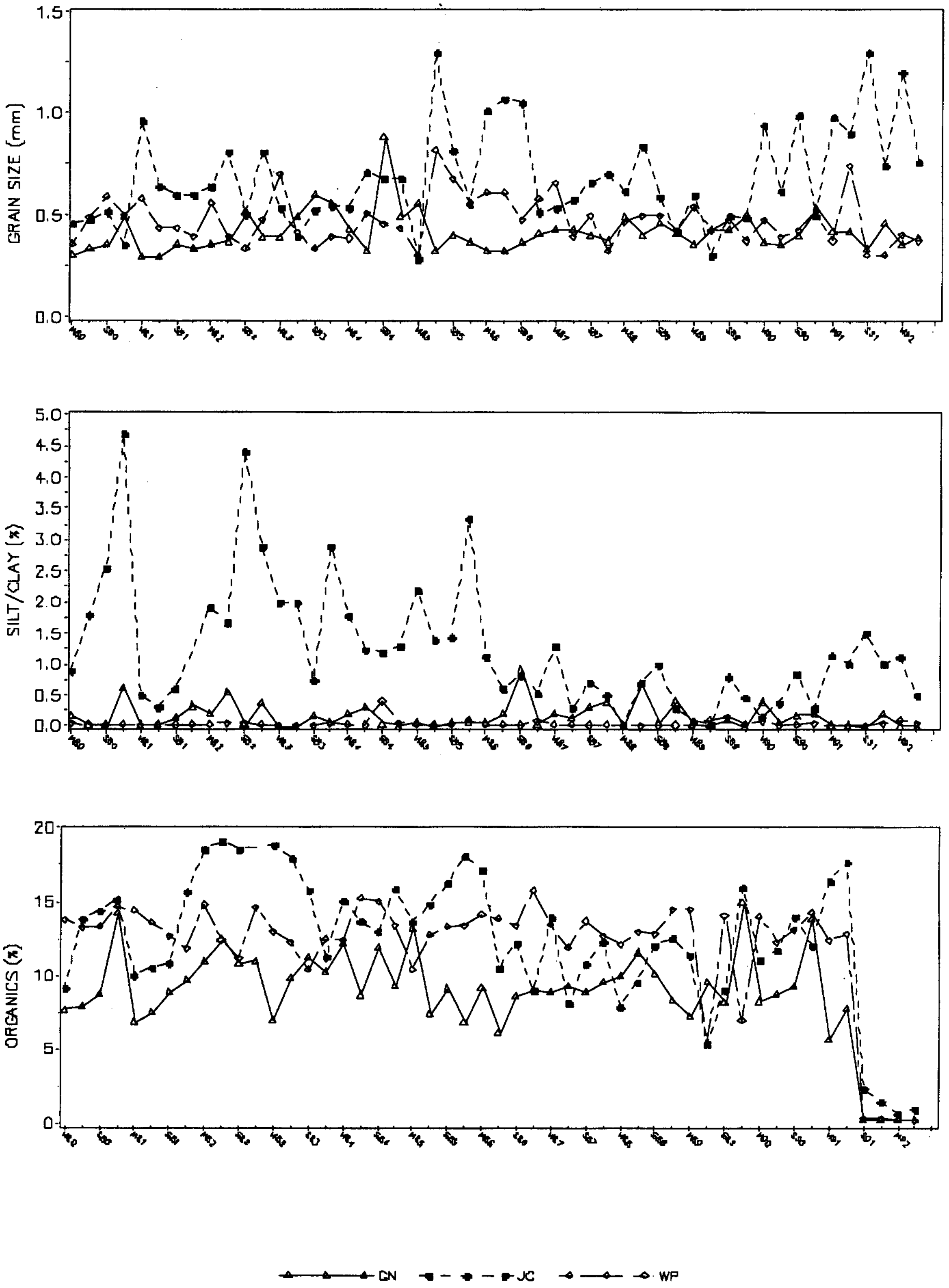

Figure 2. Sediment mean grain size (mm), silt/clay content (%) and organic content (%) at intertidal sand stations from September 1980 through December 1992.

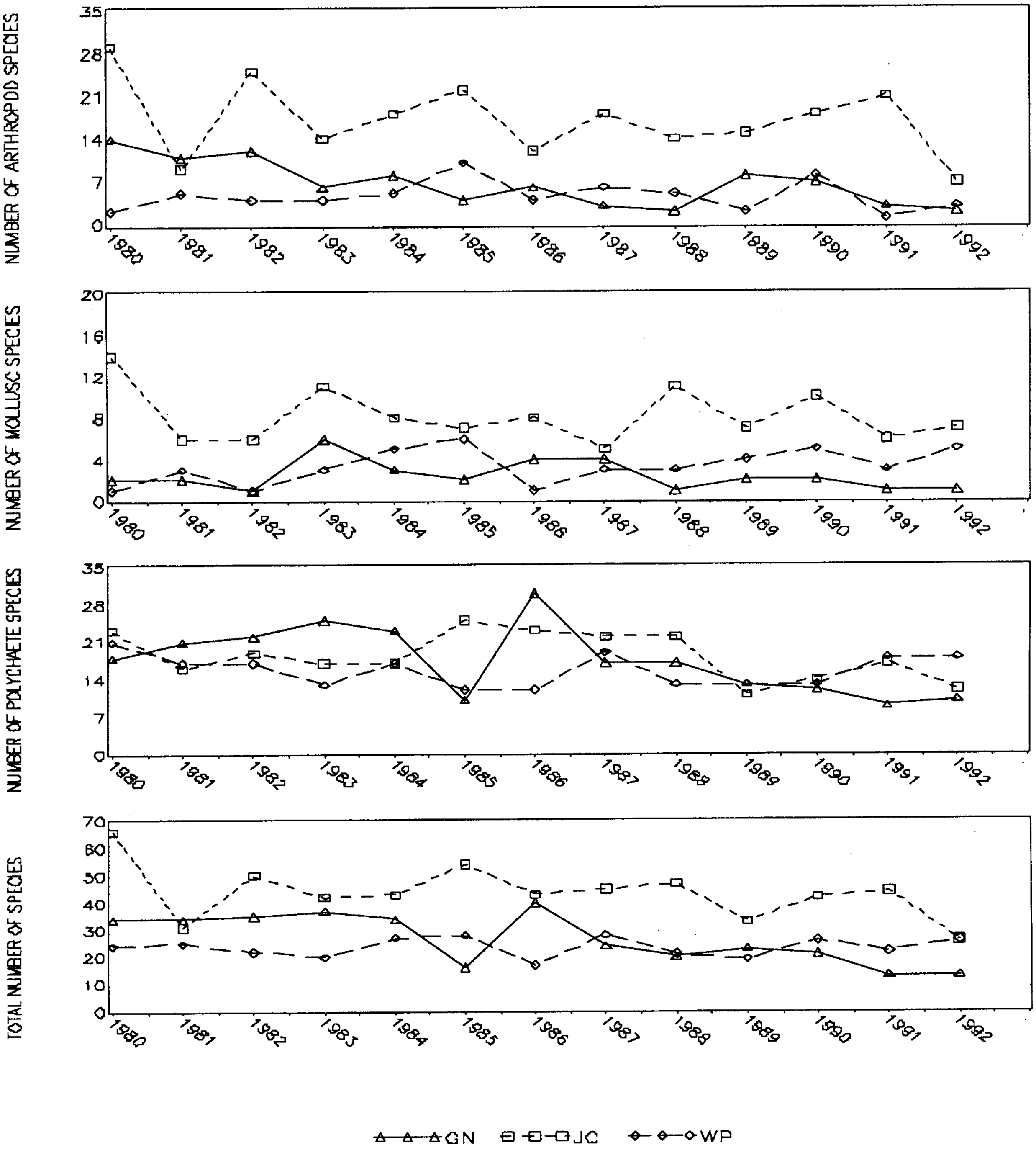

Figure 3. Total numbers of species within major taxonomic groups, and annual total collected at intertidal sand stations 1980-1992.

sczelkowii and *H. diversicolor*, and the amphipod, *Gammarus lawrencianus*. A majority of the JC annual collections contained a similar suite of dominants and exhibited relatively consistent similarities.

Discussion

Sandy beaches within LIS are relatively protected compared to open coastal environments, but wave-induced stress and the resulting sedimentary characteristics appear to be a major factor in structuring macroinvertebrate communities. The numbers of species and individuals was generally higher at the sheltered station. Similar patterns have been observed in other studies of sheltered versus exposed intertidal and shallow subtidal communities (Croker 1977; Withers and Thorpe 1978; Mauer and Aprill 1979; Oliver *et al.*1980; Knott *et al.* 1983; Brazeiro, 2001) Of the major invertebrate groups, oligochaetes, arthropods, and molluscs were dominant components of the sheltered community while rhychocoels were primarily abundant at exposed sites. Many of the dominant polychaete species found during this study have also been reported as co-dominant members of open ocean wave exposed beach assemblages characterized by Haustoriid amphipods. For example, Orbinids, particularly *Leitoscolopolos fragilis*, have frequently been reported as a dominant in clean intertidal sands in Delaware (Maurer and Aprill 1979) and South Carolina (Dexter 1969) and at a sandy site in Barnstable Harbor (Whitlatch 1977). *Paraonis fulgens* is frequently found only in clean, well-drained, sandy habitats (Croker 1975; Whitlatch 1977; Tourtellotte and Dauer 1983) and was exclusively found at our more exposed stations in this study.

Eelgrass beds provide a buffer from wave scour and can be a major source of organic material, enhancing food resources and affording protection at low tide. Several studies have established the link between declining eelgrass abundance and changes in the sedimentary environment and infaunal communities (Dexter 1947; Rasmussen 1973, 1977; Thayer and Phillips 1977). Eelgrass decomposes slowly and thus provides a long-lasting substratum for development of microfloral communities (Burkholder and Doheny 1968; Thayer *et al.* 1980; Rizzo 1990). The abundance of food (i.e., microfloral species colonizing the surface of detritus and sediments) is considered to be a limiting factor to marine oligochaete populations (Caspers 1980; Giere and Pfannkuche 1982; Knott *et al.* 1983; Rossi and Underwood, 2002). Because the JC community is typically dominated by small deposit-feeding organisms such as oligochaetes, the local eelgrass population may indeed influence the composition and abundance of this intertidal sandy beach community.

Temperate intertidal beaches are rigorous environments subjected to widely varying physical conditions and the invertebrate communities are often regarded as highly variable, physically controlled assemblages (Holland and Polgar 1976). Many macroinvertebrates, particularly intertidal species, have adapted to this extreme environment and are highly tolerant of the fluctuating conditions or exhibit high resiliency, despite frequently occurring adverse environmental conditions (Boesch 1974; Oliver *et al.* 1980; Mauer and Aprill 1979; Dauer 1984). Over this long-term study, the composition of both exposed and sheltered intertidal communities remained relatively stable, although the means of maintaining this stability varied over the exposure gradient. Exposed communities included more species that were resistant to the unstable environment. Strong burrowers and species capable of ingesting relatively large sand particles dominated the clean, highly mobile sediments of exposed beaches. Species inhabiting the more sheltered site were smaller, deposit-feeders that could rapidly colonize an area following periods of severe environmental stress.

Conclusions

Intertidal communities showed consistent, long-term spatial relationships among stations; patterns of abundance and community composition at WP were similar to GN, while the community at JC remained dissimilar, both in terms of species composition and abundance. These spatial differences

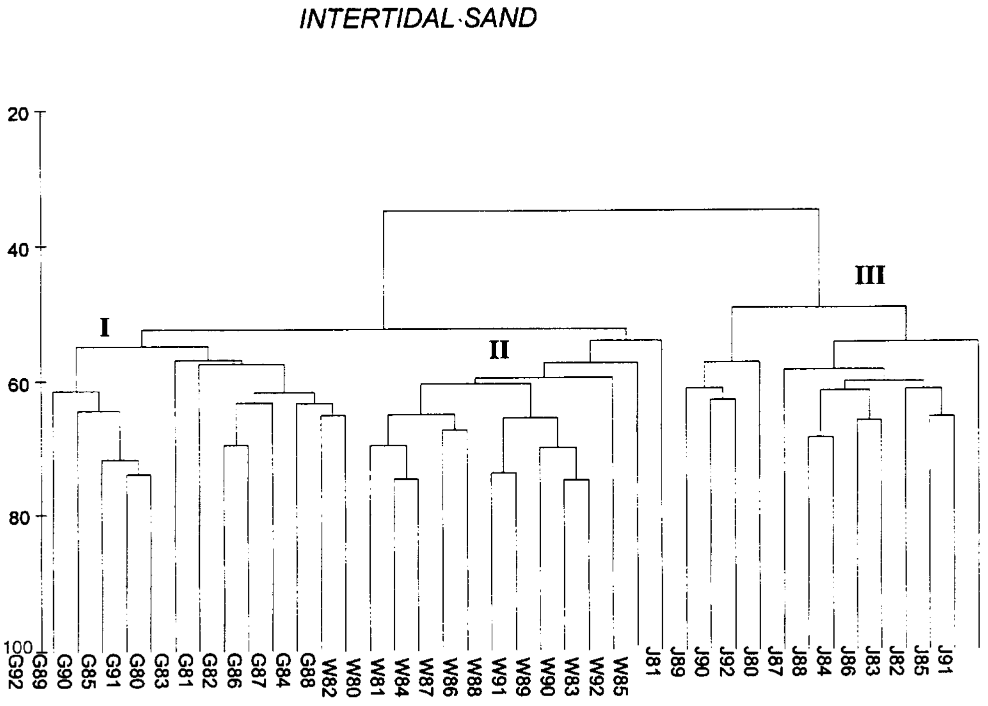

Figure 4. Clustering dendrogram of density data of annual collections at intertidal sand stations 1980-1992.

were attributed to site-specific differences in the degree of exposure to wind and wave-induced sediment scour.

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Appendix 1. Taxa sampled at the Giants Neck (GN), Jordan Cove (JC) and White Point (WP) intertidal sand stations from 1980-1992. Occurrence is expressed as presence in the total number of sampling periods.

Taxon	March 1980-1992			June 1980-1992			Sept. 1980-1991			Dec. 1980-1991		
	GN	JC	WP	GN	JC	WP	GN	JC	WP	GN	JC	WP
Nemertea	69	36	65	89	43	97	92	30	69	78	42	79
Oligochaeta	52	100	58	76	100	75	87	99	80	45	98	70
Polychaeta												
Ampharetidae												
Ampharete spp.					0.8							
Capitellidae												
Capitella spp.	0.8		2.3	21	26	20.7	73	74	31.7	17	25.8	20.8
Mediomastus ambiseta	6.9	0.8			0.8		0.8	3.3		8	1.7	0.8
Heteromastus filiformis					1.5			0.8				
Cirratulidae								0.8				
Tharyx acutus	0.8	0.8	0.8				2.5	0.8	0.8	1.7		0.8
Tharyx spp.	5.4	0.8										0.8
Cirratulus grandis							0.8					
Cirratulus spp.							0.8					
Caullereilla spp.			1.5			0.8						
Monticellina dorsobranchialis									0.8			
Cossuridae												
Cossura longocirrata										0.8		
Glyceridae												
Glycera capitata							0.8					
Lumbrineridae												
Scoletoma tenuis	0.8	0.8	0.8	0.8								0.8
Scoletoma fragilis			0.8									
Maldanidae				0.8	0.8							
Clymenella torquata												0.8
Opheliidae												
Ophelina aulogaster			1.5			2.3			3.3			2.5
Ophelia bicornis			0.8			4.6			3.3			0.8
Ophelia acuminata						0.8						
Ophelia spp.						0.8			1.6			
Nephtyidae												
Nephtys spp.						0.8			0.8			
Nephtys ciliata												0.8
Nereidae												
Nereis virens	0.8	0.8					5.0	4.2	0.8		1.7	
Nereis succinea		6.9			3			19			6.7	
Hediste diversicolor		72.3		28	75	1.5		87			78.3	
Nereis spp.				24.6	24	17.7	3.3	12			0.8	

Taxon	March 1980-1992			June 1980-1992			Sept. 1980-1991			Dec. 1980-1991		
	GN	JC	WP	GN	JC	WP	GN	JC	WP	GN	JC	WP
Orbinidae												
Leitoscoloplos fragilis	38	0.8	49.2	37	1.5	26.2	93	18	96.7	62	11.7	79.2
Leitoscoloplos acutus	1.5		0.8			0.8	11		5.0	3.3		3.3
Leitoscoloplos robustus							0.8	0.8	0.8		0.8	0.8
Leitoscoloplos spp.	3.8			2.3	0.8	3.0	31	4.2	31.7	23	5.8	9.2
Paraonidae												
Paraonis fulgens	41		61.5	85	0.8	92.3	65		83.3	47		58.3
Paraonis gracilis										0.8		
Aricidea catherinae	2.3		6.2			14.6	2.5	3.3		1.7		1.7
Aricidea spp.			0.8									
Paradoneis lyra											0.8	
Cirrophorus lyriformis												0.8
Pectinariidae												
Pectinaria gouldii								0.8				
Phyllodocidae												
Eteone heteropoda				0.8	1.5		10	0.8	1.7		0.8	
Eteone longa								0.8	4.2			
Eteone lactea				0.8								
Eteone spp.							0.8					
Eulalia viridis							0.8					
Eumida sanguinea										2.5	2.5	
Eumida spp.											0.8	
Phyllodoce arenae									0.8			
Phyllodoce mucosa									0.8			
Polynoidae												
Lepidonotus squamatus	3											
Lepidonotus spp.		0.8										
Harmothoe imbricata												
Harmothoe spp.		5.4	0.8									
Harmothoe extenuata	0.8											
Spionidae	2											
Polydora socialis			0.8	0.8		0.8						
Polydora cornuta	1.5	4.6		45	49	16.2	25	79	19.2	1.7	15.8	0.8
Polydora quadrilobata	1.5											
Polydora caulleryi	0.8			0.8						0.8		
Polydora spp.		0.8										
Spio spp.	4			1.5								
Spio setosa	1.5	0.8	0.8	3	2.3	2.3		0.8				
Spiophanes bombyx				3		0.8			1.7			1.7
Spio filicornis	2.3					0.8						
Marenzelleria viridis	40	74	22.3	92	98	46.2	44	88	5.0	18	70.0	4.1
Scolelepis squamata				1.5		6.2	3.3			1.7		0.8
Streblospio benedicti		1.5			1.5		1.7	44			9.2	
Pygospio elegans		32	1.5		20	13.8	6.7	3.3	5.0		16.7	6.7
Prionospio steenstrupi						0.8		0.8				
Prionospio heterobranchia								0.8				

Taxon	March 1980-1992			June 1980-1992			Sept. 1980-1992			Dec. 1980-1991		
	GN	JC	WP	GN	JC	WP	GN	JC	WP	GN	JC	WP
Syllidae												
Exogone hebes	2.3		21.5	1.5		13.8			4.2	0.8		14.2
Exogone dispar		2.3			6			17		0.8	7.5	
Parapionosyllis longicirrata	2.3	1.5	4.6	11	3.1	31.5	4.2	2.5	21.7	2.8	3.3	25.0
Streptosyllis arenae	2.3	7.7	23.1	1.5	6.2	36.9	12	2.5	32.5	10	4.2	58.3
Sphaerosyllis erinaceus			3.8	1.5		6.9				2.5		
Syllides verrilli					0.8	1.5	6.7	38	1.7		10.8	0.8
Brania clavata		1.5	0.8					5			10.8	
Autolytus spp.					0.8							
Hesionidae												
Microphthalmus aberrans	4			0.8			1.7	2.5		5		
Microphthalmus sczelkowii	4	1.5		29	0.8	0.7	25	25	0.8	8	8.3	1.7
Dorvillidae												
Schistomeringos caecus	3			8			4.2			3.3		
Schistomeringos rudolfii			0.8									
Protodorvillea gaspeensis	0.8			1.5						0.8		
Mageleonidae												
Magelona rosea	0.8						4.2		12.5			2.5
Magelona spp.												0.8
Sabellaridae												
Sabellaria vulgaris		0.8									0.8	
Sabellidae		0.8										
Potamillla reniformis					1.5					0.8	2.5	
Fabricia sabella		5.4			7			1.7			0.8	
Euchone elegans		0.8										
Chone infundibuliformis		1.5			0.8						2.5	
Jasmaneira spp.											0.8	
Archiannelida						4.6	1.7		0.8	0.8		5.8
Arthoropoda												
Amphipoda					5.4			4.2			1.7	
Ampeliscidae	0.8	0.8						0.8				
Ampelisca abdita				0.8						0.8		
Ampelisca verrilli		0.8										
Ampelisca vadorum											0.8	
Calliopiidae												
Calliopius laeviusculus				0.8			0.8					
Corophiidae												
Corophium spp.				2.3	4	0.8	2.5	8.3			5.0	0.8
Corophium acherusicum		1.5			7			2.5			0.8	
Corophium acutum		0.8			2.3			3.3	0.8		1.7	
Corophium insidiosum					12	0.8		2.5				0.8
Corophium tuberculatum					0.8			4.2			1.7	
Erichthonius brasiliensis								0.8				
Erichthonius rubicornis								0.8				

Taxon	March 1980-1992			June 1980-1992			Sept. 1980-1992			Dec. 1980-1991		
	GN	JC	WP	GN	JC	WP	GN	JC	WP	GN	JC	WP
Dexaminidae												
Dexamine thea								6				
Gammaridea	5											
Gammarus lawrencianus	2.3	1.5		3.8	48	3.0		6		0.8	25.8	0.8
Gammarus mucronatus				0.8	14			41			2.5	
Gammarus spp.		3.1			8.5	0.8	0.8	0.8				
Gammarus tigrinus										0.8		
Gammarus daiberi		0.8				0.8						
Ampithoidae												
Ampithoe spp.										0.8		
Ampithoe valida					0.8			3.3				1.7
Ampithoe rubricata					0.8						0.8	0.8
Cymadusa compta					0.8							
Aoridae					0.8			1.7	0.8		0.8	
Leptocheirus pinguis	1.5	0.8	0.8	0.8	5.4	0.8	0.8	10		0.8	2.5	
Leptocheirus plumulosus	3	0.8						7		0.8	0.8	
Leptocheirus spp.				0.8				1.7				
Microdeutopus gryllotalpa		0.8			8		0.8	18	1.7		5.8	
Microdeutopus anomalus		0.8			2.3			1.7				
Unciola serrata					0.8	0.8						
Liljeborgiidae												
Listriella barnardi					0.8			2.5			0.8	
Melitidae												
Elasmopus levis					0.8							
Oedicerotidae												
Monoculedes edwardsi					1.5							
Pontoporeiinae												
Amphiporeia virginiana						0.8						
Photidae												
Microprotopus raneyi		1.5			0.8	0.8	0.8	8	0.8	0.8		
Phoxocephalidae												
Trichophoxus epistomus				0.8								
Paraphoxus spinosus					0.8							0.8
Phoxocephalus holbolli			0.8		0.8							
Melitidae												
Melita dentata								2.5				
Lysianassidae												
Psammonyx nobilis												
Ischyroceridae												
Jassa falcata	0.8	1.5		2.3	2.3	1.5		0.8			0.8	2.5

Taxon	March 1980-1992			June 1980-1992			Sept. 1980-1992			Dec. 1980-1991		
	GN	JC	WP	GN	JC	WP	GN	JC	WP	GN	JC	WP
Hyalidae												
Hyale nilssoni	0.8				0.8	0.8				3.3		
Hyale plumulosa										0.8		
Stenothoidae							0.8					
Stenothoe minuta	0.8				0.8					1.7	0.8	
Stenothoe gallensis								0.8				
Haustoriidae										1.7		
Haustorius canadiensis	6			1.5		3.1				3.3		2.5
Neohaustorius biarticulatus	20		6.9	8		1.5				36		3.3
Protohaustorius wigleyi				0.8								
Talitridae												
Orchestia grillus					0.8							
Orchestia spp.						0.8						
Decapoda												
Crangon septemspinosus								6	5.0			
Majidae												
Libinia spp.										0.8		
Xanthidae												
Dyspanopeus sayi					0.8			5			0.8	
Eurypanopeus depressus								0.8				
Portunidae												
Carcinus maenas					6.2			12.5			1.7	0.8
Xiphosura												
Limulus polyphemus								0.8		0.8	0.8	
Mysidacea							1.7					
Heteromysis formosa		0.8				0.8				0.8		
Neomysis americana									0.8			
Caprellidae												
Caprella penantis									0.8		3.3	
Caprella linearis											1.7	
Caprella spp.											1.7	
Isopoda							0.8					
Armadillium vulgare								0.8			0.8	
Edotea triloba	0.8				5	0.8		23				
Edotea spp.								0.8				
Philoscia vittata	0.8	0.8	1.5									
Idotea baltica					2.3		0.8	5			2.5	
Idotea phosphorea					1.5			0.8				
Idotea filiformis								0.8				
Erichsonella filiformis					6.2							
Jaera albifrans					7							
Chelura terebrans					0.8							

Taxon	March 1980-1992			June 1980-1992			Sept. 1980-1992			Dec. 1980-1991		
	GN	JC	WP	GN	JC	WP	GN	JC	WP	GN	JC	WP
Cirolanidae												
Politolana concharum				0.8								
Ptilanthura tenuis							0.8					
Tanaidacea												
Leptochelia savignyi								1.7		0.8	0.8	
Cumacea							0.8					
Oxyurestylis smithi				1.5	2.3	0.8	0.8		1.7	0.8		
Leucon americana					0.8							
Leptocuma minor					1.5							
Cyclaspis varians					1.5							
Chordata	0.8											
Hemichordata												
Saccoglossus kowaleskii				2.3		3.8			17.5			2.5
Harpacticoidia	0.8	0.8		0.8	7							
Hutchinsonella macracantha								0.8				
Mollusca					0.8							
Gastorpoda		1.5									0.8	0.8
Nassariidae												
Ilyanassa obsoleta					0.8							
Nassarius trivittatus											1.7	0.8
Lacunidae												
Lacuna vincta	0.8	4.6		1.5	22	3.1	0.8	2.5				0.8
Columbellidae												
Mitrella lunata	0.7	1.5	2.3		1.5	0.8				0.8	7.5	1.7
Anachis lafresnayi		0.8			0.8						3.3	
Anachis avara			0.8								1.7	
Cerithiopsidae												
Cerithiopsis greeni												0.8
Pyramidellidae												
Odostomia bisuturalis	0.7							0.8				
Odostomia seminuda		0.8										
Turbonilla elegantula		0.8				0.8						0.8
Odostomia spp.			0.8									
Turbonilla nivea			0.8									
Muricidae												
Urosalpinx cinerea		0.8										
Littorinidae												
Littorina littorea					0.8						0.8	

Taxon	March 1980-1992			June 1980-1992			Sept. 1980-1992			Dec. 1980-1991		
	GN	JC	WP	GN	JC	WP	GN	JC	WP	GN	JC	WP
Melampodidae												
Melamphus bidentatus											0.8	
Calyptraeidae												
Crepidula fornicata				0.8			0.8	1.7			0.8	
Crepidula plana		0.8		1.7			0.8				10.0	
Crepidula convexa		0.8						3.0			1.7	
Crepidula spp.											0.8	
Trochidae												
Margarites spp.				0.8	0.8							
Naticidae												
Euspira spp.				0.8								
Hydrobiidae												
Hydrobia totteni						0.8		0.8				
Scaphandridae												
Acteocina canaliculata				0.8								
Bivalvia												
Teredinidae		0.8										
Veneridae												
Gemma gemma		25	10.0	2.3	11	3.1		41			32.5	7.5
Lyonsiidae												
Lyonsia hyalina		0.8										
Mytilidae												
Mytilus edulis		15.4	3.8	2.3	9.2	7.7					5.0	
Tellinidae												
Tellina agilis		0.8		0.8		2.3	6.7		0.8	4.2		1.7
Nuculidae												
Nucula annulata		0.8	2.3			0.8					1.7	3.3
Solenidae												
Ensis directus		0.8					0.8					
Myidae												
Mya arenaria		3.8					2.5	36			4.2	
Mactridae												
Spisula solidissima							1.7					
Spisula spp.							2.5	4.2				
Mulinia lateralis										0.8		
Montacutidae												
Mysella planulata											0.8	0.8
Semelidae												
Cumingia tellinoides									0.8			
Edwardsia elegans					0.8							
Loligo pealeii					0.8							

Diurnal Variability in the Surface Mixed Layer in Central Long Island Sound

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Abstract

Long Island Sound (LIS) experiences episodes of mixing during the stratification-dominated summer months and a complete overturn during fall/winter. A one-dimensional numerical turbulence model (General Ocean Turbulence Model, GOTM) is used to diagnose the response of water column structure in central Long Island Sound to both diurnal and synoptic period meteorological forcing during the period from 08/04/99 to 09/28/99. Results discussed here focus on diurnal variations in the surface mixed layer during this period, and provide insight into the response to the sea breeze. The model was forced by net surface heat and momentum flux estimated from available meteorological observations, and by a barotropic tidal current. The mixed layer is seen to deepen rapidly beginning in the afternoon; mixed layer deepening continues through the night but at a reduced rate. Mixed layer depths ranged from 4m to 6m during the period considered. The maximum and minimum in SST are approximately coincident with the zeros in net heat flux. SST declines monotonically throughout the evening and increases rapidly in the morning. The maximum diurnal variation in SST was approximately 2°C during the period considered. Analysis of the Turbulent Kinetic Energy (TKE) balance showed that shear production was very sensitive to the surface wind stress. During periods of positive net heat flux there was a competition between shear production and buoyancy production. During nighttime cooling buoyancy production made a weak positive contribution to the TKE balance.

Introduction

Interest in the factors contributing to an early overturn of the water column and elevated bottom water temperatures during fall 1999 was the initial motivation for examining water column structure in Long Island Sound during this period. CTDEP Station D3 (located at 41°11', -72°85') was considered as a representative west central Long Island Sound site. Hydrographic observations were available from 08/04/99 to 09/28/99. Wilson and Hao (2003) have shown that a sequence of synoptic events contributed to a complete mixing of the water column by 08/29/99, more than two weeks before Hurricane Floyd. During the first part of this period, however, surface forcing was associated primarily with the diurnal sea breeze. It is the water column response to this forcing that we want to discuss here. The available hydrographic data did not resolve either the diurnal or semi-diurnal fluctuations in water column structure. Our strategy was to use the observations for initial conditions for a 1D turbulence model subject to forcing which included the diurnal sea breeze and semi-diurnal tide. The model would allow us to diagnose the transient mixed layer response. The model also affords the option of relaxing against available hydrographic observations.

Model

We used a 1D community turbulence model GOTM which is described in detail by Burchard and Bolding (2001). It offers a second moment closure recently refined by Canuto *et al.* (2001). The model is based on one-dimensional (vertical) momentum, heat, and salt equations. The required forcing includes the components of the barotropic pressure gradient and the components of the surface momentum flux. The barotropic pressure gradient can be specified in terms of the components of depth averaged current; this was taken from a 2D tidal model. The components of surface wind stress were computed from hourly observations of wind speed and direction at 88 meters elevation at the meteorological tower at Brook Haven National Laboratory using scripts in the SEAMAT Air-Sea Toolbox.

The model requires, in addition, specification of both the short wave radiation and the sum of the long-wave, latent and sensible heat fluxes. Long-wave flux, and sensible and latent heat flux contributions were calculated using scripts from the SEAMAT Air-Sea Toolbox. The long-wave flux requires SST, air temperature, relative humidity, and cloud cover. Sensible and latent heat fluxes require SST, air temperature, relative humidity, wind speed. All meteorological data used were hourly observations from Brook Haven National Lab with the exception of cloud cover measured from LaGuardia Airport. Hourly observations for SST were unavailable. The strategy used was iterative; we first forced with estimated fluxes using linearly interpolated (in time) SST from CTD observations at Station D3 which exhibited no diurnal variations. We then used the model's SST to recalculate the surface heat fluxes.

Results

Figure 1 shows model results for temperature and Vaisala frequency within the upper 10 meters of the water column for a four-day period beginning on 08/04/99. The mixed layer is seen to deepen rapidly beginning in the afternoon; mixed layer deepening continues through the night but at a reduced rate. Mixed layer depths ranged from approximately 4m to 6m during the period considered. SST reaches a maximum in early evening and then declines monotonically throughout the evening and nighttime; it increases rapidly in the morning. The maximum diurnal variation in SST was approximately 2 °C during the period considered. Figure 2 shows time series of the net heat flux and wind stress magnitude. The time series values are normalized by the maximum values for the period: 840 W/m^2 and 1.2×10^{-4} N/m^2. The maximum and minimum in SST are approximately coincident with the zeros in net heat flux.

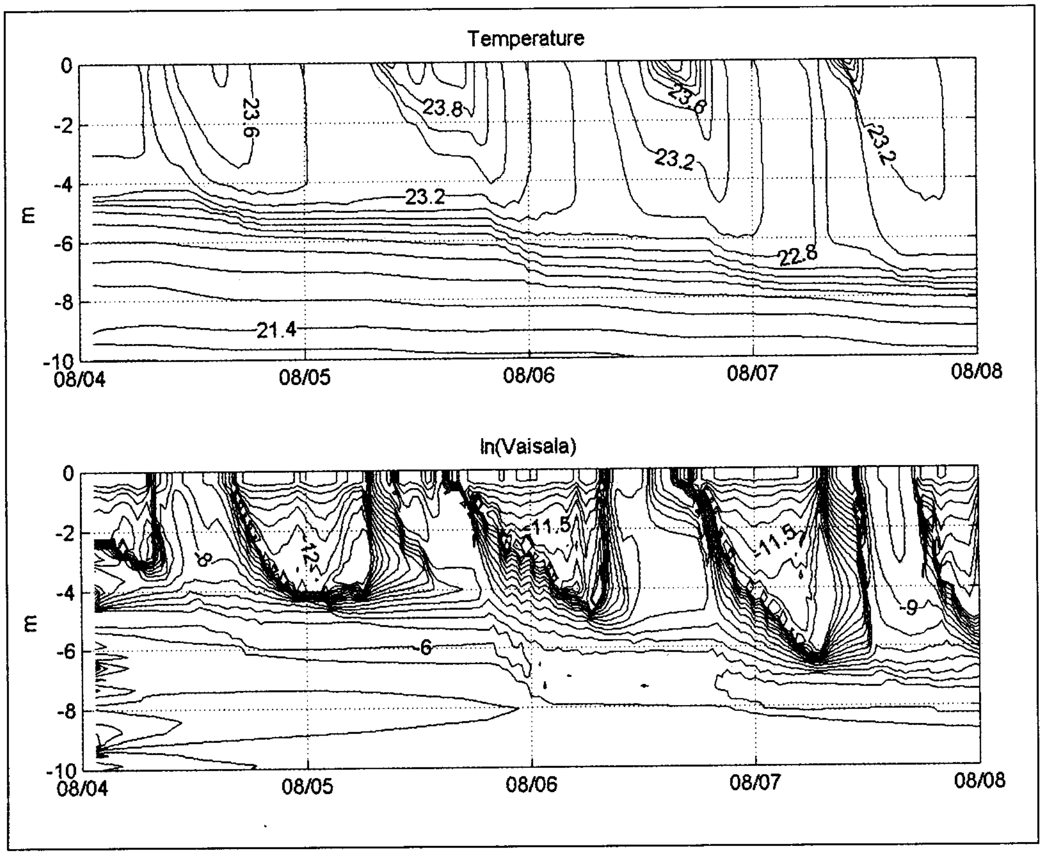

Figure 1. Temperature (°C) and ln of the Vaisala frequency (s^{-1})

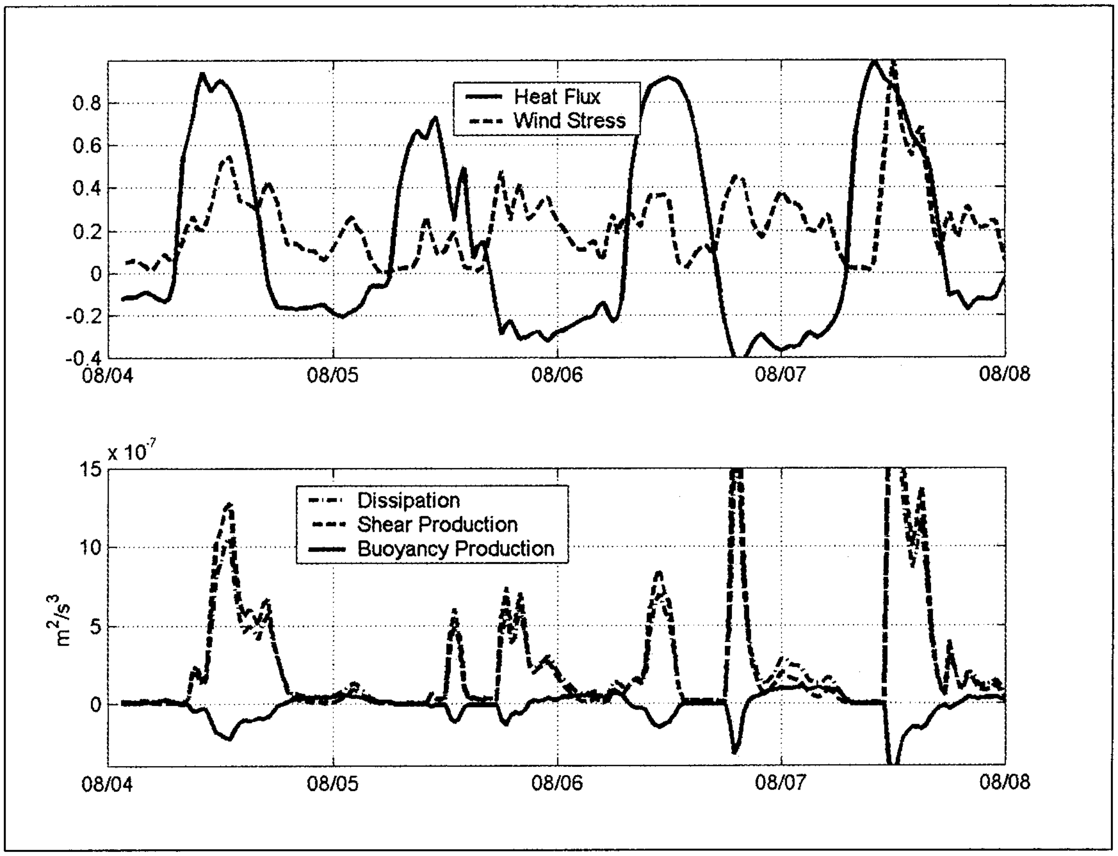

Figure 2. Normalized net heat and momentum flux and TKE terms.

Figure 2 also shows time series of dominant terms in the TKE balance at 2 meters. Shear production is very sensitive to the surface wind stress. Buoyancy production is dependent on the net heat flux. During periods of positive net heat flux there is a competition between shear production and buoyancy production. During nighttime cooling buoyancy production makes a weak positive contribution to the TKE balance.

The depth-dependent structure of both the shear production and buoyancy production are shown in Figure 3. There is strong diurnal variability in both production terms within the mixed layer. The ratio of the sum of buoyancy and shear production to the dissipation is of order unity.

The bottom mixed layer occupies a major fraction of the water column. Shear production exhibits strong quarter-diurnal variability within the bottom mixed layer. Characteristics of the interaction of the surface and bottom mixed layers can be investigated with the model.

Acknowledgment

We gratefully acknowledge support from New York Sea Grant.

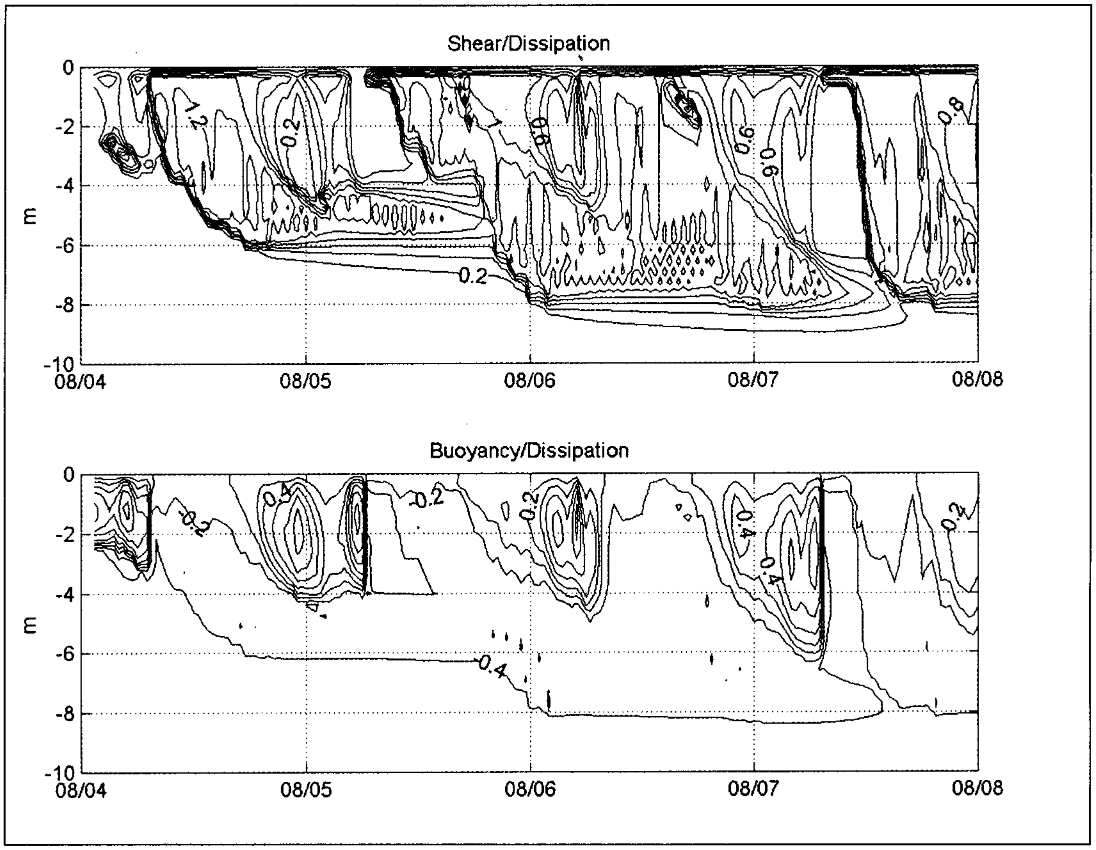

Figure 3. Shear production and buoyancy production.

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Tidal Marsh Loss in the Quinnipiac River Estuary

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Many coastal areas have experienced significant loss of tidal wetlands, and this problem is expected to increase in severity as climate change leads to an acceleration in sea level rise. Aerial photographs taken over the last several decades show substantial loss of vegetation from certain areas within the extensive Quinnipiac River tidal marshes, CT. Classification and analysis of color infrared photos for a 53-ha study area showed an increase in mudflat of 17 ha (36%), over the period 1974-2000. This loss of vegetation was accompanied by significant changes in vegetation composition, from predominantly *Typha latifolia* (85% of vegetated area) to predominantly *Phragmites australis* (61%). *Pluchea purpurascens* now appears to dominate in some areas formerly vegetated by *T. latifolia* but not yet converted to mudflat. Several possible causes of mudflat development have been suggested, ranging from local effects (hydrologic and water quality changes) to regional mechanisms (sea level rise). In order to assess the mechanism of marsh loss, we have begun collecting data on sediment dynamics at sites corresponding to the three different vegetation regimes (*P. australis, T. latifolia, P. purpurascens*/emerging mudflat). These data consist of: radiodated sediment cores, short-term sediment deposition rates (as measured by rubber sediment collectors), and hydroperiod. The erosion taking place in this marsh has the potential to lead to large fluxes of nitrogen and trace metals from the marsh to the river, and ultimately to Long Island Sound.

Episodic Sediment Resuspension and Dispersion in Estuaries

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Aperiodic resuspension of estuarine sediments affects a variety of physical, chemical, and biological processes active in the vicinity of the sediment-water interface. The extent of this influence varies as a function of the spatial and temporal scales of the resuspension. These in turn vary in response to the character of the resuspension process. In the typical estuarine setting the resuspension process is the result of a variety of factors both natural and man-made acting collectively and individually. Discrimination of the relative importance of each of these factors is best realized by analysis of relatively long-term time series observations of near-bottom sediment transport system. Analyses of data obtained by in-situ instrument arrays deployed at a variety of locations within Long Island Sound, both shallow and deep water, are used to define the spatial and temporal scales of influence of persistent tidal flows, high energy storm events and mechanical dredging. Mass balance calculations indicate that both tidal and storm induced resuspension can have system wide impacts. In contrast, mechanical dredging effects are most often confined to the immediate vicinity of the project area. The role of each within the sediment transport regime of the Sound varies as a function of the associated time scales. Tidally mediated resuspension dominates near-bottom mass flux due to persistence. Storm induced perturbations, while orders of magnitude larger than background values, are generally of secondary importance due to their short duration. The influence of dredging is yet more limited due to limited dispersion, short duration, and low recurrence frequency. The implications of these results within evaluations of sediment transport dynamics and/or marine resource management are discussed.

Micronuclei in Peripheral and Red Blood Cells of *Mustelus canis* and *Squalus acanthias* Representing Two Orders of Sharks from the Long Island Sound and Coastal Waters of the Norwestern Atlantic

Borucinska, J.D., A. Czachorowska, S. Mookherjea and M.Habib

Two species of sharks, the smooth dogfish (*Mustelus canis*) and the spiny dogfish (*Squalus acanthias*) were examined for the presence micronuclei (MN) in peripheral red blood cells (RBC). This study was part of a pilot survey of the health status of wild shark populations in which micronuclei and histopathological lesions were used as bioindicators. Twenty nine *M. canis* were collected between April and June 2000 in the Long Island Sound. The collections were done during survey cruises of the Marine Fisheries Division, Connecticut Department of Environmental Protection. Nineteen *S. acanthias* were collected between September and October 2000 in the coastal Atlantic waters off Massachusetts, USA. These fish were collected with the help of the Division of Marine Fisheries, Commonwealth of Massachusetts. The sharks were manually restrained and a blood sample was taken from the caudal vein. Four blood smears were prepared from each shark, air dried, fixed in 70% ethanol, and stained with the Feulgen reagent for the micronucleus assay. Five thousand RBC were examined for each shark. Micronuclei were found in 69% *M. canis* and 35% *S. acanthias*. *Mustelus canis* had 1.9% of RBC with MN (a total of 179 MN), *S. acanthias* had MN in 0.2% of RBC (a total of 10 MN). *Mustelus canis* had a higher total number of microscopic lesions than *S. acanthias*. The results of this small pilot study indicate that sharks caught within the Long Island Sound have worse bioindicator indices of their health than sharks from open coastal waters of the northeastern Atlantic. More studies are needed to fully evaluate the health status of LIS sharks and its causal relationship with their environment.

The Surface Elevation Table (SET) and Marker Horizon Approach for Evaluating Wetland Sediment Elevation Dynamics in Coastal Wetlands

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Given the potential for coastal wetland submergence from rising sea levels, wetland managers need to determine if their marshes are building vertically at a pace equal to sea-level rise. Measures of vertical accretion are often used for this purpose. But measures of vertical accretion alone often overestimate elevation gain, and therefore underestimate the potential for submergence, because they do not fully account for local subsurface influences on elevation such as compaction and decomposition of underlying sediments. Therefore a method was developed using direct measures of elevation change from a surface elevation table (SET) in combination with measures of vertical accretion from soil marker horizons, to calculate both an elevation deficit (sediment elevation vs. sea level) and the amount of shallow subsidence (subsurface influence on elevation). With knowledge of the separate influence of surface and subsurface processes on sediment elevation, managers can develop more appropriate management practices to enhance sediment elevation. The SET is a mechanical leveling device that attaches to a benchmark and measures sediment elevation with high precision. The SET has undergone numerous design improvements during the past decade, including the development of deep (> 10 m) and shallow (< 1 m) benchmarks to aid in determining which subsurface process is controlling sediment elevation. A recent innovation is the use of a sonic sensor to provide automated, continuous readings of sediment elevation that can be used in conjunction with the mechanical SET. An overview is presented of the design improvements to the SET and its applications in measuring wetland sediment elevation dynamics.

The Genetic Stock Structure of Winter Flounder Larvae in Long Island Sound: Spatial Distribution and the Impact of Larval Entrainment

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Winter flounder (*Pseudopleuronectes americanus*) is one of a number of coastal American flatfish that faces intense fishery pressure and thus has been the focus of management efforts. This species has experienced dramatic declines in population numbers over the past three decades with concomitant decreases in commercial and recreational fishing landings. The genetic stock structure of winter flounder larvae in Long Island Sound has not been previously characterized. Stage 1 (yolk-sac) and 2 (pre-flexion) larvae were collected from several locations in Long Island Sound known to be nursery areas for winter flounder in the spring of 2001. Genetic variations among larvae were characterized through the use of 6 microsatellite loci that had been previously reported to be highly polymorphic and heterozygous in winter flounder. Gene frequency differences among microsatellite alleles were used to characterize population structure. Substantial genetic differences were seen among the putative source populations. These genetic differences appeared to be geographically based and provide evidence of genetically distinct spawning populations that appear to be temporally stable. These differences were used to characterize the most likely sources of winter flounder larvae entrained at the Millstone Nuclear Power Station as well as settled juvenile winter flounder collected in the Niantic River. Samples were classified to the most likely geographical source population through use of a neural net learning algorithm (NeuroShell®Classifier, Ward Systems, Inc.). This approach had high precision (i.e., >98% correct assignment of known larvae to correct source population) and accuracy in classifying the likely geographical sources of samples. These results impact winter flounder management issues.

Harbor Seals in Long Island Sound: a Survey of Potential Haul-Out Sites and Effects of Human Disturbance on Behavior

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Long Island Sound is an important over-wintering site for North Atlantic populations of harbor seals (*Phoca vitulina concolor*). The abundance of harbor seals in LIS is increasing. However, the number of protected haul-out sites is limited and human activities along the densely-populated shoreline have the potential to disturb harbor seals from their haul-outs. Haul-out sites are important areas for harbor seals to rest and thermoregulate. Surveys of potential haul-out sites and weather-related hauling-out patterns were conducted from 1996 to 2002 in Norwalk, Connecticut. The results provide evidence that harbor seals are selective of their haul-out site habitat. Of the 16 islands and reefs surveyed, only two locations, Sheffield Island Ledges and Smith Reef, were used by seals as haul-out sites in Norwalk. This study also illustrated the influence that weather conditions, particularly wave height and wind speed, have upon the number of seals hauled-out. In addition, observations were conducted in order to determine if human activities near haul-outs affected harbor seal behavior. Observations were made during ground-based surveys and from boats that remained at least 200 m from the haul-outs. Seals showed signs of disturbance by human activities that occurred at distances as great as 160 m. Small boats that approached within 60 m of a haul-out caused seals to flush from the rocks. Repeated disturbance from haul-outs may cause harbor seals to abandon a site. Long-term surveys suggest that harbor seals may have abandoned haul-outs near Norwalk Harbor due to increased boat traffic.

Development of a System for Continuous Monitoring of Pallial Cavity Pressure and Valve Gape in the Oyster *Crassostrea virginica*

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We have developed an optical biomonitor capable of continuous short- and long-term recording of pallial cavity pressure and valve gape in bivalve molluscs. The pressure sensor, which is 2 cm long and 5 mm in diameter, consists of a PVC casing with a reflective latex diaphragm at its distal tip. The lower half of the sensor casing penetrates the shell and mantle of the oyster and protrudes approximately 2 mm into the suprabranchial chamber. Functionally, it is a simple amplitude modulated displacement transducer that records changes in light reception caused by movement of the diaphragm. The valve gape sensor is attached to the right valve during experimental trials and directly measures the distance between two fibers by the extinction of light. With this arrangement, we are able to examine the relationship between changes in valve gape and pressure, in response to measured changes in environmental parameters such as food concentration. This device allows us to explore more thoroughly the mechanisms available to bivalves for controlling pumping rate, and expand the scope of our understanding about the responses of bivalve molluscs to changes in environmental parameters. Results of laboratory trials have revealed multiple patterns of association between pumping and valve gape and have established that the system can record changes in these parameters in real time.

Long-Term Monitoring of Benthic Epifaunal Recruitment in Long Island Sound: Expansion of a 10-Year Monitoring Effort

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As global biological invasion events increase, through range expansions and introductions of exotic species, it is important to understand the spatial and temporal processes controlling epifaunal community dynamics. Our current monitoring program focuses on temporal and spatial recruitment and population dynamics for a number of native and invasive hard-substrate benthic animals at four sites in eastern Long Island Sound. Expansion of our monitoring efforts will add to the extensive 10-year recruitment record established for the Avery Point site and further our understanding of the patterns of exotic species introductions and native recruitment in marine waters. Analysis of the ten-year epifaunal recruitment record from Avery Point indicates there is a strong correlation between the onset of recruitment with increasing spring water temperatures in invasive ascidian species but not for native species. This suggests that as mean ocean temperature increases invasive species will be capable of starting recruitment earlier. Potentially, early recruiting species will be able to out compete natives for space. Data from the summer and fall 2001 indicate species presence, recruitment magnitude and temporal recruitment patterns are site-specific. Invasive species comprised 40% of the epifaunal recruitment. Recruitment of both native and invasive species was correlated to water temperature; however, the strength of the associations is species and site-specific. Our 2001 monitoring efforts helped identify the range extension of the bryozoan *Bugula neritina* from warmer southern waters, and the presence of a previously unobserved colonial Didemnid ascidian.

A One-Box Model of Exchange between Long Island Sound and Shelf Waters

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and C.A. Edwards, Ocean Sciences Department, Univ. of California at Santa Cruz, Santa Cruz, CA

The exchange between Long Island Sound (LIS) and adjacent waters is estimated using conservation of water and salt together with monthly-averaged CT DEP survey data for 1991-1999. Modeling the salt exchange as a constant transport coefficient times the salinity difference between adjacent basins, we find using inverse methods that at least 7400 m^3/s of water must be exchanged. The variability around this long-term average exchange is significant and structured, indicating increased transport in spring. When different transport coefficients for April-June are allowed, our estimation procedure yields $9500 \pm 400 m^3/s$ for the spring and $6200 \pm 300 m^3/s$ for the rest of the year. Admitting the possibility of exchange through the East River does not appreciably change these estimates. Estimates of the volume exchange based on the Knudsen approach, assuming two-layered advective transport can be up to ten times larger than the above results. We argue that this is due to our assumption that the exchange is dispersive, inversely proportional to the difference in mean salinities of adjacent basins rather than the smaller difference in upper- and lower-layer salinities at the boundary. Our estimates are consequently a lower bound. It is clear that the relative importance of the two mechanisms must be established in order to properly exploit the data.

Effects of Nutrient Inputs on Phytoplankton Growth Rates in Long Island Sound

Gobler, C.J., Sañudo-Wilhemly, S.A., Buck, N.J., Sieracki, M.E.

Long Island Sound (LIS) experiences annual summer hypoxia within the bottom waters of its western extent. Although overgrowth of phytoplankton caused by nutrient loading profoundly affects hypoxia in estuaries, nutrient-phytoplankton dynamics in LIS have not been well studied. To understand how nutrient loading impacts the growth of phytoplankton in LIS, nutrient (N, P, and Si) addition experiments were conducted at three stations in LIS (west, central and east) and in the East River (ER) during July 2000 and April 2001. During July experiments, we observed a gradient in the N limitation of phytoplankton growth across LIS. ER phytoplankton showed no response to N, P, or Si additions, western LIS (WLIS) phytoplankton were mildly N-limited (50% biomass increase during N additions), and central and eastern LIS (CLIS and ELIS) phytoplankton yielded large (> 3-fold) increases in phytoplankton biomass in response to experimental N additions. Experiments conducted in April yielded different results. While ER phytoplankton remained nutrient replete, WLIS algal communities were stimulated by N or Si additions (1.5 and 3-fold increases in particulate organic carbon, respectively). Moreover, although CLIS phytoplankton remained N-limited (2-fold biomass increase during N additions) in April, ELIS phytoplankton did not respond to nutrient additions. This result suggests that seasonal variation in flow rates of the Connecticut River can impact the degree to which ELIS phytoplankton are nutrient limited. Finally, size- and species-specific responses of phytoplankton to nutrient additions indicated that N loading can stimulate the growth of large diatoms, which are known to sink rapidly and thus may exacerbate hypoxia across LIS.

Free Amino Acids Composition of Several *Porphyra* Species from the East U.S. Coast

Hariskov, S.D., S. Miller, C. Yarish and R. Hamilton. Marine Biotechnology Laboratory, University of Connecticut, Stamford, CT 06905

The free amino acid content of five *Porphyra* species from the East US coast were determined. The experiment included four blade cultures of *P. leucosticta*, *P. umbilicalis*, *P. linearis*, and *P. amplissima*, as well as four conchocelis cultures of *P. leucosticta*, *P. linearis*, *P. purpurea*, and *P. amplissima*. The corresponding blade and conchocelis cultures were from the same strains and were grown under the same controlled conditions. The free amino acid determination was performed using ethanol extraction and precolumn derivatization with 6-aminoquinolyl-N-hydroxysuccinimidyl carbamate (AQC). High performance liquid chromatography (HPLC) for reverse-phase analysis was used for qualitative and quantitative assay of the free amino acids. CHNS/O Analyzer was employed in the investigation of the total nitrogen content of the selected cultures. The inspected blade cultures exhibited higher total FAA content than the corresponding conchocelis cultures, except for *P. amplissima*. The taurine values observed in *P. umbilicalis* (2853±763 mg/100g D.W.) and *P. linearis* (2654±918 mg/100g D.W.) were significantly higher than values reported in the literature for Asian *Porphyra* species. Another objective of the study was to test the hypothesis for possible relationship between total FAA content and total nitrogen values. Higher total FAA values were found to correlate with higher total nitrogen values both in blade and conchocelis cultures, except for *P. amplissima*. The above-mentioned properties of the described US *Porphyra* species could trigger interest for use of those species in integrated aquaculture and bioremediation.

Effects of Anoxia on Algal Photosynthesis and Nutrient Release on Coastal Pond Sediments

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Coastal eutrophication increases the likelihood of anoxia in coastal ponds by stimulating the growth of algae, which consume oxygen at night during summer months. We examined the photosynthetic response of a green alga common in coastal ponds (*Ulva lactuca*) and several species found on the rocky coastline (the red alga *Chondrus crispus* and two kelps, *Laminaria digitata* and *Alaria esculenta*). Photosynthesis and respiration rates were taken daily from samples that were subjected to either conditions of dark and anoxia or normal light and aeration. Photosynthetic rates declined after five days of anoxia in *Ulva* and after three days in *Chondrus*. Photosynthesis began to fail after only one day in both kelps. Respiration rates were unaffected in all four species. Another experiment examined nutrient concentration in cylinders containing coastal pond mud cores, various amounts of *Ulva lactuca*, and seawater. Water samples were taken at four- or eight-hour intervals at different heights above the sediment during 24-hour incubations in dark and light conditions and analyzed for nutrient concentrations. Ammonium and phosphate increased with time under dark conditions with increased algal abundance, and exhibited highest concentrations near the sediment. Under light conditions, these two nutrients decreased as time progressed in the cylinders containing algae, with a rapid reduction at heights further from the sediment and with increased layers of algae. Nitrate and nitrite were rapidly consumed in cylinders containing algae in dark and light conditions. While the kelps were intolerant of anoxia, it appears that *Ulva lactuca* is tolerant of prolonged anoxia and utilizes nutrients released from pond sediments during anoxic events.

The Fishes of the Hudson River National Estuarine Research Reserves: Sticklebacks (Gastrosteidae) in Decline

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The composition and distribution of the fishes in the estuarine marshes of the Hudson River are poorly understood. The objectives of my study were: 1) to quantitatively identify fish species found within the open water and vegetated habitats of the four Hudson River NERRs; 2) to determine the breeding cycles of the four-spine (*A. quadracus*) and nine-spine (*P. pungitius*) sticklebacks within each marsh. Bi-weekly sampling of the four NERRs (encompassing over 100 miles) of the Hudson River began in late April (2002), ending in September/October (2002) with the purpose of collecting fishes across the wide salinity gradient. Collection methods include two-person seine, fyke nets and a throw trap. Few *A. quadracus* were found and no *P. pungitius* were collected. Other sampling this summer indicated that the four-spine stickleback remains locally abundant in Connecticut. Analysis of a long-term sampling program (the Hudson River Estuary Monitoring Program) revealed that the abundance of the four-spine stickleback has decreased in the Hudson River, particularly since the late 1980's.

Corroborating a General Model of Estuarine Responses to Nitrogen Loading

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A model of estuarine responses to nitrogen loading has been developed to assist with planning and land-use management decisions. It is an empirically-based numerical simulation linking land-use and N-loading to ecologically important and socially relevant endpoints of water quality, hypoxia, and eelgrass habitat. Developed initially for Childs River, Quashnet River and Sage Lot Pond in Waquoit Bay (Cape Cod MA), we modified it to make it easily applicable to new sites. So far, we have applied the general model to 6 new sites: Apponogansett Harbor, Buttermilk Bay, Red Brook Harbor, Sippican Harbor (in Buzzards Bay, MA), Ninigret Pond and Quonochontaug Pond (in southern Rhode Island). The only changes to the model were site-specific descriptions of bathymetry, and forcing functions; all ecological parameters were unchanged. Simulations were compared to recent field data from all sites, and results were evaluated for chlorophyll stock, phytoplankton primary production, planktonic net community production and respiration, DIN, DIP, water clarity, macroalgae stock, and total system net P and R. The CLUE model (Changing Land Use and Estuaries) simulates reasonably well the overall patterns of ecologically important variables related to the eutrophication in a range of shallow coastal sites. It is general, dynamic and predictive. Process-based simulations often require site-specific "tuning" yet our unchanged general model still achieves enough generality for use across a wide range of New England sites. Our hope is that the model can encourage scientifically aware decisions at the local level.

The Importance of Individual and Environmental Influences on the Reproductive Output of Tautog (*Taùtoga onitis*) in Long Island Sound

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Tautog (*Tautoga onitis*) are nearshore demersal fish that have been steadily declining in Long Island Sound probably due to their slow growth, long life span, high fishing pressure, and spawning behavior that enhances catchability. Recently, a fishery management plan was drafted by the Atlantic States Marine Fisheries Commission to provide for the conservation, restoration, and enhancement of tautog stocks. Estimates of the reproductive potential (i.e. seasonal fecundity) of remaining stocks are urgently needed to assess impacts and develop suitable recovery plans for the species. We estimated seasonal fecundity for female tautog in Long Island Sound using both field and laboratory experiments conducted in 2000 and 2001. We examined the influences of individual size and timing within the season on both batch fecundity (number of eggs released per spawning event) and spawning frequency (the probability of an individual spawning on a given day). Reproductive output was greatest during the month of July when females of all sizes spawned almost daily and released their largest batches of eggs. Seasonal fecundity increased with individual size with large (500 mm) females spawning 20-50 times as many eggs over a season as small (250 mm) females. An average (350-400 mm) female spawned about 7 million eggs in 2000 compared to about 10 million eggs in 2001 suggesting that other factors besides individual size affect reproductive potential. Seasonal fecundity estimates for Long Island Sound tautog were higher than previously reported for the species elsewhere; environmental and regional influences on seasonal fecundity will be discussed.

Phytoplankton Community Diversity and Population Dynamics in Long Island Sound: an Update on the Project Progress

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In an effort to study species diversity and population dynamics of phytoplankton along the western-eastern nutrient gradient of Long Island Sound, we are investigating 1) species composition of 18 stations throughout the Sound for larger-sized phytoplankton using a light microscope; 2) species composition of the small-sized phytoplankton (< 10 µm) using molecular analysis; 3) physiological and molecular characteristics of some specific groups of phytoplankton isolated from the Sound. Some of the results and current progress will be presented with discussion on future directions.

Do Marine Aggregates Increase the Accessibility of Small Particles to Benthic Suspension-Feeding Bivalves?

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Aggregations of organic and inorganic particles are common in coastal waters such as Long Island Sound. These marine aggregates (a.k.a. marine snow, organic aggregates, flocs, amorphous detritus) vary in size, shape and composition over a range of temporal and spatial scales. Aggregates are formed in nature via biological (e.g., zooplankton feeding, bacterial mediation, diatom bloom dynamics) and physical (e.g., coagulation, differential settling, macrophyte fragmentation) processes. Also common in coastal ecosystems are benthic, suspension-feeders such as bivalves. Suspension-feeding bivalves actively pump water through their mantle cavity and extract particles for food. Since marine aggregates are often enriched with microorganisms, aggregates may increase the accessibility of small ($<2\mu m$, otherwise poorly retained, picoplankton to these suspension feeders. To test the hypothesis that the presence of aggregates increases the amount of small particles ingested by bivalves, experiments were designed using artificial aggregates generated on a rolling table. Fluorescent beads (1 μm and 10 μm) were incorporated into these aggregates in order to track the fate of small particles. Preliminary results for the blue mussel (*Mytilus edulis*), marsh mussel (*Geukensia demissa*) and sea scallop (*Placopecten magellanicus*) will be reported.

Estimation of Nonpoint Nitrogen Loads and Yields from Monitored and Unmonitored Basins Draining to Long Island Sound 1988-1998

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To supplement information on non-point nitrogen loads entering Long Island Sound from the Connecticut, Housatonic, Thames and Connecticut coastal basins, and to understand their annual variation, nitrogen loads from monitored basins were estimated using a log-linear regression model that related streamflow and time trends to constituent load at 28 water-quality/streamflow stations. Predictions were corrected for retransformation bias using the Minimum Variance Unbiased Estimator. Total nitrogen load estimates at monitoring stations were used along with ancillary data to develop another multiple-linear regression model that relates total nitrogen yield to land use characteristics, time, and point source discharges of nitrogen. This model was used to predict non-point nitrogen yields from monitored basins with point sources and from unmonitored basins in coastal Connecticut. Average non-point nitrogen yields from monitored and unmonitored basins ranged from 1,100 - 15,000 pounds per square mile per year. An analysis of serially correlated error was used to estimate confidence intervals on these yield estimates. Estimates of non-point nitrogen load from monitored and unmonitored basins were summed to determine the non-point nitrogen load from each of six Long Island Sound Management Zones for water years 1988-98. Annual estimated non-point nitrogen load from the study area ranged from 21 million pounds during 1995 to 50 million pounds in 1990.

The Frequency of Chimerism amonġ Colonies of *Hydractinia symbiolongicarpus*

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Hart & Grosberg (1999), using RAPD polymorphisms, have reported young colonies of *Hydractinia symbiolongicarpus* could fuse at a rate of up to 6.9%. This result is of unusual interest in that it implies that the dermersal larvae of *H. symbiolongicarpus* remain in kin groups at appreciable frequencies from the time of fertilization until settlement. The band sharing protocol used to generate this result is sensitive to allele frequency and to assumptions regarding the transmission genetics of fusibility. Both issues are incompletely known. A more direct approach is available. We have sampled two populations, Long Island Sound (Old Quarry Harbor, Guilford, CT) and Barnstable Harbor, MA, searching for hermit crab shells that bore two or more newly recruited *H. symbiolongicarpus* colonies. These colonies were explanted from the shells, reared in the laboratory, and subsequently tested for fusibility. The results of these fusion tests will be reported and compared with the results reported Hart and Grosberg (1999).

An Analysis of Epifaunal Communities Associated with *Phragmites australis* and *Spartina alterniflora* Stems

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The invasion of the common reed *Phragmites australis* has been of concern in northeastern US salt marshes dominated by the cordgrass *Spartina alterniflora*. The effects of the plant on nekton and vertebrate wildlife have been documented in the literature, but little is known about its impact on stem-dwelling meiofauna (e.g., nematodes and copepods) and macrofauna (e.g., annelids and amphipods). In this study, the epifaunal communities associated with *P. australis* and *S. alterniflora* stems have been analyzed for 3 years, at 3 study sites, to determine if significant numerical and/or compositional differences exist in these communities. Stems of the two plants were collected monthly or semi-monthly from salt marshes in northern NJ and Long Island, NY during the 2000, 2001, and 2002 growing seasons. Dead and live stems of both types were collected for the first two seasons. The data reveal significant differences in the abundance and composition of the epifaunal communities. At most samplings, *S. alterniflora* stems supported a higher density of animals than *P. australis* stems, with dead *S. alterniflora* stems being the most inhabited vegetation type. Most taxa (e.g., copepods, annelids, and insect larvae) were found differentially on *S. alterniflora* stems, and this grass also tended to support a more diverse assemblage than *P. australis*. The reasons for these differences are not known, and may range from differences in epiphytic communities to differential predation by grass shrimp. Because stem epifauna may serve as a link between microbial and macrofaunal trophic levels, these findings may have important implications in *P. australis*-invaded salt marshes.

A Cohort-Based Method for Estimating Net Transport of Larvae in Estuaries

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In estuaries, there is often retention or net movement of larvae in an up-estuary direction, against the mean flow. We are addressing how estuarine circulation and larval behaviors serve to transport larvae of the bay anchovy *Anchoa mitchilli*, which have been reported to display net up-estuary migration. Larvae exhibit a preference for deep water and rise into shallower water at night. Both behaviors should promote up-estuary movement. To test this prediction, we have developed an approach to quantifying population-wide rates of up-estuary distribution. As part of the 1998 Estuary Monitoring Program conducted for the Hudson River Utilities, anchovy larvae were sampled biweekly over the entire length of the estuary. Data on larval concentration, larval size distribution, and age-size relationships were combined to yield along-river distributions of cohorts of larvae sharing the same hatch dates. Cohorts of larvae show net movement upriver, for the first 35 days of life. Among larvae older than 35 days, cohort movement varies among dates. Immigration episodes cause apparent downriver shifts in cohort distribution among older larvae at times.

Phytoplankton Dynamics in Long Island Sound: Influence of Environmental Factors on Naturally-Occurring Assemblages

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Phytoplankton are the most important primary producers fueling coastal marine food webs. Human activities, such as eutrophication, pollution, habitat disruption, and selective removal of species by fishing, can significantly alter phytoplankton abundance and species composition, and subsequently change micro-grazer and larger consumer feeding, growth, and reproduction. In this study, we are examining spatial and temporal dynamics of phytoplankton assemblages (e.g., community structures, population abundance, species composition) in Long Island Sound (LIS). Water samples are being taken seasonally along an expected eutrophication gradient (from west to east) to examine which environmental factors (e.g., nutrients, temperature, grazer abundance) most influence the physiological condition and abundance of dominant phytoplankton species. Flow cytometry and immunological techniques are being used to probe the physiological states of individual phytoplankton cells, as well as to characterize plankton assemblages based upon cellular functions. A variety of fluorescent markers including Annexin V-*fluor* (detects cell viability), Nile red (detects lipid production), and ELF-97 (determines phosphate limited cells) are being tested. Preliminary data concerning how nutrients and other environmental factors influence the composition and cellular processes of phytoplankton assemblages will be discussed. Such information is vital for improving the management of LIS and developing strategies to protect and restore its living marine resources.

Sand Shrimp Predation on Juvenile Winter Flounder Across a Latitudinal Gradient

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From laboratory feeding experiments, the sand shrimp *Crangon septemspinosa* has been implicated as an important predator of post-settled winter flounder. Direct evidence supporting that this predator-prey interaction occurs in field populations, however, is lacking. The following investigation analyzed the stomach contents of sand shrimp collected over a four-month period (Apr-Jul) from three northwestern Atlantic estuaries: Narragansett Bay, RI, Niantic River, CT, and Sandy Hook Bay, NJ. The Öuchterlony immunoassay was used to detect the presence (or absence) of winter flounder in the diet of sand shrimp. This technique utilizes the highly specific recognition capabilities of antibodies to identify immunogenic moieties (flounder antigens) present in the stomach contents of predators. Results indicate that shrimp are possible sources of intense predator-induced mortality for juvenile flounder. There was a significant and positive linear relationship between the incidence of flounder in the stomachs of shrimp and the body size of the shrimp. The importance of shrimp in regulating flounder abundance in Sandy Hook Bay was minimal due to the absence of large shrimp (> 40 mm). Conversely, in Narragansett Bay and Niantic River predation rates were high as a result of the presence of large shrimp during peak winter flounder settlement periods - when fish are small and extremely vulnerable to predation. Thus, sand shrimp predation rates on juvenile winter flounder depend on the relative spatial and temporal overlap between the settlement of juvenile winter flounder and the movement patterns of large, primarily gravid female, shrimp that migrate into shallow estuarine habitats to reproduce and feed.

Sea Level Rise in Long Island Sound (LIS) Over the Last Millennium

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We studied cores from salt marsh islands in the mouth of the Connecticut River (Great Island) and the Housatonic River (Knells Island). Cores were sliced in 2 cm intervals, dated with ^{210}Pb, ^{137}Cs and ^{14}C, and benthic foraminifera were used as sea level indicators. The records go back 600 years (Great Island) and 1500 years (Knells Island). Both locations show evidence for enhanced fresh water discharge around 1900 and 1950 AD, well-documented wet periods in the climate history of Connecticut. The relative sea level rise (RSLR) curve from Knells Island shows little change between 500 and 1000 AD, then the rate of RSLR accelerates until ~1600 AD to ~2.5 mm/year. From 1600 to 1700 AD, the curve is flat, then the rate increases to ~1.7 mm/year, with an acceleration to ~3 mm/year in the last 100 years. The Great Island RSLR curve shows a rate of 1.7 mm/year from 1400 AD on, with a short slow-down at ~1700 AD, and a slightly faster rate of 2.3 mm/year in the last 300 years. These data are similar to other records from the Connecticut shore of LIS, with ~1 mm/year for the last 1000 years, and 2.5-3 mm/year for the last 200-300 years. Many curves show a slight decrease in rate of RSLR around 1500-1600, correlated with the coldest stretch of the Little Ice Age. The Knells Island core shows an acceleration at ~1000 AD, which may correlate with the onset of the Medieval Warm period. A pronounced, short slow-down in the rate of RSLR occurred around 600 AD in several cores. We tentatively correlate this episode with a coeval cold snap recorded in the GISP2 ice core record. The data thus suggest a direct link (no significant lag time) between climate change and rates of RSLR on the northeastern US seaboard.

Mercury in Connecticut and Long Island Sound Sediment

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Mercury is a common contaminant in sediments from the Sound and in marshes and wetlands in Connecticut. Contamination levels in most sediment cores range from 50 to 600 ppb Hg, with peak values dated at about 1960-1970. Modern sediment concentrations are lower by about 40% compared to these peak concentrations. Surface sediments of the Sound show an increase in Hg concentration from east to west, which we relate to additional sources (sewage treatment plants) and deposition of fine-grained material in the western section of the Sound. Marshes along the Housatonic River have much higher Hg concentrations (up to several ppm Hg) than elsewhere, and wetlands along the Still River have Hg concentrations in excess of 50 ppm Hg. The high Hg concentrations in the Still River Basin are related to the long period of hat-making in Danbury. Mercury profiles from a core near New Milford show up to 100 ppm Hg in deposits that probably formed around 1900 AD, the peak in hat-making activities in Danbury. We surmise that the Housatonic River sediments are a significant source of Hg for the Sound. The Housatonic River basin is prone to major floods and during these floods the Hg-bearing sediments are washed out and deposited in the Sound. We estimate that currently about 35% of the Hg in LIS surface sediments is derived from waste water treatment plants and that up to 25% of the Hg in the Western Sound comes from the Housatonic River basin.

* For more, see "Mercury contamination chronologies from Connecticut wetlands and Long Island Sound sediments", *J.C. Varekamp et al.* in: *Environmental Geology* 43:268-282.

Establishment of a Surface Elevation Table (SET) Site Array on the Tidelands of Long Island Sound: the Barn Island Sites, a Sound-wide Database on the Web, and a Call for Volunteers

Warren, R.S., Connecticut College; Nels Barrett, Natural Resources Conservation Service; C. Warren, Numbers to Knowledge and R. Rozsa, Connecticut Department of Environmental Protection

Large areas of tidal marsh have been lost in sub-estuaries of the western Sound over the past three decades. Grasses first appeared stressed, followed in a few years by large-scale die-out. Destabilized sediments are then eroded and marshland is converted to tide flat or open water. Air photos and ground observations suggest that similar changes may be occurring in some mid-Sound systems, and in the Eastern Sound, published information documents changes in high marsh vegetation consistent with increasing hydric conditions. On much larger scales, similar patterns are documented in Louisiana and the Chesapeake, where an imbalance between marsh elevation increase and sea level rise is an important causal factor. Information is limited and ambiguous on marsh elevation relative to LIS sea level rise; addressing this data gap, the LIS Fund is supporting establishment of 9 SET sites on the Barn Island marshes by Connecticut College's Center for Conservation Biology and Environmental Studies. These are the first in what is hoped will be a Sound-wide array of SET stations. A Web-based data base for this array is also being developed and will be available to all cooperating investigators; Version 1.0 will be presented. Technical and regulatory aspects of SET installation in Connecticut will be discussed, along with a call for volunteers to install and adopt sites in the central and western reaches of LIS.

POSTER PRESENTATIONS

MYSOUND - Still Monitoring Your Sound

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The MYSOUND network of monitoring stations, funded by the EPA as part of the EMPACT Program, has entered its third year of providing real-time water quality data to a broad user community of educators, scientists, students and recreational users of Long Island Sound. The original network of four stations (Eastern Sound Offshore, Lower Thames River, Bridgeport Harbor, and Western Sound) measured temperature, salinity and dissolved oxygen at the surface and near bottom of the water column. We have retrofitted two of these stations (Eastern Sound Offshore and Western Sound) with additional sensors to measure suspended material concentrations (SMC). Additional funding this year from EPA has enabled us to deploy a new buoy in Central Long Island Sound (CLIS) equipped with meteorological instruments and a non-directional wave module for determining wave height and period, in addition to the water quality parameters and SMC at surface and bottom. Unlike the previous monitoring stations, this buoy is using satellite technology to beam the data to a central server from which the data is retrieved, increasing reliability. The CLIS Buoy fills the gap that existed in the Central Sound between our already established Eastern Sound Offshore Buoy and the Western Long Island Sound Buoy. Concurrent SMC data from these three axial buoys is presented showing the response of these three buoys to Sound to wind and wave events. We also examine this past summer's dissolved oxygen record from the Western Long Island Sound data buoy, originally deployed to record the occurrence and duration of anoxic conditions that may have impacted local fisheries resources.

Surveying Salt Marsh Vegetation: A Comparison of the Braun-Blanquet and Point-Intercept Methods

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Salt marsh vegetation is often surveyed to investigate changes over time that occur after restoration efforts or environmental perturbations. Two methods used for salt marsh vegetation monitoring are the Braun-Blanquet cover scale method and the Point Intercept method as described in Monitoring Salt Marsh Vegetation (Roman *et al* 2001). Researchers may select one method over the other depending on such factors as time limitations, the need for accuracy, and simplicity. In this study, vegetation from nine fringing marshes along Casco Bay, Maine was studied and the two methods were compared. We examined how much time each method required in the field, and how often the results obtained using the Point Intercept method coincided with those obtained using the Braun-Blanquet cover scale method. Depending upon the purpose of your study, you may find our research helpful in determining the vegetation monitoring method that is right for you.

New Measurements of Oxygen Production and Consumption in Long Island Sound

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Local new primary production driven by inputs of anthropogenic nitrogen is assumed to be the major source of organic matter responsible for hypoxia of bottom waters in Western Long Island Sound (LIS). To date, various processes that characterize ecosystem responses to nutrient fluxes have been modeled (e.g. HydroQual 1996, 1999; Edwards et al. 2000). Simulated predictions of the rates of primary production and planktonic respiration have been neglected however, at least partly due to a dearth of direct measurements. Primary production and oxygen consumption were measured bi-weekly on four occasions in July and August 2002 at eight stations throughout middle and western LIS. Oxygen production of the surface water was measured across an artificial light gradient in order to characterize the physiological photosynthesis-irradiance function for the phytoplankton community over space and time. This will allow calculations of integrated water column productivity. In addition, the P-I curves will provide a basis for evaluating the photosynthesis formulation of ecological models. These field studies will continue through summer 2003. Ultimately, these data will be useful in the evaluation of LIS's response to recent decreases in nitrogen input as well as in comparison with predictions of new and existing bio-physical models.

Effect of Peat Structure on Creekbank Drainage in a Reference and a Tidally-Restored Marsh

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Tide gates have restricted saline waters from inundating many Northeast U.S. coastal marshes, encouraging replacement of *Spartina* spp. (*Spartina*) with *Phragmites australis* (*Phragmites*). As *Phragmites* replaces *Spartina*, peat derived from *Phragmites* accumulates above existent *Spartina* peat. Upon tidal restoration, however, *Spartina* replaces *Phragmites* and *Spartina* peat accumulates above *Phragmites* peat. Consequently, a *Spartina-Phragmites-Spartina* peat profile develops within the marsh subsurface. *Phragmites* peat is friable with large rhizomes and may have higher hydraulic conductivity than *Spartina* peat thereby permitting higher creekbank drainage on ebb tide. Replicate seepage meters were installed in three peat layers (*Spartina alterniflora-Phragmites-Spartina patens*) along the creekbank of a restored salt marsh and within the dominant peat layer (*Spartina alterniflora*) of a nearby reference marsh in Branford, CT. Meters were sampled on 12 separate tidal cycles in both marshes. Contrary to our hypothesis, drainage was significantly lower in *Phragmites* peat (43.0 $L \cdot m^{-2} \cdot 6h^{-1}$) relative to *S. alterniflora* (90.4 $L \cdot m^{-2} \cdot 6h^{-1}$) and *S. patens* (138.9 $L \cdot m^{-2} \cdot 6h^{-1}$) peat at the restored marsh. *Phragmites* peat exported 52.4% and 69.0% less porewater on ebb tide compared with *S. alterniflora* and *S. patens* peat, respectively. Drainage was significantly less in *S. alterniflora* peat at the reference marsh (48.4 $L \cdot m^{-2} \cdot 6h^{-1}$) compared with *S. alterniflora* peat at the restored marsh. Drainage per $1m^2$ of creekbank at the restored marsh (122.0 $L \cdot m^{-2} \cdot 6h^{-1}$) was significantly greater relative to the reference marsh (48.4$L \cdot m^{-2} \cdot 6h^{-1}$). It is possible that the interface between *Phragmites* and *S. patens* peat is a zone of high conductivity and therefore may be a preferential pathway for creekbank drainage and nutrient export.

Anthropogenic Heavy Metal Contamination in Tidal Marsh Sediments, Fletcher's Creek, Milford, CT

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Fletcher's Creek tidal marsh is part of Silver Sands State Park, a 47-acre recreational beach and salt marsh facility along Long Island Sound in Milford, CT. The area had been a dumping site for local inhabitants since the 1920's, used by the town of Milford as an unregulated landfill since the end of WWII and officially closed in 1977. Anecdotal information indicates that, in addition to regular household waste, hazardous materials including asbestos, lead paint, pesticides, oil, battery acid, freon, toluene, PCB's and radioactive medical waste were discarded at the site. During restoration of the Fletcher's Creek tidal marsh channel system in 1999, dredging, boardwalk construction and clearing of Phragmites uncovered bedded debris at the surface. The debris field was mapped, refuse samples taken to be dated, and water samples collected from tidal channels, pools and seeps in 2000, and tested spectrophotometrically for select heavy metals. The affected area was initially determined to be 3.72 square kilometers, lying 242m south of the fenced-off landfill, extending 60-100m beyond the mapped '0' limit of landfill waste, and now projected to underlie the area up to the current strandline. Bedded debris exposed in channels occurs to an average depth of 2 meters. Dated materials exposed on the surface cluster between 1964-1968. XRF analyses (winter 2001) of sediment from the debris field and within tidal channels indicate elevated concentrations of heavy metals (including chromium, copper, iron, lead, manganese, mercury, nickel, selenium and zinc) of probable anthropogenic origin. Levels exceed those of tested water samples and are consistently above pre-industrialized background.

Bird Species at Millstone Point from April 2001 through August 2002

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Bird species were noted during daily noon walks around Millstone Power Station, which includes forest, riparian, estuarine, seacoast, marsh, industrial, and grassland habitats. A record was kept of species identified each month. A total of 123 species was observed. The number of species observed per month was remarkably stable, with an average of 50 species and a range of 36 (perhaps spuriously low) to 59. Nineteen species were seen regularly year round. Twenty eight were seen regularly in the summer only, while twelve were seen regularly in the winter only. The data suggests that this area has a fairly stable number of niches with a partial seasonal turnover in type of niche from summer breeding opportunities to winter refuges.

Flushing Time of the Pawcatuck River Estuary

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The Wood-Pawcatuck River begins in central Rhode Island and drains into the Pawcatuck River estuary on the CT-RI border, eventually into Little Narragansett Bay, Westerly, RI. With many houses and marinas along the river, eutrophication and pollution are major concerns. Further, future development may reduce the fresh water input altering the estuarine habitat. Detailed CTD surveys were completed in the estuary, to delimit the present estuarine habitat, and to compute flushing. River flows from the USGS HCDN stations in Westerly and Wood River Junction, RI, were pro-rated to estimate the total inflow entering the estuary. Flows were near historical lows. CTD profiles were contoured, and salinity cross sections at low and high tide portray the extent of estuarine habitat for these river flows. Flushing time is more illusive. Dyer's Estuaries: A Physical Introduction (1973) describes several methods of determining flushing rate. Bathymetric area was calculated from a US Navigational Chart of Fisher's Island Sound (13214), at mean lower-low water. The flushing time is the time it takes for the incoming river-water to replace the existing fresh water in the system. Utilizing the Fractional Fresh Water Method, the Pawcatuck estuary has a flushing time of 2.3 days. The Tidal Prism Method includes only tidal exchange, assuming water entering at high tide will displace the volume of the river. Multiplying the surface area of the river by the tidal amplitude yielded a tidal exchange of 3.57 tidal cycles, or 1.8 days. While rapid, an exchange of 2-3 days is reasonable for shallow coastal estuaries.

Quantification of Volatile Reduced Sulfur Compounds in Connecticut Surface Waters Influenced by Long Island Sound

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Volatile reduced sulfur compounds (VRSCs), such as hydrogen sulfide (H_2S), carbonyl sulfide (COS), carbon disulfide (CS_2), methyl sulfide (MS), methyl disulfide (DMS), and dimethyl disulfide (DMDS), are important in the cycling of sulfur and in global climate regulation. These VSCs have been recognized as a significant source of atmospheric sulfur. In addition, organic sulfides can affect fate and bioavailability of trace metals by serving as strongly bound complexing agents. The ambient concentrations of most VRSCs are typically below conventional limits of detection in fully oxygenated surface waters, but can be quantified by methods that we have developed. In addition, they can accumulate in the hypolimnia of tidal ponds and deep water basins to much higher levels. We will present data from studies where we concentrate VRSCs on a chemical trap after purging samples taken at several sites along the mixing zone of two Connecticut river estuaries, the Saugatuck and the Norwalk. Separation, identification and quantification of these compounds can be realized using gas chromatography coupled with pulsed flame photometric detection (PFPD). In addition, we will present a vertical profile of VRSCs from Frash Pond, a tidal basin in Stratford, CT. Biological production has been identified as an important source of VSCs in the ocean and coastal marine environments. Therefore, we will also present chlorophyll a concentrations taken in situ at each sampling site in order to illustrate the relationship that may exist between biological productivity and VRSCs.

The Invisible World of the Seashore: an Interactive Guide to the Plankton Communities of Long Island Sound

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The Invisible World of the Seashore is a digital interactive guide to shallow water planktonic communities. This computer driven DVD program incorporates live video footage of organisms along with detailed biological descriptions. It will allow students to learn in a dynamic and interesting way about the ecological importance of these communities and to appreciate the beauty of these microscopic organisms. Students will be able to explore these bizarre invisible worlds and study life forms that would otherwise have remained completely hidden. This program will allow students to jump quickly in a non-linear fashion from phytoplankton to holoplankton to meroplankton. They will view the many truly bizarre but beautiful larval forms such as the pilidium, actinotroch, cyphonautes, bipinnaria, ophiopluteus, zoea, megalops, and many others. This program is also a valuable teaching resource providing instructions with instant access to a living reference collection of life forms that are extremely difficult to obtain.

Diel Variability in Gross Community Production Rates for Western Long Island Sound

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The bottom waters of Western Long Island Sound (WLIS) experience hypoxic conditions during summer months. This condition arises due to the formation of a strong pycnocline, which prevents atmospheric oxygen from reaching the deeper waters of WLIS. Introduction of excess nitrogen from human land usage also magnifies the hypoxia problem through eutrophication. A first time study was conducted during the summer of 2002 on the rates of gross community production and respiration throughout the water column in WLIS. The measurements of these rates will be used to create a new bio-geo-physical model of LIS. This model will provide a better understanding of how biology and physics affect hypoxic conditions in WLIS. All eight stations used in the study could not be sampled simultaneously; therefore they were sampled at various times of day. It was not known how physiological diel variation would affect the PI (photosynthesis vs. irradiance) curve throughout the day. I conducted an experiment to offer insight on if and how community production rates varied throughout the day. A 20 L seawater sample was collected from one station to do three sets of light/dark incubations and chlorophyll samples at various times of day. The seawater was incubated in the same manner as the aforementioned experiment using an artificial light gradient in a seawater bath. Titrations were then conducted at the Avery Point lab using a computer controlled titrator. PI curves for the sets of incubation were then compared. The experiment has been conducted three times thus far and the first two experiments show two contrasting patterns. The third set of incubations is currently being analyzed to confirm any effect/non-effect of diel variability.

Tidal Wetland Loss in Western Long Island Sound: a Reflection of Recent Sea Level Rise?

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In the late 1980's the CT DEP received reports of a "dying" salt marsh in western Long Island Sound (LIS) along the Five-mile River, Darien, Connecticut. The wetland was polyhaline low marsh wherein the normally tall (i.e., 1.8 m) and dense *Spartina alterniflora* was stunted, less than 0.3 m tall, sparse, and with praemose leaf tips. Today mud flat has replaced more than half the vegetated area of ca. 1970. Similar patterns of low marsh loss have since been documented in other western LIS, high tidal range (ca.2 m) systems. The largest single area of documented LIS tideland loss is in the mid-estuary position of the Quinnipiac River where over 80 ha of mesohaline marsh, dominated by *Typha angustifolia* and *Phragmites australis* (haplotype M), are converting to peat flat. Various causal factors that could drive such changes, ranging from chemical spills to goose eat-out, have been eliminated. Here we propose that a key factor contributing to these losses is an imbalance between marsh accretion and recent sea-level rise, which has increased hydroperiods beyond the limits tolerated by *S. alterniflora, T. angustifolia*, and *P. australis*. Characteristics of submerging LIS marshes, rates of marsh loss, and sea level trends over the past quarter century are also presented. An array of sediment erosion table stations, the first of a proposed Sound wide network, have been installed at Barn Island to assess elevation change and sediment accumulation rates.

Efficient and Effective Ecosystem Restoration Through an Inter-Municipal Partnership

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Challenges that accompany ecosystem restoration projects often include one or more of the following: assessing and prioritizing restoration needs, securing funding, utilizing the project as an educational tool, monitoring the success of the restoration project, and protecting the restored system. Only a few of these challenges are essential to the "construction" of a project, but all are critical to the ultimate success of a project. The Manhasset Bay Protection Committee's inter-municipal approach to restoration addresses all of these challenges and more as it gathers the necessary resources from member municipalities, their communities, and related community organizations. Future inter-municipal agreements throughout the Long Island Sound region may be the key to more successful habitat restoration projects.

Distribution of Sediment Metal Contamination in Bridgeport Harbor, Connecticut

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Bridgeport Harbor is one of the largest ports of entry in Connecticut and plays a pivotal role in the transportation infrastructure and future economic development of the state. Bridgeport Harbor has been historically characterized by industrial development and is urbanized throughout its watershed. Sources of metals to the harbor include the Pequonnock River, industrial discharges, municipal wastewater and atmospheric deposition. During June 2002, thirty two surface (0-6 cm) sediment samples were collected from thirteen stations (6 inner harbor, 5 outer harbor and 2 external to the harbor) to define a high-resolution spatial distribution of iron (Fe), copper (Cu), lead (Pb), zinc (Zn) and nickel (Ni) contamination. Sediment metal concentrations within the harbor varied widely and ranged from 31-340 mg/kg for Pb, 43 to 520 mg/kg for Zn, 7.9 to 180 mg/kg for Ni, 28 to 500 mg/kg for Cu and 0.71 to 9.6 % for Fe. Highest sediment metal concentrations were measured at the inner harbor stations, particularly in the lower reaches of the Yellow Mill Channel. Sediment metal contents were highest in the fine-grained, inner harbor sediment and were lowest in the coarse-grained sediment located outside the mouth of the harbor in Long Island Sound. Copper, Zn and Pb contents were highly correlated with both the iron and organic carbon content of the surface sediment (coefficients of determination > 0.73). Mean sediment metal concentrations of the inner harbor were 3.1 (Fe) to 5.8 (Cu) times their respective concentrations at stations located outside the mouth of the harbor.

Mumford Cove: A System in Rebound

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Mumford Cove, a small (0.5 km^2) Connecticut estuary, has a history of changing nutrient inputs and corresponding eutrophication over the last 57 years. Unlike most sites currently under investigation, the anthropogenic nutrient load to Mumford Cove has been greatly reduced, resulting in a shift from an algal dominated community to one dominated by a vascular plant, eelgrass (*Zostera marina*). From 1945 to 1987, a wastewater treatment facility discharged into the cove. At the time of the facility's removal, the cove supported a near monoculture of the green algae *Ulva lactuca* (320 g d.w. m^{-2}). Applied Science Associates monitored the cove during 1988 and 1989, reporting a return to low nutrient levels and a reduction in the biomass of *U. lactuca*. They also predicted the return of *Z. marina*, based on the observation of a few seedlings. The site was revisited in 1999. *Ruppia maritima*, a vascular plant, was present at 2 of the 4 originally monitored sites (18 and 97 g d.w. m^{-2}). *Z. marina* was present at a third site (158.73 g d.w. m^{-2}). Additional methods were used to document the extent of macrophytes, resulting in a map and further estimates of biomass. In 2002, vegetation in Mumford Cove was surveyed once again. In a survey of 21 stations, *Z. marina* was present at all but one station. Biomass ranged from 2 to 1037 g d.w. m^{-2} with an average of 330 g d.w. m^{-2}. At the original 4 stations surveyed in 1988, *Z. marina* was present at 3 of the stations (7, 116, and 210 g d.w. m^{-2}). Within 15 years after a reduction in anthropogenic nutrient loading, the physical characteristics of Mumford Cove allowed the system to recover from 40 years of disturbance to what we theorize was the initial trophic status of the bay.

A Ferry-Based Marine and Atmospheric Observing System for LIS

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A recent agreement between scientists at SUNY and the Port Jefferson Ferry Company along with a research grant from NY Sea Grant has resulted in the opportunity to collect measurements of several significant environmental parameters during the ferry's routine crossing. This tremendous opportunity provides the means to uniquely address a number of regional scientific, environmental and societal issues. These involve:

* the understanding of circulation and stratification of the LIS and their relationship to water quality and the development of hypoxia in the Sound,

* the development of a historical record of climate forcing for the LIS spanning nearly 100 years,

* the role each of the above may have played in the periodic poor-yield lobster years, of which 1999 was catastrophic to the local fishing community,

* determining the relative contributions to circulation and stratification variability in the LIS from synoptic weather events and low-frequency climate variability.

Moreover, it provides the project's partners the needed observational platform to:

* develop and validate a state-of-the-art coastal modeling and observation system for the LIS,

* contribute sorely needed observational resources to the NWS which has no means to obtain over-water observations of the LIS. The agreement and funding also provide for the installation of an educational system on the ferry which will display and describe the data being collected to the passengers and educate the audience on the pertinent environmental issues being addressed by the project. In addition, there will be a web site developed to provide the data so that other researchers and educators can take advantage of this tremendous opportunity.

Influence of Epiphytic Algal Coverage on Fish Predation Rates in Simulated Eelgrass Habitats

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We conducted laboratory experiments to determine how increased canopy structure due to epiphytic algal growth in eelgrass beds affected fish predation rates on invertebrates. The mummichog *Fundulus heteroclitus* was used as the predator and the amphipod *Gammarus mucronatus* was used as the prey species. Experiments were conducted in 38-l glass aquaria. Coverage groups consisted of no vegetative cover, 300 blades of eelgrass, 300 blades of eelgrass + low epiphytic biomass, 300 blades of eelgrass + high epiphytic biomass. Epiphytic cover of the blades was qualitatively determined by adapting the methods used for field assessment of percent coverage of eelgrass wasting disease. The epiphytic cover was quantified by taking a random sample of 20 blades from each coverage category and measuring the wet biomass of the epiphytic algae. The low epiphytic cover group had a biomass of 0.11 g/blade (+/- 0.02 g) Three hundred blades that had been sorted by epiphytic coverage category were placed together in clusters of 10 and evenly distributed in the sand at the bottom of the aquaria. The overall density of the blades was calculated to be 2307.7 m^{-2}. The predation rate was measured as the number of *G. mucronatus* consumed/fish/h, and was calculated as an indicator of foraging efficiency. The predation rate was found to be the highest in the no vegetative cover group, in which the amphipods had no refuge from the predators, and lowest in the high epiphytic cover group. Canopy structure had a highly significant effect on the predation rate.

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