FINAL REPORT

MONITORING OF BOTOM WATER AND SEDIMENT CONDITIONS AT CRITICAL STATIONS IN WESTERN LONG ISLAND SOUND

VOLUME 1 of 2

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1.0 INTRODUCTION

1.1 Background

Long Island Sound is one of the best-known, best studied, and economically important estuaries along the east coast of the United States (Kaputa,N.P.& C.B.Olsen,2000; LISS,1994; McCall,P.L.1977; Paskevich,V.F.& L.J.Poppe,2000; Rhoads,D.C., 1994; Rhoads,D.C.,et al.,1978; Riley, 1952; Welsh, B.L.&F.C.Fuller,1991;Welsh, B.L,et al.,1994). The main sources of freshwater to the Sound are the Connecticut, Housatonic, and Thames Rivers in Connecticut; the primary source of marine water to the Sound is the Atlantic Ocean, which flows into LIS at its eastern end through the Race and at its western end via the East River. The Sound, bordered by Long Island, New York on the south, Connecticut on the north, and New York City on the west has always provided important resources to the people who lived along it (LISS,1994). It is home to a diverse assemblage of plant and animals, many of which are commercially important.

The location of Long Island Sound, adjacent to one of the largest metropolitan areas in the world, makes it subject to all the coastal problems associated with large human populations, including over-fishing, point source pollution (e.g. sewage disposal, power plant effluents), non-point source pollution (e.g. pesticide and fertilizer run-off, road residue run-off), oil tanker spills, the introduction of non-indigenous species, loss of wetlands to development and sea level rise, and the development of moderate-to-severe hypoxia during certain times of the year (Kaputa,N.P.& C.B.Olsen,2000).

From historical times, the Sound has been used as a source of food, recreation, and commerce. Up until recently, Long Island Sound was known to be an important commercial source of both shellfish (e.g. oysters, scallops) and lobsters. The Long Island Sound lobster fishery was worth over $30 million in 1998. In fact, total lobster landings in Long Island Sound increased by almost 900,000 between 1996 and 1998 (CT. DEP Bureau of Natural Resources, Marine Fisheries Office, 2000). The massive lobster die-off during the late Summer-Fall of 1999 effectively shut down the lobster fishery, especially in Western Long Island Sound. The majority of experts agree that the die-off most likely resulted from a deadly synergy of environmental and biological stressors, including but not limited to, warmer than normal water temperatures, hypoxia, and a parasitic paramoeba. Many lobstermen also believe that pesticides and other anthropogenic contaminants may also have played an important role in the lobster mortality.

Lobster die-offs of varying degrees have occurred previously in Long Island Sound, usually during the Fall (CT DEP Bureau of Natural Resources, Marine Fisheries Office, 2000) and have been attributed to the occurrence of stressors, such as hypoxic conditions and/or high bottom water temperatures, that weaken the lobsters and make them more susceptible to disease and death. The die-off of 1999 was itself preceded, in 1997 and 1998, by higher-than-normal lobster mortality in the Greenwich and Stamford areas that went mostly unreported until the massive die-off of 1999.

Although the final cause of death in the massive lobster die-offs appears to be a biological pathogen, adverse environmental conditions may play a significant role in weakening the lobsters, making them more susceptible to infestation by biological pathogens. One common environmental thread that runs through all of the lobster die-offs in LIS, both the massive ones and the more minor ones, appears to be the presence of hypoxic/anoxic waters (and their associated metabolites).

The chemical basis of hypoxia is described in a simple manner in the paragraphs that follow.

1) Water is capable of carrying a certain amount of dissolved oxygen within it. The amount of dissolved oxygen that water can hold – known as its saturation point – depends on the temperature of the water – simply put, colder water holds more oxygen than warmer water. Normal oxygen saturation for LIS waters is approximately 7 mg/l-dissolved oxygen.

2) Oxygen is renewed in marine waters in two general ways – phytoplankton productivity in the upper water column (photic zone) introduces oxygen into water via photosynthesis and surface wind mixing of the waters facilitates oxygen exchange with the atmosphere, renewing the oxygen content of surface waters.

3) Oxygen is consumed in marine waters in several ways. The organisms that live in the marine environment consume oxygen during respiration, plant respiration in the photic zone also consumes a certain amount of oxygen although significantly less than what is produced by the phytoplankton, and, finally, the sediments consume oxygen via bacterial degradation of organic matter.

4) This last point is extremely important for sediments that are high in organics, such as those of Western Long Island Sound. Such sediments, in contact with oxygen, will consume oxygen via aerobic bacterial decomposition of organic matter. This sediment oxygen demand can become quite large if bottom water oxygen renewal is limited by either water temperature or the existence of a stratified water column which prevents vertical mixing, ultimately setting the stage for the onset of hypoxic and perhaps even anoxic bottom (within 2 cm of the sediment-water-interface) waters.

Hypoxia in Long Island Sound is a problem of long-standing. Its main cause appears to be nitrogen enrichment from both point and non-point sources of pollution (CT. DEP Bureau of Natural Resources, Marine Fisheries Office, 2000;Kaputa,N.P.& C.B.Olsen,2000; Welsh, B.L.&F.C.Fuller,1991;Welsh, B.L,et al.,1994). Beginning in 1988 and continuing to today, a comprehensive monitoring of western LIS waters for physical, chemical, and biological data was undertaken. The data gathered over the years have been used, among other things, to assess the effectiveness of nutrient-reduction efforts in the Sound and to evaluate long-term trends in water quality.

The collapse of the lobster fishery in WLIS during September of 1999 served to raise concerns in the scientific and regulatory communities about the environmental processes that regularly occur in WLIS. Although hypoxic conditions in 1999 were less severe and affected a smaller area than they had in 1998, they continued for 50 days – a relatively long time (Figure 1.1). The majority of the hypoxic stations were located in the western part of the Sound and the Narrows. Water temperatures in 1999 were also up to 3 ^{o}C warmer than those of 1997 – meaning that the bottom waters were already holding less oxygen than normal The sudden tremendous rainfall of late August and early September broke up an otherwise dry summer, rapidly freshening the waters of LIS and contributing to the breakup of an otherwise weak stratification.

Most experts agree that the massive die-off observed in 1999 did not result from any one particular factor, but was the result of the co-occurrence of numerous factors that weakened and stressed the lobsters beyond their limits. Sediment profile images (SPI) taken immediately after the lobster die-off showed the presence of an extremely shallow (< 1 cm deep) Redox Potential Discontinuity (RPD) in the sediments underlying the majority of lobster fishing grounds in the western Sound. Such an RPD is usually indicative of a recent hypoxic/anoxic event.

The summer of 1999 was not the first summer during which WLIS experienced severe, widespread occurrences of hypoxic conditions and loss of fisheries – it was simply the most widespread and the most economically costly occurrence. Although many other factors (e.g. neoparamoebae, high water temperature, pesticide spraying, bait degradation products) may have contributed to the die-off, bottom water hypoxia was recorded early in the summer (CTDEP monitoring data) and during the massive die-off (as recorded by SPI).

In the Spring of 2000, the USEPA and NEIWPCC authorized a four month study (August-November) of 36 sites in WLIS. The majority of these sites coincided with stations previously sampled immediately following the die-off in October of 1999. In addition to SPI work, the USEPA-NEIWPCC study also analyzed bottom waters (< 5 cm above the bottom) for dissolved oxygen, hydrogen sulphide, and ammonia. Although interesting data was gained from this study, the timing of the study was such that it missed most of the months when WLIS is most susceptible to hypoxia (June-August). This project sought to address this lack of information, particularly as it relates to the release of hydrogen sulphide and ammonia into WLIS bottom waters.

One of the issues central to management of the lobster fishery in western Long Island Sound is the condition of bottom waters with regards to hypoxia. The onset of hypoxia in western Long Island Sound results from the interplay of many different factors, including air and water temperature, rainfall, currents, amount and type of organic matter, initial bottom water oxygen levels, anthropogenic inputs, and degree of stratification. The individual contribution made by any of these factors can vary from year to year.

The work presented here evaluates the role of hypoxia and anoxia (and related sulphide and ammonia releases from the sediment) as a structuring influence on the benthic environment and communities of western Long Island Sound, especially as they pertain to lobster habitat. It is hypothesized that long-term exposures to low oxygen and high levels of ammonia (and perhaps hydrogen sulphide) may have induced a physiologically-stressed state in the lobsters that died in 1999, weakening their immune system, and setting them up for disease.

1.2 Study Description and Objectives

The study presented in this report was conducted by Dr. Carmela Cuomo, of Yale University, and Mr. Raymond Valents, of SAIC, during the year 2002. It was conducted as a follow-up to the 2000 RARE-funded study, in order to obtain a more complete picture of bottom water and sediment dynamics in WLIS over the course of a year. The 2002 study was performed under contract to the New England Interstate Water Pollution Control Commission (NEIWPCC), with funding provided by the USEPA Long Island Sound Program, as well as with funding provided by Connecticut Sea Grant, New York State Sea Grant, Connecticut Department of Environmental Protection, and the New York State Department of Environmental Conservation.

The study consisted of several tasks. A series of field surveys, begun in April and continued through December of 2002, was conducted in order to obtain sediment profile images and bottom water data (dissolved oxygen, hydrogen sulphide and ammonia) from 12 sampling stations in WLIS (Figure 1.2). These sites, identified as "critical sampling sites", were culled from 36 total stations previously sampled during surveys conducted by SAIC for the NYSDEC immediately following the massive lobster die-off in 1999 and by SAIC and the PI as part of the USEPA-NEIWPCC RARE study conducted from August-November of 2000. The objective of these field surveys was to examine overall benthic habitat quality, as revealed by SPI photographs, bottom water chemical conditions, and benthic organisms. The data obtained from this study was to be compared to that collected from CTDEP water quality monitoring surveys and other sources for the same time period to look for points of agreement and divergence. Finally, the data obtained in this study was to be compared to the conditions known to be in existence during the summer and fall of 1999 in order to further understand the role that hypoxia, as well as ammonia and other potentially toxic metabolites present in the bottom waters at the time, may have played in the lobster mass mortality.

During the course of this study, a severe lobster mortality occurred in Central and Eastern Long Island Sound. The PI and her team were asked to investigate the conditions in this area during this ongoing mortality event. Therefore, in addition to the three tasks outlined above, the investigators conducted a field survey of a portion of Central and Eastern Long Island Sound during August, 2002.

This report presents the results of the water quality measurements and benthic data collected under Task 1, a brief summary of the SPI data collected under Task 1, the results of Tasks 2 and 3, and the results of the additional Central and Eastern Long Island Sound investigation. The complete SPI report can be found in a separate companion report (Volume 2).

2.0 METHODS

2.1 Station Locations and Sampling Activities

A total of 12 stations, generally following a north-south transect across the Eastern and Western Narrows of Long Island Sound, were sampled over the course of the study (Figure 2.1). These sampling stations were chosen from all of the sites sampled during the 1999 and 2000 work, also conducted by SAIC and the PI. The sites selected represented a mixture of conditions ranging from relatively low dissolved oxygen, high ammonia, high hydrogen sulphide bottom waters and extremely shallow aRPD to normal dissolved oxygen levels, lower ammonia, and lower sulphide levels in the bottom waters and a deeper aRPD. Table 2.1 provides station coordinates and sampling times. Due to problems with the winch motor on the NMFS ship, the R/V Loosanoff, samples were unable to be obtained in April.

All of the stations denoted in red in Figure 2.1 were occupied during the three SPI surveys undertaken, as well as during the seven water- sampling cruises taken. Benthic samples were taken concurrent with the REMOTS® deployment. Station A5 is routinely sampled by the Interstate Sanitation Commission and Station B3 is regularly sampled by the state of Connecticut's Department of Environmental Protection's water quality monitoring program. The stations listed in Table 2-1 were occupied during three survey cruises conducted on the following dates: May 19th, September 10th, and December 18th. The May survey work was conducted aboard the R/V John Dempsey, owned and operated by the CT DEP. Water chemistry surveys for August, September, and October were conducted aboard the R/V Loosanoff, owned and operated by the National Marine Fisheries Service located in Milford, CT. Water chemistry surveys of both WLIS and ELIS were conducted onboard the R/V Sea Wolf, owned and operated by the State of New York and Stony Brook University's Marine Sciences Research Center. Each vessel's GPS was used in positioning each station. Surveys were supposed to be conducted monthly, beginning in April and continuing through June; bi-monthly sampling was supposed to take place in July, August, and September. This project initially relied upon the availability of the R/V Loosanoff, which, unfortunately, experienced unforeseen technical difficulties that made it impossible to use for the majority of the early cruises. The R/V John Dempsey was unavailable as a substitute ship, except for the first cruise, thus samples for June and July were unable to be obtained. The R/V Sea Wolf was generously made available to us by the state of NY Department of Environmental Comservation for the remainder of the REMOTS® cruises and the additional survey work in Eastern Long Island Sound.

During each cruise, the P.I., or her technician, was responsible for obtaining discrete samples of seawater from within both 5 cm and 1 meter of the seafloor, using the Cuomo-Dobkowski (patent pending) bottom water sampler, a sampler designed specifically for this purpose. These water samples were fixed in the field and later analyzed for dissolved oxygen, ammonia, and hydrogen sulphide, using spectrophotometric methods.

Detailed descriptions of the chemical methods employed are provided in the following sections. Detailed descriptions of the methods employed for the collection of SPI are provided in Volume II of this report.

2.2 Dissolved Oxygen Measurements

Samples of bottom water (within 5 cm of the sediment-water-interface and at a distance of 1 m above the sediment-water-interface) were taken using the new version of the Cuomo-Dobkowski bottom water sampler (patent pending). This new sampler has been configured so as to prevent any sediment from getting trapped in the water samples. The samples were collected on water chemistry cruises, as well as coincident with the SPI during REMOTS® sampling surveys. The sampler was initially lowered to 1 meter above the bottom at each station using each vessel's small hydraulic winch and a sample was taken there; although, on the first cruise on the R/V Loosanoff, the sampler was deployed and collected by hand using pull lines as the hydraulic winch was not functioning. After the 1 m above the bottom samples were collected, the sampler was again lowered – this time to the bottom. The sampler was allowed to sit on the bottom for approximately 1 minute prior to filling in order to assure that any sulphides or ammonia accidentally released from the bottom upon contact of the sampler with bottom sediments would be cleared away by bottom currents. Water samples were taken prior to the REMOTS® deployment at every station in order to ensure clean, uncontaminated water samples.

Once the sampler reached its destination (1 m above the bottom or at the sediment-water-interface) water was drawn into the sampler and it was returned to the surface. Upon retrieval, the seawater was extracted from the samplers and treated with manganous sulphate and alkaline iodide solutions. Care was taken to insure that no air bubbles were trapped in the glass sample vials. Samples were stored in the ship's refrigerator until the cruise was over.

Upon docking, the samples were transferred to a cooler and brought back to the laboratory at Yale where they were analyzed for dissolved oxygen using a modification of the Winkler technique (Strickland and Parsons, 1977). This method is considered highly accurate unless oxygen concentrations fall below 0.1 mg/l, at which point the method will underestimate the amount of oxygen present by up to 0.0015 mg/l – an amount that is not considered significant. Preserved samples were treated with concentrated sulfuric acid and samples were stoppered and mixed until all the precipitate dissolved. Samples were treated with a thiosulphate solution and titrated using a starch solution in order to detect the end-point. A minimum of three replicates were analyzed per sample and six samples were taken per station – three at bottom and three 1 meter above bottom.

Standardization of the method was carried out immediately prior to the analyses using blanks and fresh reagents each time.

Section 2.3 Dissolved Sulphide Measurements

Samples for dissolved sulphide were obtained in the same manner as those for dissolved oxygen. Once the Cuomo-Dobkowski (patent-pending) was retrieved, water samples were extracted and treated with sodium hydroxide and zinc acetate in order to chemically precipitate out any sulphides present as zinc sulphides. The "fixed" samples were then stored in a cool, dark place for later analysis in the laboratory, using modifications of Fonselius (1969) and Cline (1969). Fonselius's (1969) method is appropriate when working with concentrations of sulphide < 300 umol/l whereas Cline's (1969) method is appropriate for use with waters containing higher concentrations ($\leq$ 1000 umol/l).

Preserved samples were treated with ferric chloride and a solution of N,N-dimethyl-p-phenylenediamine dihydrochloride. Color development occurs within 60 minutes if sulphides are present. Samples were read on a spectrophotometer set at 670 nm. Three replicates were analyzed per sample and six samples were taken per station – three at bottom and three 1 meter above the bottom.

Standardization of the methods was carried out prior to each set of analyses using a series of standards prepared from a freshly-made working solution. The exact concentration of sulphide in each of the standards was calculated using the formula:

$$\text{umol/l } S^{2=} = C \cdot D/E$$

where C= ml of working solution, D= concentration of sulphide in the working solution (umol/l) and E= volume of the flask + volume of the added reagents in ml.

Section 2.4 Dissolved Ammonia Measurements

Samples for dissolved ammonia were obtained in the same manner as those for dissolved oxygen. Once the Cuomo-Dobkowski (patent-pending) was retrieved, water samples were extracted, placed in sample tubes, and frozen for later analysis. A modification of the Parsons et al. (1984) technique was used in the analysis and involves treatment of the sample in an alkaline citrate medium with sodium hypochlorite and phenol in the presence of sodium nitroprusside, resulting in the production of a blue indophenol color when ammonia is present in the sample. The sample is then read on a spectrophotometer set to 640 nm. This method measures the sum of both NH_4^+ and NH_3. Three replicates were analyzed per sample and six samples were analyzed per station (three at bottom and three 1 m above the bottom). This method detects quantities of ammonia in the 0.1 – 10 umol/l range.

The method was standardized using freshly prepared ammonia standards and three blanks (de-ionized water). Ammonia concentrations were calculated as follows:

$$\text{umol N/l} = F \cdot E$$

where E= corrected extinction from blank and F= the calibrated standard value.

Section 2.5 Benthic Sampling and Identification

One grab sample was taken at five selected sites in May and December and at six sites in September to ground-truth the benthic communities identified in the sediment-profile images. The sites chosen represented areas that previous SPI studies had identified as ranging from anoxic-azoic to Stage III healthy communities. Each sample was sieved on board the research vessel; all material retained on sieves $\geq$ 250 um mesh was retained and preserved in 10% formalin and stained with rose bengal. Samples were returned to the laboratory where they were picked under an Olympus® dissecting microscope, identified, enumerated, and recorded. They are archived at Yale University's Peabody Museum. Quality assurance and quality control procedures employed in this study are documented in a Quality Assurance Project Plan (SAIC & Cuomo, 2002).

Section 2.6 Eastern Long Island Sound Lobster Mortality Cruise

The PI and a team of two SAIC scientists trained in the use of the REMOTS® system provided the necessary equipment to conduct a field survey in Eastern Long Island Sound in August, 2002 in response to a lobster die-off event. The survey was comprised of 1 field day, during which SAIC's REMOTS® sediment profile camera was lowered to the seafloor multiple times at each of 3 sampling stations and three replicate sediment profile images suitable for subsequent analysis and interpretation were obtained. Following each field survey, the REMOTS images were analyzed using SAIC's computer-based image analysis system. Standard measurements include grain size, depth of the redox-potential discontinuity (RPD), infaunal successional stage, presence/absence of methane bubbles or surficial bacterial mats (hypoxia indicators), and Organism-Sediment Index.

Bottom water chemical conditions in Eastern Long Island Sound were assessed, by the PI and her technician, at ten stations (Figure 2.1) during the August cruise, using the same methods described in Sections 2.2-2.4. Unlike the work in WLIS, near-bottom samples were not obtained because of time constraints.

3.0 RESULTS

This section presents the results of the chemical and benthic sampling undertaken in WLIS as part of this project, as well as the results obtained on the additional cruise to ELIS. The results from the REMOTS® sediment-profile imaging are presented in Volume II of this final report.

Section 3.1 Dissolved Oxygen Measurements

Near-bottom dissolved oxygen averages increase from 0.9695 mg/l in August to 3.331 mg/l in September (Figure 3.1). Interestingly, average values then decreased to 1.258 mg/l in October and finally rose to 4.117 mg/l in December. Bottom water dissolved oxygen averages display a similar trend (Figure 3.2), although the bottom water data set begins with May values. These, as expected, are high (5.621 mg/l) and represent a time before hypoxic conditions are established in WLIS. The values decrease to their lowest (0.959 mg/l) in August and increase in September (2.557 mg/l). This increase is followed by an October decrease (1.360 mg/l) paralleling the near-bottom water conditions. Finally, the bottom waters increase to 4.609 mg/l in December revealing a full reoxygenation of the bottom waters and a decline in sediment microbial activity associated with the onset of cooler temperatures.

Section 3.2 Dissolved Sulphide Measurements

Near-bottom sulphide measurements increased from an average low of 0.019 umol/l sulphide in August to an average high of 0.442 umol/l sulphide in October (Figure 3.3). Lower values of 0.278 umol/l and 0.236 umol/l were recorded in September and December, respectively. Bottom water sulphide levels followed a similar trend (Figure 3.4). Average levels recorded at all stations in May were 1.330 umol/l; levels reached an average low of 0.101 umol/l in August. Measurements reveal an increase in bottom water sulphide values through September and into October, when levels reach an average high of 0.506 umol/l. December saw a decrease in dissolved sulphides, with average measured levels of 0.275 umol/l.

Section 3.3 Dissolved Ammonia Measurements

Average near-bottom ammonia levels increased steadily from August (0.383 umol/l) to October (2.198 umol/l) then underwent a decline in December (0.615 umol/l) (Figure 3.5). Average bottom ammonia levels for May were measured at 1.219 umol/l (Figure 3.6). These levels decreased steadily through July and August (0.784 umol/l and 0.731 umol/l, respectively) and then underwent a three-fold increase in October (2.101 umol/l). Measured levels then decreased to 0.576 umol/l in December.

The average data for October under-represents the true state of the majority of stations in WLIS; at four stations (10, 14, 21, and 24) ammonia was undetected within the limits of the method. These four stations lie towards the easternmost end of the sampling sites. The average measured ammonia level for sites in WLIS for the month of October excluding these four is 3.035 umol/l .

Section 3.4 Benthic Sampling and Identification

The results of the benthic sampling are shown in Figures 3.7-3.9 and Table 3.1. There is little variation in species composition across all stations. In addition, the density of organisms per square meter remains low. From a functional perspective, the organisms that dominate the benthos in WLIS from the months of May through December are part of a typical late Stage I-very early Stage II community.

Station 10, the easternmost station sampled, was occupied in May by errant polychates (Nepthys), as well as by small spionid polychaetes, and head-down deposit feeders (Clymenella torquata). By September, the site was still occupied by some errant polychaetes, as well as by small spionid polychaetes, although there were lower densities of both of these groups. Head-down deposit-feeders were absent from the September sampling, replaced by yet another near-surface deposit-feeder, Pherusa sp. The December sampling revealed a benthos dominated by the tubiculous amphipod, Ampelisca abdita, as well as by small surficial spionid and capitellid polychaetes. Errant polychaetes declined in number.

Station B3, located within the center of the study area, was unable to be sampled in May because of time constraints. However, it was sampled in both September and December. The

September sample was composed primarily of extremely low numbers of small surface-deposit feeding polychaetes. The same site, sampled in December, had a similar faunal density and composition. Little change occurred over six months at this station.

Station A5, located southwest of B3, was occupied in May by small, surface deposit-feeding polychaetes and ampeliscid amphipods. Numerically, Polydora was the dominant organism at this site. In September, the majority of these species were gone and densities of remaining organisms (Nepthys and Nereis) were extremely low. By December, both organismal diversity and density had slightly increased and the benthos consisted of errant and small, surface feeding polychaetes and tubiculous amphipods.

Station 32, situated within the Western Narrows, was occupied in May by ampeliscid amphipods, oligochaetes, and a variety of surface and deposit feeding polychaetes. A few deep deposit feeders were also identified, although they were in the numerically minority at this site. The data for September, once again, show ampeliscid amphipods and small surface deposit feeding polychaetes dominating the benthic fauna. The deeper deposit feeders were not found during this sampling. The December sampling showed the site to be dominated by spionid polychaetes and oligochaetes. Ampeliscid amphipods were present, but in lower densities than in September. Errant polychates were also present, albeit in small numbers.

Station 29 is the westernmost site sampled and occurs in the Western Narrows. The benthic fauna was dominated by ampeliscid amphipods and errant polychates. Small numbers of surface deposit feeding polychaetes were also identified at this site. By Septembert, there was little change in species composition, although spionid polychaetes dominated numerically. This held true for December, as well. Species composition did not change appreciatively but the numerically dominant organisms were oligochaetes.

In addition to the five stations sampled above, two other stations were sampled at different times. Station 14, located west of Station 10, was only sampled in September. This site was occupied primarily by low numbers of near-surface and surface deposit feeders.

Station 19, located southwest of Station 10, was only sampled in May. At that time, its fauna consisted of a diverse mixture of surface and deep deposit feeding polychaetes. Ampeliscid amphipods were present in significant numbers.

Section 3.5 Summary of REMOTS® Sediment-Profile Imaging

REMOTS® surveys conducted in WLIS during 2002 are consistent with those of previous surveys conducted in 1999 and 2000. There continues to be a correlation between eutrophication (as manifested in seasonal hypoxia) and compromised benthic habitat quality in WLIS. Surface sediments that appear highly anoxic were visible in the profile images in all three of the 2002 surveys. Similar to previous years, the stations in the Western Narrows generally exhibited more severe degradation in benthic habitat quality than those in the Eastern Narrows.

Surface dwelling, opportunistic benthic organisms (Stage I) were widespread in all three of the 2002 surveys, although some of the samples stationed in the Western Narrows appear to be azoic.

Complete results are presented in Volume 2 (SAIC Report 640) of this report.

Section 3.6 Eastern Long Island Sound Lobster Mortality Cruise

Sub-Section 3.5.1 Dissolved Oxygen

Bottom water dissolved oxygen levels in ELIS in August, 2003 ranged from a low of 1.839 mg/l at station T7 to a high of 4.241 mg/l at station T5 (Figure 3.10). Average bottom water dissolved oxygen levels were 3.305 mg/ l. These numbers were very similar to measurements taken in WLIS at the same time.

A CTD cast, taken at the same time by Dr. Robert Wilson, from the MSRC at Stony Brook, NY, and at the same stations, revealed dissolved oxygen levels at 1 meter above the bottom. These levels were slightly higher than those measured chemically right at the sediment-water interface but were within the same range.

Sub-Section 3.5.2 Dissolved Sulphide

Bottom dissolved sulphide levels for ELIS in August, 2003 ranged from a low of 0.076 umol/l at Station T8 to a high of 0.607 umol/l at Station T4 (Figure 3.11). The average level for all stations sampled was 0.317 umol/l, which is higher than that measured in WLIS.

Sub-Section 3.5.3 Dissolved Ammonia

Bottom water dissolved ammonia levels for ELIS in August 2003 ranged from a low of 0.193 umol/l at station T6 to a high of 0.357 umol/l at station T2 (Figure 3.12). The average value for dissolved ammonia at all stations was 0.287 umol/l, lower than the average for WLIS stations during the same time period.

Sub-Section 3.5.4 REMOTS® Sediment-Profile Imaging

Sediment profile images were obtained at three stations in ELIS during the August sampling cruise. The sediments at these three stations consist primarily of fine-grained silt-clay or mud with admixtures of very fine sand. The images showed evidence of darker, reduced sediments at depth overlain by oxidized sediment. Stage I was the dominant successional stage at Station A1, while the other two stations showed feeding voids at depth, which may be representative of Stage III communities.

4.0 DISCUSSION

Section 4.1 Seasonal Trends in Dissolved Gas Concentrations

Sub-Section 4.1.1 Dissolved Oxygen

According to the CT DEP Hypoxia Monitoring Program, the 2002 hypoxic event began on June 26, 2002 and ended on August 28, 2002. Data collection for this project began in May, 2002. Samples collected in May revealed bottom water DO measurements above 4.3 mg/l at all stations. Such levels qualify as meeting or exceeding the interim management goal of the US EPA. These levels reflect the combined effects of a well-mixed water column and the cooler water temperatures associated with spring.

Unfortunately, due to lack of problems with the ship, the PI was unable to obtain samples for the months of June and July so it is not possible to know whether bottom water DO correlates with the DO measurements taken by the CT DEP monitoring program. Correlations were attempted with the EMPACT buoys, however, archived data from this time was not available.

The present study does indicate that bottom water and near-bottom water hypoxia was moderately severe ($\leq$ 2.0 mg/l) at all stations in August, 2003. Bottom water DO levels at all stations sampled by the PI were $\leq$ 1.5 mg/l, except for Station 14 – the easternmost station. Station 14 had average bottom water DO measurements of 2.3 mg/l. These data are consistent with the data collected by the CT DEP and plotted on their hypoxia maps for this same time for this region.

Near-bottom dissolved oxygen at all stations sampled in September 2002 had concentrations ranging between 2 and 5 mg/l – falling within the moderate – excellent scale of the DEP. Bottom water dissolved oxygen levels were between 1.8 and 3.0 at all stations except for Stations 10 and 14 – the two easternmost stations – which had readings above 4.0 mg/l. These levels (below 3.0 mg/l) fall within the moderately severe to moderate hypoxic scale of the CT DEP. The hypoxia maps produced by the CT DEP for the same time period show no station below 3.5 mg/l DO and the majority of stations in LIS with DO levels above 4.8 mg/l. Thus, for this time period, a difference begins to emerge between processes occurring at the sediment-water interface that control true bottom water dissolved oxygen levels and those that occur higher in the water column. This pattern was similar to that observed by the PI during the 2000 study.

Near-bottom and bottom water DO measurements for October reveal a return to hypoxic conditions at and near the sediment-water interface. Near-bottom levels at all stations, except for Station 14, were less than 2.0 mg/l – well within the moderately severe hypoxia definition of the CT DEP. Station 14 had measured dissolved oxygen concentrations slightly above 2.0 mg/l. Similar conditions were recorded by the PI in the fall of 2000.

The DO measurements for December were all above 3.0 mg/l except for station 24, which had average bottom water DO measurements of 2.8 mg/l, placing all stations within the marginal-excellent water quality framework of the monitoring program. Given the arrival of cooler water temperatures, strong mixing, reduced plankton inputs to the sediments, and a cooling of sediment temperatures leading to a drop in bacterial decomposition rates, it is not unexpected that bottom water DO levels should rise at this time of year.

From the data collected, it appears that near-bottom and bottom water dissolved oxygen levels rise during the winter in LIS and potentially remain elevated through the late Spring. Increasing water temperatures, coupled with the aerobic water column degradation of the Spring phytoplankton bloom, and the development of a stratified water column during the summer months, contribute to the onset of hypoxic water column conditions in WLIS during July and August. These conditions, when severe, may, as in the present study, reach the very bottom waters of WLIS. The breakdown of stratification in the late summer-early Fall (September for the 2002 study), however, results in a return of oxygen to bottom and near-bottom waters in WLIS. The return to hypoxic bottom water conditions in the early-mid Fall (October for the 2002 study) appears to be caused by anaerobic decomposition on and within the sediments of WLIS and diffusive processes occurring at the sediment-water-interface. As both water and sediment temperatures continue to drop during the late Fall and early Winter, bacterial decomposition rates in the sediments begin to slow and the bottom waters return to an oxygenated state. Thus the overall pattern of dissolved oxygen in the bottom waters in WLIS appears to be bimodal – high DO in the winter and early spring, low DO in the summer, a brief return to higher levels in the late summer-early Fall as stratification breaks down, and a return to low levels in the Fall as benthic processes dominate.

Sub-Section 4.1.2 Dissolved Sulphide

Bottom water sulphide measurements taken in May were split according to location. Levels at Stations 10,14,19,24, 32, A5, and B3 were all below 1 umol/liter. Those taken at Stations 21, 27, and 29 had levels ranging from 1-3 umol/l whereas those at Stations 28 and 31 had levels between 3 - 4.5 umol/l. These stations, all of which are located in the westernmost area of the study site, had the highest sulphide levels measured during this study, although levels measured in the Fall of 2000 were higher. Near-bottom sulphide levels at all stations in May were not able to be obtained due to problems with the ship's winch.

August values for both near-bottom and bottom water sulphides at all stations remained $\leq$ 1 umol/l. This strongly supports the idea that the main contributors to hypoxic conditions during the summer months in WLIS are water stratification, water temperature, and water column aerobic decomposition of organic matter.

Sulphide in near-bottom waters and bottom waters remained below 1 umol/l through September at all stations sampled, although average values increased at almost all stations. The same trend was seen in October, with average sulphide values increasing at almost all stations sampled, but still remaining below 1.0 umol/l. December values returned to September levels.

Sulphide levels also appear to cycle over the course of a year in WLIS. Elevated levels of bottom water sulphide in the spring are most likely associated with rising sediment temperatures and the anaerobic decomposition of sediment organic matter and newly deposited dead plankton from the spring plankton bloom. These levels are reduced during the summer because less organic matter is reaching the bottom because of stratification and less of a gradient in oxygen exists between the reduced sediments and the overlying water column – diffusive processes out of the sediment are somewhat dampened. Sulphide levels rise in the fall, once stratification is broken down. This Fall increase is most likely due to three factors – the arrival of dead plankton from the Fall bloom stimulates decomposition, the warm temperature of the sediments which stimulates bacterial production, and increased diffusion of sediment sulphides into the overlying water. As sediment temperatures drop over the remainder of the Fall and into the Winter, bacterial decomposition rates slow down, no new plankton is added to the sediment, and the oxygen chemically diffuses into the uppermost cm of sediments, effectively cutting off sulphide release.

It should be noted that sulphide levels measured in the Fall of 2002 were significantly lower than those measured in 2000. Such yearly discrepancies are to be expected and most likely result from a combination of factors, including sediment and water temperature, amount and depth of bioturbation, and amount of labile organic matter in the sediments.

Sub-Section 4.1.3 Dissolved Ammonia

Ammonia was also detected in bottom waters in May at all stations in concentrations ranging from near 0 up to 3.0 umol/l; the highest concentration was recorded at Station 28 in the Western Narrows. August samples of both bottom and near-bottom waters contained little detectable ammonia, supporting the concept that summer hypoxia in WLIS is a process driven primarily by processes occurring in the upper water column.

September bottom water levels were similar to August levels; near-bottom water levels, however, underwent an increase in detectable ammonia. Stations 27 and 29 had recorded ammonia levels between 1.0 and 2.3 umol/l, whereas all other stations remained under 1.0 umol/l.

Both average near-bottom and bottom water ammonia levels increased at a majority of stations during October. Ammonia was measured in sampled waters at levels ranging from 1.6-5.2 umol/l at all stations, except Stations 10, 14, and 21 (the easternmost stations), where no ammonia was detected.

Ammonia levels in December, for both near-bottom and bottom water samples, returned to levels similar to those of August – detectable but under 1.0 umol/l at all stations sampled.

Yearly ammonia trends in WLIS bottom waters follow those of sulphides. Winter is a time of reduced sedimentary organic matter decomposition, so little to no ammonia is released from sediments. Ammonia levels begin to increase in the Spring, following the plankton bloom. Summer ammonia levels decrease, most likely because stratification reduces the amount of fresh organic matter reaching the bottom of WLIS and there is a lower diffusive gradient between the overlying hypoxic waters and the pore waters of the sediments. The breakdown in stratification in the Fall, coupled with the Fall plankton bloom, results in an increased production of ammonia within the sediments and an increase in the diffusive release of ammonia into the overlying water column. As sediment and water temperatures cool down in the late Fall, ammonia production in the sediments falls off and ammonia release into the bottom waters declines.

Section 4.2 Dissolved Oxygen, Sulphide and Ammonia Concentrations and Their Implications for Lobster Health

The data from the present study reveals a decoupling of processes controlling bottom water chemical conditions in WLIS over the course of a year. It appears that in the summer, when water temperatures are at their warmest and stratification is present, hypoxia is a process driven from the "top-down". That is, hypoxia in WLIS in the summer is primarily driven by density stratification, aerobic decomposition of plankton in the water column, and lack of bottom water oxygen renewal owing to this stratification. Such information is not new and has formed the underlying basis for the ongoing successful water quality monitoring program established by EPA and carried out by CT DEP. This gradual depletion of oxygen from the bottom waters of WLIS, however, explains the lack of lobsters and other mobile bottom-dwelling organisms *during the summer months* in WLIS. Organisms capable of migrating out of areas of low oxygen, such as flounder and lobsters, undoubtedly do. In fact, lobstermen have repeatedly stated that there were no lobsters in LIS during the hypoxic summer period preceding the die-off in 1999. It is likely that the lobsters and other mobile organisms migrated to more oxygen-rich areas of LIS.

The more interesting question regarding the lobster die-off of 1999 concerns the timing of the event – it occurred not during the peak of the hypoxia - but in the fall of 1999, after the water column had become remixed. Data from this and the 2000 study strongly suggest that the fall is the time when sediment processes determine bottom water chemistry, as opposed to surface processes, although the two remain connected. In other words, in the fall, bottom water hypoxia and anoxia result from sediment-driven processes – a "bottom-up" effect. This effect is not captured by routine water column monitoring measurements as it is generally confined to within 0.25 meters or less of the sediment-water interface. More important, however, is the fact that lobsters and other benthic organisms live within the zone where this effect is occurring.

The release of reduced end-products of anaerobic organic matter decomposition into an otherwise oxic water column, such as appears to occur in WLIS in the fall, produces several effects. First, the reduced end-products undergo oxidation reactions in the presence of oxic bottom waters, resulting in a reduction in the local dissolved oxygen levels. Secondly, organisms present in the area are going to encounter both sulphides and ammonia, along with oxygen, in the bottom waters. Lobsters can tolerate sulphides better than ammonia; both of these, however, have been shown to physiologically stress the lobsters potentially rendering them less able to leave the area and, thus, more likely to be exposed to chronic, low-level, physiological stressors. Thirdly, both ammonia and sulphide have been shown to negatively affect lobsters, especially ammonia. Lobsters become sluggish in the presence of even low-levels of ammonia. Lobsters exposed to warm temperatures, low dissolved oxygen, and ammonia at the concentrations measured in this and previous studies undergo significant mortality (A. Draxler, NMFS, personal communication).

Section 4.3. Relationships Among SPI, Dissolved Gases, and Benthic Organisms

Of particularly great concern is the result of the benthic survey done during this study. The benthic survey revealed a depauperate benthos in WLIS. Few deep-deposit-feeding organisms were found at any time during the course of this study. SPI reveal a bottom that was severely compromised and one that remained in a functionally hypoxic to anoxic state throughout the year. The benthic survey data supports the REMOTS surveys, although it casts doubt on the meaning of the feeding voids visible in some SPI at a few in WLIS. The majority of benthic organisms found and identified in the grab samples consisted of small, surface dwelling polychaetes and tubiculous amphipods – so-called Stage I-early Stage II members. Few to no late Stage II and III class organisms were found in any of the samples. This implies that WLIS is not undergoing benthic development over the course of a year but has rather arrived at a state where the conditions are such that the benthic communities remain relatively undeveloped. This has long-term implications for carbon cycling and sediment-organic-load within WLIS. Bioturbation by Stage II and later organisms contributes to oxidation of the sediments and, as a by-product of this, to the aerobic breakdown of sedimentary organic matter. Since aerobic decomposition is a much more efficient process that anaerobic organic matter decomposition, the presence of Stage II and III benthic communities actually decreases the organic carbon content of the upper sediments, making them less likely to undergo significant anaerobic decomposition and become a major source of sulphide and ammonia to the bottom waters. It appears that WLIS has potentially entered into a negative feedback loop – and organic carbon delivery to the bottom (in the form of plankton, dead or dying benthos, and/or bait), coupled with sediment and water temperature will play vital roles in determining the extent and strength of the spring and fall release of sulphide and ammonia and other end-products of anaerobic organic matter decomposition. It appears that it is the release of these end-products that are exerting a strong structuring influence on the benthic communities in WLIS, including the lobster populations.

Section 4.4 Implications for the Overall WLIS System

The presence of ammonia and sulphide in near-bottom and bottom waters suggests that anaerobic decomposition processes are dominant over aerobic processes in near-surface sediments of WLIS. This is supported by both the REMOTS images and the benthic samples. The REMOTS images from all three surveys showed sediments that were highly anoxic exposed either directly at the sediment-water interface or covered by a thin veneer of oxidized sediment. The benthic data reveals the absence of an abundance of deep deposit-feeders and the dominant presence of small, surface-dwelling organisms – the organisms that make up the early – late stage I communities of LIS. Many stage I organisms (e.g. capitellid polychaetes) have been shown to be tolerant of reduced conditions, such that they might be able to survive in areas where reduced metabolites are defusing out of the sediments even as overlying waters are oxygenated (Vismann et al., 1998; Cuomo 1984,1985). This, combined with the chemical data, strongly suggests that the organic-rich bottom sediments of WLIS are compromised and remain in a functionally hypoxic-anoxic state for the majority of the year.

5.0 CONCLUSIONS AND RECOMMENDATIONS

Section 5.1 Conclusions

The CT DEP Hypoxic Monitoring Program reported that the summer of 2002 was one of the worst recorded hypoxic events to ever occur in LIS. DEP hypoxia maps reveal significant regions of LIS with water column dissolved oxygen levels $\leq$ 1.0 mg/l (Figure 5.1). This is a functionally hypoxic-borderline anoxic state. Water temperature trends for the same time reveal higher than normal conditions at both the Battery in WLIS and the Race in ELIS commencing in July and lasting until mid-September (Figure 5.2). The results of the study reported on here were consistent with these observations. Near-bottom and bottom dissolved oxygen levels remained relatively low throughout the duration of this study and paralleled the DEP values. Near-bottom levels of DO recorded in this study were consistent with those measured by the DEP at approximately the same depth in the water column. Bottom water DO measurements were consistently lower than near-bottom waters and represent the influence of sediment oxygen demand, as well as lack of bottom water oxygen renewal, in the system. Highest levels for bottom water and near-bottom water DO were recorded in May and December, when colder water was present and WLIS was well mixed.

The measured levels of sulphide and ammonia do not exhibit a simple correlation to low oxygen, however; they do exhibit somewhat of a reverse correspondence. The highest levels of ammonia release from the sediments were measured in October whereas the highest level of sulphide release was measured in May. Both of these events were accompanied by a local bottom water decrease in dissolved oxygen levels, however, the overall oxygen content of the water column remained relatively high ($\geq$ 4.5 mg/l). The low levels of bottom DO measured in October apparently result from the consumption of oxygen by the re-oxidation of anaerobic organic matter degradation products actively diffusing into the water column from the surface sediments. While the trends are the same as 2000, both sulphide and ammonia levels were detected in concentrations well over 1.0 umol/l in the fall of that year, compared to the lower levels detected this year. Nevertheless, since there was no sudden return to stratified conditions recorded for this time period, it becomes necessary to invoke some other explanation on this fall hypoxic occurrence. The data strongly support the idea that fall hypoxia is driven by processes occurring within the sediments and at the sediment-water interface.

Section 5.2 Recommendations

Yearly inputs of organic matter to the sediments need to be monitored, in conjunction with surface and sediment temperatures, and benthic community development in WLIS in order to formulate a management plan that may allow for the existence of some commercial fishery in

WLIS. More importantly, these factors and their products (sulphide and ammonia release from sediments, bottom water and water column hypoxia and anoxia, sediment-oxygen demand, stratification) need to be monitored and need to form the basis of a management plan for WLIS in an effort to prevent WLIS from becoming a dead zone akin to that which exists in the Gulf of Mexico at the present time.

6. Literature Cited:

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

McCall,P.L.1977. Community patterns & adaptive strategies of infaunal benthos of Long Island Sound. J.Mar.Res.35:221-266.

Paskevich,V.F.& L.J.Poppe, eds.,2000. Georeferenced Sea-Floor Mapping and Bottom Photography in Long Island Sound. U.S.Geological Survey Open-File Report 00-304.CD-ROM.

Rhoads,D.C.,P.L.McCall,&J.Y.Yingst,1978. The ecology of seafloor disturbance. Amer. Sci. 66:577-586.

Rhoads,D.C.,1994.Analysis of the contribution of dredged material to sediment and contaminant fluxes in Long Island Sound. Special Tech. Report. DAMOS Contribution #88.USACE, New England Division.

Riley,G.A.,1952.Hydrography of Long Island and Block Island Sounds – Bull. Bingham Oceanogr. Coll. 13:5-39.

Welsh,B.L. & F.C.Fuller,1991.Mechanisms controlling summertime oxygen depletion in western Long Island Sound – Estuaries 14:265-278.

Welsh,B.L.,R.I.Welsh,&M.L.DiGiacomo-Cohen,1994.Quantifying hypoxia and anoxia in Long Island Sound in: K.R.Dyer & R.J.Orth,eds.,Changes in Fluxes in Estuaries: Implications from Science and Management, Olsen & Olsen, pp.131- 137.

FIGURES