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INTRODUCTION

In his Plenary Presentation to the 1992 Long Island Sound Research Conference Dr. Donald Squires began by stating that "Less is known about Long Island Sound than most other major estuaries in the United States because, comparatively, it has been the subject of less research than the others." He provided a brief history of research in the Sound and concluded that this apparent deficiency was due primarily to the absence of "a continuing stream of research funding which permitted the development of a program with long term goals". Beyond this he pointed to the need for systematic study and a continuing and well designed monitoring program which in time would allow assessment of both the current state of the Sound and the sense and magnitude of any slow gradual changes that might be occurring, such as those associated with global climatic change.

A review of the papers, abstracts and posters presented in these *Proceedings* suggests that the past twelve years of effort has resulted in some improvements in the research climate relative to that described by Dr. Squires. A continuing monitoring program directed at both water quality and the management of fisheries resources in the Sound is being conducted by the State of Connecticut Department of Environmental Protection with funding from the U.S. EPA. The developing data set from the regular shipboard surveys of this program is already finding use within several investigations such as the study of salt transport and mixing presented by Gay and O'Donnell. These observations are supplemented by higher frequency data provided by instrument arrays mounted on the public ferries crossing in the vicinity of Bridgeport and New London. This work, sponsored by New York and Connecticut Sea Grant Programs, complements both scientific studies of transport, meteorology and water quality and a variety of educational programs. Codiga describes the application of some of these data within his studies of residual circulation in the eastern Sound. Together with the developing observational network initiated with funding from EPA and currently supported by NOAA (see LISICOS.uconn.edu) and the variety of remote sensing platforms both airborne and satellite, these shipboard surveys promise to provide a valued and regular measure of the "pulse of the Sound". Such measurements directly support the increasing number of "cause and effect" studies in the Sound dealing with e.g. eutrophication, acute lobster mortality, the effects of existing and planned benthic disturbance resulting from cross-Sound cables and pipelines, harmful algal blooms, toxics, and salt marsh dynamics. Each of these subjects are treated to some extent in these *Proceedings*.

The range and diversity of study detailed in these *Proceedings* as well as the variety of agency involvement, both State and Federal, appears indicative of a growing public awareness of the value of Long Island Sound and the sensitivity of the ecosystem to the range of anthropogenic insults. The studies in general are more systematic than those found in the 1992 *Proceedings* and are intended to contribute to increased understanding of Sound dynamics as opposed to many of the earlier studies that simply used the Sound as a laboratory. This slight but significant paradigm shift has facilitated closer collaboration between the States of New York and Connecticut as well as the multiple groups of investigators interested in the Sound. Although the regular, continuing, source of research support discussed by Squires in 1992 remains to be established this commonality of interest certainly benefits efforts to develop such funding. It's likely that this subject will again be discussed in future *Proceedings*.

W.Frank Bohlen
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PAPERS

Soft-bottom benthic habitat mapping in LIS and the critical need to incorporate biota-based indicators of habitat quality

Raymond M. Valente
SeaRay Environmental, 902 Riverview Place, St. Marys, GA 31558
phone: (912) 729-1019; email: rvalente@tds.net

INTRODUCTION

Historically, benthic ecologists have sampled the seafloor blindly, using grabs or cores to extract relatively miniscule volumes of sediment across an area of interest. This has resulted in a limited ability to elucidate spatial patterns in seafloor physical characteristics and associated biological communities. Advances in ship-based acoustic survey techniques (principally side-scan sonar and single/multibeam bathymetry) now facilitate the routine production of high-resolution maps providing unprecedented insight on how seafloor topography and physical characteristics can vary at multiple spatial scales over large areas (Kenney et al. 2003). With the newfound ability to target benthic sampling more precisely and evaluate population distributions within a broader geographic or "benthoscape" context, benthic habitat mapping (BHM) has become an area of intense focus by resource agencies in the U.S. (e.g., NOAA, USGS) and countries worldwide.

This paper discusses some current issues related to BHM in general and critically evaluates BHM efforts in Long Island Sound over the past decade. The overall objective is to clarify the important distinction between *descriptive* mapping approaches, that mainly attempt to define or classify different habitat types based on observed substrate-community associations, and *interpretive* approaches that attempt to address the more subjective and elusive concept of benthic habitat quality. It is suggested that a more balanced combination of the two approaches is required going forward to address critical LIS management needs.

GENERAL BENTHIC HABITAT MAPPING ISSUES

The definition of BHM is highly subjective

It is clear from the worldwide literature that no universal definition of BHM exists. One reason is the remarkably wide variety of instrumentation and techniques that have evolved for characterizing the seafloor (Kenny et al., 2003; Solan et al., 2003). At a purely logistical level, therefore, BHM can encompass a disparate array of activities and technologies. The choice of techniques ideally should depend on each project's objectives, particularly with respect to the scale and distribution of seafloor features across the study area, the required resolution of the resulting maps, and the specific organism(s) of interest. Unfortunately, it may instead be dictated by investigator bias and/or the availability of funds and specialized equipment. To a large extent what constitutes the description of a benthic habitat is dictated by the resolution of the survey or sampling methods employed.

Substrate does not equal habitat

Given the ease with which data on seafloor bathymetry and texture can be collected rapidly over broad areas with modern geophysical techniques, the concept of benthic habitat is sometimes equated with bottom sediment or substrate type (Greene et al. 1995; Valentine and Schmuck 1995). This simplified line of thought is intuitive, because the habitat occupied by benthic organisms is the sedimentary environment (Figure 1A). However, the concept of habitat is most meaningful when defined in relation to the needs and preferences of specific organisms. Benthic habitat, therefore, is more than substrate: it is formed from the intersection of components that include substrate, species, and the species tolerances and preferences (Figure 1B).

By this reasoning, it is insufficient to describe a particular area of bottom as, for example, "soft mud habitat," but rather "soft mud that represents habitat for X or Y community of organisms". The latter recognizes that two areas of similar soft muddy sediment within a given region can represent distinct benthic habitats based on the presence of either X or Y community of infauna.

Habitat for whom?

The evolving consensus is that successful BHM requires an integration of geophysical and biological techniques, and many projects focus on characterizing the composition and distribution of benthic macroinvertebrate communities because of their importance in the structure and function of soft-bottom environments. Hence considerable effort is directed at examining substrate-community relationships across the "benthoscape" (Zajac et al. 2000) or devising *ad hoc* classification schemes to identify distinct "biotopes" (Sotheran et al. 1997; Brown et al. 2002; Frietas et al. 2003).

Such descriptive studies are valuable in elucidating whether certain habitats might be relatively rare or unique within a region and, chiefly on that basis, possess special significance or value. However, with the narrow focus on characterizing sediment-infauna associations, most BHM efforts have made only limited progress to date in directly addressing their ultimate objective: to identify benthic habitats most important or "essential" to commercially-valued species of bottom-dependent fish and shellfish.

Certain benthic habitats, even ones classified as degraded on the basis of benthic community structure, may well be of significant value to fish/shellfish due to the seasonal or longer-term abundance of one or more key infaunal prey species. For example, benthic habitat quality throughout much of western LIS has been classified as degraded based on long-term monitoring of infaunal communities during summer months, yet for many years this area also has supported an abundant lobster population and fishery. This example illustrates the daunting challenge that still remains for many BHM programs: to identify the often elusive set of factors that combine to make benthic habitats "essential" to specific fishery resources.

BENTHIC HABITAT MAPPING IN LIS

Resource managers in LIS currently have a pressing need for tools or approaches to address, for example, questions about the impact of cross-Sound cable and pipeline installation on benthic habitats and/or the effects on habitat quality of Sound-wide nutrient reduction efforts. The primary BHM effort in LIS over the past decade, undertaken by the U.S. Geological Survey and CT Department of Environmental Protection, has resulted in a series of laudable papers (Journal of Coastal Research vol. 16, 2000) and GIS-based data products (USGS 1998, 2000) that significantly advance the state of knowledge regarding the Sound's seafloor environments.
As part of these efforts, Zajac and others (1998; 2000) used historical datasets to describe and map associations between soft-bottom substrates and benthic assemblages at a variety of spatial scales (Figure 2).

While the USGS/CT-DEP efforts of the 1990's are extremely valuable in elucidating soft-bottom habitat complexity and variability, there are two drawbacks with respect to the present-day management needs: 1) the benthic community studies were based on historic Sound-wide datasets (i.e., >20 years old) that may not accurately reflect current conditions, and 2) there was little apparent integration of the various studies. Such integration would facilitate, for example, an evaluation of the spatial patterns in benthic communities relative to the strong west-to-east gradients in anthropogenic impacts that either were observed in these studies or already known to exist (i.e., impacts revealed through measurements of sediment chemical contaminants, organic carbon, and *Clostridium perfringens* spores; faunal shifts in benthic foraminifera; and recurring patterns of seasonal hypoxia).

USING THE INDEX APPROACH TO ASSESS BENTHIC HABITAT QUALITY

In addition to improved integration and assessment of existing datasets, there remains in LIS and elsewhere the need for wider acceptance and/or greater use of indicators that incorporate the concept of biotic integrity and, by extension, habitat *quality*. Karr (1981; 1991) was one of the first to advocate a "biota-based" approach, based on the principle that the integrity of resident benthic (or fish) communities serves as a better indicator of the health or quality of water or sediment habitats than physical/chemical measurements of contaminants, toxicity, nutrients, dissolved oxygen, etc. A somewhat bewildering array of benthic indices has since evolved, most of which are based on a common reductionist approach by which complex indicator datasets are distilled into one simple, easily understood numerical value or metric (Diaz et al. 2004).

The Benthic Index of Biotic Integrity (B-IBI) is one such index, in which multiple measures of benthic community structure and function (such as diversity, abundance of pollution-indicative or pollution-sensistive taxa, number of opportunistic taxa, etc.) are combined into a single number designed specifically to characterize the degree of anthropogenic disturbance to soft-bottom benthic habitats along a continuum from "highly disturbed" to "non-disturbed". Over the past 15 years, the B-IBI approach has been developed and applied widely in estuaries throughout the U.S. as part of several on-going, large-scale environmental monitoring and assessment programs (e.g., Weisberg et al. 1997; Engle and Summers 1999; Llanso et al. 2002a and b). While accounting for variations in benthic community structure due to natural habitat factors like salinity and sediment grain size, the B-IBI approach examines the full range of different habitat types and responses of infauna to known anthropogenic disturbances across the chosen region of interest (e.g., within a given watershed, biogeographic province, or major estuary like Chesapeake Bay or Long Island Sound). This is useful because it allows the results from any single location (e.g., a given embayment, cove, or pipeline corridor) to be interpreted within a broader geographic or "ecosystem" context.

SUGGESTED DIRECTIONS

The benthic index concept has its drawbacks, including the confusing proliferation of multiple indices in recent years and the reliance on a reductionist approach that oversimplifies complex ecosystems. From a purely pragmatic perspective, however, scientists and resource managers must get better at developing and "selling" metrics or indicators that are easy to understand and readily communicated to legislators and the public. This is essential to sustain funding for the monitoring programs that are the foundation of effective ecosystem management.

Considerable effort already has been spent developing and validating the B-IBI for the Virginian Province, within which LIS is located (Paul et al. 2001; Schimmel et al. 1999), but this index appears to have been overlooked or ignored in past benthic mapping efforts and on-going monitoring programs in LIS. As demonstrated in Chesapeake Bay, the B-IBI approach can be quite effective in providing managers with a uniform tool for evaluating site-specific impacts, identifying problem areas, and monitoring progress toward estuary-wide restoration goals (Llanso et al. 2003).

Although not a perfect solution, the Virginian Province B-IBI represents a practical, logical starting point for meeting the present-day needs of LIS managers. To be adopted for routine use in the Sound, this existing index may only require minor refinement or validation, which could be accomplished efficiently through GIS-based integration of the several large, existing LIS datasets of recent vintage (e.g., EMAP and National Coastal Assessment; USGS studies). Regardless of the particular index that might be selected and used, effective management requires not only descriptive maps of benthic habitat variety and distribution, but also interpretive maps of habitat quality.

REFERENCES

Brown, C.J., K.M. Cooper, W.J. Meadows, D.S. Limpenny, H.L. Rees, 2002. Small-scale mapping of sea-bed assemblages in the eastern English Channel using sidescan sonar and remote sampling techniques. Est. Coast. Shelf Sci. 54: 263-278

Diaz, R. J., M. Solan, and R. M. Valente. 2004. A review of approaches for classifying benthic habitats and evaluating habitat quality. J. Env. Man. 73:165-181.

Engle, V.D. and J.K. Summers. 1999. Refinement, validation and application of a benthic condition index for Northern Gulf of Mexico estuaries. Estuaries 22: 624-635.

Freitas, R., A.M. Rodrigues, V. Quintino. 2003. Benthic biotopes remote sensing using acoustics. J. Exp. Mar. Biol. Ecol. 285/286: 339-353.

Greene, H. G., M. M. Yoklavich, D. Sullivan and G. M. Cailliet. 1995. A geophysical approach to classifying marine benthic habitats: Monterey Bay as a model. pp. 15-30 In: Applications of Side-scan Sonar and Laser-line Systems in Fisheries Research: Workshop Proceedings. Special Publication No. 9, Alaska Dept. of Fish and Game, Juneau, AL.

Karr, J.R, 1981. Assessment of biotic integrity using fish communities. Fisheries 6:21-27.

Karr, J.R, 1991. Biological integrity: a long-neglected aspect of water resource management. Ecol. Appl. 1: 66-84.

Kenny, A.J., I. Cato, M. Desprez, G. Fader, R.T.E. Schüttenhelm, J. Side, 2003. An overview of seabed-mapping technologies in the context of marine habitat classification. ICES J. Mar. Sci. 60, 411-418.

Llansó, R.J., L.C. Scott, D.M. Dauer, J.L. Hyland, D.E. Russell, 2002a. An estuarine benthic index of biotic integrity for the Mid-Atlantic region of the United States. I. Classification of assemblages and habitat definition. Estuaries 25: 1219-1230.

Llansó, R.J., L.C. Scott, J.L. Hyland, D.M. Dauer, D.E. Russell, F.W. Kutz, 2002b. An estuarine benthic index of biotic integrity for the Mid-Atlantic region of the United States. II. Index development. Estuaries 25: 1231-1242.

Llansó, R. J., D. M. Dauer, J. H. Volstad and L.C. Scott. 2003. Application of the benthic index of biotic integrity to environmental monitoring in Chesapeake Bay. Env. Mon. Assess. 81(1-3):163-174.

Paul, J.F., K.J. Scott, D.E. Campbell, J.H. Gentile, C.S. Strobel, R.M. Valente, S.B. Weisberg, A,F. Holland, J.A. Ranasinghe, 2001. Developing and applying a benthic index of estuarine condition for the Virginian biogeographic province. Ecol. Indicators 1: 83-99.

Solan, M., J.D. Germano, D.C. Rhoads, C. Smith, E. Michaud, D. Parry, F. Wenzhöfer, B. Kennedy, C. Henriques, E. Battle, D. Carey, L. Iocco, R. Valente, J. Watson, R. Rosenberg, 2003. Towards a greater understanding of pattern, scale and process in marine benthic systems: a picture is worth a thousand worms. J. Exp. Mar. Biol. Ecol. 285/286: 313-338.

Sotheran, I.S., R.L. Foster-Smith, J. Davies, 1997. Mapping of marine benthic habitats using image processing techniques within a raster-based geographic information system. Est. Coast. Shelf Sci. 44: 25-31.

Schimmel, S. C., S. J. Benyi and C. J. Strobel. 1999. An assessment of the ecological condition of Long Island Sound, 1990-1993. Env. Mon. Assess. 56:27-49.

U. S. Geological Survey. 2000. Georeferenced sea-floor mapping and bottom photography in Long Island Sound. Open-file Report 00-3004 (CD-ROM), USGS-Woods Hole, MA.

U.S. Geological Survey. 1998. Long Island Sound Environmental Studies. Open-file Report 98-502 (CD-ROM), USGS-Woods Hole, MA.

Valentine, P. C. and E. A. Schmuck. 1995. Geological mapping of biological habitats on Georges Bank and Stellwagen Bank, Gulf of Maine. pp. 31-40 In: Applications of Side-scan Sonar and Laser-line Systems in Fisheries Research: Workshop Proceedings. Special Publication No. 9, Alaska Dept. of Fish and Game, Juneau, AL.

Weisberg, S.B., J.A. Ranasinghe, D.M. Dauer, L.C. Schaffner, R.J. Diaz, J.B. Frithsen, 1997. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. Estuaries 20, 149-158.

Zajac, R.N. 1998. Spatial and temporal characteristics of selected benthic communities in Long Island Sound and management implications. Final Report. Office of LIS Programs, State of Connecticut, Dept. of Environmental Protection, Hartford, CT.

Zajac, R. N., R.S. Lewis, L.J. Poppe, D.C. Twichell, J. Vozarik and M.L. DiGiacomo-Cohen. 2000. Relationships among sea-floor structure and benthic communities in Long Island Sound at regional or benthoscape scales. J. Coast. Res. 16:627-640.

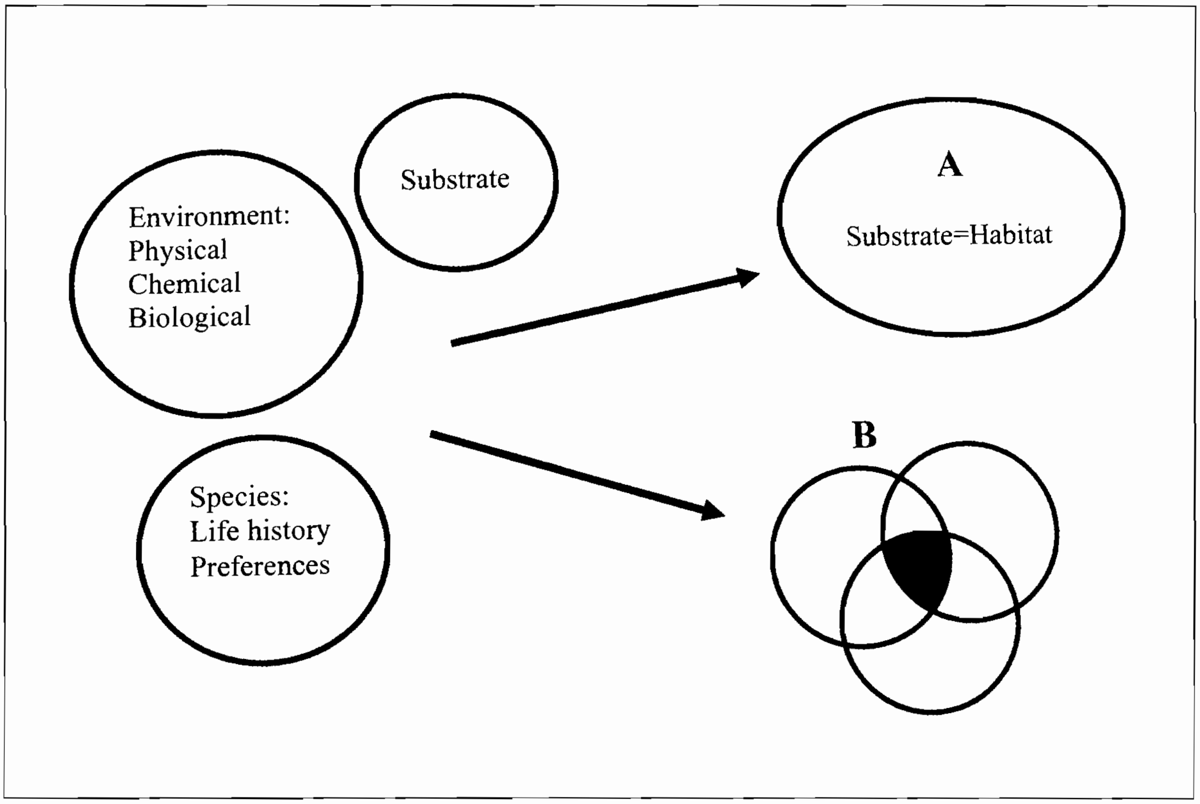

FIGURE 1. Diagram showing the relationship between physical substrate and benthic habitat. A) Simplistic view that equates substrate with habitat. B) Realistic view that combines environmental tolerances and preferences of species with substrate characteristics to define subsets of space that are habitat for the identified species (from Diaz et al. 2004).

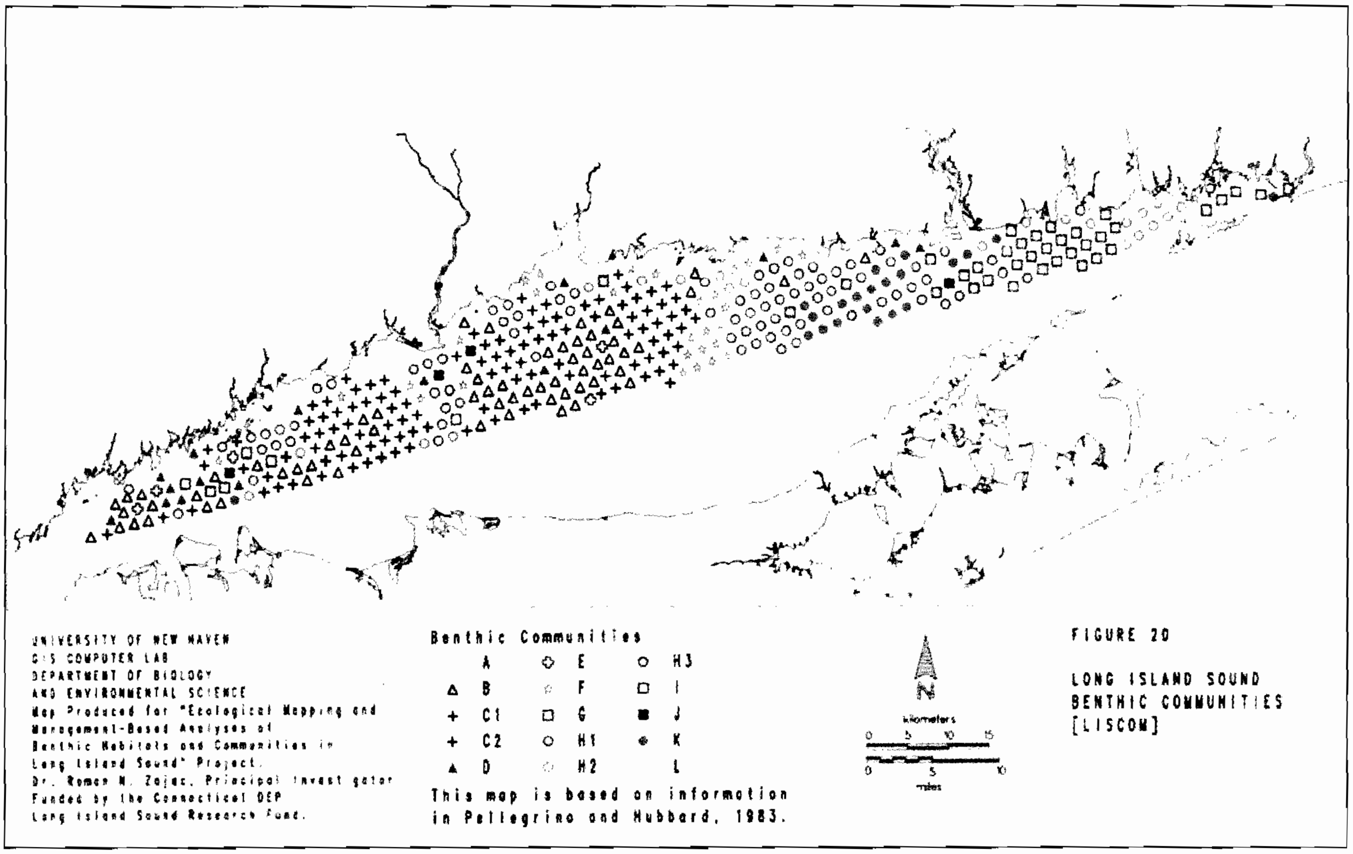

FIGURE 2. Descriptive map showing the spatial distribution of benthic communities in the northern half of LIS (based on sampling conducted in the late 1970s and early 1980s) and used to evaluate substrate-community relationships (Zajac 1998; Zajac et al. 2000).

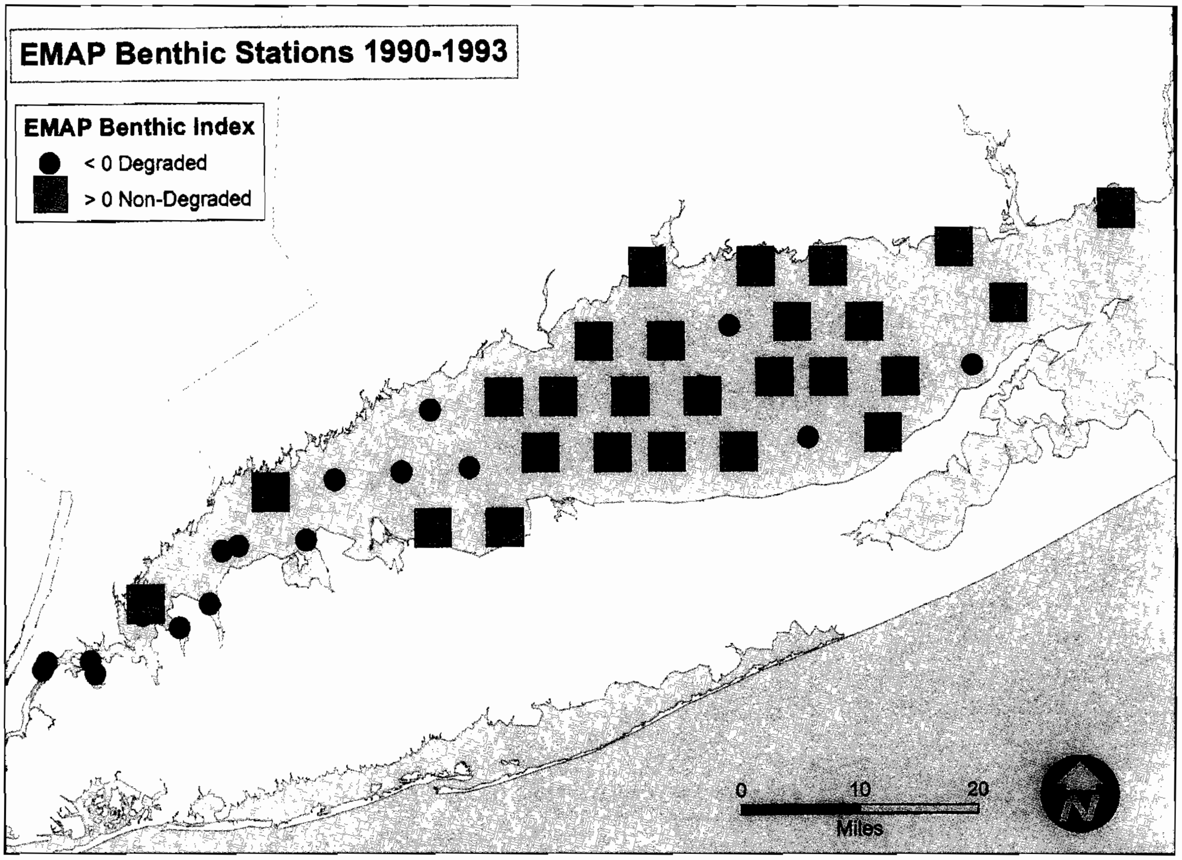

FIGURE 3. Interpretive map showing the presence of degraded versus non-degraded benthic habitat conditions, based on calculation of B-IBI values, at stations sampled from 1990 to 1993 by the U.S. EPA's Environmental Monitoring and Assessment Program (EMAP).

Effect of oxygen concentrations on fluxes of dissolved organic matter, nutrients, iron, and manganese over the sediment-water interface

Victor A. Arias[1], Annelie Skoog[1], Sergio Sanudo[2] and Aaron Beck[2]

[1] Department of Marine Sciences, University of Connecticut, Avery Point Campus, Groton, CT 06340
[2] Marine Science Research Center, Stony Brook University, Stony Brook, NY 11794-5000

INTRODUCTION

The release of nitrogen and phosphorus compounds to coastal waters is one of the principal causes of eutrophication. One of its consequences, anoxia in continental shelf sediments, is persistent during the summer in coastal regions characterized by high primary production and high organic matter delivery to sediments. Anoxic sediments have been detected in the western sections of Long Island Sound.

The cycling of organic matter in coastal regions, including dissolved organic mater (DOM) in marine sediments, plays an important role in organic carbon remineralization and preservation in sediments (Burdige *et al.*, 1992; Burdige and Zheng, 1998). Decomposition of organic matter in sediments makes the sediment-water interface an important site for nutrient regeneration, and acts as a source of continual supply of remineralized organic constituents to overlying water (Burdige and Homstead, 1994). The magnitude and relative importance of this nutrient source can be determined by direct measurements in short-term incubations of bottom water and underlying sediments (Aller, 1980a; Aller, 1980b). In the experiment reported here, three mesocosms were kept under oxic conditions, and an additional three mesocosms were allowed to reach anoxic conditions in order to study the effect of anoxia on sediment-water fluxes of compounds affected by sediment-surface redox-conditions.

METHODS

Organic-matter-rich sediment was collected at the south side of the New London Ledge Lighthouse at 41° 18' 35" N and 72° 04' 66" W. A total of 31 gallons were collected by scuba divers at a depth of 15 m. The top 2 cm of the sediment was collected using small shovels and the sediment was stored in acid-washed plastic bags during the trip back to shore.

A total of six mesocosms were prepared, each one is made of 45 cm long sections of 29.2 cm diameter Plexiglas tubing, and holds a volume of 30.2 L. The volume of sediment incubated in each chamber was estimated as 13.6 L. The chambers were equipped with 5-cm-long and 0.7-cm-diameter, Teflon-coated, magnetic stir bars. An electric motor connected to an adjustable power supply controller revolved the magnetic stir bars at 60 r.p.m. Each chamber was connected by Teflon tubing to a flexible polyethylene bag containing seawater, to compensate for the volume removed during sampling. An electrode to determine temperature and oxygen concentration was screwed into the lid closing each chamber. Each lid has a sampling probe with a septum, designed to avoid the exchange of oxygen with the atmosphere. Each lid has two adapters, where tubing to oxygenate the chamber was attached.

Three mesocosms were incubated under regulated, oxygenated conditions. Oxygen was diffused into the chambers through an FTP-Teflon tubing. Oxygen was supplied under pressure using industrial grade oxygen and a manometer at 25 Mpa. pH was monitored twice daily to make any needed adjustment to keep the pH of natural seawater. Another set of three mesocosms was incubated under unregulated oxygen conditions. Respiration of bacteria and benthic infauna was allowed to bring oxygen levels to zero and to a lower pH.

The time between sediment sampling and start of the experiment was the shortest possible to avoid sulfide formation in the sediment, and was usually less than 6 hours. The sediment was homogenized before addition to the mesocosms. Samples for bacterial counts and determinations of OM were collected before start of the experiment. Mesocosms were kept at 16°C and 60% humidity in an environmental chamber under dark conditions for 33 days.

Mesocosm-water samples were collected using 100-mL glass syringes for carbon, nitrogen, and inorganic nutrient concentration determinations. Polypropylene syringes were used for collection of iron samples. Samples were filtered through muffled 0.7-µm-pore-size, glass-fiber filters (GF/F), stored in 20-mL, glass, scintillation vials and frozen until analysis. Samples for iron were acidified and stored at 4°C in acid-washed polyethylene bottles. Sampling was carried out daily for one week and every other day for the following two weeks until levels of oxygen reached zero in the mesocosms without oxygen addition (Fig. 1a). Tubing for diffusing oxygen to the anoxic mesocosms was added at this time to return the mesocosm to oxic conditions. Sampling was carried out every other day until the oxygen concentration had reached 7 ppm.

One of the mesocosms supposed to become anoxic had a crack, and therefore never became completely anoxic. The results from this mesocosm are not included in the following discussion.

RESULTS AND DISCUSSION

Dissolved organic carbon (DOC), and dissolved organic nitrogen (DON) were released to the overlying water during anoxic conditions (Fig. 1). The relative DON increase was larger than the DOC increase, which agrees with a demonstrated preferential release of N relative to C during early decomposition of organic matter. Dissolved inorganic nitrogen (DIN) was released from sediment to overlying water during oxic conditions, mainly in the form of nitrate. This release was smaller than the very large release of ammonia released from sediment to overlying water during anoxic conditions (Fig 2).

Ironoxydroxides (FeOOH) form mixed precipitates with phosphate and DOM. FeOOH also has high surface activity, which makes it likely that DOM sorbs to the FeOOH surface. Reductive dissolution of the FeOOH phase would release coprecipitated phosphate as well as DOM. The DOC concentration variations in overlying water of the anoxic mesocosms matched the concentration variations of dissolved iron, and indicate that FeOOH dissolution released DOC and Fe^{2+}. The Fe^{2+} concentration decrease after day 13 in the anoxic mesocosms was probably caused by precipitation of a reduced iron mineral phase. It is likely that this reduced phase was an iron-phosphate mineral, such as vivianite. The precipitation of vivianite would also explain the delay (release of iron on day 7, release of phosphates on day 13, fig. 2) in the release of phosphates in the anoxic incubation.

Under anoxic conditions, Mn^{2+} was released earlier than Fe^{2+}, and stayed in solution for the remainder of the incubation (Fig. 3). This behavior can be explained by the higher electron activity or redox potential of the manganese reduction relative to the iron reduction, and by a slower oxidation rate of Mn (II) with respect to the Fe (II) oxidation rate.

In the oxic mesocosms, the manganese concentration variation was quite different from that of iron. Mn^{2+} precipitated under oxic conditions, probably as MnO_2. Iron kept a steady concentration during the incubation under oxic conditions.

The major, commonly accepted, electron acceptors for biogenic decomposition of organic matter in marine sediments are listed in Table I. A sequential occurrence is expected as a function of the redox potential and the different microbiological successions. The products of decomposition of organic matter build up in the sediment and are released with different magnitudes to the overlying water.

In the upper few oxic centimeters of the sediment, the major decomposition pathway is aerobic respiration. In the oxic chambers, this process was identified by nitrate release and nitrification of ammonia released from the sediment (Fig. 2).

Under anoxic conditions, product release rates increased by several orders of magnitude (Fig. 2) as evidenced by the large ammonia release from sediment to overlying water (Fig. 2) during anoxic conditions. In addition, nitrate reduction and denitrification can be inferred from the decrease in nitrate and nitrite concentrations once oxygen is depleted (Fig 2). Reduction of iron and manganese oxides is responsible for the release of Fe^{2+} and Mn^{2+} under anoxic conditions. Mn reduction occurs first and is followed by iron reduction and a subsequent release of phosphates. Since we did not determine sulfide concentrations sulfate reduction could not be identified, but must certainly have occurred.

Table I. Sequence of Progressive Reduction

Aerobic respiration	$\{CH_2O\} + O_2 = CO_2 + H_2O + HNO_3 + H_3PO_4$	$\Delta G° = -29.9$
Denitrification	$\{CH_2O\} + NO_3 + H^+ = CO_2 + N_2 + H_2O$	$\Delta G° = -28.4$
Nitrate reduction	$\{CH_2O\} + NO_3 + H^+ = CO_2 + NH_4^+ + H_2O$	$\Delta G° = -19.6$
Manganese reduction	$\{CH_2O\} + MnO_2(s) + H^+ = CO_2 + Mn^{2+} + H_2O$	$\Delta G° = -23.9$
Iron reduction	$\{CH_2O\} + FeOOH(s) + 2H^+ = CO_2 + Fe^{2+} + H_2O$	$\Delta G° = - 9.9$
Fermentation	$\{CH_2O\} + H_2O = CO_2 + CH_3OH$	$\Delta G° = - 6.4$
Sulfate reduction	$\{CH_2O\} + SO_4^{2-} + H^+ = CO_2 + HS^- + H_2O$	$\Delta G° = - 5.9$
Methane fermentation	$\{CH_2O\} = CO_2 + CH_4$	$\Delta G° = - 5.6$

CONCLUSION

Increases in the concentrations of a sequence of redox-reaction products were observed in the mesocosms. Anoxic conditions at the sediment surface caused release of inorganic nutrients, DOM, iron, and manganese to the overlying water. There appeared to be a preferential release of N-containing compounds, such as ammonia and DON, from anoxic sediment to overlying water. Many coastal areas are N-limited, and our data show that anoxic conditions may increase the importance of sediment as an N-source. Anoxia in coastal waters is often caused by eutrophication and may be further exasperated by N-release from anoxic sediments, possibly indicating a positive feedback loop. This is particularly important in areas, such as Long Island Sound, where regions of seasonal anoxia have been found.

REFERENCES

Aller R. C. (1980). "Diagenetic processes near the sediment-water interface of Long Island Sound II; Fe and Mn." Advances in Geophysics 22: 351-415.

Aller, R. C. (1980). "Diagenetic processes near the sediment-water interface of Long Island Sound I; Geochemistry - Decomposition and Nutrient Element Geochemistry (S, N, P)." Advances in Geophysics 22: 237-349.

Burdige, D. J., M. J. Alperin, et al. (1992). "The role of benthic fluxes of dissolved organic carbon in oceanic and sedimentary carbon cycling." Geophysical Research Letters 19(18): 1851-1854.

Burdige, D. J. and J. Homestead (1994). "Fluxes of dissolved organic carbon from Chesapeake Bay sediments." Geochimica et Cosmochimica Acta 58(16): 3407-3424

Burdige, D. J. and S. L. Zheng (1998). "The biogeochemical cycling of dissolved organic nitrogen in estuarine sediments." LIMNOLOGY AND OCEANOGRAPHY 43(8): 1796-1813.

Hedges, J. I. and R. G. Keil (1995) "Sedimentary organic matter preservation: an assessment and speculative synthesis." Marine Chemistry 49:81-115.

Henrichs, S. M. (1995). "Sedimentary organic matter preservation: an assessment and speculative synthesis - a comment." Marine Chemistry 49: 127-136.

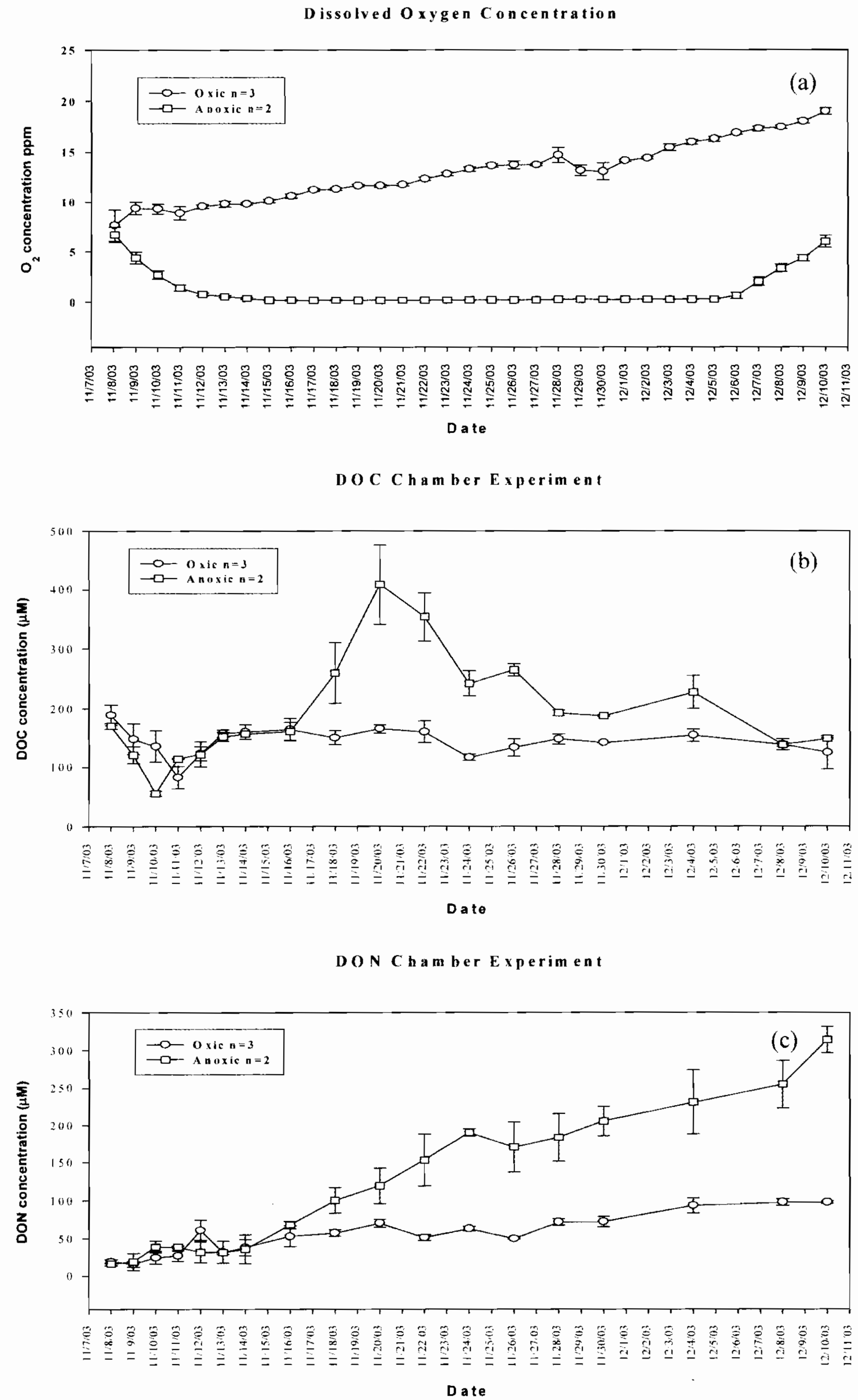

Fig.1. Average of dissolved oxygen concentration, dissolved organic carbon (DOC), and dissolved organic nitrogen (DON) for oxic and anoxic chambers.

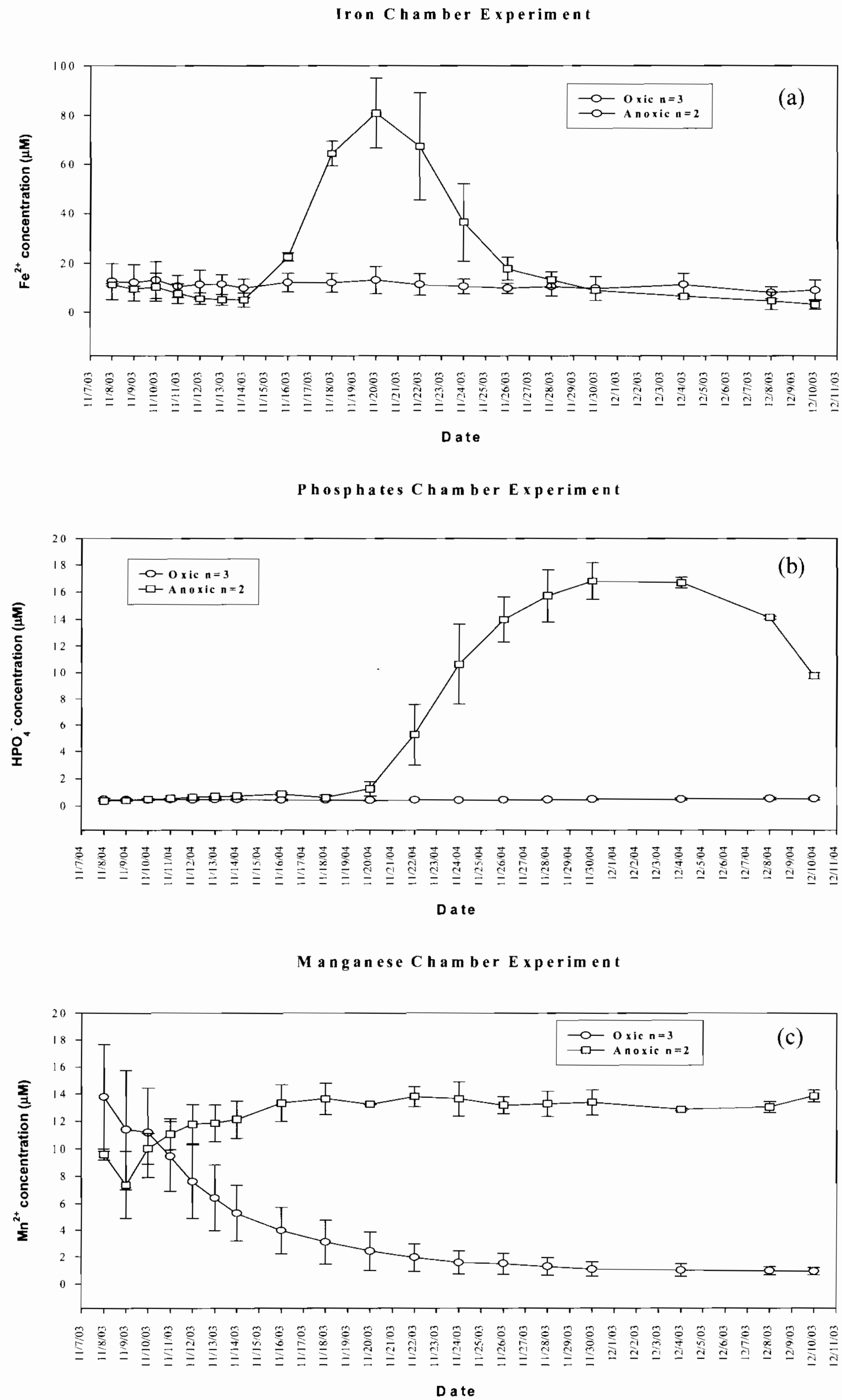

Fig.2. Average of dissolved iron (Fe^{2+}) concentration, phosphates concentration, and dissolved manganese (Mn^{2+}) concentration for oxic and anoxic chambers.

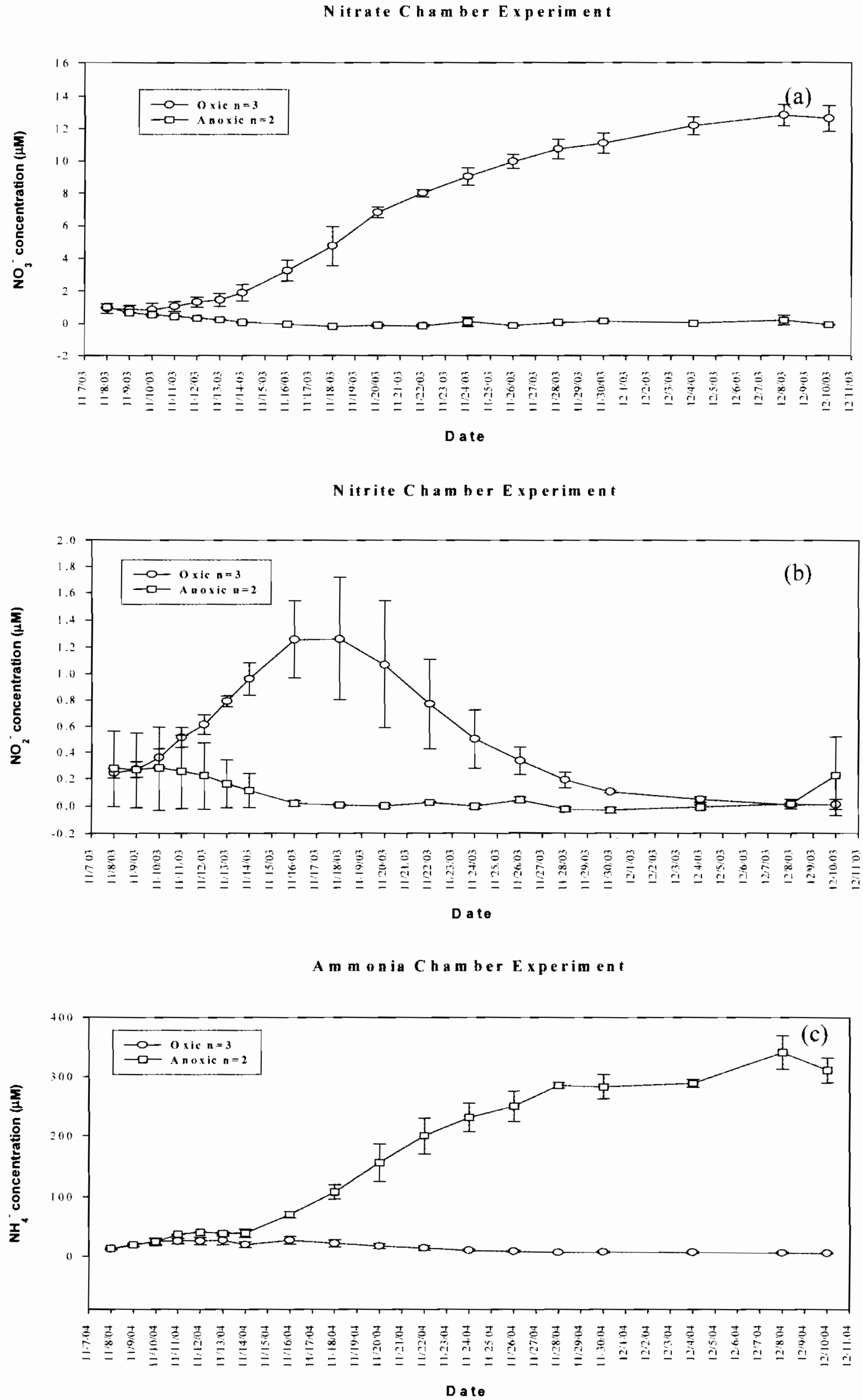

Fig.3. Average of nitrate concentration, nitrite concentration, and ammonia concentration for oxic and anoxic chambers.

Observations on the Effects of Recurring Deposition and Resuspension Cycles and the Stability of Bottom Cohesive Sediments

W. Frank Bohlen, M.M. Howard-Strobel and D.R. Cohen
University of Connecticut. Department of Marine Sciences.Avery Point.Groton, Connecticut 06340

Bohlen@UCONN.edu

ABSTRACT

Field measurements of suspended sediment concentrations, current and wave energy, and the physical characteristics of bottom sediments within nearshore areas adjoining Branford, Connecticut have shown that despite frequent resuspension and deposition due to aperiodic high energy events the bottom sediments can be characterized as relatively "tough" or stable with low water content. Similar behavior has previously been shown to occur in laboratory studies (Sills, et.al., 2003) where the density of bottom sediments progressively increased following a series of resuspension/deposition cycles. Comparisons with data obtained in a lower energy setting with similar sediment grain size characteristics indicate higher water content in the lower energy regime. We propose that sediment stability or bottom fabric is ultimately determined by the frequency of resuspension/ deposition characteristic of a given area. The observations on which this is based imply that relative water content may serve as a proxy for the assessment of the stability of cohesive sediment deposits. In addition, the data indicate that any significant disturbance of the bottom such as mechanical cut and fill operations will be unstable until a sufficient number of resuspension/deposition cycles have occurred to reestablish the fabric of the sediment column.

INTRODUCTION

The character and stability of the sediment column in coastal and estuarine waters is governed by a variety of factors including composition, grain size, infaunal biological activity, degree of consolidation, and stress history. The combination ultimately determines the quality of the deposit as a biological habitat and the role of the bed as a sink or a source within the regional sediment transport regime. Despite this importance, assessments of the factors governing sediment column formation and fabric have been limited by the paucity of field data providing concurrent measurements of interfacial dynamics and the associated deposit response. Such studies require time-series observations of flow and sedimentation characteristics over a time sufficient to provide a number of resuspension/deposition cycles due to both average ambient and aperiodic high energy transport events and to allow measurable deposition/erosion to occur. In temperate latitudes displaying evident seasonal cycles in such factors as water temperature and streamflows satisfaction of these criteria typically require a minimum of twelve months of observations obtained at a frequency greater than 2 observations per hour.

In the fall of 2001 a field investigation of the sediment transport regime along a portion of the nearshore adjoining Branford, Connecticut (Fig.1) was initiated. As part of this study two instrumentation arrays were deployed to monitor wind wave and tidal current characteristics and the associated suspended material concentrations. The implications of the data obtained by this investigation relative to the fabric of the local sediment column are the subject of this paper.

METHODS AND PROCEDURES

The analysis of the sediment transport regime in the area to the west of the Thimble Islands (Fig.1) employed a combination of moored instrumentation, to provide time series observations of hydrography and concurrent suspended material concentrations, shipboard observations, to provide increased spatial coverage, and laboratory analyses of recovered water and sediment samples. Two instrument arrays were deployed within the Connecticut nearshore in September, 2001 (Fig. 1). Each array contained a suite of instrumentation to detail , water temperature and conductivity and suspended material concentrations(using an optical backscattering probe with output calibrated using sediment obtained from the study area) at two points on the vertical, near surface and near bottom. In addition, each array contained a near bottom current meter, and a sensor to monitor tidal height and, higher frequency, surface wave conditions. The majority of the sensors were programmed to be burst sampled four (4) times each hour. The wave gage was sampled once every three (3) hours. All data were internally recorded for downloading during array servicing. The arrays were maintained on station for a period of thirteen months.

The time series observations were supplemented by aperiodic shipboard surveys of water temperature, conductivity and suspended material concentrations at a network of stations (Fig.1). Nine surveys of these stations were conducted during the course of the thirteen month study. On four (4) of these surveys core samples of the upper 1 foot (30cm) of the sediment column were obtained by divers. The variety of samples obtained during these surveys were returned to the laboratory for analysis.

The sediment core samples were extruded and sectioned to provide cm resolution over the vertical. A portion of each section was used in the determination of water content, bulk density, and weight loss on ignition. On two occasions (February, 2002 and September, 2002) the remainder of each section was retained for selected radionuclide analysis. Percent water content was estimated by wet weighing, drying at 70°C for 24-48 hr and re-weighing. The dried samples were combusted at 550°C for 0.5hr and re-weighed to determine weight loss on ignition, a measure of the percentage of combustible organics in each sediment sample. This combination of data was used to calculate bulk density for each segment.

By-weight suspended material concentrations in each sample were determined by vacuum filtration through dried and pre-weighed Nuclepore filters (47 mm dia - 0.45: pore size) mounted in standard Millipore apparatus. The laboratory data provided a continuing check on the performance of the moored instruments and a basis for re-calibration, if necessary.

RESULTS AND CONCLUSIONS

The time series observations of hydrography and suspended material concentrations indicate that sediment transport in the study area is dominated by the surface wave field which serves to aperiodically resuspend deposited materials increasing the mass of sediment in suspension to concentrations in excess of 300 mg/l at the inshore nearbottom (Fig.2). In the absence of wave induced stress, concentrations fall to less than 10 mg/l with time-series data providing no indication of a sensitivity to the ambient tidal flows. The observed response indicates that the primary role of the tidal currents in the area is to transport wave resuspended materials.

Despite the frequency and magnitude of wave induced resuspension in the study area the radionuclide data indicate long term average deposition rates ranging from 2.5 to 7.5 mm/yr. These values equal or exceed those observed in lower energy wetlands and inshore waters in the area (Nydick, et.al., 1995). Analyses of the physical characteristics of sediments found within the upper 20cm of the sediment column (Table 1) indicate that the materials are primarily fine grained silts containing a variety of organic detritus and shell hash and minimal sands. Samples obtained by divers showed a high degree of consolidation and displayed no tendency to flow in the absence of lateral constraints. Laboratory analyses indicated water contents ranging from a low of approximately 37% to a high of 67% (Table 1). Associated bulk densities ranged from near-surface values (0-1cm) of 1.23 gm/cc to 1.59 gm/cc at depth (14-17cm). Analyses indicated that these values displayed only minor seasonal variability despite active bioturbation.

DISCUSSION

The fact that long term deposition rates in the study area are relatively high despite active recycling of materials along the sediment-water interface appears to be the result of consolidation and binding of the fine grained materials under the effects of high shear stresses induced by the energetic wave field. The combination favors an increase in sediment column density and subsequent resistance to erosion resulting in relatively high stability despite the fine grained nature of the deposit. Such increases in density following an erosion/deposition cycle have previously been observed in the laboratory (Fig. 3). If the mass of laboratory materials shown this figure was allowed to accrete at some finite rate, exposure to a continuing cycle of erosion/deposition might be expected to produce a column of sediment with bulk density values ranging from approximately 1.225 gm/cc at the surface to 1.325 gm/cc at depth. Overburden would further increase the densities at depth yielding a distribution that very closely approximates that observed in the study area.

In addition to affecting bulk density, cyclic erosion and deposition also favors a decrease in sediment column water content. When compared to cores of similar sediments obtained in a relatively low energy bayou in Louisiana (Bohlen, et.al.,1999) the materials found in the study area display evidently lower water content. Values in the bayou range from approximately 65% to 76% and sediments flow freely when unsupported. The water contents of the sediments within the Connecticut nearshore are significantly lower with values approaching those found in sands or deep within consolidated fine grained sediment deposits. The occurrence of the low water content values within less than 5cm of the sediment water interface is indicative of a rapid rate of consolidation induced by the cyclic working of sediments by the wind wave associated stress field. There appears to be no other explanation for these characteristics. Biologically mediated factors produced by infaunal burrowing might contribute some degree of binding and stabilization through the production of mucal polysaccharides but such activity could not be expected to lead to reductions in water content. In fact, burrowing might rather increase water content through irrigation.

The observed characteristics of the sediment column produced in the presence of an energetic wave field suggest that mechanical cut and fill operations, such as construction related excavation and subsequent infilling, conducted in such areas may find it difficult to re-establish pre-project condition s if filling makes use of the same sediment type. Simple one-time placement of fine-grained sediments as fill will tend to produce a deposit which is subject to mass erosion when exposed to aperiodic high levels of shear stress. It will be only after several of these cycles that the deposit will experience consolidation sufficient to increase bulk density and the associated erosion resistance. Some of this mass erosion may be avoided by sequential placement of fill over a period of time rather than as a one-time filling operation. The length of this process needed to insure stability might be specified analytically using the combination of expected wave climate and sediment characteristics.

REFERENCES

Bohlen,W. F., Cohen, D. R. and M. M. Howard-Strobel 1999 An investigation of the sediment transport regime within Olsen's Bayou-Calcasieu River - Southwestern Louisiana. Rpt. prepared for: Browning-Ferris Industries and Conoco Inc. University of Connecticut. Department of Marine Sciences. Groton, Ct. 25pps + Tables and Figs.

Nydick, K. R., Bidwell, A. B., Thomas, E., and J. C. Varekamp 1995 A sea-level rise curve from Guilford, Connecticut USA. Mar. Geology 124:137-159

Sills, G., Lintern, G., and L. Jenice 2003 Effects of erosion on sediment beds. Program Abstracts. Intercoh 2003 Oct. 1-4, 2003 Virginia Institute of Marine Sciences. Gloucester Point, Va. pp.57.

Table 1. Sediment Core Analyses - October, 2001
Branford, Connecticut Study Area

Islander East Cores October, 2001 Physical Analysis

Sample	Sed Wet	Sed Dry	% Water	Sed Post Comb	% Loss	Bulk Density
Core 1						
0 -1 cm	24.40	10.26	57.95	9.70	2.30	1.32
1-2 cm	25.01	11.85	52.62	11.27	2.32	1.38
2-3 cm	24.40	11.13	54.39	10.55	2.38	1.36
3-4 cm	24.92	11.26	54.82	10.65	2.45	1.36
4-5 cm	24.72	11.54	53.32	10.92	2.51	1.37
5-7 cm	25.04	12.10	51.68	11.50	2.40	1.39
7-9 cm	27.52	14.94	45.71	14.34	2.18	1.47
9-11 cm	28.48	16.95	40.48	16.40	1.93	1.55
11-14 cm	29.05	17.99	38.07	17.43	1.93	1.59
14-17 cm	29.17	18.17	37.71	17.58	2.02	1.59
Core 2						
0 -1 cm	21.75	7.24	66.71	6.72	2.39	1.23
1-2 cm	25.13	11.98	52.33	11.38	2.39	1.39
2-3 cm	25.81	12.08	53.20	11.47	2.36	1.38
3-4 cm	27.83	16.12	42.08	15.57	1.98	1.53
4-5 cm	27.46	16.79	38.86	16.24	2.00	1.57
5-7 cm	27.03	16.28	39.77	15.71	2.11	1.56
7-9 cm	27.51	16.60	39.66	16.04	2.04	1.56
9-11 cm	25.02	11.91	52.40	11.29	2.48	1.38
11-14 cm	25.10	11.34	54.82	10.70	2.55	1.36
14-17 cm	26.25	13.70	47.81	13.09	2.32	1.44
17-20 cm	26.09	14.86	43.04	14.26	2.30	1.51
Core 3						
0 -1 cm	23.27	8.06	65.36	7.58	2.06	1.25
1-2 cm	21.91	7.23	67.00	6.80	1.96	1.24
2-3 cm	24.95	11.95	52.10	11.47	1.92	1.39
3-4 cm	27.11	14.63	46.03	14.19	1.62	1.48
4-5 cm	25.30	11.95	52.77	11.44	2.02	1.39
5-7 cm	26.30	13.42	48.97	12.92	1.90	1.43
7-9 cm	28.19	16.42	41.75	15.98	1.56	1.54
9-11 cm	25.20	12.14	51.83	11.61	2.10	1.40
11-14 cm	25.73	12.93	49.75	12.37	2.18	1.42
14-17 cm	23.63	10.18	56.92	9.63	2.33	1.33
17-20 cm	24.67	11.51	53.34	10.89	2.51	1.37
Core 4						
0 -1 cm	24.48	10.09	58.78	9.57	2.12	1.32
1-2 cm	24.45	10.52	56.97	9.97	2.25	1.33
2-3 cm	24.29	10.34	57.43	9.80	2.22	1.33
3-4 cm	26.04	12.28	52.84	11.68	2.30	1.38
4-5 cm	25.28	13.23	47.67	12.64	2.33	1.44
5-7 cm	25.24	13.14	47.94	12.55	2.34	1.44
7-9 cm	26.47	14.40	45.60	13.81	2.23	1.47
9-11 cm	24.54	12.52	48.98	11.95	2.32	1.43
11-14 cm	25.19	12.23	51.45	11.64	2.34	1.40
14-17 cm	25.85	12.96	49.86	12.32	2.48	1.42
17-20 cm	24.71	11.92	51.76	11.37	2.23	1.40

Islander East cores October, 2001.
All weights in grams.
% Water = ((sed wet - sed dry) / sed wet) * 100
% Loss: % Loss on ignition = (100 - %water) * (sed dry - sed post combustion) / sed dry
Bulk Density (g/cc) = 260 / 100 + 1.6 (% water + % loss on ignition)

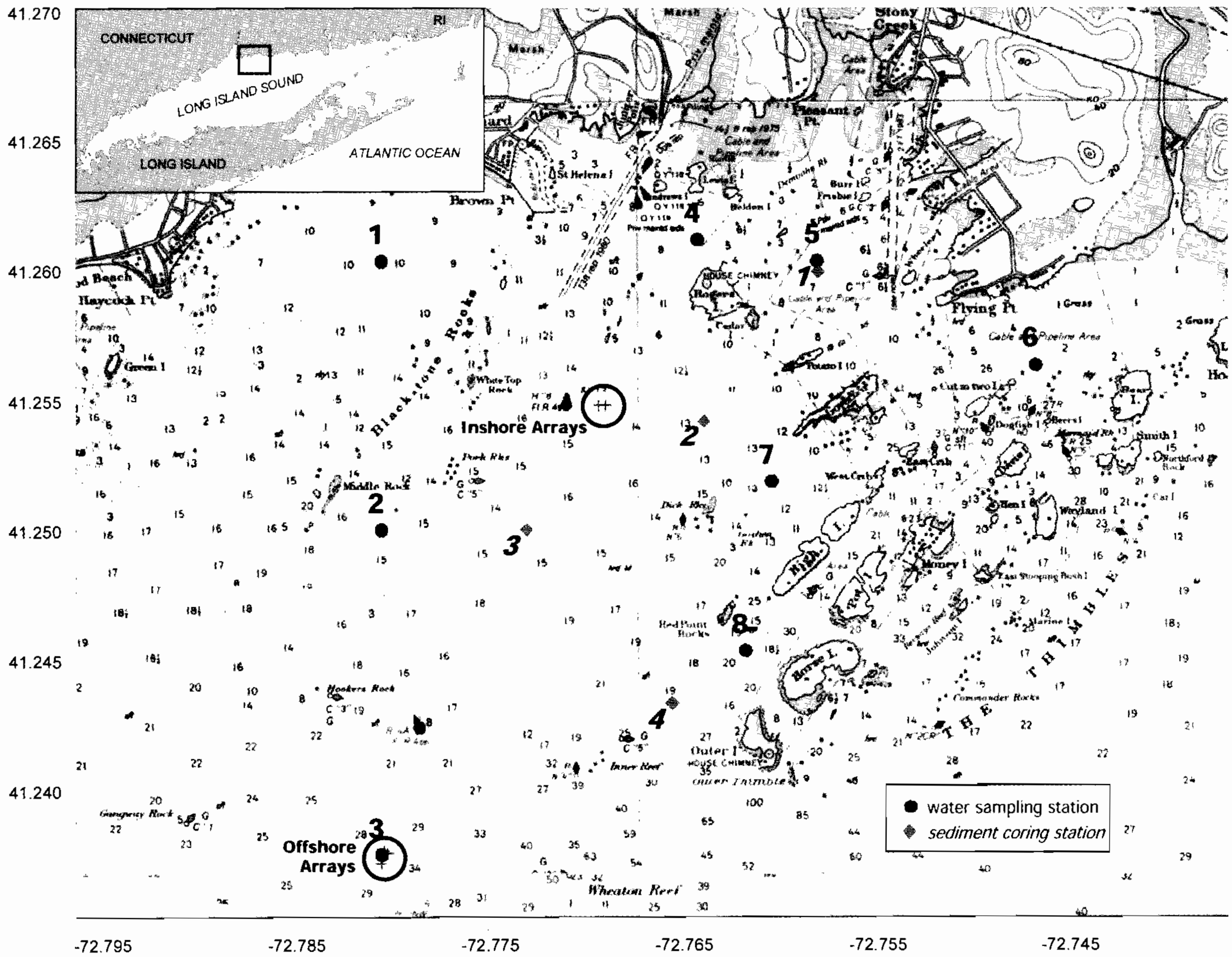

Figure 1. Study Area

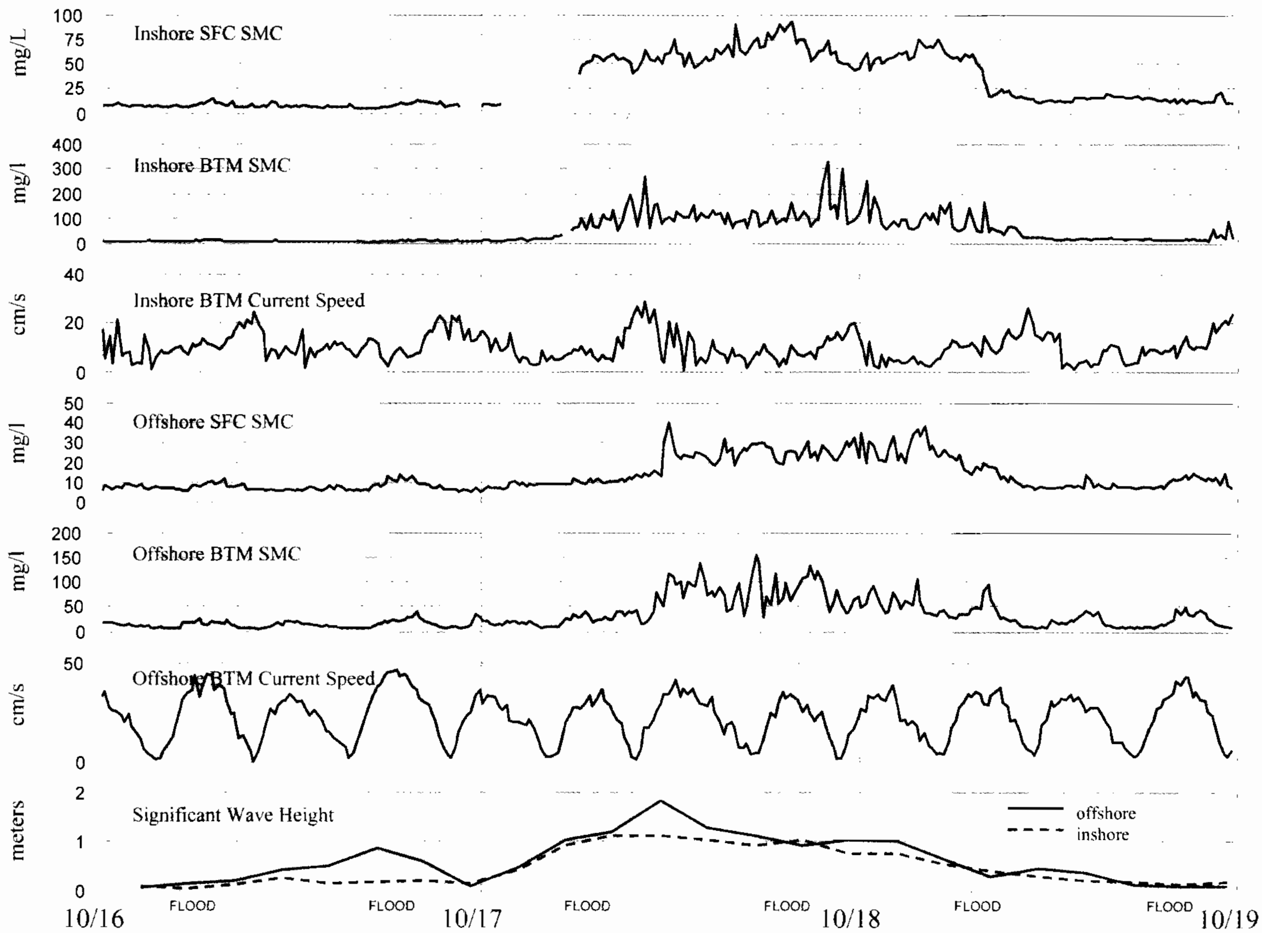

Figure 2. The Relationship(s) Between Tidal Current, Significant Wave Height and SMC - October, 2001 Branford, Connecticut Study Area

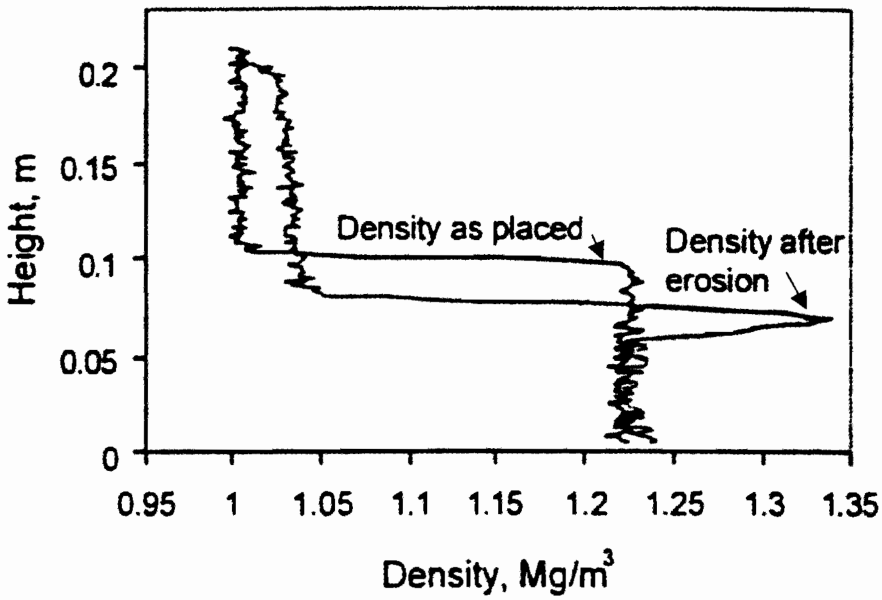

Figure 3. Laboratory Data Showing Changing Bulk Density Following Erosion/Deposition Cycling (from: Sills, et al., 2003).

The late Pleistocene-Holocene History of Long Island Sound

Johan C. Varekamp[1], Ellen Thomas[2], Maya Groner[1]
[1] Earth & Environmental Sciences, Wesleyan University, Middletown CT 06459
[2] Department of Geology and Geophysics, Yale University, New Haven CT 06511

INTRODUCTION

The early history of Long Island Sound (LIS) reflects the interplay between rising global sea levels, rising of the land as a result of glacial rebound, and fill-in by sedimentation of a depression formed after the retreat of the Wisconsinan Ice Sheet. The main events in this history are, in sequential order, retreat of the ice sheet from Long Island onto the Connecticut mainland, establishment of Glacial Lake Connecticut at the site of modern LIS and southern CT, establishment of Glacial Lake Hitchcock in central Connecticut, Massachusetts, New Hampshire and Vermont, drainage of Glacial Lake Connecticut and subsequent erosion/fluvial dissection of its lake bottom, early transgression of the sea into LIS, drainage of Lake Hitchcock, and fill-in of LIS by the sea and marine sedimentation (Lewis and Stone, 1991, Lewis and DiGiacomo-Cohen, 2000, Stone et al., 2005). During post-glacial times, loess was deposited in Connecticut and on Long Island (Kundic and Hanson, 2003).

The age of these events has been established with radiocarbon dates of peat and detrital organic matter in sediments on land and in LIS, in combination with counting varves deposited in the glacial lakes (see Ridge, 2003 and Stone et al., 2005 for listing and references). Many authors give ages in radiocarbon years, summarized and calibrated into calendar years by Kundic and Hanson (2003), and updated by Ridge (2003). The Harbour Hill-Fishers Island-Charlestown moraine indicates the position of the ice margin on the northern edge of Long Island at 21,300 calendar years BP (Ridge, 2003). The Captain Island-Norwalk Islands-Old Saybrook-Wolf Rocks moraine marks the limit of the ice sheet in southern Connecticut and LIS at about 20,400 calendar years BP (Ridge, 2003). Glacial Lake Connecticut started to form between these two moraines, between 21,300 and 20,400 calendar years BP. Varved sediments were deposited in this lake coeval with the formation of glacio-fluvial deltas along its northern shore (Lewis and Stone, 1991). Glacial Lake Connecticut may have drained around 18,000 calendar years BP (~15.5 radiocarbon years, Lewis and Stone 1991), although exact ages are not available.

Lake Hitchcock started to form when the ice had retreated further to the north, and a morainal dam formed close to Rocky Hill, CT at about 19,100 calendar years BP (Ridge, 2003). Rittenour et al. (2000) studied the lake development between varve years 2868 and 6900, assigning calendar ages of 17,500 to 13,500 years for this interval. The same varves were dated at 18,300 to 14,100 calendar years by Ridge (2003). Lake Hitchcock persisted for at least 4100 years according to the New England varve chronology (Antevs, 1922), draining around 13,500 calendar years BP (Rittenour et al., 2000), updated to 14,100 calendar years BP by Brigham-Grette et al. (2001) and Ridge (2003). In contrast, Stone et al. (2005) suggest an age of 13,500 radiocarbon years BP (16,500 calendar years BP) for the end of the stable phase and draining of Lake Hitchcock.

The age of the first marine invasion in LIS was estimated by Lewis and Stone (1991) and Kundic and Hanson (2003) at 14,300 calendar years BP (12,455 radiocarbon years BP), but Stone et al. (2005) argue (p. 12) that the early transgression (flooding of the existing channels) may have started between 15,000 and 16,000 14C years BP. After the marine invasion of the fluvial channels, the main marine unconformity cut into the upper lake beds and estuarine channel fill deposits (Lewis and Stone, 1991). A marine deltaic morpho-sequence was built over this unconformity between 13,000 and 9500 radiocarbon years BP (Stone et al., 2005; corresponding to 15,600-11,300 calendar years BP) to the west of the mouth of the Connecticut River, with its abundant sediment possibly provided by erosion of the now dry Lake Hitchcock beds. Stone et al. (2005) argued that sea level in LIS did not rise much during the delta building phase because the rates of crustal uplift and absolute sea level rise were similar. Sea level in LIS then rose sharply during melt water pulse 1b (Fairbanks, 1989) around 9500 radiocarbon years BP.

Disagreements about timing of events (e.g., draining of Lake Hitchcock by Stone et al. 2005 versus Rittenour et al., 2000) and assumptions about timing of glacial rebound suggest that the time frame of the post-glacial events is not yet rigorously defined. To address this, we dated the onset of the marine transgression in LIS with targeted radiocarbon dates on material from LIS sediment core LISAT12.

METHODS AND MATERIALS

In 1984, R.S. Lewis and co-workers collected 13 vibracores in LIS using the RV *Atlantic Twin,* and described the core lithologies (Thomas, 1989). Core LISAT12 was obtained near the Long Island coast to the southwest of the mouth of the Connecticut River (41o07.70'N, 72o28.80'W; point 25, map 2784, Stone et al., 2005) at a water depth of 37 m. We took 40 samples from core LISAT12 (curated at the Woods Hole Oceanographic Institution), and submitted material from 8 depth intervals for radiocarbon dating at NOSAMS, WHOI. We submitted mollusk (carbonate) shell fragments from all 8 samples. From 5 samples we also submitted hand-picked macrofloral remains, small twigs and leaf fragments, which are abundant in these sediments.

The level of the marine transgression in core LISAT12 is indicated by red, varved lake beds from Glacial Lake Connecticut overlain by grey sands and silts with abundant oysters (*Crassostrea virginica*; Szak 1987). The transgression interface is deformed as a result of coring, sloping between 540 and 562 cm depth-in-core, giving a mean depth of the marine transgressive interface of ~ 42.5 m below modern mean sea level, close to the maximum reported depth of the main transgressive interface (Stone et al., 2005). The sedimentary environment at 550-350 cm depth in the core is interpreted as intertidal to shallow subtidal, with abundant oysters, foraminifera and diatoms (Szak, 1987). Marine silt is present from 350 to 150 cm depth, and the upper 150 cm is coarser-grained, containing common eroded foraminifera and mollusk shell fragments. This upper section is interpreted as reworked, possibly sand-wave facies material (Fenster, 1995), as present in the modern environment near the LISAT12 coring site (Knebel et al., 1999).

The results for ^{14}C measured by AMS were corrected for the measured $\delta^{13}C$ values. Mollusk (carbonate) data have been calibrated with the 1998 marine data set of the CALIB program, version 4.3 (Stuiver et al., 1993, 1998), using the standard 450 year marine reservoir effect. Our work on LIS ^{14}C ages (last 150 years of LIS history) indicates that this is a reasonable approximation for LIS, although variations of $\pm$ several 100 years exist (Groner, 2004). The ^{14}C ages of macrophytes were calibrated with CALIB using the 1998 atmospheric decadal data set (Stuiver et al. 1993, 1998). The ages reported here are our best estimate means when multiple age calibrations occurred, and a detailed error analysis will be presented elsewhere. The currently accepted age of the LIS marine transgression (12,455 ^{14}C years BP) is based on a ^{14}C date from a depth of 487 cm in core LISAT12 (Stone et al., 2005). This sample was 'bulk organic material including shells' (*verbatim* Stone et al., 2005, p. 65; radiocarbon date GX-18094, Krueger labs), but the Krueger laboratory report states that carbonate was dissolved prior to analysis. We calibrated this datum point with the 1998 atmospheric decadal data set and will discuss it as an organic matter age.

RESULTS

The data (Table 1) show an age profile through the marine section of core LISAT12 from 69 cm to 540 cm depth. The carbonate ages show a smooth, curvilinear relation between calendar age and depth-in-core (Figure 1). The ^{14}C ages from terrestrial macrophytes are all substantially older than the carbonate ages from the same samples, with differences of 1400 to 3300 years, except for the sample at 239 cm, where the organic carbon and carbonate ages are very close. We fitted a polynomial to the age-depth carbonate data points and obtained a 'virtual carbonate age' for sample GX-18094 (Stone et al., 2005) of 9767 BP, with a corresponding calibrated ^{14}C age of the bulk organic matter of 14,722 calendar years BP, a difference of ~5000 years.

The mollusk ages have an error of $\pm$ several 100 years as a result of the uncertainty in the ^{14}C reservoir and errors in the ^{14}C determination (+/- 50 to 75 years) as well as the usual calibration uncertainty. Nonetheless, this total error is substantially smaller than the differences with the calibrated ^{14}C ages from coexisting terrestrial organic matter (thousands of years). The abundant wood and leaf fragments in the marine silts are substantially older than the associated carbonate ages, and the radiocarbon age data set from bulk organic matter of several LISAT cores (Lewis, unpublished data) shows a very poor correlation between age and depth-in-core. We therefore argue that macrophyte organic carbon ages represent the time that plants died, and not of sedimentation in LIS: the wood fragments must have resided in soils or periglacial lake sediments on land for several 1000 years prior to arrival in LIS (see also Ridge, 2003). The sample at 239 cm depth is unusual with its concordant ages, and supposedly this organic matter was transported rapidly to LIS after plant die off. Bulk organic matter in the marine sediment and varved lake beds also contains abundant terrestrial plant debris and likewise gives the age of formation of the original organic matter. An event chronology based on such ^{14}C ages provides a history that may be several 1000 years too old.

DISCUSSION

Our carbonate ^{14}C ages provide a new look at the Holocene history of LIS, indicating that the marine transgression at the site of core LISAT12 occurred only at about 10,000 calendar years BP, much later than previously proposed (14,000 - 16,000 calendar years BP, Stone et al., 2005). The ages of several other post-glacial events are also not well constrained (e.g., the draining of Glacial Lake Connecticut, the timing of isostatic rebound, the draining of Lake Hitchcock), and the late Pleistocene-Holocene history of LIS may have to be reconsidered (Figure 2).

We assume that Glacial Lake Connecticut started to form at about 20,000 calendar years BP and Lake Hitchcock started to form at 19,100 calendar years BP (Ridge, 2003). Varve counts in sediments recovered in vibracores from LIS (Szak, 1987: Reimer, 1986) combined with the thickness of lake beds (> 150 m, Stone et al., 2005) suggest that Glacial Lake Connecticut persisted for 3500-6500 years, depending on varve thickness variations between cores. The lake thus must have drained between 16,500 and 13,500 calendar years BP, a range overlapping with the age of draining of Lake Hitchcock (14,100 calendar years, Ridge, 2003). We speculate that, if Lake Hitchcock drained catastrophically, the flood may have led to the failure of the barrier retaining Glacial Lake Connecticut, with both lakes draining into the Atlantic Ocean almost simultaneously (around 14,000 calendar years BP). LIS was then dry for about 4000 years, which period included the cold and windy Younger Dryas interval (12,800 to 11,500 calendar years BP). During the Younger Dryas and till the time of inundation, the dry bottom of LIS may have been one of the source areas for e.g., the Windwood loess deposits on Long Island (Kundic and Hanson, 2003) that cover an age of 14,000 to 8500 calendar years BP.

The ocean invaded LIS starting at ~10,000 calendar years BP, possibly coinciding with melt water pulse 1b of Fairbanks et al. (1992), rather than with the older melt water pulse 1a as proposed by Stone et al. (2005). The sedimentary record of core LISAT12 indicates that the core site remained in the inter-tidal to shallow sub-tidal flat environment for about 1000 years (Szak, 1987), with the sedimentation rate roughly equal to the rate of relative sea level rise (about 0.5 cm/yr). This sequence may record the last stages of meltwater pulse 1b. The rate of isostatic rebound at this time is not exactly known; it may have been positive with limited variation over these 1000 years (Stone et al., 2005), so that true rates of sea level rise were probably higher than the estimated sedimentation rate in this core (see Fairbanks et al., 1992).

CONCLUSIONS

Over the last 20,000 years, the glacial ice sheet retreated, and lakes, including Glacial Lake Connecticut formed, followed by draining of the lake and the formation of a dry basin, followed by marine transgression and formation of modern LIS. We argue that the timing of these consecutive events should be revised, based on the new ^{14}C dates on carbonate from core LISAT12 in eastern LIS, because the ^{14}C ages derived from terrestrial organic matter in marine LIS sediments are up to 5000 years too old. We explain these discrepancies as a result of intermediate-term storage of terrestrial organic material in lake deposits or soils, and subsequent re-sedimentation in LIS. We speculate that the sudden drainage of Lake Hitchcock around 14,100 calendar years BP might have triggered the drainage of Glacial Lake Connecticut. The LIS basin was then dry for a 4000 year period that included the Younger Dryas, and may have served as a sediment source for loess deposits in Connecticut and Long Island. We date the marine incursion at 10,000 calendar years BP, which most likely coincided with melt-water pulse 1b of Fairbanks et al. (1992), occurring at a time when LIS was still experiencing glacial rebound.

REFERENCES

Antevs, E., 1922, The recession of the last ice sheet in New England. Am. Geograph. Society Res. Series v.11, 120p.

Brigham-Grette, J., Rittenour, T.M., and Mann, M.E., 2001, Drainage history of Glacial Lake Hitchcock and paleoclimatic implications of late quaternary sediments and terraces in the central CT valley. GSA, Abstracts with Programs, v.33, p.14.

Fairbanks, R. G., 1989. A 17,000-year glacio-eustatic sea level record: influence of glacial melting rates on Younger Dryas events and deep ocean circulation. Nature, v. 342, p. 637-642.

Fairbanks, R.G., Charles, C.D., and Wright, J.D., 1992, Origin of global meltwater pulses. In: Radiocarbon after four decades, ed. Taylor, R.E., p. 473-499

Fenster, M.S., 1995, The origin and evolution of the sand sheet facies, eastern Long Island Sound. PhD thesis, Boston University, p. 325.

Groner, M., 2004, Carbon dynamics in Long Island Sound: a study using stable and radioactive isotopes. Senior undergraduate thesis, Wesleyan University, Middletown CT, USA, p.138

Knebel, H., Signell, R.P., Rendigs, R.R., Poppe, L.J., and List, J.H., 1999, Seafloor environments in the Long Island Sound estuarine system. Marine Geology, v. 155, p. 277-318.

Kundic, V., and Hanson G.N., 2003, Geochemistry of Loess on Long Island. http://pbisotopes.ess.sunysb.edu/lig/Conferences/abstracts-04/kundic/kundic.htm

Lewis, R. S., and Stone, J. R., 1991, Late Quaternary Stratigraphy and Depositional History of the Long Island Sound Basin. J. Coastal Res., Spec. Issue, v. 11, p. 1-23.

Lewis, R.S., and DiGiacomo-Cohen, M.L., 2000, A review of the geologic framework of the Long Island Sound Basin with some observations relating to postglacial sedimentation. J. Coastal Research, v.16, p. 522-532.

Reimer, G.E., 1986, Sedimentology and interpretation of environment of deposition of rhythmically laminated silt and clay facies of LISAT6 core, Long Island Sound. Report from Rutgers University.

Ridge, J.C., 2003, The last deglaciation of the northeastern United States: a combined varve, paleomagnetic and calibrated ^{14}C chronology. In: Geoarcheology of landscapes in the glacial northeast, eds. Cremeens, D.L. and Hart, J.P., New York State Museum Bulletin, v. 497, p. 15-45.

Rittenour, T.M., Brigham-Grette, J., and Mann, M.E., 2000, El Nino-like climate teleconnections in New England during the late Pleistocene. Science, v. 288, p. 1039-1042.

Stone, J.R., Schafer, J.P., London E.H., DiGiacomo-Cohen, M.L., Lewis, R.S., and Thompson, W.B., 2005, Quaternary Geologic Map of Connecticut and Long Island Sound Basin. Scientific Investigations MAP 2784 with explanations, p. 72.

Stuiver, J.R. and Reimer, P.J., 1993, Extended ^{14}C database and revised CALIB 3.9 ^{14}C calibration program. Radiocarbon, v. 35, p. 215-230.

Stuiver, J.R., et al., 1998, INTCAL98 radiocarbon age calibration 0-24,000 BP. Radiocarbon, v. 40, p. 1041-1083.

Szak, C., 1987. The nature and timing of Late Quaternary events in eastern Long Island Sound. MSc Thesis, University of Rhode Island. p. 75.

Thomas, H. F., 1989, Final Summary Report for a study of 'The geologic framework of Southern New England: Vibracoring of Long Island Sound', Minerals Management Service, Department of the Interior, Subagreement MMS/AASG.

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TABLE 1. Radiocarbon age data for core LISAT12. Carbonate ages were calibrated with the 1998 marine data set and a 450 year reservoir; terrestrial organic carbon ages were calibrated with the 1998 atmospheric decadal data set (both in the CALIB version 4.3 program, Stuiver and Reimers, 1993, 1998).

Depth (cm)	^{14}C(Carbonate) Age, years BP	Calibrated Age, years BP	^{14}C (Corg) Age, years BP	Calibrated Age, years BP	AT Corg-Carb
69	7890	8351			
169	7720	8168			
239	7990	8414	7180	7973	-441
355	8590	9007	10250	12021	3014
414	8960	9598	10750	12874	3276
445	9220	9833	10250	12021	2188
487*			12455	14722	4955
497	9110	9804	9840	11226	1422
540	9220	9833			

*sample GX-18094 (Stone et al., 2005)

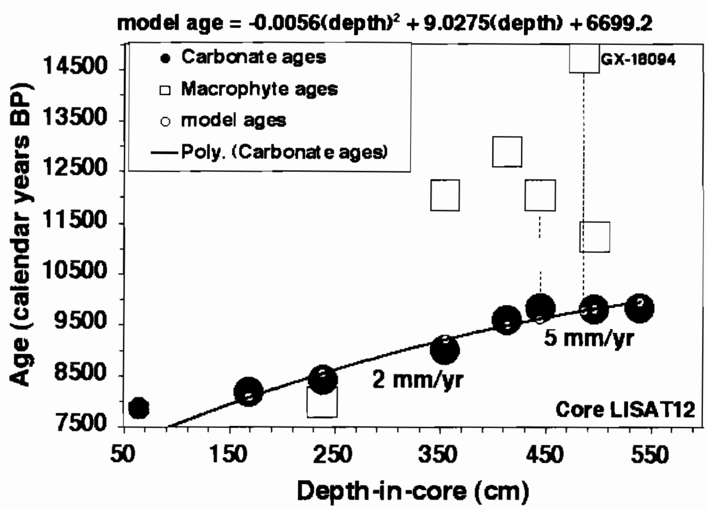

FIGURE 1. Calibrated radiocarbon ages of samples from core LISAT12. The size of the symbols approximates the compounded errors. Sedimentation rates for the bottom and intermediate interval are shown as well as the interpolation polynomial for the carbonate ages (heavy curve and 'model age' expression at the top). All macrophyte ages (except for one) are substantially older than carbonate ages from the same depth interval.

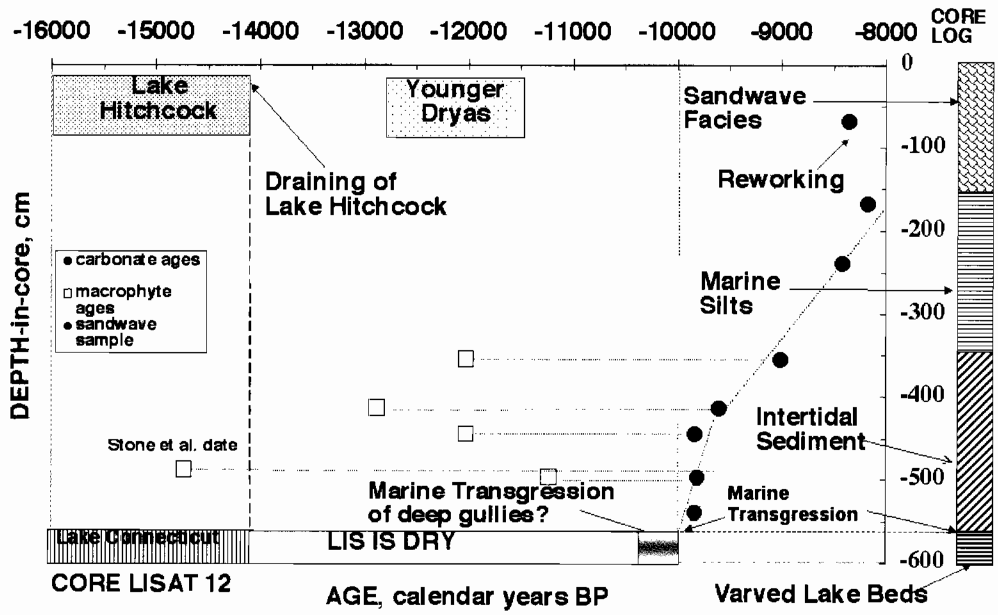

FIGURE 2. Chronology of the events that shaped LIS. Calibrated carbonate and organic matter ages in samples from core LISAT 12 are indicated by filled circles and open squares. Lake Hitchcock and Glacial Lake Connecticut both formed prior to 16,000 calendar years BP. Lake Hitchcock drained around 14,100 calendar years BP, which may have led to the drainage of Glacial Lake Connecticut directly afterwards. LIS was then dry for close to 4000 years, including the Younger Dryas period. The sea started to invade LIS initially only in the deepest channels, with the main incursion occurring around 10,000 calendar years BP. The marine sequence built up with initially intertidal deposits, followed by marine silts and reworked coarser deposits.

The Transport, Dispersion and Decay of Dissolved Materials in Western Long Island Sound

James O'Donnell
Department of Marine Sciences, University of Connecticut

ABSTRACT

A simple model of the transport of a decaying dissolved material from the East River to western Long Island Sound is presented. Solutions are computed numerically and their general characteristics are discussed. The sensitivity of solutions to the choice of dispersion coefficient and decay rate is illustrated. The method of superposition allows the simulation of a series of releases of insecticide as occurred in the 1999 with results that suggest that laterally and vertically averaged concentrations in the range 0.1-0.5 PPB may have occurred in the western most 10-20km of the Sound.

INTRODUCTION

The unusually low landing of lobsters by fisherman in Western Long Island Sound in the fall of 1999 raised fundamental questions about the direction and rates of transport of material through the East River and the distribution of insecticides entering the Sound from the East River. Though the mechanisms that are responsible for the motion of water in estuaries (tides, wind and density variations) are fairly well understood, computing the consequences of their interaction and the movement of material is difficult even with the largest modern computers. But considerable fundamental understanding about the interactions has emerged in the last two decades through the study of observations and mathematical models and these allow useful estimates to be made with limited observations and modest computing effort.

This paper first summarizes the characteristics of the circulation in the East River. Subsequently a simple approach to estimating the extent of the area of the western Long Island Sound impacted by the sudden release of a dissolved tracer whose concentration decays with time is developed. Characteristic of the solution are discussed and examples of numerical solutions that apply to insectcide distributions are presented.

CIRCULATION IN THE EAST RIVER

The circulation throughout LIS and the East River is dominated by the astronomical tides, see Jay and Bowman (1974). There is also a mean circulation that has substantial variation in both the vertical and the horizontal direction and a characteristic magnitude of approximately 10 cm/s, see Bodgen and O'Donnell (1997) and Viera (2000). Figure 1a shows a map of the bathymetry of the western Sound and the location of the acoustic current profiler "S1" that was maintained by the National Oceanic and Atmospheric Administration's (NOAA) Long Island Sound Observation Program (LISOP) just North of Willetts Point, NY, between 13-Jul-1988, and 22-Apr-1989. The vertical structure of the mean flow in the north direction (with positive towards LIS) at each level during this period is shown in Figure 1(b). Note that in the top third of the profile the flow is directed towards LIS and that the rest of the water column flows towards New York Harbor (NYH).

Blumberg and Pritchard (1999) exploited a combination of data like that shown in Figures 1b together with a mathematical model to show that the mean (average of many tidal cycles) transport of water in the East River was from LIS to NYH. They estimated the mean volume transport as 310 m^3/s. Their analysis also showed that this was the net results a flow of 260 m^3/s in the upper water column towards LIS and a counter flow of 570 m^3/s in the lower layer towards NYH. Since this circulation carries with it whatever is dissolved in the water, there is a net transport of salt out of LIS and a transport of freshwater into LIS explaining why the western end of LIS is fresher that the eastern end (see Kaputa and Olsen, 2000) despite the fact that the Connecticut River, the largest single source of freshwater, is located in the eastern Sound.

This long term average, or residual, flow is not the only mechanism by which dissolved materials are transported horizontally. Though the motion driven by tides is almost periodic, the velocities are much larger than those due to the non-tidal processes and fluid is moved considerable distances during a 12 hour tidal cycle. The 60 cm/s amplitude current oscillation at Willets Point, for example, can cause a parcel of water released at the mooring to oscillate approximately 4 km either side of its initial position. Since there is complex bathymetry in this area significant shear dispersion is to be expected. An excellent introduction to the physics and mathematics of this process can be found in the monograph by Fischer et al. (1979).

MODEL

A simple model that includes the effects of both the mean westward advection and the dispersive flux to the east associated with the complex and unsteady tidal currents is developed in the monograph of Fischer et al. (1979). The approach has been applied in many estuaries including San Francisco Bay, (Monismith et al., 2002) and Long Island Sound (Torgersen et al. 1997). Taking as the distance from the location of S1 in Figure 1a along the centerline of western LIS (approximately 52°) the evolution of the lateral and vertical average concentration, $C(x,t)$, of a dissolved material in a channel of cross-sectional area $A(x)$, with a steady volume flux, $Q(x)$, flowing through it can be described by

$$A\frac{\partial C}{\partial t} + Q\frac{\partial C}{\partial x} = \frac{\partial}{\partial x}\left\{ KA\frac{\partial C}{\partial x} \right\} - \alpha AC \qquad (1)$$

where K is the coefficient describing the rate of dispersion and α is the rate of decay of the material due, for example, to photolysis or radioactive decay. The geometry of the coastline and bathymetry of the western half of LIS (see Figure 1) can be used to estimate $A(x)$. As is clear in Figure 2a, a simple linear function, $A(x)=10000(4+7x/8)$ for $x > 0$ and $A(x)=40000$ for $x \leq 0$ captures the pattern well.

The best available estimate of the westward flux in LIS is that of Blumberg and Pritchard (1997) who suggest $Q=-310\ m^3/s$. This is within 25% of the prior estimate of Jay and Bowman (1975). The dispersion coefficient K is an empirically determined parameter and can be estimated from tracer experiments or inverse analysis if property distributions. However, a reasonable range of values can be established using prior work in similar environments and nearby locations. For example, Torgersen et al (1997) used naturally occurring radio-tracers in LIS to obtain the estimate $K = 50 m^2/s$ in western LIS. Fischer et al. (1979) provide a convenient summary table (their 7.2) of values obtained in a range of estuaries. This suggests that $50\ m^2/s$ is on the low end of the range. The value $200\ m^2/s$ is more in keeping with values used in other large estuaries and is closer to the value recently estimated by Gay et al (2004) using salinity observations. Solutions presented here will incorporate values in this range.

The decay coefficient, α, is determined by the chemistry of the dissolved constituent. Malathion©, a popular insecticide, is reported to have half-life, $T_{1/2}$, in the range 62-430 hours (Berkman, 2002, and www.epa.gov/oppsrrd/op/malathion/efedrra.pdf). The decay rate and the half life are related by the simple equation $\alpha = -\ln(0.5)/T_{1/2}$ so values of α in the range $(0.45\text{-}3.1)\ 10^{-6}\ s^{-1}$ will be explored.

Solutions of equation (1) require that we specify the initial distribution of the concentration. However, since the concentration, C, appears in each term of (1) the actual value is irrelevant. We may use unity and subsequently multiply the solution by the initial concentration of the tracer. In order to simulate the dispersion of materials introduced to the East River during a storm, we assume that the material enters rapidly (in a few hours) and is mixed thoroughly throughout the East River in the interval -10km to 2km. Note that this segment has a volume of approximately 4 x 10^8 m^3. The mass injected must be divided by this volume to obtain the concentration scale.

Boundary conditions are also required and we will assume that the dispersion flux to the west is small relative to the advection in the East River and take the derivative $\partial C/\partial x = 0$ at $x = -10$ km.

At the eastern boundary we will also require $\partial C/\partial x = 0$ at $x = 80$ km. The solution to this problem was computed numerically using an explicit, spatially centered finite difference approach as outlined in Fischer et al. (1979) and the computation and visualization software MATLAB©.

SOLUTIONS

Note that character of the solutions is determined by coefficient values, principally K and α. Together these determine the length scale, $L_s = \sqrt{K/\alpha}$, for variations in the concentration field. Since solutions to equation (1) tend to have exponential character one should expect that the influence of the source should be significant several times L_s from its source. Table 1 shows plausible values of L_s range between 4 and 20 km.

TABLE 1. Concentration field Lengthscale (m) for combinations of parameter values.

α (s^{-1})	K (m^2/s)= 50	100	200
5.0x10^{-07}	10000	14142	20000
1.0x10^{-06}	7071	10000	14142
3.5x10^{-06}	3780	5345	7559

To display the evolution of concentration we choose values in the middle of the reasonable range: K =100 and α=10^{-6}. These yield L_s=10 km so we should expect that the evolution of the solution at x = 0, 10, 20, 30, 40 and 50 km should contrast the response of western LIS to the abrupt release of a decaying pollutant like Malathion© in the East River and these are shown in Figure 2b. Figure 2c shows the same solutions on a log scale to reveal the longer term behavior when the concentrations are low.

The solution shows that the concentration at 10 km rises as material disperses into the region and then slowly reduces as the material decays. The maximum concentration occurs after 100 hours and is 2% of the maximum at the release point. At 20 km the maximum is achieved at 200 hours but is only 0.3%. The evolution at the more distant locations is illustrated in Figure 2c which shows the logarithm of the concentration. Clearly that the maximum concentration decreases with distance and that the time of occurrence is delayed at larger distances

The general character of solutions with other parameter choices is similar. Table 2 presents a comparison of the dependence of the maximum concentration at selected location on values of K and α. Solutions with slower dispersion have higher maximum concentrations in the western part of the domain and lower maxima to the east. Dispersion acts to spread the impact by increased dilution. The effect of slower decay is to raise the maximum concentrations everywhere and to delay the time that the maximum arrives at any location.

TABLE 2. Concentration maxima and time of occurrence.

Length scale, L (km)	10	7	14	17	12
Dispersion, K (m^2/s)	100	50	100	150	150
Decay, α (s^{-1})	10^{6}	10^{6}	$0.5\text{x}10^{6}$	$0.5\text{x}10^{6}$	10^{6}
Half Life, $T_{1/2}$ (hours)	193	193	385	385	193
	Maximum concentration				
x = 10	2.06%	0.63%	2.58%	3.88%	3.24%
20	0.34%	0.05%	0.54%	1.01%	0.69%
30	0.07%	0.01%	0.15%	0.35%	0.19%
40	0.02%	0.00%	0.05%	0.13%	0.06%
	Time of maximum concentration (hrs)				
x = 10	111	146	131	107	92
20	234	320	284	228	190
30	363	504	451	360	293
40	495	692	628	502	399

SIMULATIONS AND CONCLUSION

The solutions presented so far demonstrate the sensitivity of the choice of model parameters. Practical use of the model requires the specification of the initial concentrations and the aggregation of repeated releases to simulate a series of release events. Since the model, equation (1), is linear the effect of multiple discharges can be simulated by multiplying the response to a unit source, $C(x,t)$by the initial concentrations and adding the lagged products. If the discharges occurred on days 0, r_1, r_2, and r_3, and the initial concentrations were $c_{0,1,2,3}$, then the model prediction for the combined effect, C_s is the sum

$$C_s(x,t)=c_0C(x,t)+c_1C(x,t-r_1)+c_2C(x,t-r_2)+c_3C(x,t-r_3) \qquad (2)$$

The utility of the approach can be illustrated by simulating the consequences in LIS of four inscecticide applications along the shores of the East River that occurred in September 1999. Following Philips (2002) and Berkman (2002), initial concentrations in the East River can be estimated as $c_{0,1,2,3}$ {11.5, 7.4, 32.2, 15.3} PPB, and the time delays of the series of events can be taken as $\tau_{1\text{-}3}$={3, 8, 14} days. Using K=150 and $\alpha=10^{-6}$ (equivalent to $T_{1/2}$=193 hrs), equations (1) and (2) have the solution shown in Figure 3 for x = 0, 10, 20, 30, and 40 km. All three panels show the same numbers. The axes in the central panel (b) were chosen to illuminate the evolution at 10-30 km. The right panel (c) shows the evolution on a logarithmic scale so that the maxima at all locations can be determined. The important difference between this simulation and the previous solutions is that the concentrations in PPB are shown and the combined effect of the series of rainfall events is represented. Since the half life is similar to the spacing of the storms, there is a substantial additive effect. Table 3 shows the maxima at each location. It is clear that there is substantial transport to the east and high concentration values persist for several days.

These calculations suggest that it is plausible that lateral average concentrations in the range 0.1-0.5 occurred in western LIS as a result of insecticide spraying. The model could easily be extended to include the effect of degradation of the material before it entered the East River by modifying $c_{0,1,2,3}$ appropriately. This may reduce the predicted concentrations. It should be noted, however, that values measured at a particular location and depth are likely to be substantially different from the cross-sectional average. It is likely that there would be areas of zero concentration and other areas with significantly higher concentration.

TABLE 3. Simulated maximum concentrations and times of occurrence.

Half Life (hrs)	$T_{1/2}$=62.4		$T_{1/2}$=91.2		$T_{1/2}$=110.4	
Distance (km)	**Time**	**Max (PPB)**	**Time**	**Max (PPB)**	**Time**	**Max (PPB)**
0	192	15.889	192	16.035	192	16.115
10	252	0.318	259	0.427	263	0.484
20	310	0.040	326	0.069	335	0.088
30	368	0.006	462	0.015	478	0.021
40	468	0.001	525	0.004	548	0.006

REFERENCES

Berkman, C.E. (2002) The effect of impervious urban surfaces upon the concentrations of malathion and malaoxon following aerial application in the Long Island Sound Area during September and October, 1999. Report to of Jones, Verras and Freiberg, L.L.C. of New Orleans, LA.

Blumberg, A.F. and D.W. Pritchard, 1997. Estimates of the transport through the East River, New York. J. Geophys. Res. 102, 5685-5703.

Bogden, P.S. and J.O'Donnell, 1997. Generalized Inverse analysis of current measurements from a moving ship: estimating the tidal and non-tidal flows in Long Island Sound. J. Marine Res. 56, 989-1014.

Fischer, H.B., E.J. List, R.C.Y. Koh, J. Imberger, and N.H. Brooks, 1979. Mixing in Inland and Coastal Waters. Academic Press, 483pp.

Gay, P.S., J. O'Donnell and C.A. Edwards (2004), Exchange Between Long Island Sound and Adjacent Waters J. Geophys. Res. 109, C06017, doi:10.1029/2004JC002319.

Jay, D. A. and M. J. Bowman. 1975. The Physical Oceanography and Water Quality of New York Harbor and Western Long Island Sound, Technical Report 23, Ref. # 75-7, MSRC-SUNY, Stony Brook

Kaputa, N. P. and C. B. Olsen, State of CT. 2000. Dept. of Env. Protection, Long Island Sound Ambient Water Quality Monitoring Program: Summer Hypoxia Monitoring Survey '91-'98 Data Review

Monismith, S. G., W. Kimmerer, J. R. Burau, and M. T. Stacey. 2002. Structure and flow-induced variability of the subtidal salinity field in northern San Francisco Bay, J. Phys. Oceanography, 32, 3003-3019.

Phillips, K.J. (2002) Pesticide Investigation Impact to the water column of Long Island Sound due to pesticide spraying at selected ares during September 3rd-Octrober 16, 1999. Report to of Jones, Verras and Freiberg, L.L.C. of New Orleans, LA.

Torgersen, T., E. DeAngelo, and J. O'Donnell, 1997. Calculation of horizontal mixingrates using 222Rn and the controls on hypoxia in western Long Island Sound, 1991. Estuaries, 20, 328-345.

Viera, M. E. C. 2000. The Long-Term Residual Circulation in Long Island Sound, Estuaries, 23, No. 2, 199-207.

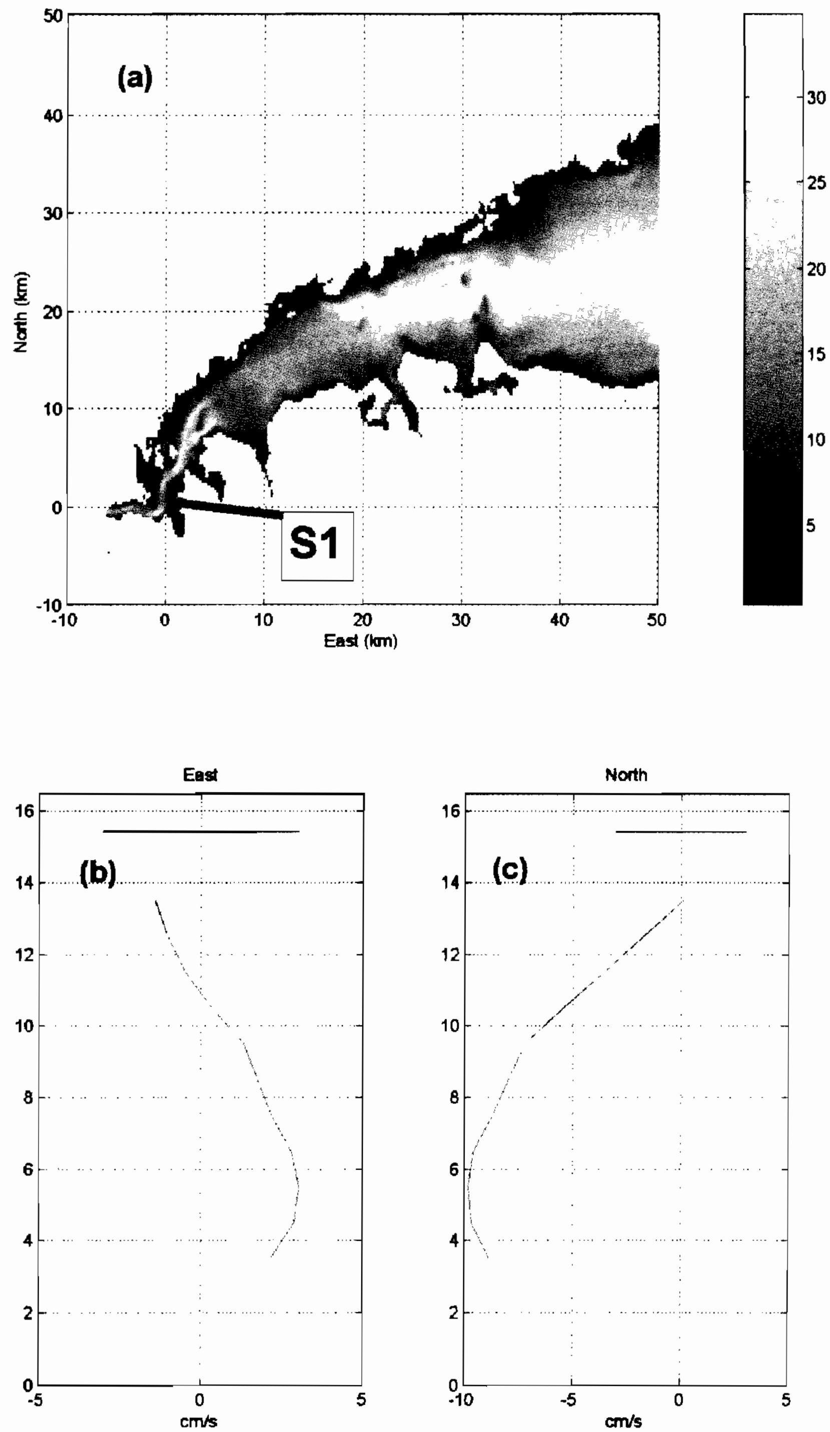

FIGURE 1. (a) Depth in western Long Island Sound and the location of LISOP station S1. (b) The vertical structure (distance above bottom) of the mean current observed at station S1 between July 1988, ands April 1989.

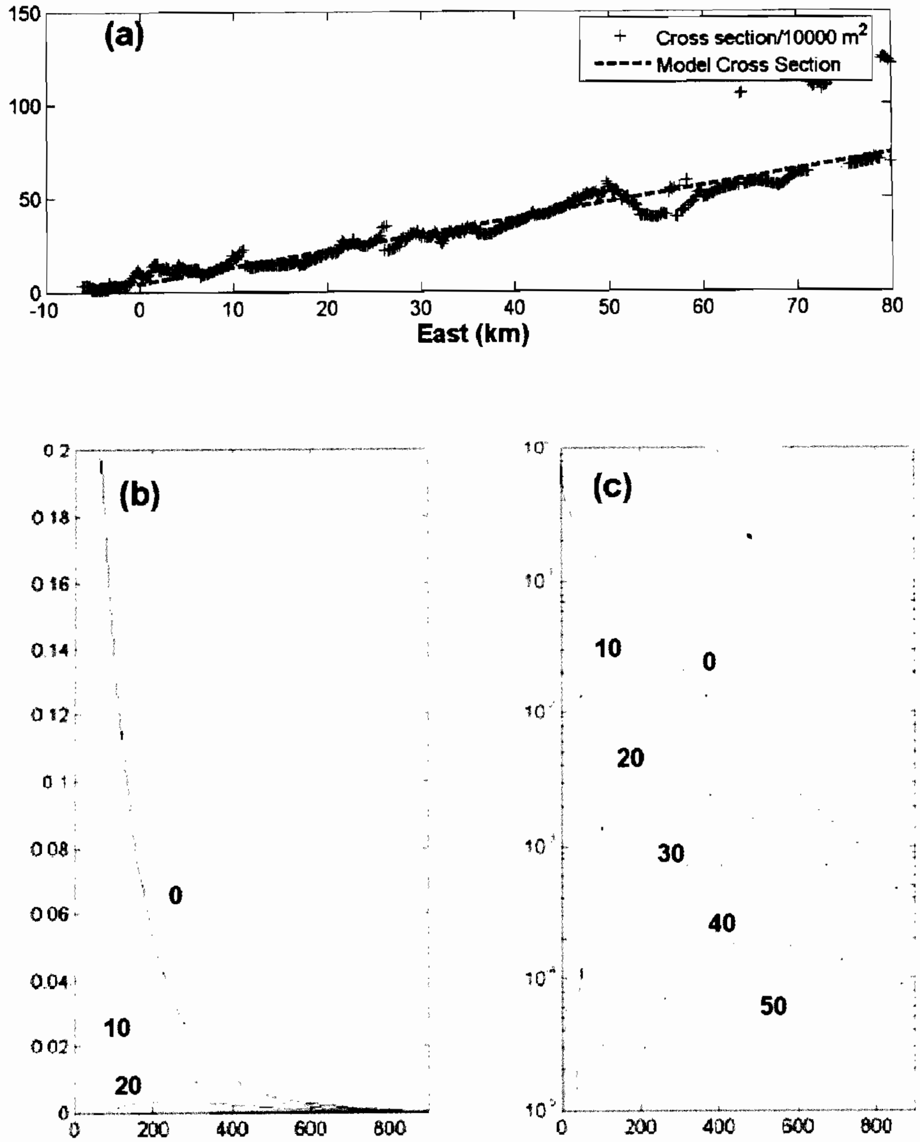

FIGURE 2. (a) Cross-sectional area of the western half of Long Island Sound and the linear approximation (dashed line) used in the model. Axes show the distance (km) from the LISOP mooring location S1. (b) Evolution of the concentration at 5 locations for K=100 and $\alpha=10^{-6}$ (equivalent to $T_{1/2}$=193 hrs). (c) A logarithmic representation of the concentration evolution shown in (b).

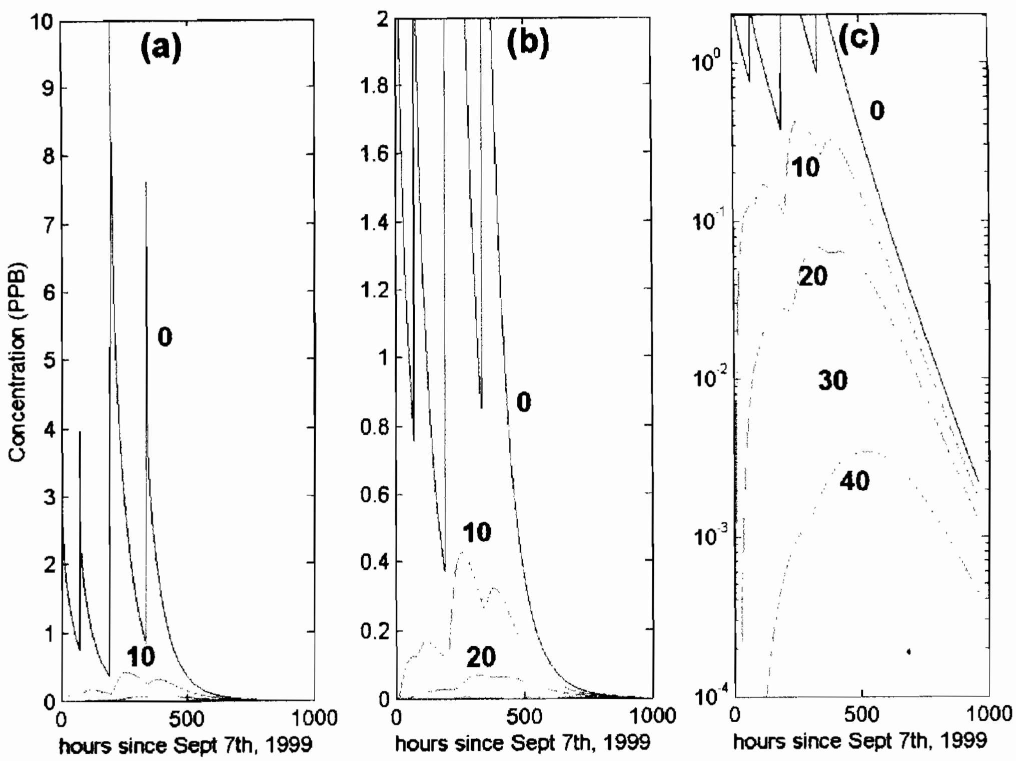

FIGURE 3. The evolution of the simulated concentration after four rainfall events with K=150 and $\alpha = 10^{-6}$ (equivalent to $T_{1/2}$=193 hrs).The right frame shows the logarithm of the concentration to illuminate the smaller values.

Hormonal Responses of Lobsters to Stresses of Long Island Sound

Laufer, Hans1,2, Demir, Neslihan1, Capulong, Christopher P.3, Pan, Xuejun1, Biggers, William J1, 4
1Department of Molecular and Cell Biology, University of Connecticut, Storrs, CT , 06269-3125
2The Marine Biological Laboratory, Woods Hole, MA , 02543
3 Department of Chemistry, University of Connecticut, Storrs, CT , 06269-3060
4 Present address: Department of Biology, Wilkes University, Wilkes-Barre, PA, 18766

Lobsters in Eastern Long Island Sound have been affected as much as 67% by shell disease that disfigures the shell and ultimately may result in their death. We have examined shell diseased lobsters for molting hormone concentrations (ecdysones), and have investigated molting hormone levels in normal and shell diseased lobsters by means of radioimmuno-assays. Ecdysteroids are low in normal lobsters and peak at about 200-300ng/ml in the hemolymph before each molt. Shorts will usually molt once a year depending on their size. Mature females, will lay their eggs 100 days following their last molt and will carry them for 170-180 days and, at times even longer, molting only after about 100 days following embryo hatching (Chang and O'Connor,1988). During the period of egg carrying ecdysteroid levels in normal females are exceedingly low (about 10ng/ml) (Laufer et al., 2003). Ecdysteroids are also higher (89.5 ng/ml) (N=76) in non-ovigerous shell diseased animals compared to 57.4 ng/ml in normal lobsters (N=210) during most of the year. The shell diseased animals were higher in 7 of 10 months of the year in which animals were available for examination. These results are statistically highly significant by a two-way analysis of variance, P=0.002. Ecdysteroid levels in shell diseased ovigerous lobsters averaged 165 ng/ml in five cases examined. The level of ecdysteroids in normal ovigerous lobsters (N=3) was 15.3 ng/ml which are substantially lower than in the shell diseased animals. The results suggest that a molt is approaching in diseased lobsters despite the carrying of embryos. Molting would lead to the loss of the brood. We conclude from these results that shell disease appears to lead to increased ecdysone levels in the blood and probably increases molting frequencies in affected lobsters. Molting hormone increases may be a protective mechanism devised by lobsters to prevent short-term effects or mild cases of shell disease. By molting the animals have a chance to regenate and repair their damaged shell. Ecdysteroids are low in normally ovigerous females, but are high in ovigerous females with shell disease, and may lead to unexpected molting, and loss of embryos (Laufer et al., 2005).

In other studies of lobsters from Long Island Sound and Vineyard Sound, we have also found that lobster blood and tissue samples contain substantial levels of alkylphenols, which are also found in sediment samples. A related alkylphenol, bisphenol A, is a known endocrine disruptor in vertebrates, that is, it interferes in the normal hormonal controls of fish and other organisms. It has been reported that trout reproduction is affected by bisphenol A since it stimulates male fish to synthesize yolk proteins. Alkylphenols in lobster tissues and blood probably result from anthropogenic environmental contaminations of industrial wastes such as detergents, lubricants, paints, and degradation of plastics and rubber tires (Biggers and Laufer, 2004). We have found that alkylphenols are toxic to larval lobsters and one has induced an intermediate stage between a larva and juvenile (Biggers and Laufer 1999 and 2004, Laufer et al., 2003).

At relatively low concentrations these alkylphenols interfere with settlement of *Capitella capitata*, an annelid worm, in a sensitive and rapid bioassay that we have developed to detect juvenile hormone active compounds, including the detection of methyl farnesoate (MF), a crustacean hormone controlling both reproduction in adults and morphogenesis, in the important transition of metamorphosis from larva to juvenile (Biggers and Laufer, 1996).

Alkylphenols are toxic at high concentrations and are endocrine disruptors at lower concentrations to lobsters and annelids (Biggers and Laufer, 2004).

We determined by gas chromatography and mass spectrometry, GC/MS, that significant numbers of lobsters in Long Island Sound and Vineyard Sound have alkylphenols in their blood and tissues. 2-t-butyl-4 (dimethylbenzyl) phenol was found in hemolymph at an average of 0.46 ± 0.11ug/ml (13 out of 14 lobsters), 2,6-bis(t-butyl)-4-(dimethylbenzyl) phenol was found in hemolymph at an average of 1.89 ± 1.14 ug/ml (11 out of 14 lobsters), 2,4-bis(dimethylbenzyl)phenol was found in concentrations of 4.03 ± 1.52 ug/ml present in all 14 lobsters analyzed, and 2,4-bis(dimethylbenzyl)-6-t-butylphenol occurred in hemolymph concentrations with an average of 10.98 ± 6.41 ug/ml(in 11 out of 14 lobsters tested).While 2,6-bis(t-butyl)-4-(dimethylbenzyl) phenol was found in concentrations of 2.55 and 6.11 ug/gram in two (N=2) and 2,4-bis(dimethylbenzyl)-6-t-butylphneol (concentrations of 23.26 and 26.89 ug/gram) samples of hepatopancreas the lobsters equivalent of a liver (Biggers and Laufer, 2004).

We have recently analyzed 15 deep-sea lobsters. Only one had low levels of blood alkylphenol contamination of 2,4-bis-(dimethylbenzyl)phenol at 0.0503 ug/ml and 2,4-bis(dimethylbenzyl)-6-t-butylphneol at 0.357 ug/ml, while batches of embryos from 5 ovigerous females had higher levels of alkylphenol contamination (up to 2.18 ug/ml).Three out of five groups of embryos tested were positive for 2,4 bis (dimethylbenzyl) phenol and one out of five groups of embryos tested positive for all four alkylphenols. Of the five batches from the deep sea lobsters embryos two out of five tested negative. The relatively larger load of alkylphenols in deep-sea embryos, compared to their mothers, suggests two possible conclusions. We already know that it takes several months to produce eggs in the lobster's body (Waddy and Aiken, 1995) and the temperature at the edge of the continental shelf is too low (circa 6-8°C) for lobster reproduction. Thus the ovigerous females probably had to migrate from contaminated areas, closer to shore, and there, they incorporated the chemicals into their eggs. Migrating and remaining in colder, deeper, and presumably cleaner waters, allowed the ovigerous mothers to become decontaminated of most of their alkylphenols (1 contaminated out of 15), while their embryos which were more isolated than the mothers, by their relatively impervious egg shells, still remained contaminated (3 out of 5). These results suggest that lobsters, given enough time in relatively uncontaminated surroundings, can become decontaminated of alkylphenols.

We conclude that alkylphenols are endocrine disruptors in both annelids and lobsters, as well as vertebrates (the latter was shown by others) and that lobsters can be decontaminated by being held in clean waters over time. The shell disease analyses indicate that the lobster responds to the disease by increase molting hormone concentrations in the blood in preparation for a molt. Ovigerous and non-ovigerous animals appear to molt more frequently than uninfected lobsters.

ACKNOWLEDGMENTS

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REFERENCES

Biggers, W. and Laufer, H., 1996. Detection of Juvenile Hormone-Active Compounds by Larvae of the Marine Annelid *Capitella* sp. I. Arch. Insect Biochem. Physiol. 32: 475-484.

Biggers, W. and Laufer, H., 1999. Settlement and Metamorphosis of *Capitella* Larvae Induced by Juvenile Hormone-Active Compounds Is Mediated by Protein Kinase C and Ion Channels. Biol. Bull. 196: 187-198.

Biggers, W. and Laufer, H., 2004. Identification of Juvenile Hormone-Active Alkylphenols in the Lobster *Homarus americanus* and in Marine Sediments. Biol. Bull.,206: 13-24.

Chang, E.S., and O'Connor, J.D., 1988. Crustacea: Molting. In Endocronology of Selected Invertebrate Types. H. Laufer and R.G.H. Downer, eds. Invertebrate Endocrinology.
Volume 2 .A.R. Liss, New York Pub.,pp.259-278.

Laufer, H., Biggers, W., Johnson, M., Demir, N., and Bagshaw, J. 2003. Hormonal Responses of Lobsters to Stresses, an Interim Report.Third Long Island Sound Lobster Health Symposium. Connecticut Sea Grant –03-02.pp.32-37

Laufer, H., Demir, N. and Biggers, W. 2005.The Lobster's Response to The Stress of Shell Disease. Shellfish Research Journal, in press.

Waddy, S.L., Aiken, D.E., and De Kleijn, D.P.V. 1995. Control of Growth and Reproduction.
Chapter 10 In: Biology of the Lobster *Homarus americanus*, J.R. Factor,ed. Academic Press, New York, pp. 217-265.

Monitoring Phytoplankton in Long Island Sound with HPLC Photopigment Profiles

Li[1], Yaqin (Judy), Olsen[1], Christine, B., Lyman[1], Matthew, J., Van Heukelem[2], Laurie, Wikfors[3], Gary, Miranda[4], Lilibeth and Lin[4], Senjie
[1]Connecticut Dept. of Environmental Protection, 79 Elm Street, Hartford, CT 06106
[2]Horn Point Laboratory, Univ. of Maryland, Cambridge, MD 21613
[3]Milford Laboratory, Northeast Fisheries Science Center, NOAA, 212 Rogers Avenue, Milford, CT 06460
[4]Univ. of Connecticut, Dept. of Marine Sciences, Groton, CT 06340

ABSTRACT

The Connecticut Department of Environmental Protection started high-performance liquid chromatography (HPLC) photopigment monitoring in 2002 with funding from US EPA National Coastal Assessment Program. Seventeen stations throughout Long Island Sound were sampled monthly for photopigments in addition to other water quality parameters such as nutrients, oxygen, temperature, salinity and phytoplankton and zooplankton species and abundance. A total of 26 pigments including various chlorophylls and carotenoids were separated and quantified by HPLC. Total chlorophyll *a* concentration obtained by HPLC and a conventional fluorometric method showed a high level of agreement. Chemtax, a matrix factorization procedure based on proposed pigment ratios was used to calculate phytoplankton class composition. The chemotaxonomical approach revealed an overall diatom dominated phytoplankton community in the Sound. In the summer, the phytoplankton community was diverse with diatoms making up 30% of the total phytoplankton while dinoflagellates, cryptophytes, prasinophytes, euglenophytes, chlorophytes and cyanophytes together contributed about 70% of the total phytoplankton. In contrast, diatoms make up about 80% of the total biomass in the winter. The chemotaxonomical analysis results were consistent with those obtained previously by microscopic methods. The chemotaxonomical approach was able to quantify a number of small-sized phytoplankton classes.

INTRODUCTION

As major primary producers for most of the estuaries, the phytoplankton community affects the nutrient dynamics, food web structure and function and overall health of estuaries. Understanding phytoplankton community structure and dynamics is essential in understanding the structure and function of estuarine ecosystems. Microscopic examination is a classic way of studying phytoplankton community composition. While this method gives detailed size and species information, the limitation is that nano- (2-20 μm) and pico- (0.2-2 μm) sized phytoplankton and other hard-to-preserve species may be missing from the microscopic method. The microscopic method also requires high taxonomic skill, is very labor- intensive and can therefore be impractical for large temporal and spatial scales.

Alternatively, phytoplankton composition may be quantified by their diagnostic pigments. High-performance liquid chromatography (HPLC) is an effective technique for separating and quantifying pigments in natural water samples. Characteristic pigments such as zeaxanthin, alloxanthin, divinyl chlorophyll *a* and divinyl chlorophyll *b*, prasinoxanthin, peridinin and phaeophytin *a* are capable of fingerprinting algal groups and recycling processes (Barlow et al. 1997). Total phytoplankton biomass (standing stock) is estimated by chlorophyll *a* and algal classes are identified by the presence of diagnostic carotenoid pigments. Quantitatively, each group can be calculated based on chlorophyll *a*:carotenoid pigment ratios (Wright & Jeffrey 1987). One of the two approaches for such purposes is a multiple linear regression analysis based on the assumption that each algal class is characterized by its main carotenoid pigment (such as Tester et al. 1995). Another approach such as CHEMTAX is through a matrix factorization from proposed pigment ratios (Mackey et al. 1997).

Long Island Sound is a temperate estuary located on the east coast of USA bordering the states of Connecticut and New York. Long Island Sound has been designated as an estuary of significance due to its value to the region including recreation, commercial fishery, etc. As other estuaries in the nation, Long Island Sound is undergoing eutrophication processes, particularly in the western part of the Sound. Low oxygen conditions (hypoxia) exist in the bottom water of Western Long Island Sound almost every summer (Kaputa & Olsen 2000).

Understanding the effect of eutrophication on hypoxia and overall water quality of Long Island Sound requires information about the spatial and temporal distribution of phytoplankton community. The objective of this project is to analyze the community structure of phytoplankton in Long Island Sound through the pigment information with the aid of conventional microscopic methods.

MATERIALS AND METHODS

Sample collection

Water samples were collected monthly from 17 stations throughout Long Island Sound (Figure 1). For HPLC pigment analysis, all of the 17 stations were sampled from April 2002 to August 2003 and 10 of the 17 stations were used thereafter. 200 or 400ml of surface water (2 meters below surface) was filtered through a Whatman GF/F glass fiber filter (0.7 µm pore size). Wrapped in aluminum foils, filters were frozen on board and transferred to a -80° C freezer after each cruise. For microscopic examination, 200ml of surface water was collected from each station and fixed with 4ml of Lugol solution (final concentration 1%). This sampling is a part of a larger sampling effort of CT DEP Long Island Sound Monitoring Program.

HPLC pigment analyses

Three ml of 95% acetone containing the internal standard, Vitamin E acetate (Fluka Chemical Corporation, Milwaukee, WI), was added (with a calibrated measuring device) to each filter. The filters (now submerged in acetone) were chilled and then disrupted with an ultra-sonic probe. The sample extracts were analyzed with methods fully described in Hooker et al. (submitted) and Van Heukelem and Thomas (2001) using a fully automated series 1100 HPLC system from Agilent Technologies (Wilmington, DE), which included an automated injector with "mix in the loop" injection programming and refrigerated auto-sampler compartment, a thermostatted column oven compartment, photodiode array detector, and computer data-station with Hewlett Packard Chemstation software. Pigments were identified by matching retention times and HPLC in-line absorbance spectra with known standards, which were either isolated from natural sources (according to Van Heukelem and Thomas 2001) or purchased from commercial vendors, Sigma-Aldrich Company (St. Louis, MI) or DHI (DHI Water and Environment Institute (HØsholm, Denmark).

CHEMTAX phytoplankton class analysis

The computer software, CHEMTAX (Mackey et al. 1997), was used to calculate phytoplankton class composition based on the pigments present in the water samples (pigment data) and the known pigment ratio for target groups (initial pigment ratio). Ten phytoplankton classes / groups including diatoms, dinoflagellates, cryptophytes and other groups were analyzed (Table 1). Prymnesiophyceae was divided into two groups, A and B, for which the former has a high but-fucoxanthin content and the latter does not. The selection of 10 target phytoplankton groups was based on pigments present in the water samples and previous and current species records from microscopic identification. For example, Prochlorophytes were not included because their diagnostic pigment, divinyl chlorophyll *a*, was not detected in collected water samples.

The initial pigment ratios (Table 1) required by Chemtax software were developed based on pigment ratios of cultured species isolated from Long Island Sound and nearby estuaries. Cultures of 23 species from 8 classes namely Bacilliarophyceae, Prymnesiophyceae, Dinophyceae, Chlorophyceae, Prasinophyceae, Cyanophyceae, Raphidophyceae and Cryptophyceae were used for developing initial pigment ratios. Cultures were collected during their exponential growth phase for pigment content analysis. Supplemental ratios were from Lewitus (personal communication 2004), Mackey et al. (1997), Gin et al. (2003) and Riegman & Kraay (2001). The Prepro settings for Chemtax were: weighting = bounded relative error by pigment; iteration limit = 500; epsilon limit = 0.005; initial step size = 10; step ratio = 1.3000; cutoff step = 1000; verbosity = normal; elements varied = 5; subiterations = 5 and weight bound = 30.

Chemtax results with the initial ratio from Long Island Sound cultured species (LIS ratio) were compared with results using the initial ratios from Chemtax or Lewitus (personal communication with Lewitus 2004). The Chemtax ratios were from the Table 1 of Mackey et al. (1997). The species used in Chemtax were mostly oceanic, while species used by Lewitus (personal communication 2004) were estuarine.

RESULTS AND DISCUSSIONS

Chemical taxonomic analysis using HPLC pigment profiles revealed that diatoms were the most abundant phytoplankton group in the Sound (Table 2). This is in agreement with previous reports (Conover 1959, Capriulo et al. 1996). A list of eight most abundant groups and their contribution to total phytoplankton pigment are presented in Table 2.

Table 2. Percentage contribution of each phytoplankton group / class to total pigment. Data are averages of all 368 samples taken from April 2002 to June 2004 throughout the Sound. (*only the major forms of dinoflagellates, which contain peridinin, are measured).

Group / class	% contribution
Bacillariophyceae (diatoms)	51.03
Cryptophyceae	14.01
Dinophyceae (dinoflagellates) *	9.14
Prymnesiophyceae (both types)	7.50
Euglenophyceae	6.85
Cyanophyceae (cyanobacteria)	4.24
Prasinophyceae	3.94
Raphidophyceae	1.98
Chrysophyceae	0.79
Chlorophyceae	0.28

The chemical taxonomic analysis apparently identifies more classes/groups of phytoplankton than the microscopic method. Based on a 2-year study from 3 stations in the Sound, Capriulo et. al. (1996) recorded 7 classes / groups of phytoplankton, namely diatoms, dinoflagellates, cryptophytes, euglenophytes, pymnesiophytes, chlorophytes and chrysophytes. Our recent microscopic analysis of water samples from over 10 stations throughout the Sound from October 2001 to July 2004 showed similar results. Both of the above microscopic studies resulted in a significant number of unidentified phytoplankton, mostly small flagellates and green algae. The chemical taxonomic method was able to identify and quantify more groups than the microscopic method because some small sized phytoplankton can be differentiated by pigment content but not by microscopic evaluation and some species are missed by microscopic evaluation because they do not preserve well.

There was a distinct seasonal pattern in phytoplankton in Long Island Sound (Figure 2). Phytoplankton diversity was higher in the summer than in the winter. Diatoms dominated in the winters and accounted for about 70-90% of the total phytoplankton biomass (as approximated by total pigments) from December to February. Although diatoms were still dominant in the summer (about 30% of total pigments), the phytoplankton community was much more diverse. From June to August, dinoflagellates and prasinophytes exhibited the highest amounts, relative to total phytoplankton. In contrast to diatoms, dinoflagellates and prasinophytes, cryptophytes did not show a clear seasonal pattern in the Sound.

Although there was a clear spatial gradient in phytoplankton biomass in the Sound, with higher biomass in the western Sound and lower biomass in the eastern Sound, as documented previously (Li et al. 2002), the community composition did not exhibit such a gradient. Figure 3 shows the distribution of the 3 most abundant phytoplankton groups at 3 stations across the Sound. The distribution of these 3 phytoplankton groups was extremely similar at all 3 stations. Caprilo et. al. (1996) reported similar results based on a 2 year sampling from one station located in the Eastern Narrow near Stamford and 2 stations located in the Central Sound near Milford and Hammonassett, respectively. Despite the similar community composition at class level, the community composition at species level was different from station to station.

The output and accuracy of the chemical taxonomic method depends largely upon the initial pigment ratios given to the software. We compared results from using our own pigment ratios obtained through species from Long Island Sound and nearby estuaries with results using ratios from Mackey et al. (1997) and Lewitus (personal communication 2004). The Mackey et al. (1997) ratios underestimated diatoms and euglenophytes and overestimated Prasinophyceae, Cryptophyceae and Haptophyceae. The two approaches (with Mackey et al. 1997 ratios and Lewitus 2004 ratios) had a good agreement for dinoflagellates. The Mackey et al. (1997) ratios are based on species from oceanic water where diatoms are not as important as in the coastal and estuarine waters. The results from using the Long Island ratios and the Lewitus (2004) ratios were very close. This is not surprising because both sets of ratios were obtained with coastal species.

As with other methods of estimating phytoplankton community composition, the chemical taxonomic approach has its inherent limitations. The method only yields composition at the class level, with a few exceptions such as for *Karenia brevis* where the species is identified by a unique pigment gyroxanthin. There is always uncertainty in the results due mainly to the change in phytoplankton's pigment concentrations, as affected by environmental conditions such as light intensity, nutrient availability as well as their physiological status. In order to improve the estimation, the initial pigment ratio matrix will be updated with more pigment data from locally isolated species.

Our next step is to further assess whether Chemtax has provided a good estimate of phytoplankton community composition by comparing the results from Chemtax to that of microscopic observation. The cell counts data from microscopic observation will be turned into bio-volume based on their shapes and sizes (Hillebrand et al. 1999). The bio-volume will then be converted to carbon and pigment based on the ratios that have been published (e.g., Menden-Deuner and Lessard 2000).

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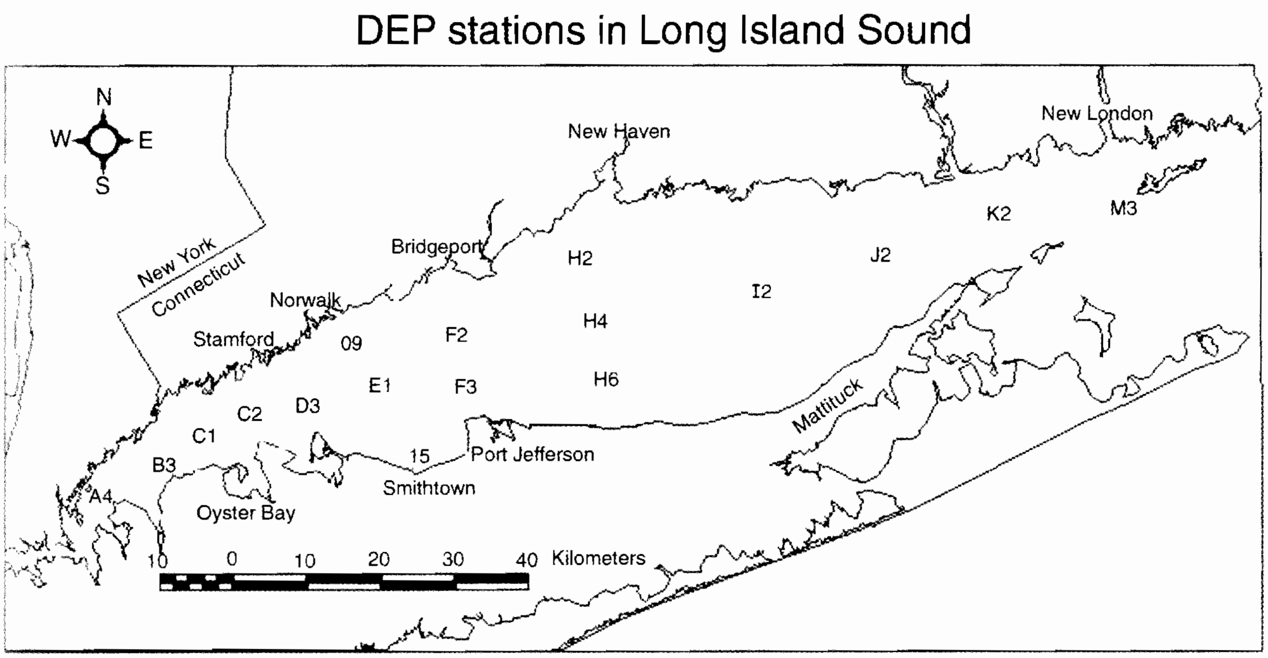

FIGURE 1. Map of sampling stations. All of the 17 stations were sampled from April 2002 to August 2003. Stations A4, B3, C1, D3, E1, F2, H4, I2, J2 and K2 were sampling stations since September 2003.

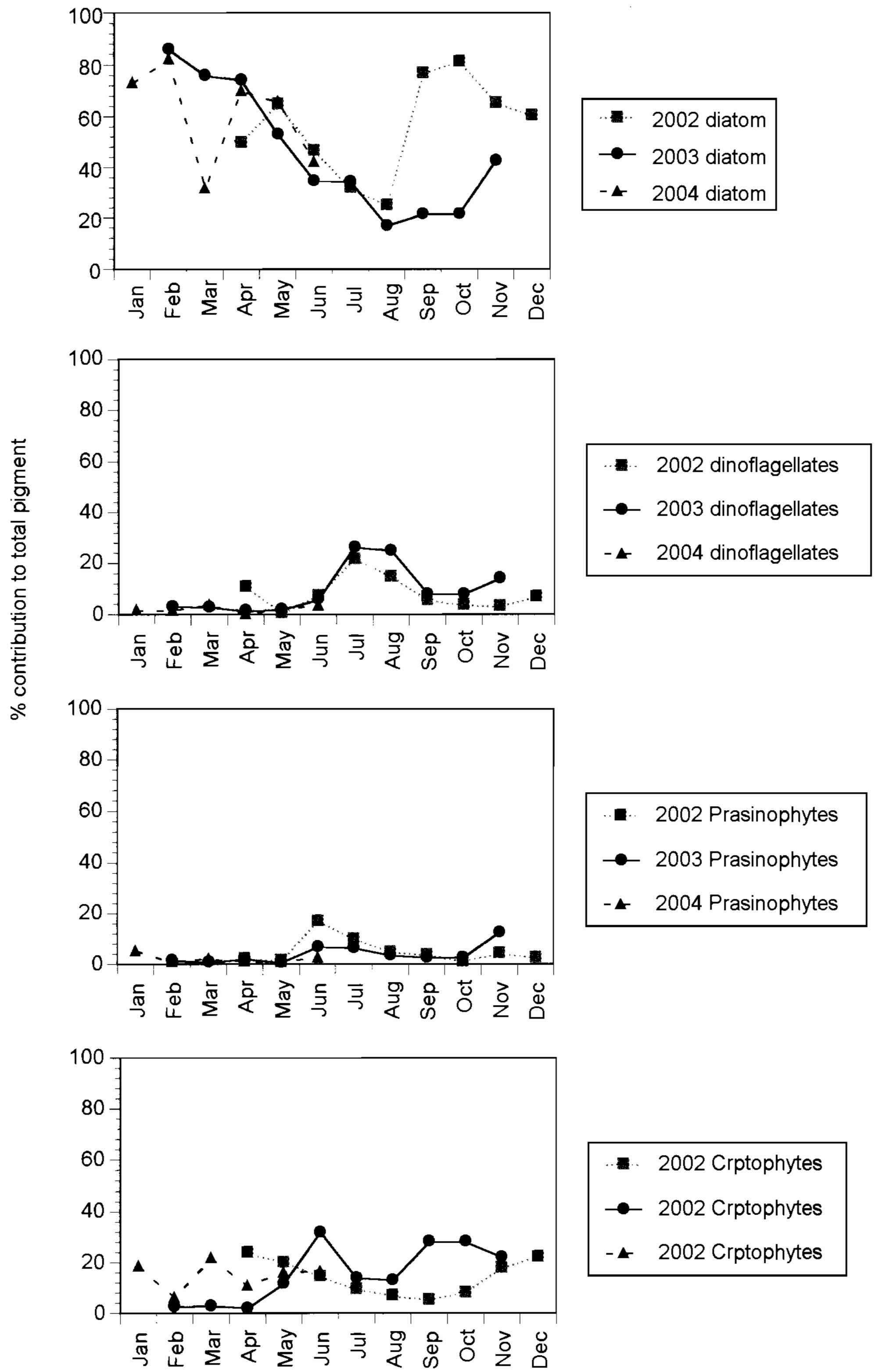

FIGURE 2. % contribution to total pigment from major phytoplankton groups. Data were average of all sampling stations for the given month.

LITERATURES CITED:

Barlow, R. G., Mantoura, R. F. C., Cummings, D. G. and Fileman, T. W. 1997. Pigment chemotaxonomic distributions of phytoplankton during summer in the western Mediterranean. Deep-Sea Research II, 44: 833-850.

Capriulo et al. 2002. The plankton food web structure of a temperate zone estuary, and its alteration due to eutrophication. Hydrobiologia 475/476 263-333.

Conover, S. A. M. 1956. Phytoplankton. In Riley, G. A., Conover, S. A. M., Deevey, G. B., Wheatland, S. B., Harris, E. and Sanders, H. [ed.] Oceanography of Long Island Sound, 1952-1954, Bulletin of the Bingham Oceanographic Collection. Peabody Museum of Natural History, Yale University.

CT DEP 2002. Long Island Sound Ambient Water Quality Monitoring Program Quality Assurance Project Plan. EPA approved through 2006.

Gin, K.Y.H., Zhang, S. and Lee, Y.K. 2003. Phytoplankton community structure in Singapore's coastal waters using HPLC pigment analysis and flow cytometry. *Journal of Plankton Research* 25: 1507-1519.

Hooker S.B., Van Heukelem, L., Thomas C.S., Claustre, H., Ras, J., Barlow, R., Sessions, H., Schluter, L., Perl, J., Trees, C., Stuart B., Head, E., Clementson, L., Fishwick, J., Llwewllyn, C., and Aiken, J. 2004. The Second SeaWiFS HPLC Analysis Round-Robin Experiment (SeaHARRE-2). NASA Technical Memorandum. NASA Goddard Space Flight Center, Greenbelt , Maryland., submitted.

Kaputa, N. P. and Olsen C. B. 2000. Long Island Sound Ambient Water Quality Monitoring Program: Summer Hypoxia Monitoring Survey 1991-1998 Data Review. State of Connecticut, Department of Environmental Protection.

Mackey, M. D., Higgins, H. W., Mackey, D. J. and Wright, S. W. 1997. *CHEMTAX user's manual: a program for estimating class abundances from chemical markers – application to HPLC measurements of phytoplankton pigments.* CSIRO Marine Laboratories Report 229, Hobart, Australia, ISBN 0 643 06040 5, 47 pp.

Menden-Deuner, S. and Lessard, E. J. 2000. Carbon to volume relationships for dinoflagellates, diatoms, and other protist plankton. Limnology & Oceanography 45: 569-579.

Paerl, H.W., Valdes, L.M., Pinckney, J. L., Piehler, M. F., Dyble, J. and Moisaander, P. H. 2003. Phytoplankton photopigments as indicators of estuarine and coastal eutrophication. *BioScience* 53: 1-12.

Riegman, R. and Kraay, G.W. 2001. Phytoplankton community structure derived from HPLC analysis of pigments in the Faroe-Shetland Channel during summer 1999: the distribution of taxonomic groups in relation to physical / chemical conditions in the photic zone. *Journal of Plankton Research* 23: 191-205.

Tanoue, E., Suzuki, K., Handa, N., Nojiri, Y. and Wong, C. S. 2002. Spatial and temporal variabilities of phytoplankton community structure in the northern North Pacific as determined by phytoplankton pigments. *Deep-Sea Research* (part 1) 48: 439 –469.

Tester. P. A., Geesey, M. E., Guo, Chunzhi, Paerl, H. W. and Millie, D. F. 1995. Evaluation phytoplankton dynamics in the Newport River estuary (North Carolina, USA) by HPLC-derived pigment profiles. *Marine Ecology Progress Series* 124: 237-245.

Van Heukelem, L. and Crystal, S. T. 2001. Computer-assisted high-performance liquid chromatography method development with applications to the isolation and analysis of phytoplankton pigments. *Journal of Chromatography* A , 910: 31-49.

Wright, S. W. and Jeffery S. W. 1987. Fucoxanthin pigment markers of marine phytoplankton analysed by HPLC and HPTLC. *Marine Ecology Progress Series* 38:259-266

TABLE 1. Initial pigment ratio matrix used for Chemtax. Pigment abbreviations: c1 = chlorophyll c1, c2 = chlorophyll c2, per = peridinin, but = 19'-butanoyloxyfucoxanthin, fucoxanthin, hex = 19'-hexanoyloxyfucoxanthin, neo = neoxanthin, pra = prasinoxanthin, vio = violaxanthin, ddx = diadinoxanthin, all = alloxanthin, dtx = diatoxanthin, lut = lutein, zea = zeaxanthin, b = chlorophyll b, a = chlorophyll a.

	c1	c2	per	but	fuc	hex	neo	pra	vio	ddx	all	dtx	lut	zea	b	a
Diatoms	9.36	7.76	0	0	58.01	0	0	0	0	10.26	0	4.13	0	0	0	100
Dinoflagellates	0.64	31.77	61.67	0	0	0	0	0	0	12.82	0	0.37	0	0	0	100
Cyanophyceae	0	0	0	0	0	0	0	0	0	0	0	0	0	14.09	0	100
Prasinophyceae	0	0	0	0	0	0	5.25	0	6.43	0	0	0	6.55	0.33	50.08	100
Chlorophyceae	0	0	0	0	0	0	5.23	0	4.70	0	0	0	14.92	0.33	27.75	100
Cryptophyceae	0	12.05	0	0	0	0	0	0	0	0	25.69	0	0	0	0	100
Prymnesiophyceae A	7.76	6.36	0	0	41.75	0	0	0	0	15.74	0	1.80	0	0	0	100
Prymnesiophyceae B	0	0	0	17.50	41.20	49.00	0	0	0	24.30	0	9.00	0	0	0	100
Raphidophyceae	4.14	7.10	0	0	39.10	0	0	0	11.7	0	0	0	0	1.89	0	100
Chrysophyceae	0	12.7	0	7.73	54.30	0	0	0	0	43.80	0	5.10	0	0	0	100
Euglenophyceae	0	0	0	0	0	0	7.70	0	0	8.60	0	5.80	2.20	2.00	85.8	100

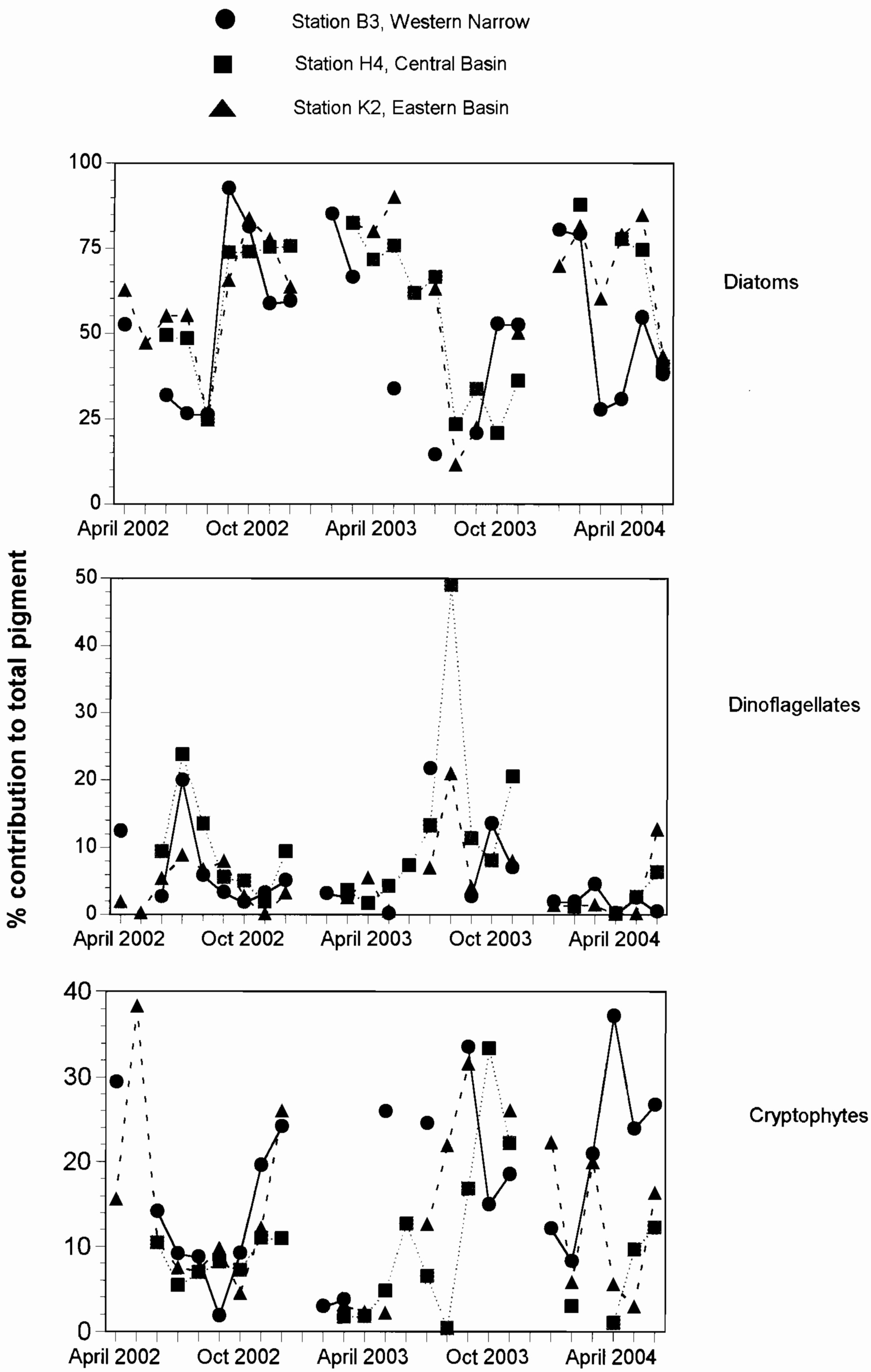

FIGURE 3. Distributions of 3 major phytoplankton groups across Long Island Sound.a

Remote sensing of suspended matter in Long Island Sound

Karl-Heinz Szekielda
The Graduate School and University Center
Ph.D. Program in Earth and Environmental Sciences
City University of New York

INTRODUCTION

Remote sensing distinction between water types in coastal waters (case 2 water) carrying high concentration of total suspended sediments (TSS), chromophoric matter (CDOM) and chlorophyll is not completely explained, especially in estuary-types of coastal environments such as the Long Island Sound. On the other hand, the correlation between reflectance and absorption of chlorophyll, CDOM and TSS at specific wavelengths, provides an opportunity to derive a distinction between contributors to the reflectance. Therefore, coastal research may take advantage of data archives of multi-spectral recordings made by aircraft derived digital imagery, the Coastal Zone Color Scanner (CZCS), SeaWiFS, Landsat and MODIS and ASTER.

Remote sensing techniques can be applied to identify marine provinces using spectral bands in the region where chlorophyll absorbs incident radiance. As reference wavelength serves a spectral region where absorption by pigments is minor and ratios can be build that relates to photosynthetic pigments. The ratios 443 nm/550 nm and 678 nm/ 667 nm are indices of chlorophyll; however, as photon penetration depth varies at applied wavelengths, each ratio responds to different depths. This allows a differentiation between various biogeochemical provinces. Bio-geochemical provinces are characterized by their variety in bio-geochemical activity due to physical and chemical changes in the marine environment. Primary forces in the biogeochemical provinces include vertical mixing rates, stratification of the euphotic zone, nutrient supply and irradiance at the sea surface. Major modifications of these forcing factors result from changes in surface circulation that defines the location and boundaries of provinces with varying primary productivity. Therefore, partitioning the coastal regions into bio-geochemical domains and provinces can be derived from seasonal cycles of plankton development, the optical field, current systems and the associated mixing rate.

Whereas in the open ocean (Case I water), color can be converted to total pigment concentrations (see for instance Morel and Maritorena, (2001), there are shortcomings in determining photosynthetic pigments in coastal regimes with Case II water. This is due to the fact that the water-leaving radiance in those regions varies in connection with the changing composition of the main contributor to the water-leaving radiance. The contribution of inelastic sources to water-leaving radiance has been subjected to modeling efforts for instance Schroeder et al. (2003), showing the important role of sunlight-stimulated fluorescence and Raman scattering of water (Waters, 1995). In Case II water, chromophoric dissolved organic compounds (CDOM), inorganic particulate matter, organic debris and phytoplankton vary in their relative compositions, and consequently, do not necessarily co-vary with the water-leaving radiance and pigment concentrations. This decoupling between spectral radiance and pigments is particularly a shortcoming in coastal waters where high pigment concentrations are present due to coastal dynamics and transport of nutrients through river effluent and discharge from waste water treatment facilities. Furthermore, the inherent variations of one class of material to another limits the use of regionally developed algorithms for retrieving mass concentrations from optical data (McKee et al., 1999).

To further elaborate on the interpretation of water-leaving spectral signatures, for categorizing biogeochemical provinces in the Long Island Sound, the following will focus on using the spectral regions where chlorophyll has its Soret band and its second absorption alpha-band in the red part of the electromagnetic spectrum. High correlation was found between chlorophyll concentration and the water-leaving radiance in this very same spectral region where sun-induced fluorescence of chlorophyll is also present, (Gower and Borstad, 1981, Bricaud et al.1991, Garcia and Maske,1996, Gower et al.,1999 , Roesler and Perry, 1995, Szekielda et al., 2003). Solar-induced chlorophyll fluorescence was used by Hoge and Swift (1987) to study the ocean color spectral variability while Gitelson (1992) analyzed the peak near 700 nm on radiance spectra and found a good relationship with chlorophyll concentrations.

RESULTS

The division of multi-spectral measurements into distinct regions through interactive analysis of two-dimensional scatter plots permits the identification of clusters that relate to specific oceanic regimes. The principle of identifying a pattern classifier to compare different depth horizons, according to photon penetration depth, has been tested over several oceanic regions with MODIS data (Szekielda, 2004) and was applied in the study of the Long Island Sound.

Figure 1 shows the chlorophyll concentration patterns as derived with the standard algorithm that is applied to MODIS data. The major problem associated with this algorithm is that it brakes down in coastal Case 2 water and no additional information is obtained to differentiate the various water constituents and their origin through river discharge, sediment erosion and eutrophication.

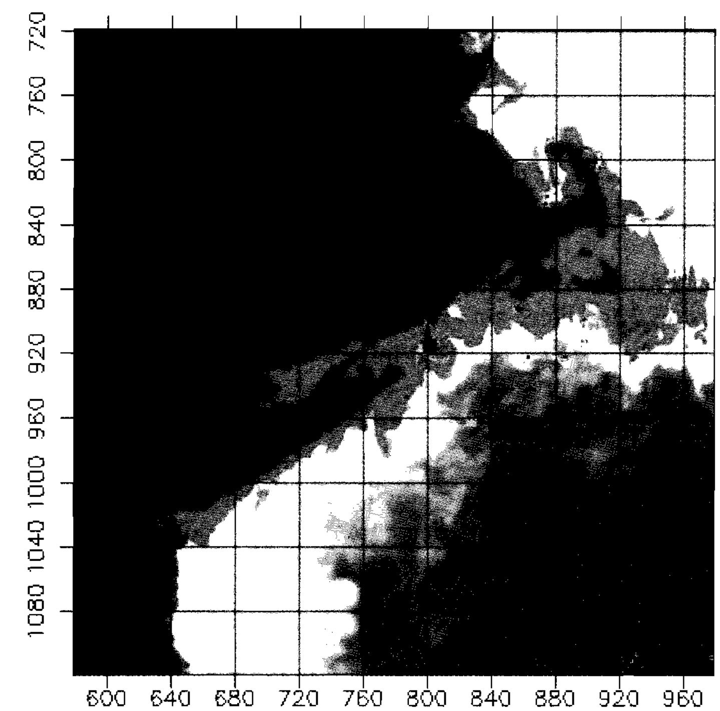

FIGURE 1: Chlorophyll distribution along the East coast of the United States based on MODIS observations on September 8, 2002.

As the components in suspended matter in the coastal environment respond spectrally differently from each other and also due to the fact that the photon penetration depth is a function of wavelength, it was found that the common chlorophyll retrieval based on the ratio 443 nm/550 nm is not necessarily correlated with ratios at longer wavelengths.

Variability in the spectral response of the biogeneous compartments and decoupling between chlorophyll, chromophoric dissolved organic compounds (CDOM), inorganic suspended matter, organic detritus and bacteria, make it attractive to test the possible recognition of biogeochemical provinces in cluster diagrams using various spectral wavelengths. This concept was proven by comparing the ratios 443 nm/550 nm with 667 nm/678 nm. The distinctive clustering that appears in spectral space allows therefore, the use of two dimensional scatter plots to design simple pattern classifiers which have the advantage of separating biogeochemical provinces based on the use of the first and second absorption band of chlorophyll. Consequently, multi-spectral satellite recordings can identify regions where high concentrations of plankton occur or where the composition of suspended and dissolved matter changes.

Figure 2 gives an example for the East coast of the United Sates and the Long Island Sound in which the clustering of data for the two ratios identified spectrally the various water masses. Typically, the open ocean responds to the chlorophyll concentration throughout the depth where about 90% of the chlorophyll is located. Coastal waters, due to their different spectral properties and high concentrations of chlorophyll, CDOM and inorganic suspended matter, absorb much higher and the major response is from the upper meters only and the first absorption band is strongly influenced by CDOM. At longer wavelengths, CDOM absorb less and the second chlorophyll absorption band at around 676 nm with the sun-induced fluorescence at around 678 nm can be used to estimate chlorophyll, particularly at higher concentrations.

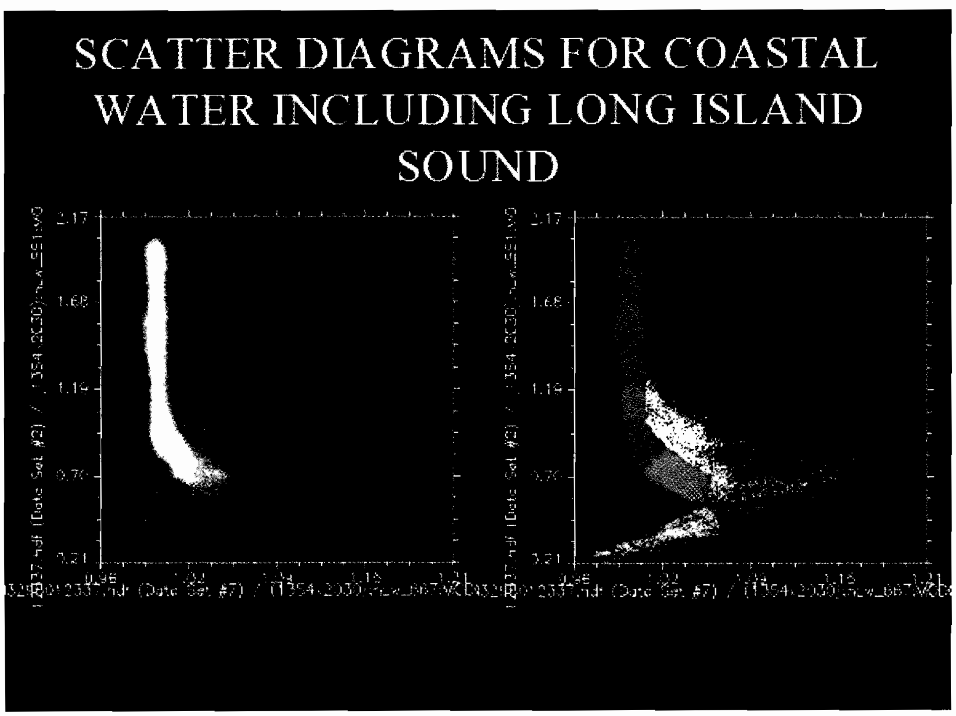

FIGURE 2: Scatter diagram for the region shown in Figure 1 for the ratio 443 nm/551 nm versus the ratio 678 nm/667 nm. Right side shows the corresponding identified cluster used to identify regions of interest presented in Figure 3.

The identified clusters were displayed as referenced regions of interest and are shown in Figure 3. In order to interpret the clusters as well as the mapped regions of interest, it is meaningful to look into the region that characterizes open ocean water. According to the low photon penetration depth, the ratio 678 nm/ 667 nm will be close to constant because the low concentration of chlorophyll and the water-leaving reflectance has been received from a short water column. The ratio 443 nm/551 nm in contrast shows the reflectance over a deeper water column and absorption of incident light by chlorophyll at the spectral region where the Soret band is located. Figures 2 and 3 with color annotation in purple identify the open ocean water with low chlorophyll concentrations. While the ratio 443 nm/551 nm decreases with increasing chlorophyll concentration, the ratio 678 nm/667 nm increases with increasing chlorophyll concentrations and as a result, an inverse relationship exists.

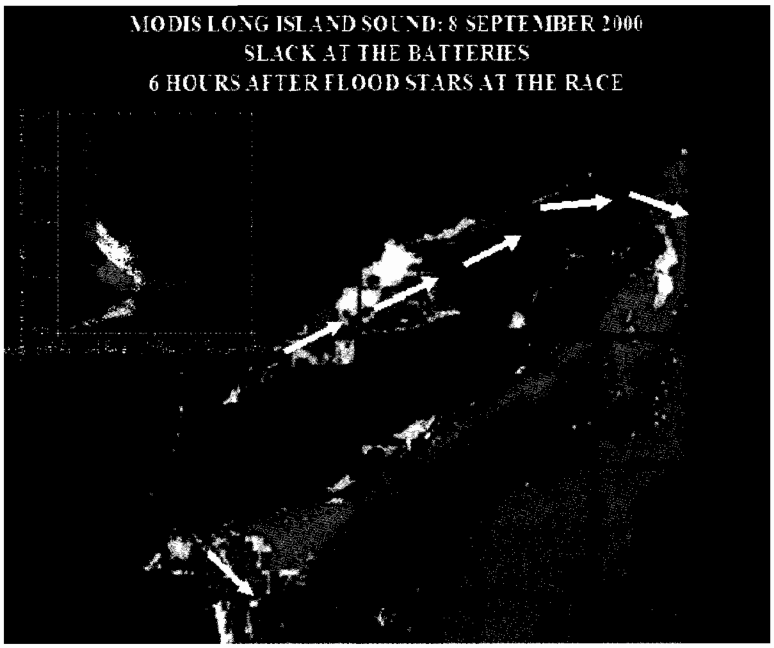

FIGURE 3: Identification of different bio-geochemical provinces based on regions of interest as shown in Figure 2.

CONCLUSION

Even though the results are preliminary, it has been demonstrated that differentiating various biogeochemical provinces can be achieved with the use of the spectral chlorophyll absorption region of the alpha-band and the Soret band. Essentially, the depth location of chlorophyll is a factor that must be considered when working at longer wavelengths. Above all, estimates of chlorophyll have to take into account that the photon penetration depth is a function of wavelength. Therefore, it can be postulated that the chlorophyll estimation based on the ratio 443 nm/551 nm is not necessarily correlated with the ratio 678 nm/667 nm, the latter being responsive to high chlorophyll concentration in the upper part of the water column only. The 443 nm/551 nm ratio is affected greatly by absorption of detritus, chromophoric dissolved organic matter (CDOM) as well as scattering by all suspended matter while the red range is much less affected by CDOM absorption. In addition, variability in the biogeneous compartment and decoupling between chlorophyll, chromophoric dissolved organic matter, inorganic suspended matter, organic detritus and bacteria may be the cause for recognition of biogeochemical provinces in cluster diagrams. Therefore, distinctive clustering appears in spectral space allowing the use of two dimensional scatter plots to design a pattern classifier. It can be concluded that the above approach can be used as separating biogeochemical provinces in connection with chlorophyll measurements based on the use of the first and second absorption band of chlorophyll.

REFERENCES

Bricaud, A., C. Roesler, and J.R. V. Zaneveld, 1995. In situ methods for measuring the inherent optical properties of ocean waters. Limnol. Oceanogr., 40 (2), 393-410.

Garcia-Mendoza, E., and H. Maske, 1996. The relationship of solar-induced natural fluorescence and primary productivity in Mexican Pacific waters. Limn. Oceanogr., 41(8), 1697-1710.

Gitelson, A., 1992. The peak near 700 nm on radiance spectra of algae and water: relationship of its magnitude and position with chlorophyll concentration. International Journal of Remote Sensing, 13, 3367-3373.

Gower, J.F., and G. Borstad, 1981. Use of the in vivo fluorescence line at 685 nm for remote sensing surveys of surface chlorophyll a p. 329-338. In: J.F. Gower [ed.], Oceanography from the space. Plenum.

Gower, J.F.R., R. Doerfer, and G.A. Borstad, 1999. Interpretation of the 685 nm peak in water-leaving radiance spectra in terms of fluorescence, absorption and scattering, and its observation by MERIS. Int. J. Remote Sensing, 20, 9, 1771-1786.

Hoge, F., and R. Swift, 1987. Ocean color spectral variability studies using solar induced chlorophyll fluorescence. Applied Optics, 26, 18-21.

McKee, D., A. Cunningham, and K. Jones, 1999. Simultaneous measurements of fluorescence and beam attenuation: instrument characterization and interpretation of signals from stratified coastal waters. Estuarine, Coastal and Shelf Science, 48, 51-58.

Morel, A., and S. Maritorena, 2001. Bio-optical properties of oceanic waters: A reappraisal. J. Geophysdical Res., 106, 7163-7180.

Roesler, C.S., and M.J. Perry, 1995. In situ phytoplankton absorption, fluorescence emission, and particulate backscattering spectra determined from reflectance. J. Geophys. Res., 100, 13,279-13,294.

Schroeder, M., H. Barth, and R. Reuter, 2003. Effect of inelastic scattering on underwater daylight in the ocean: model evaluation, validation, and first results. Applied Optics, 42(21), 4244-4260.

Szekielda, K.H., C. Gobler, F. Moshary, B. Gross and S. Ahmed, 2003. Spectral reflectance measurements of estuarine waters, Ocean Dynamics, 53, 98-102.

Szekielda, K.H., 2005. Pattern recognition of marine provinces. International J. Remote Sensing, 26, 1499-1503.

Salt Marsh Change, 1926-2003 at Marshlands Conservancy, New York

Ellen Kracauer Hartig[1,2] and Vivien Gornitz[3]

[1]*Friends of Marshlands, Inc. PO Box 237, Rye NY 10580*
[2]*Great Eastern Ecology, Inc. 2231 Broadway, Suite 4, New York NY 10024, eh291@columbia.edu*
[3]*Center for Climate Systems Research/Columbia University and NASA/Goddard Institute for Space Studies, 2880 Broadway, New York, New York, USA 10025, vgornitz@giss.nasa.gov*

Although historic rates of salt marsh accretion in the region have generally exceeded or kept pace with sea-level rise, marsh loss has been observed in parts Connecticut (Anisfeld, 2003) and southern Long Island (Hartig et al., 2002; Mushacke, 2003). In New England, low marsh cordgrass (*Spartina alterniflora*) has migrated landward (Bertness et al., 2002). Many of these coastal wetlands have experienced anthropogenic modifications that could have initiated marsh deterioration and loss, but the regional pattern of relative sea-level rise may also exacerbate these anthropogenic effects.

Marshlands Conservancy, part of Westchester County Department of Parks, Rye, NY, contains the largest remaining salt marsh in New York State outside of Long Island (N 40° 57' and W 73° 41'). The tidal range–around 2 meters–is somewhat higher than average for the region and elevations change sharply from marsh to field to forested steep slope. Changes at Marshlands Conservancy salt marshes were measured by standard aerial photo interpretation techniques using digitized black and white, color, and color infrared aerial photographs from 1926, 1974 and 2000 acquired from Westchester County Planning Department and NYSDEC. Using the 1974 to more recent aerial photography, results were compared with sites along the northern and southern shoreline of Long Island (Mushacke, 2003) and with Jamaica Bay (Hartig et al., 2002).

General land loss at Marshlands Conservancy became immediately apparent upon examination of aerial photographs (Fig. 1). Several trends were clearly identified: 1) large-scale marsh losses to the southwest and northeast of the study area (points A and E); and 2) widening of tidal channels and tributaries in the central core (points B, C, and D), culminating in enlarged tidal pools.

As of 2000, the central (core) portion of the study area contained the most remaining intact marsh (8.4 ha), down from 11.3 ha in 1926–a 26% reduction in marsh area (Table 1). The southwestern section had only 0.2 ha remaining in 2000, down from 2.4 ha in 1926–a 90% reduction, while the northeastern section went to 1.1 ha in 2000, from 3.2 ha in 1926–a 66% reduction. Average annual percent losses overall were greater during the 1974-2000 period (1.2%/yr) than in the earlier period from 1926 to 1974 (0.35%/yr). The total remaining marsh area in 2000 for all three sections was 9.7 ha, down from 16.9 ha in 1924, representing a 43% reduction (or an average loss of 0.58% yr over the entire 74 year period).

Numerous areas of erosion and flooding were detected during 2003 field surveys, with a few exceptions. Marsh expansion occurred in two places–the western cove and along a sandy area in the central core–but the size of these two *Spartina alterniflora* stands during onsite measurements in 2003 was less than a quarter acre and were not large enough to compensate for marsh loss. Field observations (May 2003) revealed considerable dissection and fragmentation of the low marsh which had been contiguous on older photographs. Geomorphological indicators of marsh loss at Marshlands Conservancy include tidal channel widening, expansion of tidal pools, ponding at the head of tidal creeks, slumping, perimeter erosion of low marsh, and conversion to mudflats.

A comparison of selected sites around the region (Table 2) indicates that the salt marshes have been disappearing at average rates of 1.2% per year (at Marshlands Conservancy, Rye, NY) to 3.1% per year (at Manhasset Bay) since 1974 (Hartig et al. 2002; Mushacke, 2003, Fallon and Mushacke, 1996). Among sites examined, Jamaica

Bay has experienced the largest total areal loss with an average rate of 1.5% per year. In several cases salt marsh islands have become completely submerged. For example, in Shinnecock Bay (southeastern Long Island) only 7 out of 13 salt marsh islands that were present in 1974 remained by 1994 (Fallon and Mushacke, 1996). The apparent submergence of these islands was partially compensated by inland migration of other salt marshes along the shoreline, suggesting that sea-level rise was a contributing factor.

The observed indications of peripheral erosion and inundation at Marshlands Conservancy (Fig. 1) could represent an early warning sign of the effects of rising sea level, although other localized effects probably predominate at present. Some likely hypotheses of marsh deterioration can be grouped into three main categories: climate change, anthropogenic-induced stresses, and ecological factors.

Climate Change—Short-term accelerations of sea level and changes in storm patterns have been suggested as potential triggers of marsh loss. During the 20th century, local Connecticut sea level has risen 21-26 cm (Gornitz et al., 2004). While this historic rise in sea level may have contributed to some marsh loss, it cannot completely explain the recently <u>accelerating</u> trends found at Marshlands Conservancy (Table 1) or at Jamaica Bay (Hartig et al., 2002), inasmuch as the overall rate of relative sea level rise has remained relatively constant throughout this period (Gornitz et al., 2002). Although short-term sea level accelerations several years in duration, superimposed on the long-term trend, appear on local tide gauge records (NOAA, 2003), their effects on salt marshes are probably not significant inasmuch as they are closely followed by several years of falling sea levels.

Anthropogenic Stresses—Most wetland losses in Marshlands Conservancy are probably linked directly or indirectly to anthropogenic activities. Dredging of a navigation channel by the Army Corps of Engineers, re-alignment of a canal at a nearby golf course property, and construction of a marina may all have strengthened tidal currents, thereby enhancing erosion. In general, upland land use changes, urbanization, and channelization may have contributed to erosion by curtailing potential sediment inputs and magnifying wave reflection and refraction. An altered sediment budget and hydrologic regime may have led to increased marsh immersion during the tidal cycle, thereby increasing waterlogging and initiation of peat collapse. The observed pattern of increased interior ponding and marsh loss is consistent with deterioration of the below-ground biomass due to waterlogging and other forcings (see below). On the other hand, accretion rates for several marshes around Long Island Sound appear to have remained steady or even increased slightly, in spite of recent urban development (Anisfeld, 2003; Kolker, 2003). Further research is needed to investigate the potential role of sediment starvation and hydrologic changes on marsh loss.

Ecological Factors—*Spartina alterniflora* helps maintain salt marsh stability through a tightly interwoven root network that increases soil strength and that promotes trapping of sediment (DeLaune et al., 1994). The apparent retreat of low marsh cordgrass could be caused by high nutrient levels in nearby waters. A sewage treatment plant is located upstream along Blind Brook northeast of Marshlands Conservancy. Runoff from fertilizers applied to the golf course and residential lawns could be another source of excess nutrients. Nitrogen eutrophication linked to adjacent shoreline development may stimulate replacement of high marsh by low marsh (Bertness et al. 2002). However, sea-level rise could also induce similar vegetation shifts.

To date, the Marshlands Conservancy site showed a ~31% (av.) reduction in salt marsh landmass between 1974 to the present, as compared to 22-62% over this time period at other Long Island marsh complexes (Table 2). The average annual submergence rate of 1.2 percent per year at Marshlands lies within the range calculated for the selected marshes within the region. However, no quantitative data records the full regional extent of the marsh loss phenomenon to date. Broad areas of fragmentation and decreases in density of vegetation, enlargement and coalescence of tidal pools, and replacement of low marsh by barren mudflats resemble descriptions of marsh deterioration in Louisiana, Chesapeake Bay, and Jamaica Bay.

While other processes may dominate currently, sea-level rise due to global warming may become the dominant cause of marsh loss in the future. The salt marshes of Marshlands Conservancy may be more vulnerable to the impacts of future sea-level rise because of several factors. These marshes are constrained on their landward sides by steep-sloping topography and existing suburban development, which limits their potential to migrate inland with rising sea levels. Given the rapid decrease in the areal extent of low marsh *Spartina alterniflora*, these wetlands could probably not survive the accelerated rates of sea-level rise anticipated due to global warming.

ACKNOWLEDGMENTS

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REFERENCES

Anisfeld, S. 2003. Wetland loss in the Quinnipiac River Estuary. Tidal Wetlands Loss Workshop, June 24 and 25, 2003. Stony Brook, NY.

Bertness, M. D., Ewanchuk, P. J., and Silliman, B. R. 2002. Anthropogenic modification of New England salt marsh landscapes. *Proc. Natl. Acad. Sci.* v. 99, p. 1395-1398.

DeLaune, R. D., J. A. Nyman, and W. H. Patrick, Jr. 1994. Peat collapse, ponding and wetland loss in a rapidly submerging coastal marsh. *Journal of Coastal Research*, 10: 1021-1030.

Fallon D. and F. Mushacke. 1996 (unpublished). Tidal Wetlands Trends in Shinnecock Bay, New York 1974 to 1995. Division of Fish, Wildlife and Marine Resources, New York State Department of Environmental Conservation, East Setauket, NY, USA.

Gornitz, V., S. Couch, and E. K. Hartig. 2002. Impacts of sea level rise in the New York City metropolitan area. *Global and Planetary Change*, 32: 61-88.

Gornitz, V., Hale, S., Larsen, K., Levine, N., Rosenzweig, C., and Sacks, L., 2004. *Bracing for Climate Change in the Constitution State: What Connecticut Could Face.* Environmental Defense, 42p.

Hartig E. K., V. Gornitz, A. Kolker, F. Mushacke, and D. Fallon. 2002. Anthropogenic and climate-change impacts on salt marsh morphology in Jamaica Bay, New York City. *Wetlands,* 22: 71-89.

Kolker, A. 2003. Marsh changes at selected sites along Long Island Sound. Tidal Wetlands Loss Workshop, June 24 and 25, 2003. Stony Brook, NY.

Mushacke, F. 2003. Tidal Wetlands Losses in Nassau and Suffolk Counties. NYSDEC website: www.dec.state.ny.us/website/dfwmr/marine/wetlands/index.html (last accessed on August 7, 2003).

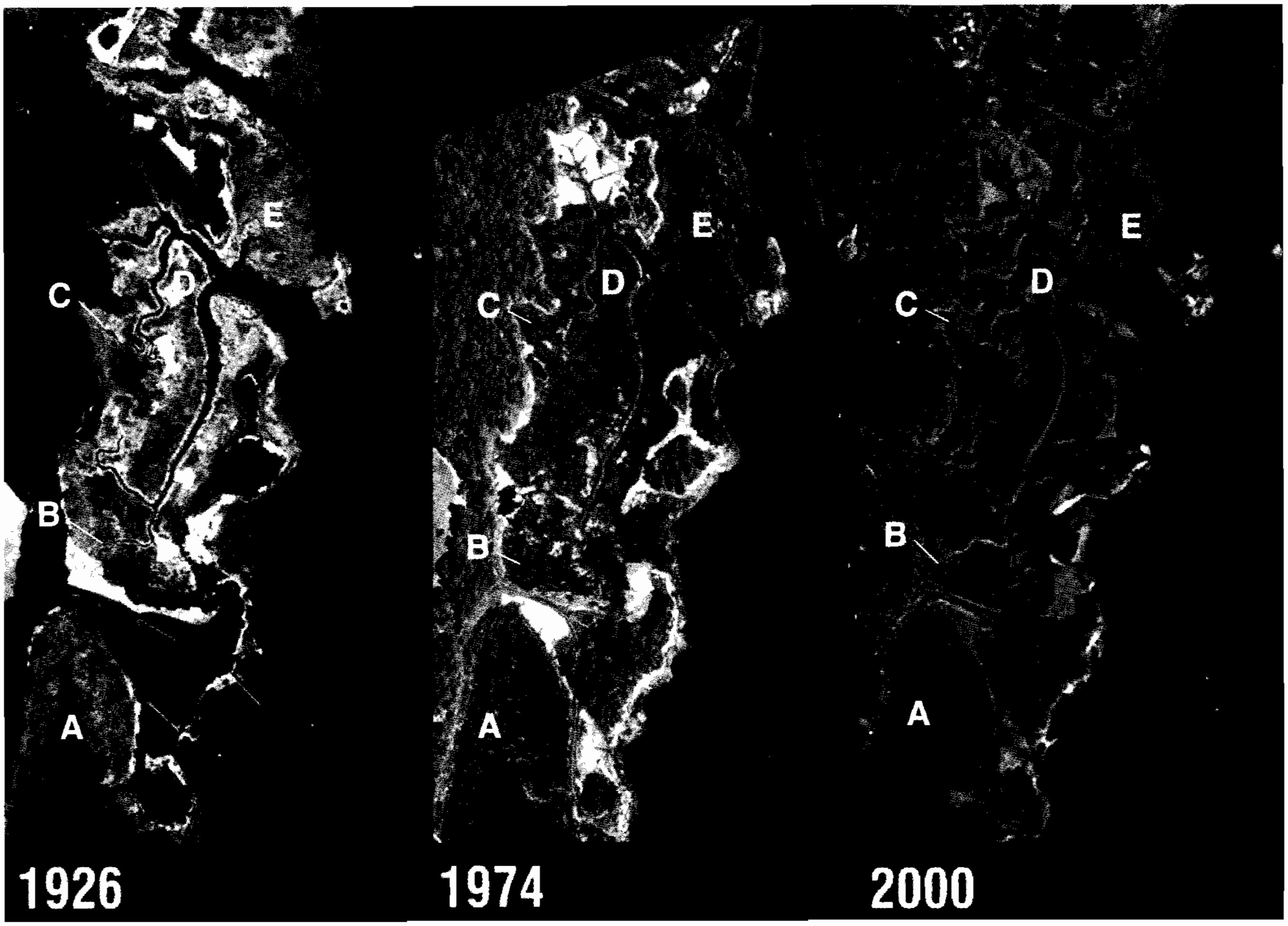
NOAA. 2003. Historic Sea Level Trends. http://coops.nos.noaa.gov/sltrends/sltrends.shtml

Location	1926 ha	1974 ha	1974 % loss since 1926	1974 % loss yr-1 since 1926	2000 ha	2000 % loss since 1974	2000 % loss yr-1 since 1974
Northeast	3.2	1.2	62.5	0.13	1.1	8.3	0.3
Core	11.3	10.9	3.5	0.07	8.4	23	0.9
Southwest	2.4	2.0	16.7	0.35	0.2	90	3.4
Total	16.9	14.1	16.5	0.35	9.7	31	1.2

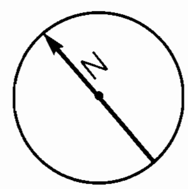
TABLE 1: Changes in salt marsh extent at Marshlands Conservancy from aerial photo analysis, 1926, 1976 and 2000. Sources: Friends of Marshlands, Inc. (1926), Westchester County Planning Department (1974), GlobeXplorer, Airphoto USA, Marshland image 2850746 (2000) at: www.globexplorer.com (last accessed in 2003).

Location	Changes in Area Hectares (Year)		Changes since 1974 % Loss Yr -1	
North Shore–Long Island				
Marshlands Conservancy	14 (1974)	9.7 (2000)	31	1.2
Manhasset Bay	10 (1974)	3.8 (1994)	62	3.1
Stony Brook Harbor Area	121 (1974)	77.0 (1999)	36	1.5
South Shore–Long Island				
Jamaica Bay	799 (1974)	495 (1999)	38	1.5
Oyster Bay Area	526 (1974)	411 (1998)	22	0.9
Shinnecock Bay–Islands	12 (1974)	7 (1995)	42	1.98
Shinnecock Bay–Shoreline	177 (1974)	181 (1995)	-2.3	-0.11

TABLE 2: Reduction in extent of vegetated salt marshes, Long Island, NY, since 1974 (Hartig et al., 2002; Fallon and Mushacke, 1996; and Mushacke, 2003).



1. Aerial photographs of study area for years 1926, 1974, and 2000. Sources: Friends of Marshlands, Inc. (1926), Westchester County Planning Department (1974), GlobeXplorer, Airphoto USA, www.globexplorer.com, Marshland image 2850746 (2000).



Seedling recruitment on plots liberated from extended wrack cover in *a Spartina alterniflora* salt marsh, Jamaica Bay Wildlife Refuge, New York

R. Stalter[1], M. D. Byer[2], A. Jung[1], G. W. Frame[2], W. S. Panagakos[3], M. Waaijer[3], and M. P. Schreibman[3]
[1]Department of Biological Sciences, St John's University, Jamaica, NY 11439 USA
[2]Division of Natural Resources, Gateway National Recreation Area, Natrional Park Service, 210 New York Avenue, Staten Island, NY 10305 USA
[3]AquaticResearch and Environmental Assessment Center, Brooklyn College, City University of New York, Brooklyn, NY USA

ABSTRACT

Under natural conditions, mats of dead vegetation, called wrack, are deposited throughout a salt marsh by tidal action. If wrack remains in place for long periods, it kills or damages the plants beneath it. The objective of the present study was to record marsh plant species recruitment during the 2004 growing season on experimental plots in five arrays located on the low marsh and creek banks of the Jamaica Bay salt marsh, areas that support pure stands of *Spartina alterniflora.* Wrack 10-15 cm deep had been placed and held in place on these arrays during portions of 2002 and 2003, then removed fiom one plot in each array at approximately monthly intervals. Seedling recruitment of three vascular plant species, *Salicornia europaea, Spartina alterniflora* and *Suaeda calceoliformis,* was greatest on plots nearly or completely devoid of living shoots of *S. alterniflora.* In May 2004, seedlings of *Salicornia* were observed in 80 percent of the plots, with numbers ranging from 0-7,611/m^2; *S. alterniflora* seedlings were present in 80 percent of the plots, at densities of 0-787/m^2. Overall survival of these seedlings measured in October 2004 was ~12 and ~31 percent, respectively. *Suaeda* seedlings were found in only one plot, 262/m^2. We predict that, although *Salicornia* may dominate in some of these plots for a time, all plots will eventually be repopulated by pure stands of *S. alterniflora*

INTRODUCTION

The primary objective of the study was to record salt marsh species recruitment during the 2004 growing season on experimental plots in arrays where *Spartina alterniflora* had been deliberately smothered and killed back by wrack in experiments conducted in 2002 and 2003. A second objective was to measure subsequent survival of these same recruited salt marsh plants following their invasion of the plots.

Work on the original study was initiated during the fall of 2002 to provide information that might explain the loss of salt marshes dominated by this species in Jamaica Bay, New York. A review of maps and photography from 1924 to 1999 indicated that *Spartina alterniflora* salt marshes have declined at a rate of 17 hectares/year (Alderson and Frame 1994). The approximate 93I hectares of marsh that existed in 1924 had been reduced to less than 445 hectares by 1999. Based on this rate of salt marsh loss, Frame and Schreibman (2002) predicted that most of the S. *alterniflora* salt marsh in Jamaica Bay might disappear by 2025.

STUDY AREA

The site of this study was Jamaica Bay Wildlife Refuge, bordering Jamaica Bay, an inlet of the Atlantic Ocean at the West end of Long Island, New York (40" 35' N Latitude, 72°, 52' W Longitude).

Jamaica Bay Wildlife Refuge is composed of 1,905 hectares of land (Stalter and Larnont 2002). Jamaica Bay experienced little change for the first two centuries following European settlement in 1651 (Black 1981). After the end of the Civil War, 1865, the Bay's fisheries were established and houses were built on land along the border of the bay. In 1890 a railroad crossed Jamaica Bay, opening the area to further development. By the early 20th Century, channels in the bay were deepened by dredging, and dredge spoil was deposited on portions of the salt marsh. Detailed history of the development of Jamaica Bay can be found in Black (1981) and Stalter and Larnont (2002).

Development of Jamaica Bay was halted in 1950 when the bay and surrounding land were transferred to the New York City Department of Parks. Jamaica Bay Wildlife Refuge was transferred to Gateway National Recreation Area, the nations first and largest urban National Park, in 1972.

The vascular flora of Jamaica Bay Wildlife Refuge, based on fifteen years of observations and collections, consists of 456 species within 270 genera and 90 families (Stalter and Lamont 2002). They described seven major plant communities, including a salt marsh community composed of four vegetation zones. Not included in the salt marsh vegetation zones was *Phraginites australis,* a fresh to brackish marsh associate that borders Jamaica Bay salt marshes in many areas (Stalter and Lamont 2002). The four salt marsh vegetation zones identified were: upper high marsh, lower high marsh, upper low marsh and lower low marsh.

The upper high marsh is dominated by *Iva frutescens, Spartina patens* and *Distichlis spicata.* These taxa are bordered by *Iva frutescens* and occasionally by *Phragnites australis.*

The lower high marsh may be populated by *Salicornia spp., Spartina alterniflora,* and to a lesser degree by *Limonium carolinianum, Distichlis spicata,* and *Suaeda calceoliformis.*

The upper low marsh is almost exclusively covered by a short form of *S. alterniflora.* The upper low marsh merges with a fourth zone, the lower low marsh, which is dominated by a tall form of S. *alterniflora.* The stature of the two forms of S. *alterniflora* may be related to environmental factors (Stalter 1968, Stalter and Larnont 2002). Our studies on salt marsh species recruitment and survival were confined to experimental plots within the lower low marsh, the site occupied by the tall form (ecophene) of *Spartina alterniflora.*

METHODS

Five arrays of permanent experimental plots were established in uniform strands of nearly pure *Spartina alterniflora* having near total cover, and free of wrack. Each array consisted of eight 2m x 2m plots in a row or two adjacent rows, roughly parallel to the water's edge. In each of these plots, a central Im x lm sample plot was marked off, surrounded by an 0.5m wide buffer zone. Each array was GPS located and permanently marked at the corners, as were the 2m x 2m plots and the Im x Im sample plots.

Wrack was collected and placed upon each array, at a thickness of 10-15cm, orienting the wrack parallel to the long axis of the array, xcept for one randomly selected 2m X 2m plot within each array that remained uncovered and-served as a control. Since wrack washed off some of the arrays, and had to be replaced repeatedly, fish netting with 2.5cm mesh was laid over the wrack-covered arrays and held in place with 15cm wire staples during November and December of 2002 and March of 2003.

The individual plots within each array were numbered sequentially. Control or experimental status and sequence of wrack removal were assigned to each plot using random numbers.

The baseline observations and initial covering with wrack of the five arrays were carried out on 24 August (Arrays B, D, F) and 5-7 October (C, E), 2002. On 28 September 2002, one plot in Array B was uncovered and sampled, and on 11 November 2002, one plot each in Arrays E and F. Beginning in June 2003, one plot in every array was uncovered and sampled at intervals of about one month through November 2003. Parameters recorded were shoot numbers by species. cover estimates, and mean height of vegetation. These same plots were sampled for seedling recruitment during the first week in May, 2004 and for seedling survival in October 2004. Vegetation was classified according to Gleason and Cronquist (1991).

RESULTS AND DISCUSSION

Ten herbaceous and one woody salt marsh species, *Iva orarria,* have been identified at Jamaica Bay salt marshes (Stalter and Lamont 2002). Five additional brackish marsh species also exist here; these are generally found in the upper, rarely flooded area of the salt marsh (Table 2). *Phragmites australis* is common at Jamaica Bay, but is not a salt marsh species. *Phragmites* is abundant at many moist soil sites including some sites surrounding Jamaica Bay salt marshes.

Wrack deposition had killed the aerial portions of most of the *Spartina alterniflora* in the experimental plots by November 2003. At least 60 days appeared to be the critical period of continuous wrack coverage needed to kill *Spartina alterniflora.* The bare plot surfaces were subsequently colonized by large numbers of *Salicornia europaea* and *Spartina alterniflora* seedlings the following spring. Both species' germination physiologies are appropriate to the soils and hydrology of the study sites. *Suaeda calceoliformis* seedlings were found in just one plot. *Spartina alterniflora* also colonized some of the plots by rhizome sprouts over the summer of 2004. No other salt or brackish marsh species invaded the barren plots. The mode of colonization of vascular plant species at the Jamaica Bay salt marsh is presented in Table 2.

Seedlings of *Spartina alterniflora* had densities of 0 to 787/m^2 in May 2004. *S. alterniflora* dominated the surrounding marsh as a monoculture (Byer et al. 2004). *Salicornia europaea* was more successful as an early marsh colonizer; its seedling densities ranged from 07611/m^2. *Suaeda calceoliformis,* density was 262/m^2 in a single plot.

Salicornia but not *S. alterniflora* seedling densities in May 2004 are significantly correlated with existing cover in the plots. *Salicornia* and both old and new *S. alterniflora* shoot numbers observed in the fall of 2004 are strongly negatively correlated with duration of wrack cover during the previous years. Yet the numbers of seedlings of both species observed previously, in the spring of that year, are not correlated with duration of the previous years' wrack cover (Table 1). A possible explanation for the disparity may be a chemical change in the soil caused by wrack cover; the longer wrack remains in place, the more is may alter the soil chemistry beneath it. The altered soil may be more injurious to the young shoots than unaltered soil, affecting seedling survival over time. Seedling germination, on the other hand, may respond favorably to the bare surface created by continuous wrack cover lasting 60 days or more, regardless of the altered soil chemistry. Further work on various soil parameters such as salinity, aeration, microbial population, pH, and available nutrients on wrack covered and control (no wrack cover) soils may provide answers to these questions.

We did not identify individual *S. alterniflora* seedlings with stakes in the spring of 2004. Therefore we could not distinguish the origins of the shoots of *S. alterniflora* shoots present in the experimental plots in the Fall of 2004 *S. alterniflora* surrounding the plots may invade plots by producing rhizomes, horizontal underground stems, which will produce above ground plants. The shoots observed in October 2004 may have been the original seedlings, sprouts from existing rhizomes that survived beneath the wrack, or rhizomes that spread into the plot from *S. alterniflora* plants in non-wrack covered ground after the wrack was removed. We will address the aforementioned problem in 2005 by marking individual *S. alterniflora* seedlings with plastic stakes. These individual plants will be monitored monthly during the 2005 growing season to determine survival rates.

Since *Salicornia europaea* is an annual, its October 2004 shoots must all have been seedlings in the spring. The S. *alterniflora* shoots observed at that time. on the other hand, could originally have been either, since it was impossible to distinguish between seedlings and sprouts at that season. When they first appear, seedlings and sprouts of this grass are very different in appearance; the seedlings are very slender and delicate whereas the vegetative shoots are much thicker and more robust. All of the plants counted as seedlings in May of 2004 were indeed seedlings. As to what proportion of the shoots present in the fall might have been vegetative sprouts, we might use observations from a concurrent study as a clue: A *S. alterniflora* marsh nearby was covered with several inches of sediment to raise its elevation to keep pace with rising sea level, thus ensuring the marsh's survival. It was expected that the now buried rhizomes would produce new shoots that would grow up through the sediment. However, most of the re-colonization was by *S. alterniflora* seedlings (which in any case accomplished the intended re-vegetation). Thus, given also that we found no vegetative sprouts in the spring, it seems likely that most of the shoots observed in the fall in the present study were of seedling origin. This was our assumption in calculating percent survival of seedlings over the summer. On this basis, rounding to the nearest whole percent, survival of *Salicornia* seedlings was ~12 percent, that of *S. alterniflora* seedlings, ~31 percent.

The present study leaves several questions unanswered (1) What is the seedling survival rate of *Spartina alterniflora,* (2) What is the rate and mode of recruitment of salt marsh species in other sub-salt marsh communities of Jamaica Bay salt marshes. (3) since water/soil salinities in each zone may be a function of daily, seasonally, and monthly rainfall, how does water and soil salinity and tidal flooding affect the kind and number of species and individuals that invade bare patches in the different vegetation zones in Jamaica Bay salt marshes.

REFERENCES

Alderson, C. and C.W. Frame 1999. Monitoring the salt marsh at Schmitt's Marina in Big Egg Marsh, Jamaica Bay, New York, using permanent plots. New York City Department of Parks and Recreation and Division Natural Resources, Gateway National Recreation Area. Draft manuscript with maps and photographs, October 1999.

Black, F.R. 198 1. Jamaica Bay: A History. Cultural Resource Management Study No. 3. Division of Cultural Resources, North Atlantic Regional Office, National Park Service, U.S. Dept. of Interior. Washington, D.C.

Byer M.D., G.W. Frame, W. Panagakos, M. Waaijer, Z. Aranbayev, Y. Michaels, R. Stalter & M.P. Schreibman 2004. Effects of wrack accumulation on *Spartina alterniflora,* Jamaica Bay Wildlife Refuge, New York City. WIT Press. C.A. Brebbia, J.M. Saval Perez & L. Garcia Andion (Editors). Pgs. 183-190.

Frame, G.W. and M.P. Schreibman 2002. Research proposal for a pretreatment biological Inventory of Big Egg Marsh in Jamaica Bay Wildlife Refuge, Gateway National Recreation Area, New York. Draft manuscript, October 2002.

Gleason, A.A., and A. Cronquist, 1991. Manual of vascular plants of northeastern United States and adjacent Canada, New York Botanical Garden, Bronx, New York 91Opp.

Stalter, R. and E.E. Lamont. 2002. Vascular flora of Jamaica Bay Wildlife Refuge, Long Island, New York. Journal of the Torrey Botanical club 129 (4): 346-358.

Stalter, R. 1968. An ecological study of a South Carolina salt marsh, Ph.D. dissertation. University of South Carolina, Columbia. 62pp.

Tanacredi, J.T. (editor). 2001. The Jamaica Bay Blue Ribbon Panel on marsh loss and Coastal Sea Level Rise: A Future Agenda for Mitigation and Pilot Investigations. Final Report July 2001.

TABLE 1. Linear regressions and correlation coefficients of some of the parameters recorded. Symbols: ns = not considered significant, * = considered significant ($P < 0.05$), ** = considered highly significant ($P < 0.01$), *** = considered extremely significant ($P < 0.0001$).

X (indep vbl)	Y (dep vbl)	N	b (slope)	r (correl coeff)	P (random prob)
Cover estimates	*Salicornia* sdlgs May 04	33	-18.869	-0.3755	0.0313 *
Cover estimates	*S. alterniflora* sdlgs May 04	33	-1.538	-0.2882	0.1039 ns
Total days wrack cover	*Salicornia* sdlgs May 04	26	-0.6099	-0.0284	0.8906 ns
Total days wrack cover	*S. alterniflora* sdlgs May 04	26	-0.1900	-0.1014	0.6222 ns
Total days wrack cover	*Salicornia* shts fall 04	40	+0.0369	+0.0300	0.8540 ns
Total days wrack cover	*S. alt.* old shts fall 04	40	-0.2766	-0.6805	<0.0001***
Total days wrack cover	*S. alt.* new shts fall 04	40	-0.1804	-0.4470	0.0038 **
Total days wrack cover	Vegetation hgt fall 04	40	-0.1598	-0.7010	<0.0001***
Total days wrack cover	Vegetation cover fall 04	39	-0.1081	-0.4500	0.0040**

TABLE 2. Mode of reproduction for herbaceous salt marsh species, Jamaica Bay, New York. S-seeds, R-Rhizome. From Stalter and Lamont (2002).

Species	**Mode of Reproduction**	
Distichlis spicata	*S*	*R*
Juncus gerardii	*S*	*R*
Limonium carolunianum	*S*	*R*
Atriplex arenaria	*S*	
Salicornia europaea	*S*	
Spartina alterniflora	*S*	*R*
Spartinan patens	*S*	*R*
Cyperus filicinus	*S*	
Eleocharis halophila	*S*	*R*
Scirpus pungens	*S*	*R*
Suaeda calceoliformis	*S*	
Suaeda maritime	*S*	
Aster subulatus	*S*	
Aster tenuifolius	*S*	*R*
Solidago sempervirens var. *mexicana*	*S*	

Formation of Aggregates in Natural Waters: Preliminary Results from Laboratory Experiments

Tzong-Yueh Chen and Annelie Skoog

Department of Marine Sciences, University of Connecticut, Groton, CT 06340

INTRODUCTION

Marine dissolved organic carbon (DOC), which contains 200-700 Gt C (1 Gt = 10^{15} g), is the largest organic carbon reservoir in the ocean (Kepkay, 2000). The concentration of DOC is greater than that of particulate organic carbon (POC) by at least an order of magnitude (Kepkay, 2000). Since the oceanic DOC pool contains a similar amount of carbon as the atmospheric CO_2 pool (~750 Gt; Siegenthaler and Sarmiento, 1993), the formation, transformation, and transport of oceanic DOC could play an important role in global carbon cycling.

Photosynthetic production is the ultimate source of organic carbon in the ocean (Druffel *et al.*, 1992), producing both POC and DOC. The microbial loop has been recognized as a major pathway in the dynamics of marine DOC (e.g. Pomeroy, 1974; Azam *et al.*, 1983). Recent studies have shown that abiotic aggregation may also be an important route for DOC transformation and transport (e.g. Kepkay 1994; Chin *et al.*, 1998). The abiotic formation of POC in fresh water systems can account for up to 25% of primary production, which is the same order of magnitude as bacterial production (Kerner *et al.* 2003).

Kepkay (1994) summarized seven different aggregation processes. Aggregation is the result of collision, occurring with a frequency dependent on particle concentration and size, and stickiness, which is a factor dependent on chemical and physical characteristics of the particle. Once the particle grows to a size of a few μm or larger, it can be colonized by heterotrophic bacteria. These large aggregates can also be consumed by protozoa and so enter the microbial loop and the grazing food chain. An additional potential fate is sinking to the deep ocean as the "spring rain" of food to the abyss (Honjo, 1997).

In this study, we used a rolling table (modified from Shanks and Edmondson, 1989) with water samples contained in glass bottles to generate laboratory-made aggregates. Rolling induces a complicated mixture of aggregation processes in the glass bottles. Small particles (usually less than few μm in diameter) ubiquitously undergo Brownian motion. In addition, there is significant shear during the first few hours, and then differential sedimentation becomes the dominant mechanism for aggregation (Jackson, 1994). There were two goals of this study: 1) to identify that collision can increase POC concentrations in natural water samples with varying salinity; 2) to demonstrate that formation of particles and subsequent microbial activities could cause a measurable shift in chemical composition of organic matter.

METHODOLOGY

Study sites and sampling

Water samples were collected in the summer of 2003 at two stations: 1) the coast of Avery Point located in eastern Long Island Sound with a salinity of 21.4-26.7 parts per thousands (ppt); 2) the Thames River estuary, CT with a salinity of 6.3-6.8 ppt. *In situ* temperature and conductivity/salinity were measured by a portable conductivity meter. Water samples were collected using acid-washed 5-L Niskin bottles and placed and delivered under sheltered and cool conditions to the lab. Control samples were collected from both stations.

Rolling-table experiments

Samples for rolling were pre-filtered using 200-μm-pore-size filters and placed in 150-ml, muffled glass bottles. These water samples were rolled at a speed of 8 revolutions per minute (rpm) at 12 °C under dark conditions for two days. Six experiments (three for coastal waters and three for estuary waters) were conducted.

Chemical and biological determinations

POC was defined as the material retained on a GF/F filter (i.e. particles larger than 0.7 μm in diameter), whereas the filtrate was defined as DOC. DOC concentrations were determined by the high temperature catalytic oxidation (HTCO) method on a Shimadzu TOC-V analyzer (Benner and Strom, 1993). POC and particulate nitrogen (PN) samples were acidified with 1N HCl and concentrations were determined using a Carlo-Erba CHN elemental analyzer. Concentrations of total hydrolysable amino acids (THAA) were determined by high pressure liquid chromatography (HPLC) using a C-18 column, a methanol/acetonitrile mobile phase and fluorescence detection (Keil and Kirchman, 1991, 1993). Bacterial cells were fixed with glutaraldehyde (final concentration, 1%) and collected on 0.2-μm-pore-size polycarbonate filters pre-treated with Irgalan Black solution. Bacterial abundance was then determined by the 4', 6-diamidino-2-phenylindole (DAPI) direct count method using epifluorescence microscopy (Hobbie et al., 1977). All determinations were done in triplicate.

RESULTS AND DISCUSSION

Initial conditions

The estuarine and coastal samples had salinities in the range of 6.3 to 6.8 ppt and 21.4 to 26.7 ppt, respectively (Table 1). The DOC concentrations in the estuarine, low-salinity samples (310-425 μM) were much higher than those in the coastal, high-salinity samples (141-171 μM), which is likely due to the high riverine organic input. The estuarine samples also had higher concentrations of POC and PN. Despite the higher DOC concentrations in the estuarine samples, the bacterial cell numbers were on the same order of magnitude at the two study sites (9.6-14.1×10^6 cells in the estuarine samples and 7.5-14.6×10^6 cells in the coastal samples). A possible explanation is that the bacterial biomass was limited by top-down control, such as protozoan grazing or viral lysis, rather than bottom-up control. Biologically-labile components, such as amino acid carbon and amino acid nitrogen (see below for the term definitions), also indicated that the total biomass at the two stations were on the same order. The particulate C/N ratios in the estuarine samples (6.65-7.26) and in the coastal samples (5.99-6.72) were close to the Redfield ratio of 6.6, indicating that phytoplankton biomass may be a major component of particulate organic matter (POM) at both stations.

Table 1 Some hydraulic and initial parameters in Avery Point coast and Thames River estuary, CT

	Unit	Brackish	Salt
Salinity	ppt	6.3-6.8	21.4-26.7
DOC	μM	310-425	141-171
POC	μM	77.5-98.3	24.1-29.9
PN	μM	10.7-14.3	3.59-4.99
Part. C/N		6.65-7.26	5.99-6.72
Bacterial #	10^6 cell	9.6-14.1	7.5-14.6
AA-C content	μM	9.56-20.2	10.3-13.9
AA-N content	μM	2.29-5.06	2.53-3.36

Aggregates Formation and Microbial Activities

To determine the increase in the POC concentration, we calculated the percentage difference between controls and rolling samples, according to the following equations:

$$\Delta POC(\%) = \frac{POC_{rolling} - POC_{control}}{POC_{control}} \times 100\%$$

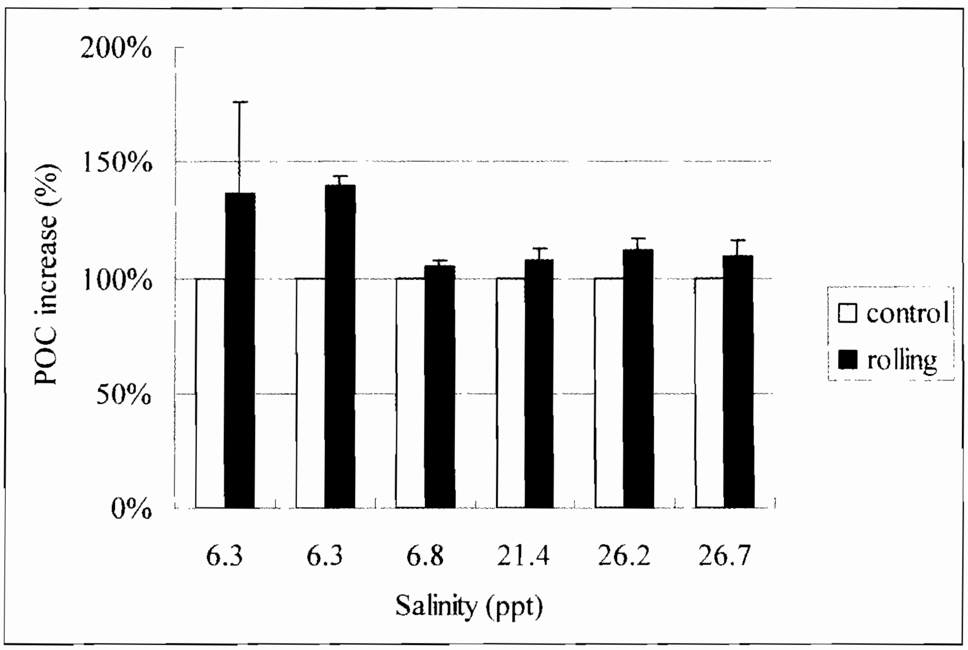


FIGURE 1 POC increase in percent after two days of rolling.

The POC concentrations increased by 5-39% compared to the controls after the two-day incubation (Figure 1). Meanwhile, bacterial cell numbers in rolled samples increased more than four times when compared to the control samples. In order to estimate whether bacterial biomass contributed a significant fraction of the POC increase, we carried out a simple calculation based on bacterial numbers (Tab. 2). The increases in bacterial cell abundance ranged from 7.3 to 56.4×10^6 cells and included both free bacteria and bacteria on the aggregates.. Bacterial-biomass carbon was calculated using a carbon conversion factor of 20 fg cell^{-1} (1 fg = 10^{-15}g; Lancelot and Billen, 1984). The increases in bacterial biomass ranged from 0.012 to 0.094 μM C during the two-day incubation, which accounted for 0.1 to 1.4% of the POC increase. This calculation demonstrated that bacterial biomass in the aggregates is not a major fraction of the POC increase and indicated that abiotic aggregation may dominate the POC increase.

TABLE 2 Calculation of fraction of bacterial biomass carbon in POC formed by aggregation

Salinity (ppt)	ΔBact. # (10^6 cells)	ΔBB (μMC)	ΔPOC (μM)	ΔBB/ΔPOC (%)
6.3	20.6	0.034	35.3	0.1
6.3	56.4	0.094	35.0	0.3
6.8	7.3	0.012	3.8	0.3
21.7	17.8	0.030	2.2	1.4
26.2	14.1	0.024	2.7	0.9
26.7	20.5	0.034	2.4	1.4

Chemical composition—amino acids and C/N ratios

Concentrations of 20 amino acids were determined. Amino acid-carbon and -nitrogen contents and amino acid-carbon and -nitrogen yields were defined according to:

AA-C (or N) content = $\Sigma\{(\#\text{ of carbon in }AA_i) \times [AA]_i\}$

AA-C (or N) yield (%) = {AA-C (or N) content/mole C (or N) in POM} × 100%
where the AA_i is the ith analyzed amino acid.

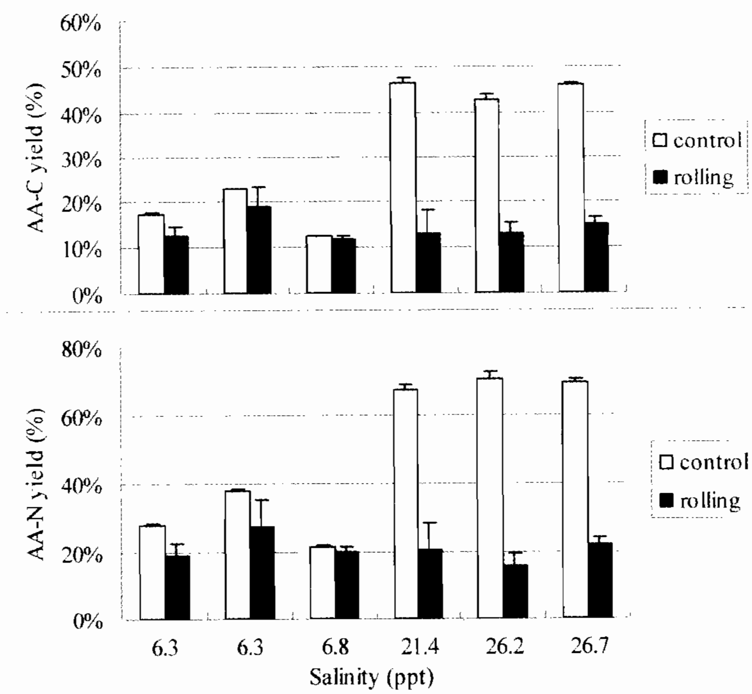


FIGURE 2. Amino acid-carbon and -nitrogen yields before and after rolling

Both AA-C and AA-N yields decreased after the two-day incubation (Figure 2). Although the AA-C and AA-N yields contributed different fractions in estuarine and coastal waters, significant decreases of AA-C and AA-N yields were observed after rolling in samples from both locations. A likely interpretation of the yield decrease is microbial degradation/modification of amino acids (Ogawa *et al.*, 2001).

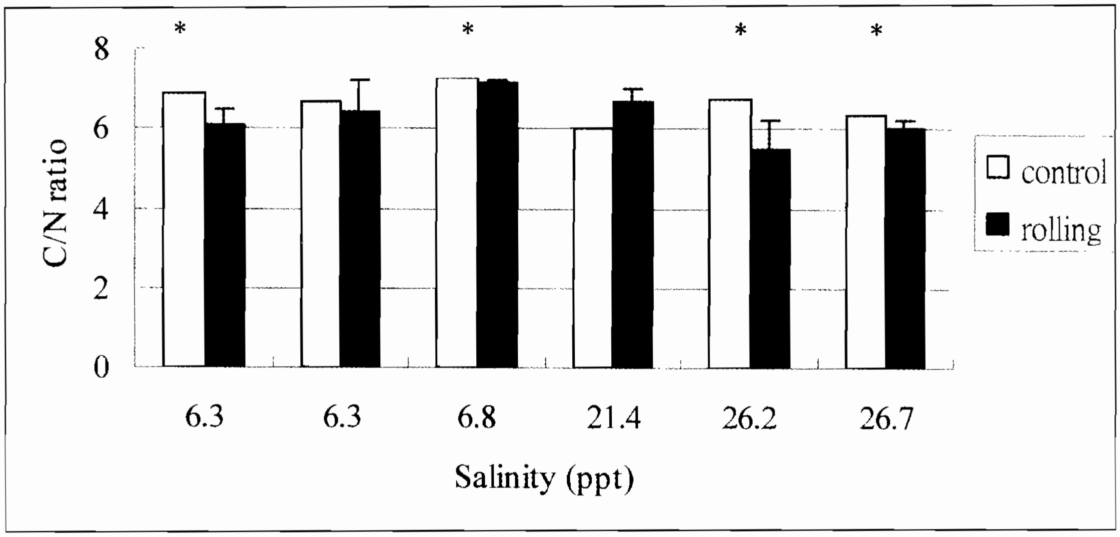


FIGURE 3. The changes of particulate C/N ratio after two-day incubation

Four of six experiments showed a statistically significant decrease in the C/N ratio (labeled with a star, $p < 0.01$) after the two-day incubation (Figure 3). This decrease changed the particulate C/N ratio from close to the Redfield ratio of 6.6 toward the bacterial C/N ratio of ~5. Since the increase in bacterial biomass accounted for less than 1.5% of the POC increase, bacterial biomass can not be the only reason for the decrease in the C/N ratio. Since there is a net increase in POC concentrations, and a net decrease on the POM C/N ratio, there must be an additional nitrogen source to the POM. There are three possible explanations for the PON increase: 1) Organic nitrogen can be obtained from microbial transformation of inorganic nitrogen; 2) bacteria transfer dissolved organic nitrogen to the particulate phase according to mass balance; 3) positively charged N-containing DOM is sorbed to the negatively charged aggregates. We are presently in the process of investigating this question further.

CONCLUSIONS

We found that laboratory-made aggregates can be formed by abiotic aggregation (in this case, through a shear gradient and differential sedimentation) in natural waters of varying salinity. POC concentrations increased by 5-39% after a two-day incubation on the rolling table. Aggregation enhanced the growth of heterotrophic bacteria and the degradation of amino acids in the particulate phase. The abiotic aggregation altered the chemical composition of POM in terms of amino acid composition and C/N ratio. Abiotic aggregation of POC may add to the export of OM from surface waters as well as increase chemical modification of surface-water OM by increasing bacterial degradation.

REFERENCES

Azam, F., T. Fenchel, J. G. Field, J. S. Gray, L. A. Meyer-Reil, and F. Thingstad (1983) The ecological role of water-column microbes in the sea. Mar. Ecol. Prog. Ser. 10: 257-263

Benner, R. and M. Strom. (1993). A critical evaluation of the analytical blank associated with DOC measurements by high-temperature catalytic oxidation. Mar. Chem. 41: 153-160

Chin, W. C., M. V. Orellana, and P. Verdugo (1998) Spontaneous assembly of marine dissolved organic matter into polymer gels. Nature 391: 568-572

Druffel, E. R. M., P. M. Williams, J. E. Bauer, and J. R. Ertel (1992) Cycling of dissolved and particulate organic matter in the open ocean. J. Geophys. Res. 97: 15639-15659

Hobbie, J. E., R. J. Daley, and S. Jasper (1977) Use of Nuclepore filters for counting bacteria by fluorescence microscopy. Appl. Envirn. Microbiol. 35: 1225-1228

Honjo, S. (1997) Marine snow and fecal pellets: The spring rain of food to the abyss. Oceanus 40: 2-3

Jackson, G. A. (1994) Particle trajectories in a rotating cylinder: implications for aggregation incubations. Deep-Sea Res. I 41: 429-437

Keil, R. G. and D. L. Kirchman (1991) Contribution of dissolved amino acids and ammonium to the nitrogen requirements of heterotrophic bacterioplankton. Mar. Ecol. Pro. Ser. 73: 1-10

Keil, R. G. and D. L. Kirchman (1993) Dissolved combined amino acids: Chemical form and utilization by marine bacteria. Limnol. Oceanogr. 38: 1256-1270

Kepkay, P. E. (1994) Particle aggregation and biological reactivity of colloids. Mar. Ecol. Pro. Ser. 109: 293-304

Kepkay, P. E. (2000) Colloids and the ocean carbon cycle. In "Marine Chemistry" (P. J. Wangersky ed.) Springer, New York, USA pp. 35-56

Kerner, M., H. Hohenberg, S. Ertl, M. Reckermann, and A. Spitzy (2003) Self-organization of dissolved organic matter to micelle-like microparticles in river water. Nature 422: 150-154

Lancelot, C. and G. Billen (1984) Activity of heterotrophic bacteria and its coupling to primary production during spring phytoplankton bloom in the southern bight of the North Sea. Limnol. Oceanogr. 29: 721-730

Ogawa, H., Y. Amagai, I. Koike, K. Kaiser, and R. Benner (2001) Production of refractory dissolved organic matter by bacteria. Science 292: 917-920

Pomeroy, L. R. (1974) The ocean's food web: a changing paradigm. BioScience 24: 499-504

Siegenthaler, U. and J. L. Sarmiento (1993) Atmospheric carbon dioxide and the ocean. Nature 365: 119-125

Shanks, A.L. and E. W. Edmondson. 1989. Laboratory-made marine snow: A biological model of the real thing. Mar. Biol. 101: 463-470

Water Quality and Chlorophyll Concentration in Salt and Fresh Water

Valerie Schoepfer, Norwich Free Academy, Norwich, CT 06360

Chlorophyll is a green pigment catalyst in most plants used for photosynthesis. Sunlight enters the plant and goes to the photosynthetic chloroplast. In the chloroplast, chlorophyll molecules grouped into photosystems hold the sun's energy. Photosynthesis changes carbon dioxide and water, which yield in the presence of light and chlorophyll, glucose and oxygen. Photosynthesis also changes energy into starch, which is used when there is no sun to make sugar needed.

Finding the concentration of chlorophyll in the water can indirectly estimate the healthy or unhealthy amount of phytoplankton present. The healthy range of chlorophyll is from less then $1 mg/m^3$ to around $30 mg/m^3$ during a phytoplankton bloom. Phytoplankton are the primary producers for the marine food chain.

Dissolved oxygen, measured in parts per million, or ppm, varies from day to night and throughout the day. During the day, the water can absorb oxygen from the air slowly or obtain it from the photosynthetic phytoplankton. At night, the phytoplankton do not release oxygen because of the lack of light needed for photosynthesis. The range of dissolved oxygen in the water can be normal (5-14ppm), stressful (3-5ppm), or lethal (0-3ppm). Salinity and temperature determine the maximum amount of oxygen the water could hold ("Tips for the Testing of Dissolved Oxygen", para. 9). If too much phytoplankton is present from the addition of nutrients, the body of water can become unhealthy. Sunlight cannot reach the lower levels of the water ("Long Island Sound" para. 12). This causes phytoplankton and the plants on the bottom to die and decompose aerobically, causing hypoxia. Monitoring the amount of phytoplankton in relation to the variables that affect the concentration is important to finding the health of the body of water. Monitoring these variables may lessen the chance of a phytoplankton bloom.

In the summer of 2003, a local pond had a large phytoplankton bloom. According to long time town residents, this has not happened in Bozrah for at least the past 70 years (Shelly). Another long time town resident said that the pond was constantly clean, except for the summer of 2003 (Shelly). She also reported that when the bridge and pipes were redone over Fitchville Pond a few years prior, the freely flowing water became more stationary with more vegetation that remained year round on both sides of the bridge, (Shelly). There was a local interest to see why the bloom happened and what could be done about it. The study was done during the summer of 2004 in the same pond as well as others to determine if the bloom would be seasonal, if one would happen in other ponds, and why.

The study focused on what variables affect the chlorophyll concentration in fresh and salt water. Some of the variables looked at were water temperature, salinity, tide, dissolved oxygen, precipitation, and season. Salt water readings were taken at ten sites at Project Oceanology on six tides in August 2003 in Baker's Cove and the Poquonock River in Fisher's Island Sound using a Millipore filtration system, centrifuge, and a Hach DC Spectrophotometer following test directions written by Howard Weiss (Weiss 525-528, 881-882). Salinity, dissolved oxygen, and temperature readings were taken with a YSI 85 Water Quality meter. Other equipment from Project Oceanology include boat time and a professional library. Fresh water tests were done in Bozrah, Montville, and Salem at seven sites on and connected to Fitchville Pond and Gardner Lake on thirteen study dates during autumn and winter of 2003/2004 using a LaMotte Water Quality Kit, multiple filtration methods, a centrifuge, and a B+L Spectronic 21 Spectrophotometer.

The hypothesis states that more chlorophyll will be present in fresh water than salt water. Higher concentrations will be at still water sites, such as a pond, rather than swift moving water, like a river. In fresh water, there will be more during the summer than during autumn and winter from higher temperatures and longer days. In the brackish water, higher chlorophyll concentrations will be found in lower salinity at the head of the river and places where temperature is higher, unlike the open Long Island Sound stations. Stations with more dissolved oxygen should indicate higher chlorophyll concentrations.

Salt water test results (Figure 1) show that flood tide chlorophyll concentrations were more regular and had a more distinct pattern than ebb tide concentrations. The upriver stations with very low salinity and warmer temperatures had the highest concentrations of chlorophyll during both ebb and flood tides. The water was shallow and clear, this might let the plankton grow quicker and be able to multiply more rapidly due to increased sun absorption. Runoff to the freshwater stations could result in more nutrients addition, causing a slight phytoplankton bloom. The salt water study took place over the course of one week which is not enough time for a pattern to form, however, there were some trends. There were no significant blooms in salt water during this study. The levels increased, but there was not enough time to determine any long range conclusions.

The chlorophyll concentrations in the open water of Long Island Sound were lower than the less saline waters of the estuary. The hypothesis was correct. More chlorophyll was measured at stations with lower salinities and higher temperatures. Dissolved oxygen remained constant and did not correlate with chlorophyll changes.

Ebb tide samples were taken on August 8, 11 and 14 2003. Over the one week study, ebb tide had an irregular pattern of chlorophyll concentrations. Flood tide chlorophyll concentrations were taken on August 11, 13, and 14. Flood tide had the most regular and consistent pattern of increase and decrease throughout the stations.

Precipitation data was compared to chlorophyll concentration data as an indicator of runoff. About .2 inches of rain fell on the 12th, which may have influenced the chlorophyll levels on August 13 and 14 by decreasing the temperature and salinity of surface waters. There was more rain on August 7 and 8, but this did not affect the chlorophyll values on August 8 or 9. In most cases, precipitation correlates to an increase in chlorophyll concentrations, but may not have been the only cause for the increase.

Dissolved oxygen levels did not seem to affect chlorophyll concentrations, which remained constant for most of the study, ranging from around 5ppm to 11.1ppm. Less saline stations had higher concentrations of chlorophyll, but salinity may not directly impact chlorophyll concentration.

Chlorophyll concentrations at stations nine and ten were low on August 8 and 11. A few days before these tests, a plane crash occurred at station 10 and released thousands of gallons of jet fuel into the water, possibly killing the phytoplankton, therefore lowering the chlorophyll and dissolved oxygen readings. The results for station 10 should be disregarded because they possibly were impacted by the crash. Station 9 was slightly protected by a boom barrier and should be regarded with caution.

Fresh water results (Figures 2,3) show that seasonal changes with decreasing day lengths and cooler temperatures may have an influence on chlorophyll concentrations. Oceanography texts indicate that there is an expected increase in chlorophyll concentrations during spring and autumn (Ross 183). As the seasons changed, the chlorophyll concentrations generally decreased, with an exception for station dates with phytoplankton blooms. However, these stations still decrease during the change in season.

Some variables were factors in determining chlorophyll concentrations. Rain seemed to be a related factor for all stations except station 2, a river with a strong current. The precipitation could have washed nutrients into the river, but would not have enough time for the nutrient input to impact the chlorophyll levels. The chlorophyll values were independent from the dissolved oxygen readings. Other factors must be looked at to determine the cause of chlorophyll value changes.

Several stations, 2, 5, and 6, had high concentrations for a good portion of the study. The sites were a known point source pollution location, a stagnant pond, and a still water lake site. Station 2 was on a bridge over Fitchville Pond. This station has reportedly had, in the past, animal waste deposited in it. On October 11, the reading was 84.836 mg/m^3. The acceptable range for chlorophyll in an oligotrophic lake is from 0-2 mg/m^3, 2 to 15 mg/m^3 for a mesotrophic lake, and 15-30 mg/m^3 for a eutrophic lake. Above this is considered a highly eutrophic lake. Both Fitchville Pond and Gardner Lake are considered mesotropic lakes according to the "Connecticut Water Quality Standards" (Rocque 15). On most other dates, the concentration was in the normal range. Station 5 was a small stagnant pond with a large goose population. Six dates out of the 13 were above 20 mg/m^3, which indicate a phytoplankton bloom. Early in the study, this pond experienced a very large bloom, covering the entire pond. The water temperature decreased and days grew shorter with the cold weather, and the concentration decreased as phytoplankton died. Station 6 was on Gardner Lake. The concentration spiked to 32.10 mg/m^3 on November 15. This day the water level was low, allowing the sun to hit more phytoplankton. Also, there was precipitation for three days prior to the test date. This could impact the nutrient runoff into the lake. On all other dates, the concentration was within the normal range. Most fresh water stations were within the normal range the entire study and the concentrations decreased as the weather grew colder. Shallow depths and nutrients from manure are potential causes for the high levels but were not tested variables.

Station seven was a small pond connected to Gardner Lake. On October 18, two large bryozoa colonies (Pectinatella magnifica) were found, about two feet in diameter. The bryozoa colonies washed away in a heavy rainstorm, did not reform and the chlorophyll readings fell and stayed low. Before it grew, the chlorophyll readings varied greatly. Whatever variable impacted the chlorophyll values at this site could also have affected the bryozoan's growth and formation.

There were several problems and uncontrolled variables in this study. When the chlorophyll values were determined with the common formula, many of the results were negative. The spectrophotometer book listed too much background material, most likely sediment in this case, on the filter as a probable cause for negative values. To eliminate the negative values, the 750 wavelength numbers were not included in the formula, leaving only the 665 chlorophyll$_a$ readings adjusted for sample volume. Using only the 665 values produced chlorophyll results higher than the previous formula's values, but had relative values and the negative numbers was eliminated. Another problem occurred when sampling in January; a few sites were frozen. Liquid water could not be collected; therefore, ice was collected, melted, and tested causing the chlorophyll concentrations to possibly be impacted. For the same reason, some stations dissolved oxygen readings were not taken and the temperature was recorded as zero degrees Celsius. The salt water dissolved oxygen, salinity and temperature were measured more precisely than fresh water because of more accurate equipment. The salt water spectrophotometer was more modern compared to the one used in fresh water. In salt water, a powerful, motorized vacuum allowed 1000mL of a sample to be filtered. Less powerful equipment was available in fresh water so in some cases only 250mL of the sample was filtered causing the numbers adjusted for sample volume in the equation to be less accurate. Pump design for home based fresh water samples presented a design challenge. Seven pumps were tried when designing the fresh water studies. The first one was a Millipore syringe with a dry seal which did not produce a strong vacuum. A small medical syringe was next, but filtering took long because of the size. A motorized aquarium pump was tried but could not generate enough vacuum to be useful. A bike pump with a large chamber was then attempted. This created too much of a vacuum and was difficult to operate. A small shop vac was for the clear samples, but did not draw water through the dirty samples. It also could not run for long periods of time or it would overheat. Next, a big wet dry vac was used with the same problems as the small vac. The pump that worked best was a brake line hand pump modified to fit the filtration unit. The water would filter quickly with this pump and worked with all types of samples. This pump was then used for the majority of sample preparation. Magnesium Carbonate ($MgCO_3$) was used for the salt water tests but not fresh water. Some previous studies suggest that adding $MgCO_3$ to a sample destroys pigments ("Chlorophyll$_a$", par. 5). Other studies show that there were no differences in the concentration of samples with and without $MgCO_3$ ("Chlorophyll$_a$", para. 5). Differences in sampling techniques between fresh and salt water would only be problematic if summer salt levels were being compared to fall and winter fresh water values. This study did not base any conclusions on that basis.

In the future, nutrient input could be studied along with chlorophyll data to determine if there is any correlation. In addition, the health of an insect population could be studied to monitor water quality. Many years of study are needed to determine the overall quality of the water. One years worth of data cannot show trends or abnormalities in algae blooms. A number of sites show evidence of nutrient runoff. In the area of study, there is no sewage treatment. There are solutions to ponds with blooms (Ramey para 23-30). The addition of copper compounds to water can reduce the amount of algae. Certain species of fish introduced to the body of water can eat the algae and control harmful blooms. Skimming the surface of the water is another way to remove some harmful algae. These possible solutions are not without side effects to the water. Now that base line year one data was obtained, follow up studies can begin to look at long term conditions.

REFERENCES

"Chlorophyll$_a$ an intro and an analytical methodology". [Online] Available http://www.lakes.chebucto.org/DATA/PARAMETERS/CHA/cha.html, Jan. 21, 2004.

"Tips for the Testing of Dissolved Oxygen". [Online] Available http://www.lamotte.com/pages/common/techtips/dotip.html. October 13, 2003.

"Long Island Sound Water Quality Report 2001". [Online] Available www.savethesound.org/report2001/intro.htm, January 24, 2004.

Ramey, Vic. "Algae" [Online] Available http://aquat1.ifas.ufl.edu/guide/2algae.html Feb 5 2004.

Rocque, Arthur J. "Water Quality Standards". Hartford Connecticut: State of CT DEP, 1997.

Ross, Davis A. Introduction to Oceanography. New York: Meridith Corporation, 1970.

Shelly, Pauline. Personal Interview. 16 May 2004.

Weiss, Howard M, Michael W. Dorsey. Investigating The Marine Environment: A Sourcebook Volume 2 and 3. Groton CT: Project Oceanology, 1979.

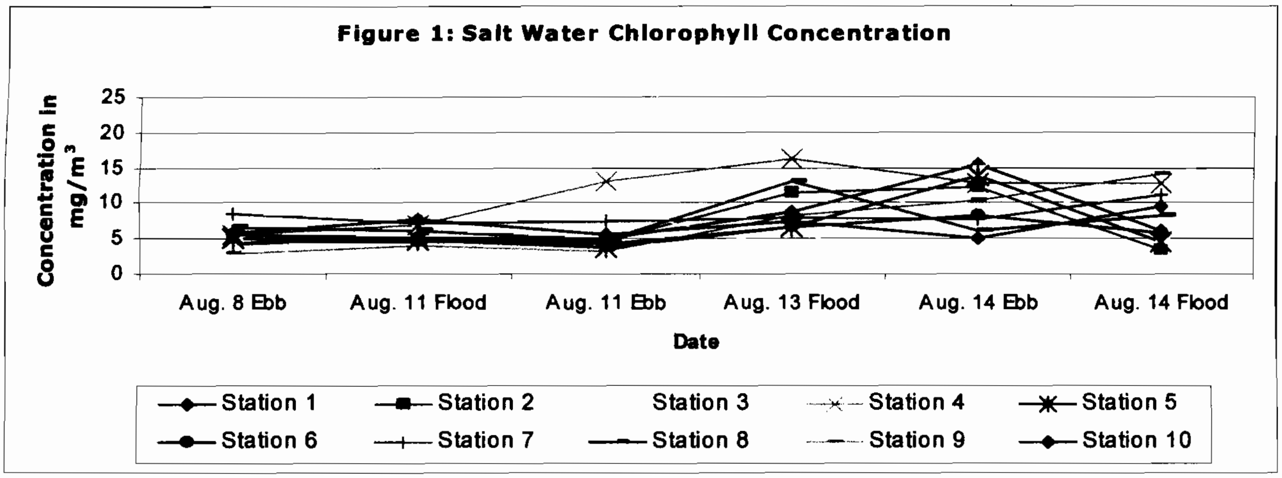


Figure 1: Salt Water Chlorophyll Concentration

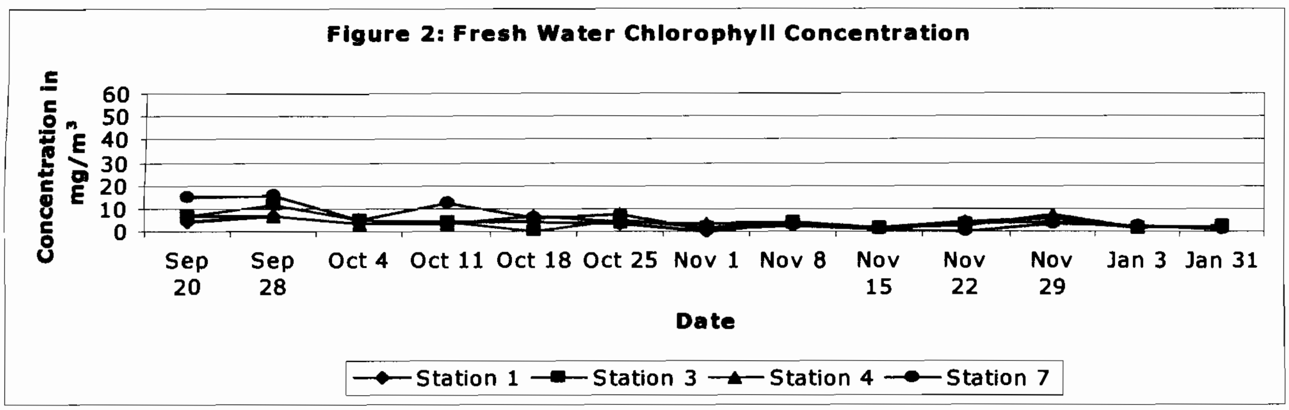


Figure 2: Fresh Water Chlorophyll Concentration

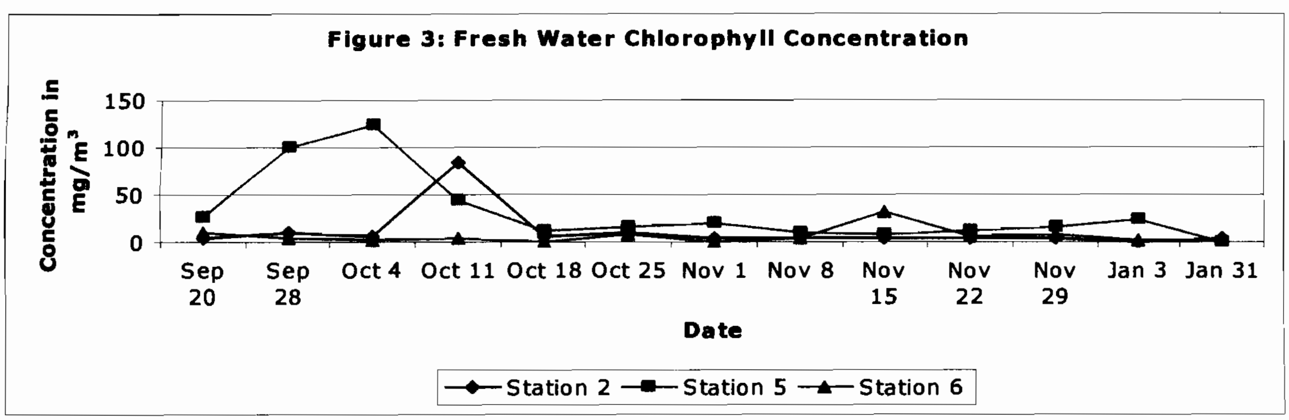


Figure 3: Fresh Water Chlorophyll Concentration

ABSTRACTS

Salt Marsh Restoration and Drowning in Long Island Sound: A Comparison of Three Connecticut Marshes

Anisfeld, Shimon, C., Mitch, Azalea and Nunnally, Sally;
Yale School of Forestry & Environmental Studies, 370 Prospect St., New Haven, CT 06511,
shimon.anisfeld@yale.edu

Salt marsh "drowning" — in which marshes become too wet to support vegetation and are converted to unvegetated mudflats — has been observed recently in several areas of Long Island Sound. At the same time, other sites have seen successful restoration efforts, in which marshes that were previously tidally-restricted have been re-opened to tidal flushing, resulting in wetter conditions and the conversion of systems dominated by Phragmites australis to healthy low marshes dominated by Spartina alterniflora.

We have examined marsh dynamics — both physical and ecological — at 3 marshes, all dominated by S. altemiflora, but exemplifying a range of conditions: a reference site (Hoadley Creek) with apparently stable conditions; a restored site (Jarvis Creek) with successful colonization by highly productive S. altemiflora, and a degrading site (Sherwood Island) in which low marsh is currently being converted to mudflat. At each marsh, we have created maps of elevation, hydroperiod, vegetation type, and productivity. We will also be installing surface elevation tables (SETs) at these sites, in order to determine short- and long-term changes in marsh elevation.

Temporal/Spatial Distributions and Emission of Elemental Mercury from Long Island Sound

Prentiss H. Balcom, William F. Fitzgerald, Chad R. Hammerschmidt;
Department of Marine Sciences, University of Connecticut, Groton, CT, USA
Chun-Mao Tseng, National Taiwan University, Taipei, Taiwan
Carl H. Lamborg, Woods Hole Oceanographic Institution, Woods Hole, MA, USA

In situ elemental mercury (Hg°) production and emissions to the local/regional atmosphere are major processes. Our investigations of the air-sea partitioning of mercury (Hg) in Long Island Sound (LIS) include twelve Hg° surveys conducted from January 1999 to March 2002 using both real-time, shipboard and laboratory analysis methods. Average surface water Hg° concentrations ranged from 100-260 fM (250- 1100% saturation), with little diurnal variability, and elevated levels in warm months and near sources of labile (reactive) Hg. Average concentrations of Hg species showed no east-west gradient for dissolved total Hg (1.7-3.4 pM), dissolved reactive Hg (0.6-1.9 pM), or unfiltered total Hg (5.9-9.5 pM). Average unfiltered reactive Hg levels (0.5-4.2 pM) were higher in eastern LIS near the large Connecticut River source. Atmospheric total gaseous Hg (TGM) levels were measured from April 2000 to March 2001 at Hammonasset State Park using an automated mercury vapor analyzer, and average weekly concentrations ranged from 1.1-2.1 ng m3. The calculated flux of Hg° from LIS ranged from 90-750 p mol m2 d', with higher fluxes associated with elevated Hg° concentrations çwarm months) and wind velocities (early spring).The average flux from LIS was estimated to be 70 kg y^{-1} which compares well with a previous estimate from 1995-1997. The evasion process removes about 30% of total Hg inputs to US (230 kg y^{-1}), remobilizing Hg and extending its lifetime in active reservoirs. Production of Hg° competes for labile Hg reactant with *in situ* biological synthesis of monomethylmercury (MMHg), such that water bodies with a large production of Hg° may have less accumulation of MMHg in sediments and smaller amounts in biota.

Sediment Dynamics in Connecticut Estuaries: ^{7}Be, ^{210}Pb, ^{137}Cs, Trace Metals, and Modeling to Investigate Delivery, Erosion, and Accumulation

Benoit, Gaboury and Hirschbeck, Matthew
Yale Environment SchoolJ2 I Sachein Street/New Haven CT 06511 Gaboury.benoit@yale.edu

Short and medium term sedimentary dynamics (delivery, scavenging, deposition, erosion, resuspension, burial, export) were investigated by means of radionuclide tracers, especially short- lived ^{7}Be (1/2 life 53 d). Atmospheric deposition supplies 27 mBq/cm^{2} of ^{7}Be in New Haven, and this value probably applies throughout the region. During a rainstorm in November 2002, two separate peaks in water column ^{7}Be were observed, each about 1% of concentration in rainfall, due to dilution. During the storm, which lasted 39 hours, there were several fill tidal cycles. Peaks in water column ^{7}Be seem more related to delivery from the atmosphere and watershed than to tidal flushing. The removal from the water column was exponential, with efold removal rate constants of 0.9 and 1.3 d^{-1} for the two peaks. ^{7}Be inventories in sediments at the site sampled followed no simple pattern. Subsequent synoptic surveys of ^{7}Be inventories at 30 sites throughout the estuary reveal a complex spatial pattern. ^{210}Pb and ^{137}Cs profiles in sediments did not follow simple, easily interpretable patterns, except at a site upstream from tidal influence, where sediment accumulation rate was 0.4 cm/yr by both methods.

Evidence of Historical Wet Period Deposits in Central Long Island Sound: Source and Transport Processes Based on Mercury Profiles and Sedimentology

Drew A. Carey, Johan C. Varekamp

Vibracores collected from central Long Island Sound (US) recovered sediments deposited prior to, and during, dredged material disposal. The weight of dredged material consolidated underlying sediments, compressing the sediment record differentially (based on overlying weight of dredged material). Three district sediment layers were found at slightly varying levels within ambient sediments (below district layers of dredged material). Each sediment layer contained coarse sand and shell fragments visually distinct from surrounding alive green silts typically found in central LIS. Results of mercury profiles and descriptive sedimentology are presented relative to background time history of mercury and sediment deposition in US. We speculate that coarse layers may represent wet period sheet wash deposits from the Housatonic delta. We predict that similar deposits should be found underlying ambient silts and thickening from the central basin to the mouth of the Housatonic.

Observed Residual Circulation in Eastern Long Island Sound: Transverse-Vertical Structure, and Exchange Transport

Dan Codiga, Graduate School of Oceanography, University of Rhode Island
Dirk Aurin, Department of Marine Sciences, University of Connecticut

Fundamental characteristics of the residual circulation in Eastern Long Island Sound (LIS), which controls pathways and fate of waterborne materials vital to the LIS ecosystem, remain largely unknown. As part of the FOSTER-LIS (Ferry-based Observations for Science Targeting Estuarine Research) program, one year of ADCP (acoustic Doppler current profiler) measurements has recently been collected by a ferry that crosses nominally 8 times daily on a transect spanning the Eastern end of LIS between New London, CT and Orient Point, NY. The unique ability of these data to isolate and remove the more energetic tidal component is exploited to investigate the transverse-vertical structure and exchange transport of the residual circulation. Westward flow in to US extends from the seafloor to near the surface in the central portion of the estuary, splitting and underlying two shallow outgoing eastward currents that deepen toward the northern and southern shores. Outward flow is predominantly in the southern portion of the transect where peak measured residual currents, excluding the narrow Plum Gut, reach ~25 cm/s. Inward flow reaches similar magnitudes deeper than ~40 m near the ~75-m deep thalweg, and weakens toward the north and south and with distance from the bottom. A recent idealized semi-analytic solution for transverse-vertical residual circulation structure has been determined using bathymetry representative of the ferry transect. It captures observed characteristics reasonably well, indicating that effects of transverse bathymetry, Coriolis, and friction are all three important in shaping the flow structure. Estimated advective exchange transports will be presented and compared to those based on salt budgets, followed by a discussion of the implications for salt transport.

The Long Island Sound Mesozooplankton Monitoring Program

Dam, Hans G., Department of Marine Sciences, University of Connecticut, Groton CT 06340-6048
McManus, George B., Department of Marine Sciences, University of Connecticut, Groton CT
Smith, Amy N., Department of Marine Sciences, University of Connecticut, Groton CT
Chen, Lihua, Department of Marine Sciences, University of Connecticut, Groton CT
Christina, Iott, Coastal Studies, University of Connecticut, Groton CT
Avery, David E., Department of Marine Sciences, University of Connecticut, Groton CT
Lyman, Matthew, Connecticut Department of Environmental Protection, Hartford CT
Olsen, Christine, Connecticut Department of Environmental Protection, Hartford CT

A comprehensive examination of the zooplankton of Long Island Sound has not been attempted since the time of Deevey's work in Long Island Sound in the 1950's. This lack of information is critical in light of the eutrophication and hypoxia problems that Long Island Sound currently experiences. Consequently, as part of the Long Island Sound Monitoring Program, the CT Dept. Environmental Protection has contracted the University of Connecticut to carry out a zooplankton monitoring program since August of 1992. Duplicate vertical tows for the entire water column are taken monthly at six stations along the West-East axis of the Sound. Samples are analyzed for total biomass (dry weight) and for taxon composition of mesozooplankton (200-2000 μm). In this talk, we will present a brief overview of the results obtained to date, including comparisons of biomass and dominant taxa among the stations, and the seasonal cycle of abundance of the mesozooplankton. We will interpret the results in the framework of the West-East eutrophication gradient in the Sound. We will also attempt to compare our results to previous investigations.

Twenty-five Years of Nitrogen Export From the Pawcatuck Watershed to Little Narragansett Bay

Fulweiler, R.W. and Nixon, S.W.
Graduate School of Oceanography, University of Rhode Island, Narragansett, R.I. 02882-1197

As part of a larger study of nutrient export from the Pawcatuck watershed to Little Narragansett Bay we measured the concentrations of dissolved inorganic nitrogen, dissolved organic nitrogen, and particulate nitrogen at the mouth of the Pawcatuck river eighty times between 12/6/01 and 11/29/02. Annual fluxes were calculated and compared to the long-term United States Geological Survey record available at this site. The beginning of the study corresponded to a thought in New England and the Pawcatuck River discharge for this study was ~60% lower than long-term mean (0.83 x 10^6 m^3 d^{-1} and 1.46 x 10^6 m^3 d^{-1}, respectively). Total nitrogen export for this study (16 x 10^6 mol y^{-1}) was significantly less than the total nitrogen export from 1976-1986 (29 x 10^6 mol y^1). However, the relationship between water discharge and TN export has not changed. The composition of total nitrogen in the Pawcatuck River seems to be shifting over this twenty five year period. Dissolved organic nitrogen appears to be increasing since the late 1970s while dissolved inorganic is decreasing. Mean dissolved inorganic nitrogen concentration in the Pawcatuck River decreased significantly from 39±12 μM in 1980 to 28±12 μM in 2002. This is despite a 40% increase in nitrogen fertilizer use and rapid development within the watershed. A possible explanation for the decrease in dissolved inorganic nitrogen is a change in agricultural practices while the increase in dissolved organic nitrogen may be driven by increased forest cover.

A Simple Semi-Empirical Model of the Salt Flux in Estuaries

Gay, Peter, S., Department of Marine Science-UCONN, 1080 Shennecossett Rd., Groton, CT 06340
O'Donnell, James, Department of Marine Science-UCONN, 1080 Shennecossett Rd., Groton, CT 06340

The distribution of salt in estuaries is determined by circulation and mixing, as are the concentrations of nutrients, contaminants, etc. Direct measurement of long-term transport of material is difficult, so interpretation of distributions can provide quantitative estimates that are useful in models of biological and chemical distributions. This paper develops a one-dimensional analytical model that represents transport as a dispersive process and exploits salinity distributions to estimate coefficients for estuarine channels with linearly varying cross-sectional area. This provides a simple way to evaluate salt flux throughout the estuary using archived hydrographic data. The model is applied to observations from the Chesapeake Bay, Delaware Bay, and Long Island Sound. Good fits to observed axial depth-averaged salinity profiles are obtained using three or more segments, each having a constant dispersion coefficient and linearly varying cross-sectional area. Inclusion of unsteady effects in the model allows the characterization of seasonal variability.

An Assessment of the Needs of Connecticut's Shellfish Aquaculture Industry

Getchis, Tessa S.
Connecticut Sea Grant College Program, Department of Extension, University of Connecticut, 1080 Shennecossett Road, Groton, CT 06340-6048

An assessment was conducted to identify the needs of Connecticut's shellfish aquaculture industry. Participants identified and prioritized issues of importance through a mail survey, personal interviews, and an industry summit. Government and public relations, or lack thereof are major constraints to the industry. Industry members are in favor of streamlining the permitting process, and developing positive relationships with regulatory agencies. Most wish to increase the acceptance of aquaculture and improve support from other users of the marine environment. A working group is being developed to address these issues through collaborative research, outreach, and education programs.

Application of Remote Sensing Technologies for the Delineation and Assessment of Coastal Marshes and their Constituent Species around Long Island Sound

Gilmore. Martha S.[1], Civco, Daniel L.[2], Prisloe, Sandy[3], Hurd, James D.[2] and Wilson, Emily H.[3]
[1]Dept. of Earth and Enviromuental Sciences, Wesleyan University, 265 Church St., Middletown, CT 06459;
[2]Dept. of Natural Resources Management and Engineering, University of Connecticut, 1376 Storrs Road, Storrs, CT 06269;
[3]Center for Land Use Education and Research, UConn Cooperative Extension System, 1066 Old Saybrook Road, P0 Box 70, Haddam, CT 06438.

Over the past century, there has been significant loss of coastal wetlands around Long Island Sound due primarily to anthropogenic disturbance and sea level rise. In addition to physical loss of marshes, the composition of marsh communities is changing as species such as *Spartina alterniflora* and *Spartina patens* are being replaced by monocultures of *Phragtnites asustralis*. Here we seek to document the extent and vegetative composition of coastal marshes using a combination of moderate resolution (30 m Landsat ETM+ and 15—60 m Terra ASTER) and high resolution (0.6 — 2.4 m QuickBird) multispectral satellite imagery and in situ field measurements of plant spectra. The moderate resolution imagery are being processed to recognize, locate and measure marshes over the whole of US. Present-day images can be compared to Landsat images collected in the past two decades to assess changes in marsh position and extent. Land cover data surrounding marshes may lend some insight into any contributors to marsh loss or change. Previous studies by this group have shown that *P. australis* is most spectrally distinguishable from other marsh species in August and September in Landsat images. High resolution satellite and in situ studies are underway at five field sites; Wheeler Marsh, Old Saybrook, Chapman Pond, and Barn Island Marsh in CT and Flax Pond in NY. In situ data are collected weekly - monthly in order to document plant phenology and corresponding changes in plant spectra. Preliminary analyses of data for Old Saybrook show that *S. patens*, *S. alterniflors*, *P. australis* and possibly *Typha angusolia* display distinct spectral signatures in processed QuickBird images taken in July 2003. Detailed spectra taken on the site in late May 2004 show *Phraginites* to have increased blue and near- infrared reviewed for reflectance relative to the other species. If such spectral differences are consistent and robust throughout the year, multispectral remote sensing satellite data may prove a powerful tool for coastal wetland management and monitoring.

Some Intriguing Observations From An Inveterate Long Island Sound Fisherman, Writer and Naturalist

Glowka, Art, Save the Sound, Norwalk, CT

I've been writing about fishing and observing western Long Island Sound for forty years. I've also been actively involved in the EPA Long Island Sound Study (LISS) since its inception. I, along with others in the academic and regulatory communities, feel that LISS is failing to address the Sound's biological destruction. LISS needs to be reassessed and changes made in both management and priorities if the Sound's ecosystem is to be restored.

The present LISS has become mired down in its singular concentration on Total Maximum Daily Load (TMDL) and Comprehensive Conservation and Management Plan (CCMP) that has morphed into 58 pages of turgid prose and 205 action items, many of which make little sense. Most meetings, whether they be Management, STAC or CAC have been reduced to confusion and frustration.

The fact is that the Sound is not improving its basic biological integrity and many bio-indicators are getting worse! Intertidal blue mussels are gone. Marshes are disappearing. Hypoxia gets worse each year. Fish populations decline and those that remain are under-nourished. Lobsters are almost non-existent in the western Sound. Then there are the questions of whether East River flow affects the western Sound and whether the SWEM model is really telling us anything. Are these catastrophes caused by erroneous assumptions or unintended consequences?

We must address the Sound as a totality of biological processes as well as the interweaving of all its living marine resources. I will offer my own observations of the Sound's physical, chemical and biological interactions along with a list of suggestions and studies that should be done. The truth is that you have to understand how something works before you can fix it.

Impact of Microzooplankton Grazing on Phytoplankton Biomass and Community Structure in the East River- Long Island Sound Ecosystem

Gobler, Christopher J., Marine Science Program, Southampton College of Long Island University, Southampton, NY

The overgrowth of phytoplankton in Long Island Sound (LIS) has contributed to the annual occurrence of hypoxia in bottom waters for decades. To date, the role of microzooplankton grazing in controlling phytoplankton biomass and altering phytoplankton community structure in LIS has been poorly studied. To determine grazing mortality rates of phytoplankton in US and the East River (ER), dilution experiments were conducted at three stations in LIS (west, central and east) and in the ER during spring and summer cruises. In the spring, levels of phytoplankton biomass were highest in western US (7 μg chlorophyll *a* L^{-1}). Concurrently, microzooplankton grazing rates were significantly correlated with levels of phytoplankton biomass across all study sites ($r^2 = 0.94$, $p < 0.05$), with the highest rates observed in western LIS (0.5 d^{-1}) Flow cytometric analysis indicated that, during spring, grazing rates on picoplankton were greater than the rates on the total phytoplankton community, whereas grazing rates on the nanoplankton were substantially lower than rates on the total community. During summer, levels of phytoplankton biomass were 3 — 7 times lower than spring, and picoplankton dominated cell abundances at all stations. Grazing mortality rates (1.3 — 1.5 d^{-1}) of the total phytoplankton community were about three times greater than rates observed during spring, although grazing was not detected in the ER. In a manner similar to the spring cruise, grazing rates on picoplankton were more than double the grazing rates on nanoplankton. In summary, results suggest that grazing by microzooplankton may influence phytopiankton biomass and community structure, pelagic carbon flux, and hypoxia in LIS.

Temporal and Spatial Variability in Photosynthetic Characteristics and Primary Production, and Testing a Formulation for Primary Production in Long Island Sound

Goebel, Nicole L., University of Connecticut. Groton, CT, USA, nicole.goebeluconn.edu
Kremer, Jim N., University of Connecticut, Groton, CT, USA, jim.kremerc2iuconn.edu
Edwards, Chris A., University of California at Santa Cruz, Santa Cruz, CA, USA, cedwardsucsc.edu

Previous studies identify decomposition of high phytoplankton stocks, encouraged by high nitrogen loads, as the primary mechanism for decreasing oxygen levels and subsequent hypoxic events in the bottom waters of Long Island Sound (LIS) during summer. Few of these studies have actually measured this rate process. We present a first time investigation of pelagic oxygen metabolic processes in Long Island Sound, based on comprehensive, site- specific measurements of net community production (Photosynthesis-Irradiance series) during summer-autumn of 2002 and 2003, with the ultimate aim of formulating a model for primary production in Long Island Sound. Each non-linear *P-I* curve was fit numerically in order to calculate photosynthetic parameters for biomass-specific rates of photosynthesis at low irradiance (a^B) and photosynthetic capacity at light saturation ($P^B m$), and community respiration (R_C) Temporal and spatial variation in these fitted parameters, chlorophyll concentrations (Chl), and environmental variables (salinity, temperature, irradiance) are investigated. $P^B m$, a^B , R_C, Chl, and thermohaline and irradiance properties varied significantly with season. During each sampling trip, R_C, Chl , and thermohaline and underwater irradiance properties varied significantly throughout the Sound forming strong along-Sound gradients, whereas $P^B m$ and a^B did not vary significantly with space. These photosynthetic and respiratory components are essential for estimating and modeling primary production in LIS. Routine measurements of phytoplankton stocks, euphotic zone, and incident irradiation are tested in an empirical relationship to our rigorous field measurements of net community production, as a formulation for phytoplankton production in an ecological model of oxygen metabolism in US. Such corroboration of this new production model with local, direct measurements of pelagic primary production not only sets our model apart from those that base this rate process on temperature and/or tuned parameters, but will enable us to merge our ecological model with an existing hydrodynamic model to yield a simplified, accurate model of oxygen production/consumption in US. This model can improve our understanding of oxygen dynamics and hypoxia in LIS and bolster its utility as a management tool for evaluating present efforts to reduce nitrogen loads to LIS.

Biogeochemical Controls on the Production and Distribution of Methylmercury in Sediments of Long Island Sound

Hammerschmidt, Chad R., Fitzgerald, William F., Lamborg, Carl H., Balcom, Prentiss H., Visscher, Pieter T.
Department of Marine Sciences, University of Connecticut, Groton, CT 06340.

Near shore marine sediments, including those of Long Island Sound (LIS), are a repository of pollution- derived "legacy Hg", and they host active communities of sulfate-reducing bacteria, the principal group of microorganisms mediating transformation of inorganic Hg to toxic monomethylmercury (MMHg). This combination in biologically productive near-shore environments results in considerable production of MMHg. Organic matter is a principal control on the distribution of inorganic Hg (Hg(II) = total Hg — MMHg) and MMHg. Partitioning with sedimentary organics governs levels of Hg(II) in pore water and influences microbial MMHg production. Potential rates of Hg methylation vary inversely with the distribution coefficient of Hg(II) and positively with the concentration of Hg(II), mostly as HgS° or an organic-Hg complex, in low-sulfide pore waters. Hg methylation is inhibited in high-sulfide sediments. Bioturbation enhances production of MMHg; subsurface peaks in Hg methylation potentials coincide with anomalies in a bioturbation index that indicate physical disturbance of sediment. Much of the MMHg synthesized in sediments is mobilized to overlying water. In site sedimentary production is the major source of MMHg to LIS, and it can account for most of the MMHg in primary producers. Hence, bioturbation may redistribute "legacy Hg" within the sedimentary column to zones where SRB actively methylate Hg(II) that is de-sorbed from the organic particulate phase, creating a potential for methylation, mobilization, and bioaccumulation of pollutant Hg that was buried during the past 150 years.

Bioaccumulation of Methylmercury in Long Island Sound

Hammerschmidt, Chad R., Fitzgerald, W. F.
Department of Marine Sciences, University of Connecticut, Groton, CT 06340.

Coastal fisheries are the major source of human exposure to toxic monomethylmercury (MMHg). Humans are exposed to MMHg principally by consumption of marine fish, most (about 75%) of which are from the coastal zone. However, and unfortunately, the bioaccumulation of MMHg in coastal marine systems has received scant scientific attention. We examined MMHg in four ecologically representative and commercially relevant fishes from Long Island Sound (US). These include alewife (*Alosa pseudoharengus*), winter flounder (*Pseudopleuronectes americanus*), American lobster (*Homarus americanus*), and bluefish (*Pomatomus saltatrix*). MMHg comprised most of the total Fig in axial muscle of finfishes. MMHg bioaccumulates in the food web of US; concentrations in suspended particulate matter, most of which is autochthonous, are greater than those in water and less than those in zooplankton. Levels of MMHg are highest in piscivorous bluefish, less in invertebrate-feeding lobster and flounder, and lowest in planktivorous alewife. Bluefish from LIS have 2-3 fold more MMHg than those of comparable size from nearby New York Bight. Preliminary results show that MMHg concentration is unrelated to fish size (length, weight) for each of the study species in LIS, likely due variations in dietary MMHg of these migratory species. The absence of agreement between size and MMHg content of coastal fishes, unlike that in many freshwater systems, suggests that human consumption advisories for Hg in marine species that are based on fish length may not reliably protect public health from the potential effects of dietary MMHg exposure.

Microzooplankton in Long Island Sound

McManus, George B., Department of Marine Sciences, University of Connecticut, Groton CT
Dam, Hans G., Department of Marine Sciences, University of Connecticut, Groton CT
Costas, Barbara A., Department of Marine Sciences, University of Connecticut, Groton CT
Thibodeau, Kristin, Department of Marine Sciences, University of Connecticut, Groton CT
Haberlandt, Katherine A., Coastal Studies, University of Connecticut, Groton CT
Lyman, Matthew, Connecticut Department of Environmental Protection, Hartford CT
Olsen, Christine, Connecticut Department of Environmental Protection, Hartford CT

Microzooplankton (20-200 μm) are the predominant grazers of phytoplankton in coastal waters and thus serve as critical links in the marine food chain. Because of difficulties in preserving, cultivating and identifying many of the species in this community, however, information on their abundance and distribution is limited. This is especially true for ciliate microzooplankton. For the past two years, the Long Island Sound Monitoring Program (CT Dept. Environmental Protection) has contracted the University of Connecticut to participate in sample analysis. Whole water samples and 64 μm mesh concentrates are collected monthly at six stations along the West-East axis of the Sound. Samples are analyzed for biovolume and taxonomic composition of microzooplankton. Data on abundance and distribution of the dominant taxa will be presented, along with other results from our lab on the diversity and biogeography of ciliate microzooplankton in Long Island Sound. Comparisons with data from other monitoring programs (e.g. Chesapeake Bay) will also be made.

Lessons Learned from NYNJ Harbor Deepening Project including Elastodynamic& Electromagnetic Measurements, Imaging, & Interpretation of Utilities, Rocks, & Sediments

Murphy William F., Ward, W. Bruce, Boyd, Beckett, Fleming, Gary, Beda, Stephanie, Fleming, Robert, and Oristaglio, Michael, Earthworks LLC, 27 Glen Rd, Sandy Hook CT 06482

The US Army Corps of Engineers and the Port Authority of New York & New Jersey are deepening the NYNJ harbor to a water depth of 50 feet or more. Key issues in the project are (a) sediments deemed unsuitable for offshore disposal, (b) undiggable rock, (e) identification and relocation of utilities, and (d) monitoring of environmental affects.

Elastodynamic and electromagnetic measurements and imaging were developed to map the bottom and subsurface in three dimensions. The strata include Holocene sands, silts, & muds, Pleistocene silt & clay varves, Jurassic diabase, Triassic sandstones & shales, Ordovician serpentine, and Manhattan schist. The turbidity in NYNJ harbor is very high. Aerial photograph—like images map sediments, bottom morphologies, habitats, rock outcrops, debris, pipelines, trenches, abutments, scours, dredge cuts, and bottom marks and grooves from shipping. The top-of-rock is mapped to a depth 40 ft below the mudline with an accuracy of 2ft. Rock diggability is measured remotely. Combinations of seismic & electromagnetic measurements have located sewer tunnels, cables, oil pipelines, and communication & power lines. All measurements are calibrated with core borings. Disturbance to the environment is monitored.

The deepening project is ahead of schedule and under budget. Scientific investigations before and during construction increase efficiency and help to protect the environment. The harbor is best investigated as a system. Spatial trends (e.g., strike and dip) are important. Local changes in properties are abrupt. The bottom composition and morphology change in time. Seasonal variations are important. Bottom marks and grooves caused by shipping persist.

Observations of the Circulation in Long Island Sound

O'Donnell, James, Bohlen, W. Frank
University of Connecticut, Department of Marine Science, Groton, CT

Since the main sources of freshwater to Long Island Sound are distributed along the north shore. We conjectured that this may create a residual (time mean) pressure gradient force directed towards the south that, in concert with the Coriolis acceleration, would transport the brackish river effluent to the west on the north side of the Sound. We conducted two observation programs aimed at detecting such a circulation. In the spring of 1995 we maintained four moored current meters at the Mattituck Sill for more than a month and conducted a series of ship surveys to augment the moored observations. In the summer of the same year we performed a similar measurement campaign in the western Sound along a section between Bridgeport, CT and Port Jefferson, NY, slightly west of Stratford Shoals.

The observations did not support our conjecture. Along the Mattituck section we found the freshest water along the coast of Long Island and the strongest flows to be to the west in a near surface jet in the central portion of the section. The westward motion was weaker and distributed across the rest of the section. The observations at the salinity or density though there appears to be slightly weaker stratification in the shallow near-shore areas. The current observations show a broad eastward flow near the surface across most of the section. This overlies a westward flow that is strongest in the deeper portions of the section. Though there is a westward motion along the northern shore and an eastward motion along the southern shore, this does not appear to be associated with horizontal density gradients.

We conclude that the observations are inconsistent with the hypothesis. Several alternative explanations are plausible. It is possible that the freshwater delivered to the eastern Sound from the Connecticut River is rapidly mixed horizontally and vertically so that a residual horizontal buoyancy gradient is never established. The residual currents observed in the shallow areas of the Sound may be produced by the topographic rectification of tidal currents. Quantitative evaluation of these ideas will require additional research.

Structure and Variability in the Longitudinal Salt Field of the Connecticut River

O'Donnell, J., Howard-Strobel, M. M., Gay, P.,
University of Connecticut, Department of Marine Sciences

The Connecticut River is the largest contributor of freshwater and associated dissolved and particulate matter to Long Island Sound. Despite this significance, fundamental estuarine processes within the river are still relatively unknown. Using data from 21 surveys and 310 CTD casts collected in the mid-1990's, we examine the axial structure of the salinity field in the Connecticut River and how the structure varies with time as a result of the affects of river discharge. Specifically, we use the position of the intersection of the two psu salinity isohaline (X2) with the bottom as an indicator of the salt field response. The location of X2 within an estuary has been shown to correlate with several ecologically significant indicators, including: benthic invertebrate populations, planktonic detrital material, larval fish survival and the estuarine turbidity maximum. We show that the response of the Connecticut River is nearly 1:1, unlike other estuaries, with implications regarding the estimation of the longitudinal dispersion coefficients with dependence on flow and X2 position.

Anthropogenic Influences on Benthic Foraminiferal Faunas in Long Island Sound

Thomas, Ellen, Varekamp, Joop, C., Thaler, Bess, and Acosta, Zirah,
Department of E&ES, Wesleyan University, Middletown, CT 06459

We used benthic foraminifera (unicellular, eukaryotic, shell-forming organisms) in ~ 2m long gravity cores in westernmost (WLIS75GGC 1; ~18 m water depth) and coastal central Long Island Sound (B1GGC1 ;~ 7 m water depth) to document changes in Long Island Sound (US) ecosystems over the last 1000 years. Age models were derived from metal pollution records and ^{14}C dating. Before European settlement, faunas were low-diversity, stable, and in B1GGC1 dominated by *Elphidium excavatum*. In WLIS75GGC1 this species was less abundant; *Elphidium incerlum* and *Buccella frigida* were common. In both cores, the absolute abundance of foraminifera and the relative abundance of *Elphidium excavatum* started to increase in the early 1800s, possibly because of increased productivity of diatoms, the main food source of *E. excavatum*. In the late 1960s absolute foraminiferal abundance decreased while *Ammonia beccarii parkinsoniana*, formerly absent or rare, became common to dominant in western US. This change could have been caused by hypoxia (possibly in conjunction with rising temperatures), but laboratory evidence contradicts this hypothesis. More probably, high N/Si resulting from strong eutrophication favored primary producers other than diatoms, making *E. excavatum* less competitive. Such a change in dominant primary producers might be expected to reverberate throughout the ecosystem. The US ecosystem thus changed significantly with the enhanced nutrient input associated with human population growth in the 1800s, and again with more severe eutrophication over the last few decades.

Predicting Dissolved Organic Carbon Distributions in Long Island Sound

Vlahos P., Dierssen, H.
University of Connecticut, Marine Science Department, Groton, CT

Allocthonous and autocthonous dissolved organic matter in coastal areas plays an important role in the net autotrophic or heterotrophic nature of the system and is important in our understanding of carbon cycling. Here we present dissolved organic carbon (DOC) concentrations from US surface waters taken from the RV Connecticut in May 2004 and compare these values to predicted values of DOC based on in situ measurements of colored dissolved organic matter (CDOM) fluorescence. Differences between the measured and predicted estimate of DOC are due to the variable composition of DOC at each station. We identify some of the sources of DOC with stable carbon isotopes. We will also evaluate methods for estimating CDOM and DOC from ocean color reflectance measurements at the sea surface. Our aim is to use remote sensing techniques in combination with measurements of CDOM fluorescence from a mooring to develop reliable predictions of DOC in US.

Developing New Techniques to investigate how nutrients control Phytoplanicfon assemblages in Long Island Sound

Wikfors*, G.H.[1], Strychar[2], K. B., Ward[2], J. E., Alix[1], J. H., and Holohan[2], B. A.
[1]National Marine Fisheries Service (NOAA), 212 Rogers Ave., Milford, CT 06460
[2]Department of Marines Sciences, University of Connecticut, Groton, CT 06340

Relationships between nutrient fluctuations and physiological status of phytoplankton species may explain the bases for variations in phytoplankton assemblages in Long Island Sound (US). We have combined flow-cytometry and immunochemical methods to examine how nutrient concentrations affect the physiological condition (metabolic activity, enzyme expression) and intracellular characteristics (e.g., lipid, protein) of individual phytoplankton cells. We are applying this approach to samples of phytoplankton and environmental measurements collected at three LIS locations seasonally (western, central, eastern LIS). At each location, two sites were sampled including an inshore site (ca. 10 m deep) and an offshore site (ca. 20 m deep). Initial results confirm the expectation that nutrient loads are highest in the western Sound and lowest in the eastern Sound. Higher concentrations of nutrients were found in water samples collected in the fall and winter, and lower concentrations were observed in the spring and summer, during and just after the spring phytoplankton bloom. Dominant phytoplankton genera in the eastern Sound include *Chaetoceros* and *Rhizosolena*, in the central Sound *Asterionella* and *Rhizosolenia*, and in the western Sound *Coscinodiscus*, *Rhizosolenia* and the ciliate *Tintinnus*. Laboratory experiments testing the ability of fluorescent probes to differentiate nutrient-replete vs. nutrient-deficient cultures of the centric diatom, *Thalassiosira pseudonana*, revealed the need for extensive method development and validation. Fully-validated protocols will be applied to field samples of natural phytoplankton communities. Continued refinement of our newly- developed techniques will permit a better understanding of how nutrient dynamics in LIS affect primary If producers and ultimately grazer populations. Such information is vital for improving the long-term management of LIS and developing strategies to protect and restore its living marine resources.

POSTERS

Tissue Metal Contents of Eastern Oysters (Crassostrea .virginica) in New Haven Harbor

Cacopardo, Matthew[1] and Breslin, Vincent T.,[2]
[1]Undergraduate Student, Biology Department
[2]Associate Professor, Science Education and Environmental Studies
Southern Connecticut State University, New Haven, CT 06515

Natural and caged oysters (placed 10-3-03) were collected from various locations in New Haven harbor known to represent a wide range of sediment metal contents. Oysters (n= 10) were harvested from each of three locations on two occasions and depurated in laboratory aquaria in filtered (0.45 μm) water for 48 hours. The soft tissue was removed and individual oyster tissues were oven dried, acid digested, and the digests analyzed for iron, copper, zinc, nickel, lead, cadmium and arsenic using flame and graphite furnace atomic absorption spectrometry. Natural oyster populations sampled on 11-3-03 from three inner and outer harbor locations yielded mean tissue metal contents ranging from 172 to 368 mg/kg dry wt. for Fe, 690-860 mg/kg dry wt. for Cu and 0.47-0.9 1 % dry wt. for Zn. For caged oysters sampled on 11-21-03, mean oyster tissue metal contents ranged from 144 mg/kg to 235 mg/kg dry wt. for Fe, 650 to 820 mg/kg dry wt. for Cu and 0.65 to 1.01% dry wt. for Zn. Tissue metal contents for caged oysters sampled from the Savin Rock and Lighthouse Point locations were similar to tissue metal contents for caged oysters sampled from the inner harbor areas (mouths of Mill and Quinnipiac Rivers) even though they are located in areas of comparatively lower sediment metal concentrations. Measured oyster tissue Cu and Zn contents were similar to results of previous studies for oysters sampled in New Haven harbor and significantly higher than previously measured oyster tissue metal contents for oysters collected from Milford and Stratford harbors. Tissue metal analysis of a second set of caged oysters sampled on 5-10-04 is currently ongoing.

Anthropogenic Eutrophication Of Long Island Sound: Effects On Diatom Communities Through Time

Cooper, Sherri R., Thomas, Ellen, Varekamp, Johan and Sangiorgi, Francesca,
Earth & Environmental Sciences, Wesleyan University, Middletown, CT

The overall goal of this research is to place recent environmental changes in Western Long Island Sound (WLIS) within an historical context. To investigate changes in eutrophication, hypoxia and anoxia, salinity, temperature and metal pollution in WLIS, paleoecological methods are used, including: fossil evidence (diatoms, dinoflagellates and foraminifera) and biogeochemical evidence (biogenic silica, metal concentrations, and various elemental ratios and isotopic compositions).

Diatom results from two sediment cores show significant declines in diversity and increases in centric:pennate ratio beginning in the 19th century. Core WLISGGC1 (2 m long) shows a change in centric:pennate diatom ratio from —1.5 to -4.0 in the early 1800s. Dominant species include Thalassionema nitzschioides, Paralia sulcata, and Cyclotella species. There is evidence of a bloom of Fseudónitzschia (10% relative abundance) in this core from the early 20th century.

These changes suggest an increase in eutrophication of the basin starting in early 1800s. This conclusion is supported by dinoflagellate data (a large increase in heterotrophic species in the early 19th century), foraminifera species and accumulation rates, and other biogeochemical data (including carbon isotope data and increasing concentrations of organic carbon and nitrogen). Recently, dominant primary producers in WLIS may have shifted from diatoms to other phytoplankton, with potential impact on all LIS biota.

Trace Metal Concentrations in Tidal Marsh Sediments, Fletcher's and Nettleton Creeks, Milford, CT

Coron, Cynthia R., Fleming, Thomas. H. and Page, Lawrence,
Earth Sciences, Southern Connecticut State University, New Haven, CT coroncl@southernct.edu

Fletcher's Creek and Nettleton Creek tidal marshes, are 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, bedded debris from unregulated dumping was exposed at the surface in a debris field of 3.72 square kilometers, lying 242m south of the fenced-off landfill, and extending a minimum distance of 60-100m beyond the mapped '0' limit of landfill waste. Bedded debris exposed in channels occurs to an average depth of 2 meters.

X-ray fluorescence analyses of sediment from the debris field and within tidal channels in the restored part of Fletcher's Creek indicate elevated concentrations of heavy metals of probable anthropogenic origin. Select ranges include Sn = 8-280 µg/g, Pb = 111-802 µg/g, Zn =153-2760 µg/g, Cu 57-1847 µg/g. Concentrations were highest in the tidal channel immediately south of the fenced landfill and parking lot boundaries, and lower in the strandline zone. Comparisons between element concentrations in sediment in the restored portion of Fletcher's Creek marsh to those in the un-restored Nettleton Creek tidal marsh are underway.

Ocean color remote sensing of Long Island Sound: Discriminating water column constituents from space

Dierssen[1],H.M., Aurin[1], D., Twardowski[2], M., Freeman[2], S., Vlahos[1], P., Branco[1], A.,
[1]University of Connecticut. Department of Marine Sciences, Groton, CT 06340
[2]WET Labs, Inc. 165 Dean Knauss Dr., Narragansett, RI 02882

The spatial patterns of phytoplankton biomass and dissolved organic matter in Long Island Sound (US) are highly variable in both time and space. In particular, eastern and western regions of LIS are subject to different physical mechanisms that create unique spatial patterns of flow and biomass. Ocean color remote sensing is an important tool for discerning large scale (1 km) features in absorption and scattering properties of the water column. However, traditional remote sensing algorithms fail in complex "Case 2" waters, such as US, that are optically controlled by constituents other than chlorophyll. In US, colored dissolved organic matter, suspended sediment, and even wind-blown pollen from terrestrial ecosystems play important roles in defining the optical properties of the water column. Here, we present bio-optical data collected in March 2004 and discuss the potential for remote sensing these various water constituents. Remote sensing imagery of LIS from the MODIS and SeaWiFS sensors will be presented.

The Analysis of Tidal Ellipse Structure in Central Long Island Sound

Glassberg, Phillip and Wilson, Robert E., Marine Sciences Research Center, Stony Brook University, Stony Brook, NY 11794-5000

Oscillatory tidal currents in coastal waters are typically described in terms of their ellipse properties. These properties include the length and orientation of the major axis, the eccentricity, and the polarization. Ellipse properties may change significantly with depth. The vertical structure of tidal ellipses in a particular area is important because it can have significant influence on both the horizontal transport, the horizontal dispersion of dissolved and suspended materials due to the combined effects of current shear and vertical mixing. The vertical structure of tidal ellipses is itself strongly dependent on the intensity of local vertical mixing with the water column, and the observed structure can be used to diagnose the characteristics of vertical mixing in a particular area. The vertical structure of tidal ellipse properties derived from ADCP moorings in central Long Island Sound is presented, and the relationships to water column stratification, vertical mixing and horizontal dispersion are discussed.

A Previously Unrecognized Moraine and Other Geologic Interpretations Derived From NOAA Bathymetric Data Collected in the Vicinity of The Race

Lewis, Ralph, S., DiGiacomo-Cohen, Mary, L.,
Long Island Sound Resource Center, UCONN, Groton, CT 06340,
Poppe, Lawrence, J., USGS,
Woods Hole, MA 02543, Smith, Shep, M., NOAA, Norfolk, VA 23510

Sharing of multibeam bathymetric data (NOAA Ship *Thomas Jefferson*, October 2003 survey Hi 1250G) between NOAA's Atlantic Hydrographic Branch and the State of Connecticut/USGS Geologic Mapping Cooperative has yielded a new geologic perspective on approximately 91 km^2 of the sea floor in the vicinity of The Race. Strong tidal currents at The Race result from the narrowing and shoaling associated with the Orient Point-Fishers Island Moraine segment which forms a northeast-trending bathymetric high that bisects the study area. Tidal scour is inferred to have stripped away up to 60 m of glacial lake deposits and locally exposed bedrock of the Avalonian Terrane northeast of The Race. Northwest-trending (~343°) and north- trending ridges and lineaments in these localities exhibit alignments and characteristics similar to coastal bedrock ridges near Alewife Cove (Waterford/New London) and at Bluff Point (Groton). Step-like features that trend northeast (50-60°) in the northwest corner of the area are inferred to be strike ridges associated with offshore extensions of the northeast-trending Avalonian bedrock in the Bluff Point area.

A remnant of the glacial delta of the Thames River lies just north and west of Fishers Island. The surface of this delta is inferred to approximate the elevation (-21 m) of the lake deposits that once extended across the study area. To the southeast, erosion of the lake deposits in western Block Island Sound has exhumed a 3 km ridge segment that trends northeastward and is inferred to be part of a previously unrecognized moraine. Bedforms and scour depressions throughout the study area indicate that net sediment transport is westward in the trough along the margin of Fishers Island and eastward in the trough bordering Little Gull Island.

Surficial Geologic Interpretation of the Sea Floor off Branford, Connecticut, on CD-ROM

Paskevich, V.F.[1], Poppe, L.J.[1], Moser, M.S.[2], DiGiacomo-Cohen, M.L.[3], and Christman, E.B.[2]
[1] U.S. Geological Survey, Woods Hole, MA 02543
[2] Atlantic Hydrographic Branch, National Oceanic and Atmospheric Administration, Norfolk, VA 23510
[3] Long Island Sound Resource Center, Connecticut Department of Environmental Protection, Groton, CT 06340

The U.S. Geological Survey (USGS), in cooperation with the National Oceanic and Atmospheric Administration (NOAA) and the Connecticut Department of Environmental Protection, has produced detailed geologic maps of sections of the sea floor in Long Island Sound (LIS). The current phase of this research program uses continuous-coverage sidescan sonar imagery to study the distribution of surficial sediments and sedimentary environments, the processes controlling these distributions, and the relation of benthic community structures to sea-floor geology. The mosaic and interpretation presently being released on CD-ROM (USGS Open-File Report 2004-1003), produced from data collected during survey H11043 by NOAA 's Atlantic Hydrographic Branch, covers approximately 41.1 km^2 of the sea floor in north-central LIS off Branford, Connecticut.

Sedimentary environments characterized by erosion and non-deposition prevail in the shallow waters on Townshend Ledge and Branford Reef, where strong tidal currents and wind-driven waves prevent the deposition of finer-grained sediment and leave lag deposits of boulders and gravel on the offshore extension of the Madison moraine. As water depth increases on the flanks of bathymetric highs, conditions favoring erosion are replaced by sedimentary environments characterized by sorting and reworking. Faint current ripples, observed on the surface of grab samples of sandy muds and muddy sands, and sediment resuspension, observed in bottom video, reflect this sorting and reworking by tidal and storm currents. Long-term deposition predominates at greater depths in the central and southern parts of the study area. In these areas, fine-grained clayey silts accumulate in the deeper waters protected from strong storm conditions. However, even in these lower energy environments, sediments are locally remobilized as evidenced by the presence of sedimentary furrows. Anthropogenic artifacts (e.g. trawl marks, dredge-spoil disposal mounds, and a pipeline/cable) are also present on the sidescan sonar imagery.

Limulus and Us: Brining Community Awareness to Our Ecological Ties to the Horseshoe Crabs of Long Island Sound

Mattei, Jennifer H., Department of Biology, Sacred Heart University, Fairfield, CT 06825

Limulus polyphemus is a "living fossil" unique to the Atlantic Coast of North America. For over 350 million years this species has existed on Earth, yet in recent years its numbers have begun to decline throughout its range. Causes for decline include harvesting as bait for the fishing industry, collection for biomedical research and the pharmaceutical industry, and habitat degradation. *Limulus* has a unique ecological role in Long Island Sound and is considered to be of critical importance to the fitness of migratory shorebird populations. Most of what is known about the ecology of the species is based upon studies conducted in the southern portions of its range. The status of the Long Island Sound population is largely unknown. Due to the fact that the adult breeding population is harvested in large numbers, the population may decline rapidly before conservation measures can be enacted.

Initial results from a four year tagging program show that the *Limulus* population of Long Island Sound ranges extensively throughout the Sound. The population has a skewed sex ratio with more males than females present on breeding beaches. Horseshoe crabs rarely come back to the same beach to spawn in consecutive years. Less than 1% of the crabs tagged at Milford Pt. have returned to spawn again. However, 9% of the horseshoe crabs tagged return to breed on the same beach more than once in a season.

More information is needed and more volunteers are required to gather enough data to help in the management and conservation of this important species.