Sediment Contamination, Toxicity and Infaunal Community Composition in St. Lucie Estuary, Florida based upon measures of the Sediment Quality Triad

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Executive Summary

The St. Lucie Estuary and adjacent watershed have been highly altered in recent years from increased human population growth, intense agriculture and related activities, and periodic release of large amounts of freshwater from Lake Okeechobee. Heavy freshwater discharge following the 1998 El Nino event was associated with observations of unusually large incidences of fish with external deformities as well as other water quality-related problems, further exacerbating concerns about the ecological conditions of the estuary and status of its renewable resources. Since the St. Lucie Estuary is hydrologically connected to the Everglades, its study is also part of the larger ecosystem restoration and freshwater management efforts in South Florida, known as the Comprehensive Ecosystem Restoration Plan.

This study, addressing a major management concern on the environmental quality of the St. Lucie Estuary, has characterized the estuary in terms of environmental toxicity and described the extent and severity of habitat degradation using the sediment quality triad approach. The study design was geographically comprehensive and offered a parsimonious sampling of the estuary, using five sampling strata: North Fork, South Fork, Convergence Zone, Middle Estuary and Lower Estuary.

The study demonstrated that the overall level of chemical contamination in the estuary was low, notably in terms of contaminants that generally occur in urbanized estuaries, such as polycyclic aromatic hydrocarbons and polychlorinated biphenyls. This was demonstrated both by chemical analyses of sediment samples collected in all parts of the estuary and also by application of a Human Reporter Gene System biomarker based on an engineered cell line, which responds to and records the presence of such compounds in sediment and other environmental samples. The severity and spatial extent of sediment toxicity varied considerably, depending on the sensitivity of the specific test employed test and life stages of the test organisms.

The whole sediment bioassay using two species of amphipods did not demonstrate significant sediment toxicity; however, using juvenile seed clams as test organisms, over 80 percent of the study area was found to be toxic. The sea urchin fertilization test showed sediment porewater to be toxic in over one-half of the study area. In comparison, NOAA's nationwide data on sediment toxicity showed about 6 percent of the studied areas, comprising of over 7,000 sq km in 25 coastal bays and estuaries, to be toxic; for sediment porewater the fraction was 25 percent. No such estimate exists for the juvenile seed clam test.

Integrating data from levels of contamination, the severity of toxicity, and changes in the benthic infaunal community structure showed that the North Fork and South Fork portions of the estuary were relatively highly degraded, whereas the lower portion of the estuary, leading to the southern Indian River Lagoon, was relatively less impacted.

Even though the overall level of contamination was low, this study has identified copper as a contaminant of concern in the estuary from several lines of evidence; (1) the observed levels of copper approached or exceeded sediment quality guidelines that have been developed for Florida coastal waters; (2) indication, from a logistic model based on nationwide data on sediment toxicity, of a more than 50 percent probability of mortality in amphipods if exposed to sediment with the high copper concentrations observed in the estuary; (3) copper-spiked sediment bioassays using juvenile seed clams, which indicated that ambient concentrations of copper in the estuary far exceeded the "lowest observed effects concentration" and approached the copper concentration that would cause 50 percent mortality in test animals; and (4) a field study on oysters deployed in the estuary, which demonstrated high copper accumulations in oysters and concomitant sub-lethal effects from a suite of biomarkers.

There are many potential sources of copper in an estuary, extensive copper use in the estuary's watershed, notably as fungicide in citrus orchards and vegetable crops, as algaecide in suburban areas, leaching of copper-based anti-fouling paints on boats, and effluents from wastewater treatment plants; however, a study on source attribution of copper entering the estuary has not been conducted. A heavy metal and toxic chemical simulation model was developed from an existing hydrodynamic model to incorporate the sources, transport, and fate of copper in the St. Lucie estuary. Its computational code has been transferred to resource management agencies in Florida.

A logical next step in the study of the St. Lucie Estuary is to synthesize information from the suite of studies that have recently been funded in the region. This could be done using "integrated assessment" protocols that focus on the management requirements of scientific data and provide a means of data integration. This approach would facilitate determining priority of information needs and formulating relevant scientific questions for managing the estuary's ecosystem, including its human dimensions.

1.0 Introduction

1.1 Background

Largely in response to increasing human population along the coast and typical of south Florida's estuaries and coastal bays, the St. Lucie Estuary and its watershed have experienced considerable alterations due to increased agricultural and urban development. Most notable in this regard is a freshwater management system consisting of a network of canals, drainage ditches and water control structures. In addition to affecting freshwater supply to the estuary, the system has also increased the drainage area of the estuary, now estimated at 2,000 sq km. The freshwater drainage also transports fungi, toxic algae, and other pathogens, contaminants, sediment and organic matter into the estuary. The C-44 canal, one of the major drainage canals, conveys flood control discharges from Lake Okeechobee to the South Fork of the estuary. Controlled freshwater releases from the lake often contribute large deposits of "muck" in the estuary that are known to degrade habitats for submerged aquatic vegetation, oysters and fish. Muck, comprised predominantly of fine grained abiotic particles, may also have adsorbed organic matter and contaminants, and its accumulation is likely to promote hypoxic conditions within the estuary. Large quantities of freshwater release from Lake Okeechobee after heavy rainfall associated with the 1997-98 El Nino-Southern Oscillation (ENSO) event reportedly produced unusually high incidences of fish disease and mortality, promoted toxic algal blooms in the estuary, formed plumes of colored water and suspended particulate matter extending to nearshore Atlantic Ocean reefs, and reduced overall biodiversity of estuarine and freshwater fish communities (Sime, 2005).

Prompted by widespread concern about the water quality and biological resources of the estuary, the St. Lucie River Issue Team (SLRIT), established under the South Florida Ecosystem Restoration Working Group, identified and recommended funding of several research projects in 2000 to address water quality-related resource management issues in the estuary and its watershed. This study, performed under a Joint Project Agreement between NOAA and State of Florida, was designed to characterize the estuary in terms of chemical contamination and its associated adverse biological effects.

The purpose of the study was to characterize the St. Lucie Estuary in terms of environmental toxicity and to describe the extent and severity of habitat degradation using the sediment quality triad approach (Chapman, et al., 1987). The study design also encompassed aspects of key environmental issues that have been defined for the estuary, including possible impacts from agricultural runoff (i.e., those from citrus groves and row crops), urbanization, and probable relationships between prevalence of fish abnormalities and severity of contamination.

The St. Lucie Estuary (SLE) in southeast Florida discharges to the Atlantic Ocean through the St. Lucie Inlet after crossing the southern portion of the Indian River Lagoon (Figure 1-1). As noted, extensive urban and agricultural drainage projects in the SLE watershed have expanded the drainage basin, now estimated to cover an area exceeding 2,000 sq km (Haunert and Stewart, 1994). Canals in the watershed include C-23, C-24, part of the Central and South Florida Flood Control Project, and the St. Lucie Canal (C-44), which provides both navigation and the release of water from Lake Okeechobee.

These changes have altered the timing (excess wet season flows, insufficient dry season flows), distribution, quality, and volume of freshwater entering the estuary (Haunert et al., 1994). The St. Lucie Estuary is hydrologically connected to the Everglades; its study is also part of the larger ecosystem restoration and freshwater management efforts in South Florida, known as the Comprehensive Everglades Restoration Plan.

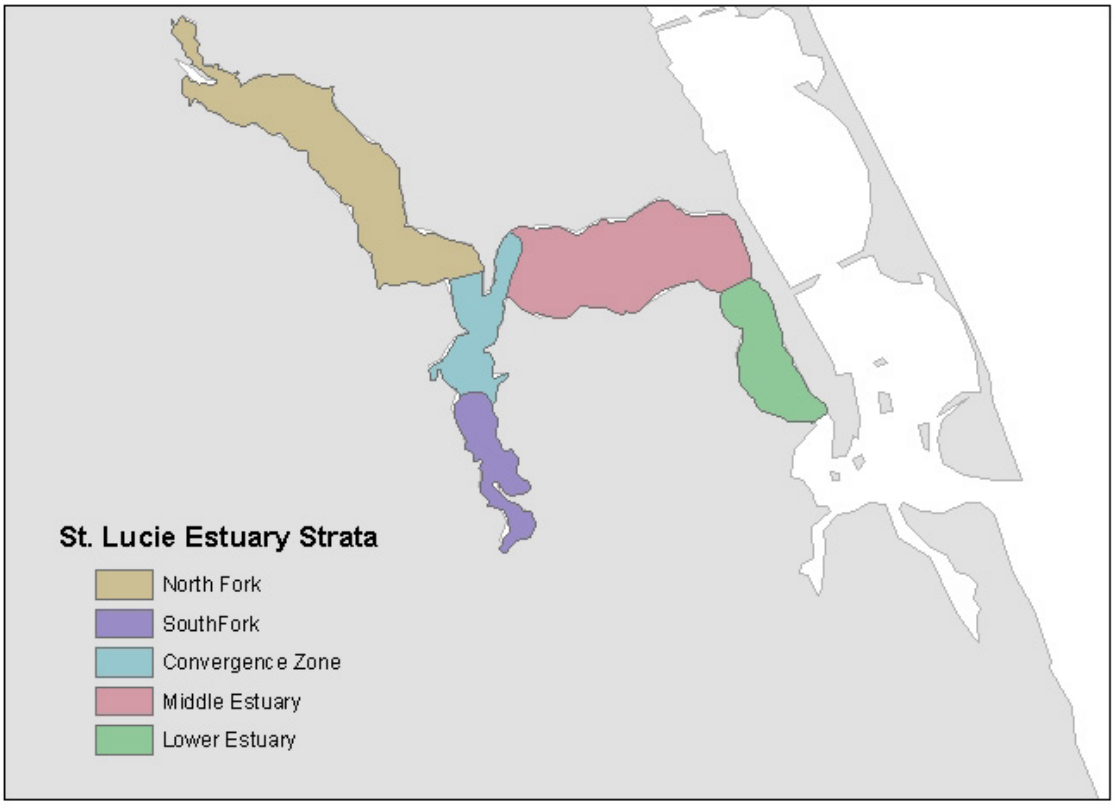

Figure 1-1. St Lucie Estuary strata. Stratum 1 (North Fork), stratum 2 (South Fork), stratum 3 (Convergence Zone), stratum 4 (Middle Estuary), and stratum 5 (Lower Estuary).

1.2 Approach

The study began in the Year 2001 and progressed through a series of phased activities. The foundation of this study was based on the sediment quality triad (SQT) approach that has previously been used by NOAA in sediment quality assessment studies nationwide (Long, 2000). Sampling sites were selected using a stratified random sampling design to allow for a geographically comprehensive characterization of the estuary in terms of chemical contamination in sediment, sediment toxicity, and benthic infauna community structure. Sediment toxicity tests were conducted exposing test organisms or a cell line to whole sediment, sediment porewater, and organic extract of sediment. Threshold Effects

Level (TEL) and Probable Effects Level (PEL) sediment quality guidelines were used to evaluate the observed levels of contaminants in terms of their toxicological significance. These guidelines were developed for Florida coastal waters (MacDonald et al., 1996).

Concurrent and subsequent investigations were conducted for further elucidation of cause-effect relationships between ambient levels of a contaminant of concern and their associated adverse biological effects. These studies included copper-spiked sediment bioassays using juvenile clams (Dr. Michael Fulton, NOAA) and a preliminary assessment of potential effects using a suite of biomarkers (Dr. Amy Ringwood, University of North Carolina at Charlotte).

In addition, a number of ancillary studies were conducted to apply or refine analytical techniques for use in the St. Lucie Estuary or to provide supporting data for the SLRIT-funded study on the incidence and prevalence of disease and deformities in fish (Dr. Joan Browder, NOAA). These ancillary studies are as follows:

- Acetyl-cholinesterase activity in fish and crustaceans, particularly in the vicinity of drainage canals (Dr. Peter Key, NOAA)

- DNA strand breakage or Comet assay on selected species of fish to evaluate severity of effects due to oxygen radicals, membrane damage, enzyme induction or inhibition, or exposure to harmful radiation (Dr. Scott Steinert, Computer Sciences Corporation)

- Preliminary study of vitellogenin expression in fish due to the presence of endocrine disrupting chemicals that are sometimes associated with effluents from wastewater treatment plants (Dr. Nancy Denslow, University of Florida)

Final reports or data from these studies are available on-line; it is anticipated that the authors will publish their findings in scientific journals or as symposium proceedings.

A hydrodynamic model for the St. Lucie Estuary has been developed, which can potentially account for the sources, transport and cycling of copper in the estuary and may be used to analyze copper-related water quality impacts of different source control scenarios. The model code, an accompanying report, and a guidance document to support acquisition of data for the model parameters have been provided to the South Florida Water Management District and the Florida Department of Environmental Protection (July 2006).

Finally, all of the data and reports produced under this study have been assembled and are being made accessible on line. The current website for accessing the data is: http://www8.nos.noaa.gov/cit/nsandt/download/bi_monitoring.aspx

2.0 Methods

2.1 Sampling sites

A stratified random sampling design was used to determine sampling sites. The study area was divided into five strata with six randomly selected sites in each stratum: North

Fork (stratum 1), South Fork (stratum 2), Convergence Zone (stratum 3), Middle Estuary (stratum 4) and Lower Estuary (stratum 5). In 2001, sediment samples were analyzed for contaminants and benthic infauna from each of the 30 sites (Table 2-1; Figure 2-1). Due to funding constraints, toxicity testing was done only at 15 of the 30 sites (3 sites per stratum). In addition to the randomly selected sites used for characterization of the estuary, additional pre-determined sites were sampled to provide chemistry and toxicity data in support of the fish disease and deformity study (Dr. Joan Browder, NOAA). The data from non-randomly selected sites were not used in the SQT assessment.

The toxicity tests included an amphipod survival assay (*Ampelisca abdita*), the sea urchin (*Arbacia punctulata*) fertilization success test using sediment porewater, and the Human Reporter Gene System (HRGS) P450 test using organic extract of sediment. In a repeat assessment of sediment toxicity to amphipods in 2003, two amphipod species were used (*Ampelisca abdita* and *Eohaustorius estuarius*). Samples were collected from the same 15 sites as in 2001.

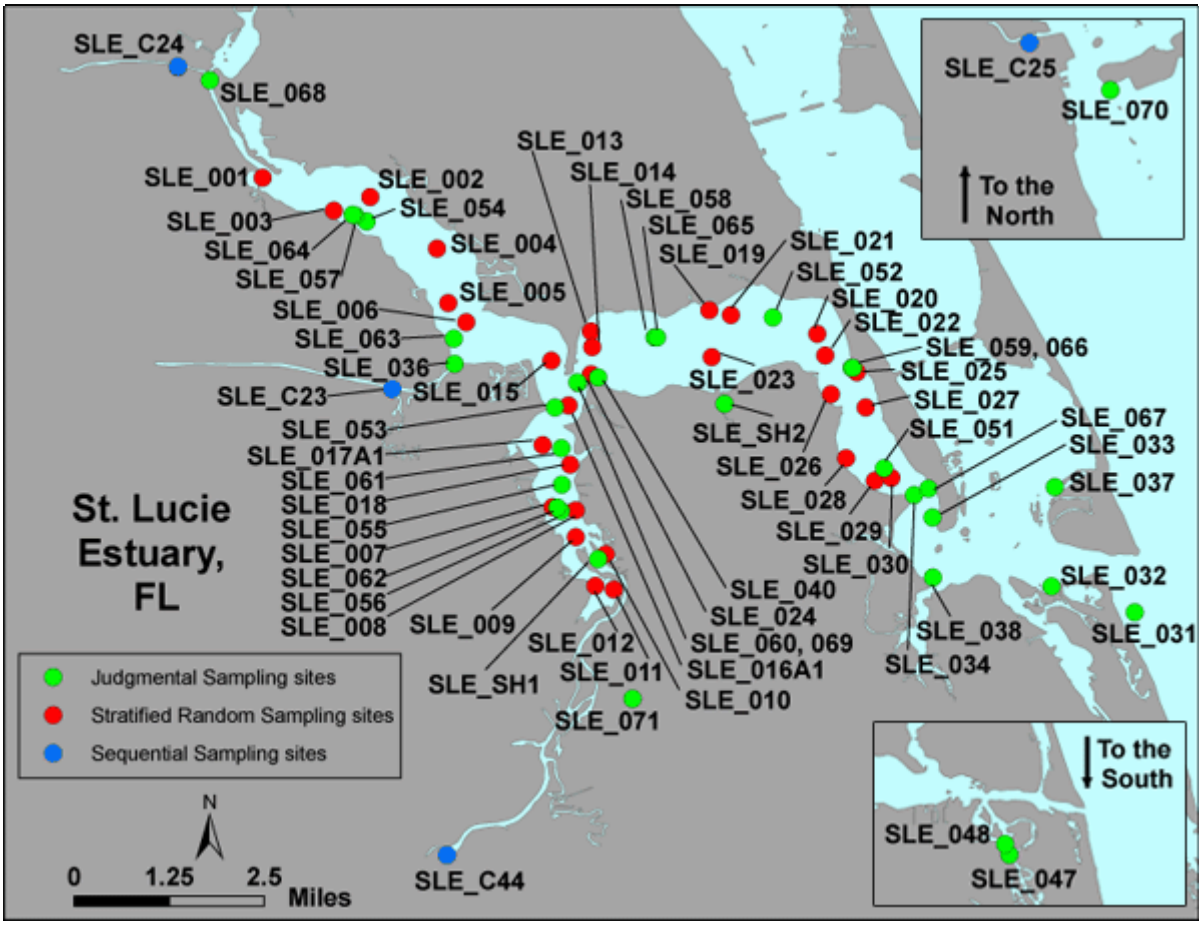

Figure 2-1. St. Lucie Estuary sampling sites.

Table 2-1. Sediment quality triad site locations in the St. Lucie Estuary, FL. Assays: amphipod survival (AMP), sea urchin fertilization (SU), and P450 reporter gene system (RGS).

Stratum	Random Sites	Assays	Latitude	Longitude	Location
1	1		27° 14' 26.8"	-80° 19' 12.0"	North Fork
1	2	Amp/SU/RGS	27° 14' 15.0"	-80° 17' 56.3"	North Fork
1	3	Amp/SU/RGS	27° 14' 4.6"	-80° 18' 21.9"	North Fork
1	4		27° 13' 38.4"	-80° 17' 10.2"	North Fork
1	5	Amp/SU/RGS	27° 13' 2.2"	-80° 17' 02.1"	North Fork
1	6		27° 12' 48.8"	-80° 16' 51.0"	North Fork
2	7	Amp/SU/RGS	27° 10' 42.0"	-80° 15' 52.2"	South Fork
2	8		27° 10' 39.8"	-80° 15' 36.5"	South Fork
2	9	Amp/SU/RGS	27° 10' 21.4"	-80° 15' 35.8"	South Fork
2	10		27° 09' 56.6"	-80° 15' 25.9"	South Fork
2	11		27° 09' 45.6"	-80° 15' 10.2"	South Fork
2	12	Amp/SU/RGS	27° 09' 48.2"	-80° 15' 23.1"	South Fork
3	13		27° 12' 41.7"	-80° 15' 28.3"	Convergence Zone
3	14	Amp/SU/RGS	27° 12' 32.0"	-80° 15' 24.5"	Convergence Zone
3	15	Amp/SU/RGS	27° 12' 21.9"	-80° 15' 55.3"	Convergence Zone
3	16		27° 11' 50.4"	-80° 15' 42.0"	Convergence Zone
3	17	Amp/SU/RGS	27° 11' 23.9"	-80° 15' 59.6"	Convergence Zone
3	18		27° 11' 11.9"	-80° 15' 40.1"	Convergence Zone
4	19	Amp/SU/RGS	27° 12' 57.7"	-80° 14' 06.5"	Middle Estuary
4	20		27° 12' 41.5"	-80° 12' 50.9"	Middle Estuary
4	21	Amp/SU/RGS	27° 12' 54.1"	-80° 13' 52.1"	Middle Estuary
4	22	Amp/SU/RGS	27° 12' 26.7"	-80° 12' 43.5"	Middle Estuary
4	23		27° 12' 25.9"	-80° 14' 02.3"	Middle Estuary
4	24		27° 12' 11.4"	-80° 15' 26.0"	Middle Estuary
5	25		27° 12' 17.2"	-80° 12' 23.5"	Lower Estuary
5	26	Amp/SU/RGS	27° 11' 59.6"	-80° 12' 42.6"	Lower Estuary
5	27	Amp/SU/RGS	27° 11' 49.6"	-80° 12' 16.70"	Lower Estuary
5	28		27° 11' 15.3"	-80° 12' 31.2"	Lower Estuary
5	29	Amp/SU/RGS	27° 11' 0.4"	-80° 12' 11.5"	Lower Estuary
5	30		27° 11' 2.4"	-80° 12' 00.4"	Lower Estuary

2.2 Field sampling

2.2.1 Chemistry and toxicity samples

Field samples were collected in May 2001 on board NOAA Ship *Ferrel,* and in shallow sites from a launch provided by the ship. Sediment samples were collected using a stainless steel Young-modified van Veen grab sampler. The sampler was deployed with a davit and winch. Only the upper 2 - 3 cm of sediment from the sediment water interface was collected for chemical analyses and toxicity bioassays. Sediments were removed from the sampler with an acetone rinsed plastic scoop and transferred to a lined, acetone rinsed HDPE (high density polyethylene) plastic bucket liner supported by a HDPE plastic bucket. A new scoop and bucket liner were used at each station. Sediments were homogenized using a Teflon® paddle prior to distribution into individual containers for chemical analyses and toxicity bioassays. The grain size and toxicity bioassay sediments were refrigerated ($4^{o}C$) and shipped to laboratories by overnight courier in cooler with blue ice. Contaminant chemistry and carbon samples were stored frozen and shipped by overnight courier in coolers with dry ice.

2.2.2 Benthic macroinfauna samples

A Young-modified Van Veen grab (area = 0.04 m^2) was used to collect bottom samples at each of the 30 stations during May, 2001. Three replicate samples were collected at 15 of the 30 sites, and single bottom water samples were collected at the remaining 15 sites. Macroinfaunal samples were sieved through a 0.5 mm mesh screen, carefully transferred to HDPE containers, and preserved with 10% formalin in the field. At the end of the 7-day field sampling, the macroinfaunal samples were shipped by overnight courier to the contractor laboratory in Mobile, Alabama. In 2003, six sites were resampled in the North and South Forks and triplicate analyses were performed.

2.3 Sediment grain size analyses

Grain size parameters that were computed included percent gravel, sand, and silt /clay at half-phi intervals using the hydrometer technique for fractions smaller than 44 mm and nested sieves for larger particle fractions. Total organic carbon (TOC) content was measured as ash-free dry weight expressed as a percentage.

2.4 Sediment contaminant chemistry analyses

The sample processing protocols and analytical methods are described in detail in Lauenstein and Cantillo (1993, 1998). The documents are available from NOAA's National Status and Trends web site (http://NSandT.noaa.gov) and the National Environmental Methods Index (http://www.nemi.gov). Metals and organic contaminants data from 2001 and 2003 are also available online (http://NSandT.noaa.gov)

2.5 Sediment toxicity assays

2.5.1 Amphipod survival

This test is commonly used in North America for assessing sediment quality, in part because the test integrates the effects of complex contaminant mixtures in relatively unaltered sediment and also because amphipods are fairly common in coastal water and in many areas they are ecologically significant. *Ampelisca abdita* is the most commonly used species in NOAA's studies. This euryhaline species occurs in fine sediments from the intertidal zone to a depth of 60 m, with a wide geographical range.

All toxicity tests were conducted in accordance with a standard guide for conducting 10-day static sediment toxicity tests with amphipods (ASTM 1993 and 2000) with additional guidance developed for testing four different species. The negative control tests were conducted with sediment collected from the Pettaquamscutt Cove, Narrow River, RI. A positive control test, using sodium dodecyl sulfate as a reference toxicant test, was used to determine the sensitivity of each batch of test organisms.

Based on statistical analyses of sediment toxicity data on amphipods, including power analysis, two criteria are used to declare a sediment sample to be toxic: first, the t-test must show that the sample survival is statistically lower than in the control, and second, the sample's mean survival must also be equal to or less than 20% of that in the control (Thursby et al., 1997)

The test was repeated with samples collected in 2003 using two amphipod species to determine their relative sensitivities to sediment-borne contaminants. The re-testing was necessitated by inadequate performance in negative controls, i.e., poor amphipod survival rate in "home" sediment.

As is required in all such testing, water properties, for example, level of dissolved oxygen, temperature, pH, total ammonia, were monitored in all test chambers.

2.5.2 Juvenile seed clam (*Mercenaria mercenaria*) test (whole sediment)

Field collected sediment samples were warmed to room temperature and press-sieved through a 212-micron mesh screen. Tests were run in pre-cleaned 16-oz glass beakers containing 60 mL of sediment and 180 mL of filtered seawater. Five replicates were used for each sample. After settling of the sediment, 30 juvenile clams were added to each beaker. Tests were run at a salinity of 30 parts per thousand and temperature of 20 °C in a chamber with a 12 hour light-dark cycle. Clams in the test beakers were fed 5 mL of *Isochrysis galbana* every 48 hours. At the end of 10 days, clams were retrieved by re-sieving the sediment through a 212-micron mesh screen, and clam mortality noted in each beaker. Site-specific mortality was evaluated in comparison to the Folly River, South Carolina reference sediment, using ANOVA and Dunnett's tests on arcsin transformed data. Additional details of the testing procedures and data analyses are provided by Ringwood and Keppler (1998) and Fulton, et al. (1999).

2.5.3 Sea urchin fertilization test

This test has been used extensively in assessments of ambient water quality, toxicity of industrial and municipal effluents, and sediment toxicity in coastal waters. It combines the features of testing sediment porewater (the phase of sediments in which dissolved toxicants may be bioavailable) and exposures to gametes or early life stages of invertebrates, which often are more sensitive than the adult forms.

Porewater testing to determine fertilization success using gametes of the sea urchin *Arbacia punctulata* followed standard procedures (USGS SOP F10.6, USGS, 1995). Porewater was extracted from sediment samples by using a pressurized pneumatic device. The extractor is made of polyvinyl chloride (PVC) and uses a 5 µm polyester filter. After extraction the sample is centrifuged, and the supernatant collected and stored frozen. Prior to commencing the experiment, samples are thawed in a water bath, and the water quality measurements are made (dissolved oxygen, pH, sulfide, and ammonia). Each porewater sample is tested in a dilution series (100%, 50% and 25%) with five replicates per treatment. A reference porewater sample collected from Redfish Bay, Texas is included with the batch of samples as a negative control.

This test, sometimes also knows as the sperm cell toxicity test, involves exposing sea urchin sperm to porewater followed by the addition of eggs. At the test's conclusion, the fraction of fertilized eggs (eggs showing fertilization membrane) is recorded. Sodium dodecyl sulfate is used as a positive control. Reduction in mean fertilization success after exposure to porewater, in comparison with the negative control, is the experimental end-point. The resulting data are analyzed to calculate EC_{50} (concentration that is effective in causing a 50% response in test organism). In addition to statistically significant differences with control sediment, a detectable significance criterion is used to determine the 95% confidence value based on power analysis of data from similar tests nationwide (Carr and Biedenbach, 1999).

2.5.4 Human Reporter Gene System (HRGS) P450 response

This test is used to determine the presence of certain toxic chemicals that are known to induce the cytochrome P450 enzyme system, including certain polycyclic aromatic hydrocarbons, polychlorinated biphenyls, dioxins and furans. The P450 Human Reporter Gene System (HRGS) utilizes a human hepatoma cell line (101L) stably transfected with a plasmid containing the firefly luciferase gene. When the cells are exposed to CYP1A1-inducing compound, luciferase is produced, and can be easily measured with a luminometer. The HRGS fold induction response gives an assessment of the total amount of CYP1A1 inducing compounds in an extract of an environmental sample. This method has been used nationwide in support of sediment toxicity assessments sponsored by NOAA (Anderson, et al., 2005a).

The test methodology has been described in detail in a number of publications (APHA 1998, ASTM 1999, EPA 1999). Benzo[a]pyrene Equivalents (B[a]PEq) were calculated

for all samples. The B[a]PEq is a measure of the CYP1A1-inducing PAHs, plus any coplanar PCBs, dioxins or furans that may be present in the sample, and is calculated as follows:

B[a]PEq (μg/g) =

((fold induction / 60) * (volume factor/ dry weight))* (dilution factor)

Fold induction is calculated as mean relative light units (RLU) produced by the sample divided by mean RLU produced by the solvent blank. The factor of 60 represents the approximate fold induction produced by 1.0 μg of B[a]P/mL. If a dilution is used, the B[a]PEq value is multiplied by the dilution factor.

The extracts were lighter in color than those tested in most estuary studies thus far. The lighter color could be the result of larger particle size, lower organic matter, or lower concentrations of PAHs. Based on experience, it was determined that 10 μL, instead of the 2 μL used previously, would be applied from each extract to the test system. Test samples that produced a coefficient of variation greater than 20% were re-tested, along with SRM samples that required dilution.

2.6 Macroinfaunal sample analysis

In the laboratory of Barry A. Vittor & Associates, Inc., benthic samples were inventoried, rinsed gently through a 0.5 mm mesh sieve to remove preservatives and sediment, stained with Rose Bengal, and stored in 70% isopropanol solution until processing. Sample material (sediment, detritus, and organisms) was placed in white enamel trays for sorting under Wild M-5A dissecting microscopes. All macroinvertebrates were carefully removed with forceps and placed in labeled glass vials containing 70% isopropanol. Each vial represented a major taxonomic group (e.g. Polychaeta, Mollusca, and Arthropoda). All sorted macroinvertebrates were identified to the lowest practical identification level (LPIL), which in most cases was to species level unless the specimen was a juvenile, damaged, or otherwise unidentifiable. The number of individuals of each taxon, excluding fragments, was recorded. A voucher collection was prepared, composed of representative individuals of each species not previously encountered in other St. Lucie samples.

Several numerical indices were chosen for analysis and interpretation of the macroinfaunal data. Infaunal abundance is reported as the total number of individuals per station and the total number of individuals per square meter (= density). Taxa richness is reported as the mean number of taxa represented in a given site location. Taxa diversity, which is often related to the ecological stability and environmental "quality" of the benthos, was estimated by the Shannon-Weaver Index (Pielou, 1966), according to the following formula:

$$H' = -\sum_{i=1}^{S} p_i (\ln p_i)$$

Where, S = the number of taxa in the sample, i = the i'th taxa in the sample, and p_i = the number of individuals of the i'th taxa divided by the total number of animals in the sample.

Taxa diversity can be calculated using ln or $\log_2$ and both methods are common in the scientific literature. The taxa diversity calculated in this report using ln, can be converted to $\log_2$ diversity by multiplying the ln taxa diversity by 1.4427. Taxa diversity within a given community is dependent upon the number of taxa present (taxa richness) and the distribution of all individuals among those taxa (equitability or evenness). In order to quantify and compare the equitability in the fauna to the taxa diversity for a given area, Pielou's Index J' (Pielou, 1966) was calculated as J' = H'/lnS, where *ln*S = H'max, or the maximum possible diversity, when all taxa are represented by the same number of individuals; thus, J' = H' /H' max.

3.0 Results

All data from chemical, toxicological and biological analyses of samples from this study are accessible on-line. Only summaries of data are presented in this report.

3.1 Sediment Chemistry

3.1.1 Metals

Sediment samples from thirty stratified random sites in the SLE were analyzed for nine metals (Table 3-1). All of the metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver and zinc) have threshold effects level (TEL) and probable effects level (PEL) sediment quality guidelines (MacDonald et. al., 1996). Overall, 36% of all metal concentrations exceeded the TEL and 1% exceeded the PEL Sediment quality guideline. Exceedences were highly variable for TEL (10% arsenic, 0% cadmium, 67% chromium, 63% copper, 7% lead, 16% mercury, 73% nickel, 3 % silver, 23% Zinc) and the PEL (3% copper, 7% mercury).

Table 3-1. Graphic depiction of TEL and PEL sediment quality guidelines and NS&T Mussel Watch Project sediment concentration quartiles for thirty stratified random sediment sites in the St. Lucie Estuary. Concentrations greater than the TEL and less than the PEL (T). Concentrations greater than the PEL (P). NS&T quartiles are represented by ∀ Quartile 1 ∀ Quartile 2 ∀ Quartile 3 ∀ Quartile 4.

Stratified Random Sampling

Stratum	NS & T site	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Silver	Zinc
1	1									
1	2	T		T	T	T		T		T
1	3			T	P		P	T		T
1	4			T	T		T	T		T
1	5			T	T	T	T	T	T	T
1	6			T	T			T		T
2	7			T	T			T		
2	8									
2	9									
2	10				T					
2	11			T	T			T		T
2	12			T	T			T		T
3	13			T	T			T		
3	14				T			T		
3	15			T	T			T		
3	16			T	T			T		
3	17			T	T		T	T		
3	18			T	T			T		
4	19	T		T	T		P	T		
4	20			T				T		
4	21			T	T			T		
4	22							T		
4	23			T	T			T		
4	24									
5	25							T		
5	26	T		T	T			T		
5	27									
5	28									
5	29			T				T		
5	30			T						

Strata comparisons using the rank sums Wilcoxon tests found significant differences between strata for copper, lead, silver and zinc (Figure 3-2; Table 3-3). Overall, metal concentrations were highest in the North Fork (Stratum 1).

Table 3-2. Results for Sums Wilcoxon statistical test. Asterisks (*) denote where a significant difference was found using the Kruskal/Wallis multiple comparison test.

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Metals	ChiSquare	Prob	Stratum 1	Stratum 2	Stratum 3	Stratum 4	Stratum 5
Arsenic	6.71	0.15	A	A	A	A	A
Cadmium	9.16	0.06	A	A	A	A	A
Chromium	8.33	0.08	A	A	A	A	A
Copper	12.12	0.02*	A	AB	AB	B	B
Mercury	5.00	0.29	A	A	A	A	A
Nickel	3.30	0.51	A	A	A	A	A
Lead	11.22	0.02*	A	AB	AB	AB	B
Silver	11.94	0.01*	A	AB	AB	AB	B
Zinc	10.16	0.04*	A	AB	AB	AB	B

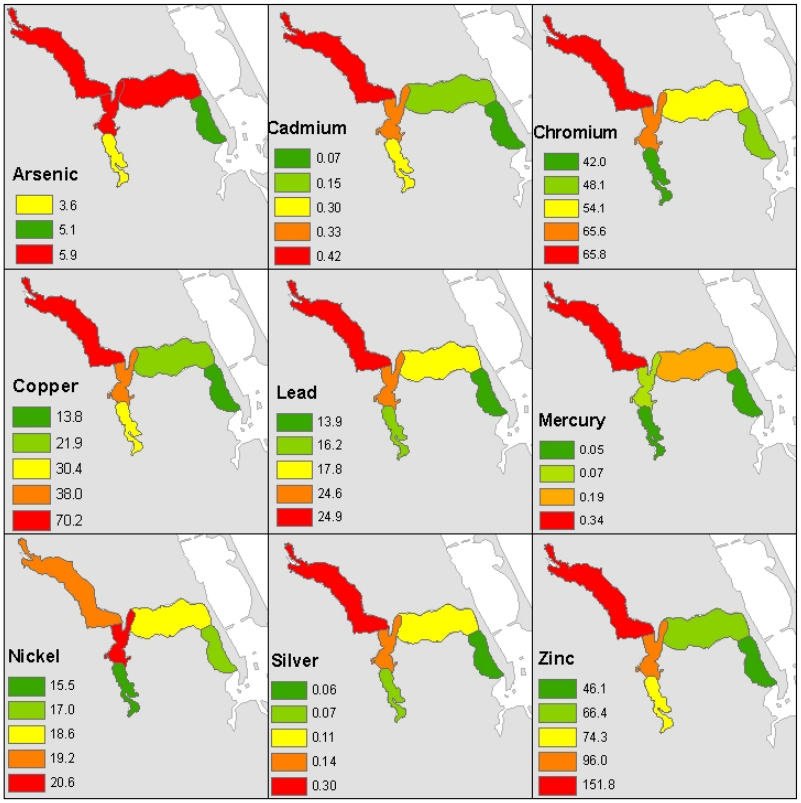

Figure 3-1. Mean metal concentrations (µg/g dry wt) by strata for stratified random sites in the St. Lucie Estuary. The North Fork and Convergence Zone strata consistently showed high metal concentrations.

3.1.2 Organic contaminants

Aggregated totals of chemical concentrations by chemical class in sediment were compared to NS&T Mussel Watch percentiles. St. Lucie sediments exhibited moderate to high levels of contamination relative to National concentrations (NS&T Mussel Watch sediment data 1996-97). Twenty seven percent and 79 percent of the St. Lucie sediment concentrations exceeded, respectively, the 75th and 50th percentiles of National concentrations (NS&T Mussel Watch 1996-1997). The observed levels of contaminants may also be compared with the ERM/ERL guidelines (Long et al., 1995), the TEL/PEL guidelines for Florida coastal waters (MacDonald et al., 1996), and the on-line database of NOAA's National Status and Trends Program (http://nsandt.noaa.gov).

Table 3-3. Graphic depiction of NS&T Mussel Watch Project sediment concentration quartiles for thirty stratified random sediment sites in the St. Lucie Estuary. NS&T quartiles are represented by ∀ Quartile 1 ∀ Quartile 2 ∀ Quartile 3 ∀ Quartile 4.

Stratum	Site	Σ Butyl Tin	Σ Chlordane	Σ DDT	Σ Dieldrin	Σ HCH	Hexachlorobenzene	Mirex	Σ PAH	Σ PCB
1	1									
1	2									
1	3									
1	4									
1	5									
1	6									
2	7									
2	8									
2	9									
2	10									
2	11									
2	12									
3	13									
3	14									
3	15									
3	16									
3	17									
3	18									
4	19									
4	20									
4	21									
4	22									
4	23									
4	24									
5	25									
5	26									
5	27									
5	28									
5	29									
5	30									

The spatial distribution of butyl tins, dieldrins and HCHs exhibited no significant difference between strata. Stratum 3 was significantly higher or amongst the highest of all contaminants, while stratum 5 was significantly lower or amongst the lowest of all the strata. (Table 3-4; Figure 3-2). For contaminants with significant differences found between strata, the convergence zone (stratum 3) had the highest mean concentrations.

Table 3-4. Results for Sums Wilcoxon statistical test. Asterisks (*) denote where a significant difference was found using the Kruskal/Wallis multiple comparison test.

Total Organic Contaminants	ChiSquare	Prob	Stratum 1	Stratum 2	Stratum 3	Stratum 4	Stratum 5
Total Butyl Tins	2.38	0.67	A	A	A	A	A
Total Chlordanes	10.35	0.03*	AB	AB	A	AB	B
Total DDTs	12.8	0.01*	AB	AB	A	AB	B
Total Dieldrins	7.96	0.93	A	A	A	A	A
Total HCHs	6.06	0.19	A	A	A	A	A
Hexachlorobenzene	5.7	0.22	A	A	A	A	A
Mirex	13.56	0.01*	AB	ABC	A	BC	C
Total PAH	11.14	0.02*	B	AB	A	AB	B
Total PCB	9.89	0.04*	B	AB	A	AB	AB

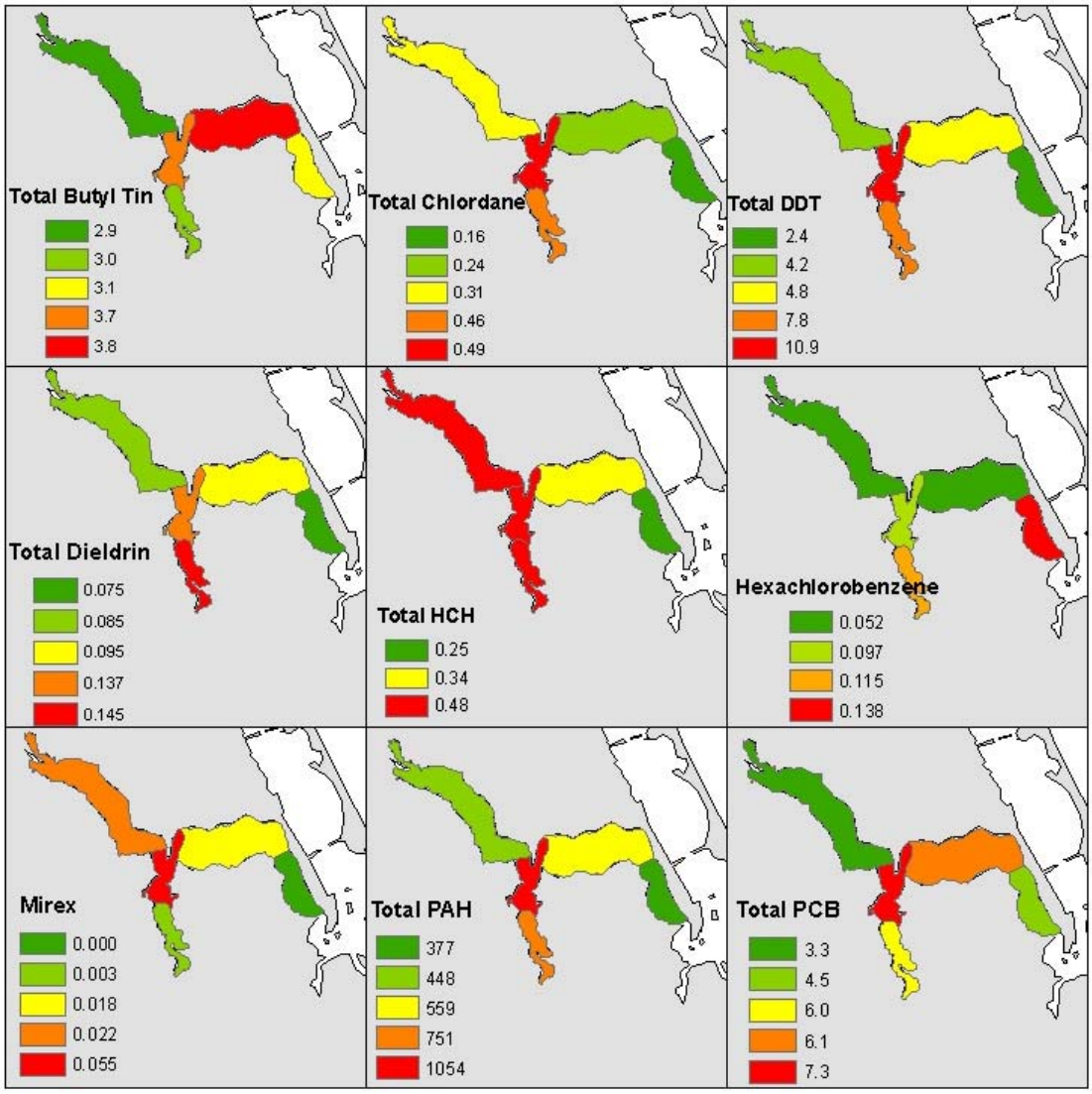

Figure 3-2. Mean concentrations (ng/g dry wt except ng Sn/g dry wt. for total butyl tin) by chemical class and strata for stratified random sampling sites in the St. Lucie Estuary.

3.2 Toxicity Tests

3.2.1 Amphipod survival (whole sediment test)

Amphipod tests were performed on sediment samples from a subset of 15 out of the 30 randomly selected sites in five strata. Results from the tests conducted in 2001 were inconclusive due to inadequate performance in the negative control tests (unacceptable levels of mortality, 5 to 22 percent, in animals exposed to "home sediment"). Those results did not meet acceptance criteria for the test.

Relatively high variability in the survival of *Ampelisca abdita* in negative control sediments is not unusual. It may be due to stress caused on the animals during their removal from tubes for counting or failure of the animals to quickly construct tubes in the experimental chambers. Another possible stress factor in the batch of animals used in this study could have been the much higher than normal ambient temperature at the collection site. Also, this species is considered to be relatively hardy to sediment contamination due to its tube-living and suspension feeding nature; in essence, the animal can isolate itself from the sediment and, in turn, from sediment-bound contaminants. The burrowing amphipod *Euhaustorius estuarius*, also recommended for use in sediment toxicity assessment studies, has shown a more sensitive response in comparison with *A. abdita* and shows tests results that are often more discerning of the range of sediment contamination. Typically, *E. estuarius* survival in negative control sediment is more uniform, ranging between 96 and 100 percent.

To re-test the sediment for amphipod toxicity and determine the relative response of these two species, parallel testing was performed in Year 2003 using 15 sediment samples, three sites from each of the five designated strata in the estuary.

The test negative control acceptance criteria for survival were met in all tests (≥ 90 % mean survival, not < 80 % survival in any one replicate). All reference toxicant tests were within the laboratory control chart limits. Mean *Eohaustorius* survival in the fine reference sediment (*Ampelisca* control sediment) was 80.0% and 86.0% in the first and second test, respectively. These values were significantly different from mean *Eohaustorius* control survival, but within the ≤ 20% absolute difference from the control recommended by some regulatory programs (USACE, 2000).

In the case of *A. abdita*, sediments from only four sites showed significantly lower survival than in the control sediment (SLE3, SLE5, SLE14, and SLE22). Mean percent survival ranged from 75% to 85%. Only one test sediment, SLE9, had mean percent survival of 80% but was not significantly less than the control survival. Replicate percent survival in sediment SLE9 was 100%, 95%, 10%, 100%, and 95%. At test termination, the replicate that had only 10% survival was carefully examined, and dark bright blue paint chips were discovered, along with an unusual odor. In the case of *E. estuarius*, sediments from seven sites (SLE5, SLE7, SLE9, SLE12, SLE15, SLE22, and SLE26) showed significantly lower survival than that in control sediment.

In terms of spatial distribution and concordance, the results from the two species were comparable, even though *E. estuarius* showed statistically significant mortality in 7 samples and *A. abdita* in 4 samples. However, in neither case, the mean survival in test sediments was equal to or less than 80 percent of the mean performance control. This criterion is based on statistical analysis of a large amount of *A. abdita* data from a variety of sources nationwide, which showed that the beta error associated with 20 percent difference from the control is less than 10 percent, i.e., power of the test is greater than 90 percent (Thursby et al., 1997).

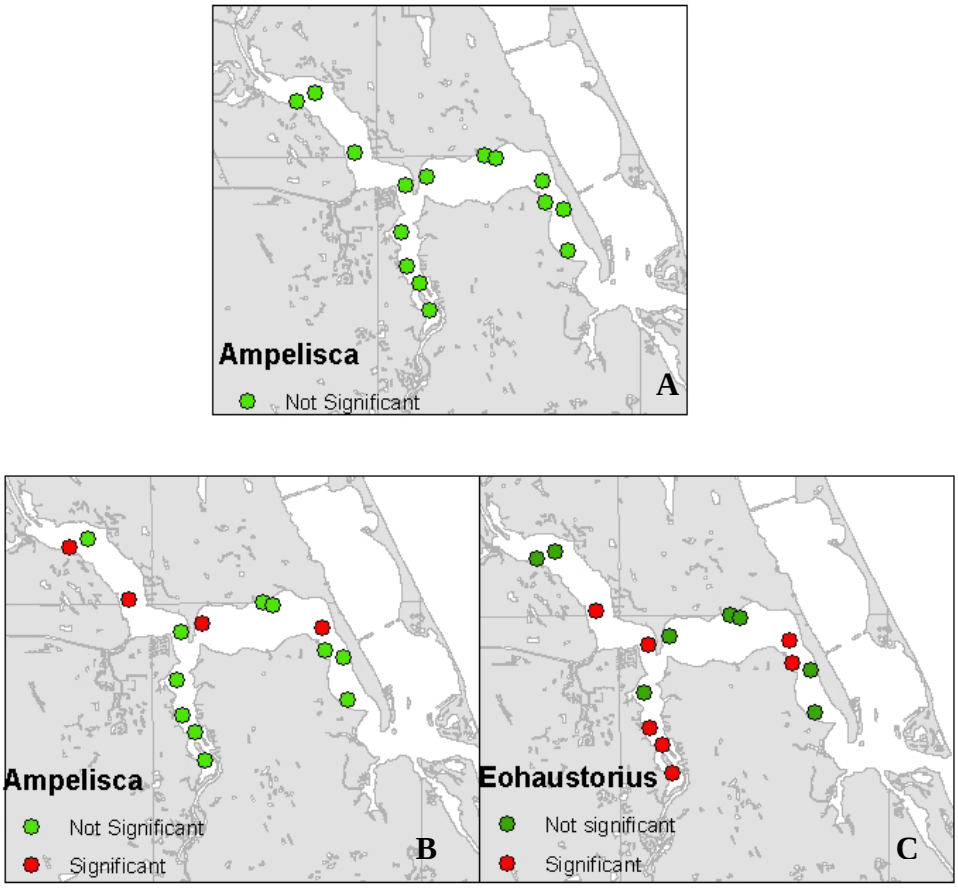

Figure 3-3. Results from 2001 (A) and 2003 (B & C) amphipod sediment toxicity tests depict significance relative to control sediment.

3.2.2 Juvenile clam survival

This test uses a more sensitive life stage of a widely distributed clam species in the southeastern United States. As such the test has been among the battery of toxicity tests that has been used in sediment quality assessment in coastal states, e.g., South Carolina,

and as part of the federal Environmental Monitoring and Assessment Program. As noted earlier, the test involves the use of 30 juvenile clams (*Mercenaria mercenaria*) in each test chamber and counting the surviving animals after 10 days of incubation in sediment. Test results showed a wide range of mortality following exposure to the St. Lucie Estuary sediments.

Thirteen out of 15 sampling sites within the estuary showed statistically significant mortality when compared to the Folly River reference sediment (Figure 3-4). For samples showing lower survival than in the controls, percent mortality ranged from 6 percent (Stations 2 and 16) to 39 percent (Station 3). It should be noted that statistical criteria for declaring a sample toxic under this test are not as rigorous as in the case the amphipod test.

Testing of sediment collected from six sites in the Year 2003 showed a similar pattern. Four of the six sites showed significant mortality. Sites 7 and 17, which showed low mortality in 2001, did not show significant mortality in 2003.

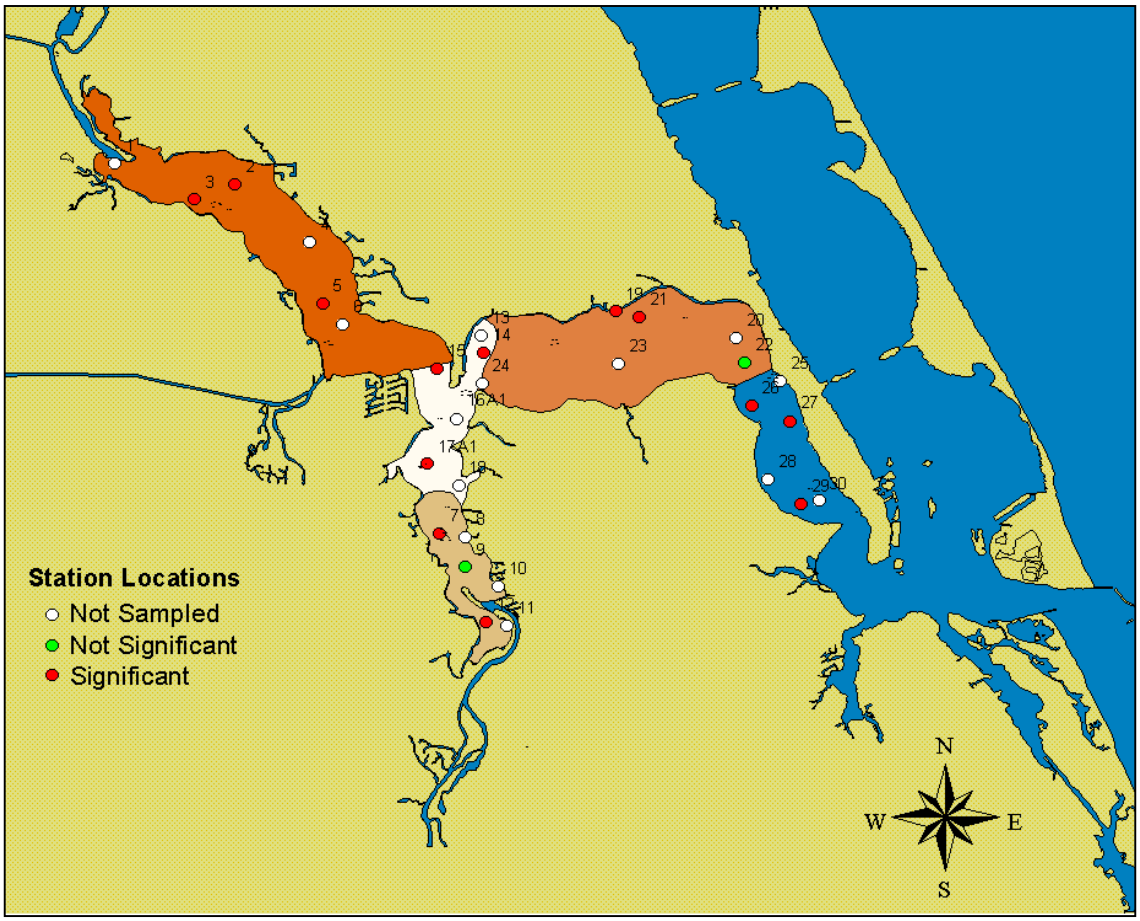

Figure 3-4 Results of sediment toxicity testing in 2001 using juvenile clam *Mercenaria mercenaria*

3.2.3 Sea urchin fertilization (sediment porewater)

Salinity of porewater samples from the 15 randomly selected sampling sites varied from 22 to 37 ppt (USGS/CERC, 2001). Initial dissolved oxygen in all samples exceeded the 80% saturation and did not require aeration. In all but two samples, sulfide concentration was below the detection limit (0.01 mg/L); at SLE-7 it was 0.01 mg/L and at SLE-19, it was 0.03 mg/L. Total ammonia ranged from 1 to 12 mg/L with the unionized ammonia levels from 31 to 308 µg/L. The generally accepted "no observed effect concentration" (NOEC) level for unionized ammonia for sea urchin fertilization tests is 400 µg/L. However, the embryological end-point for the sea urchin test (not used in the St. Lucie Estuary study) is much more sensitive to the presence of unionized ammonia, with an NOEC value of 30 µg/L (Carr et al., 2006).

Toxicity testing of the samples indicated that 8 out of the 15 sites (53%) were toxic in the undiluted porewater. All of the sites in the North Fork and South Fork showed significant toxicity and met the detectable significance criteria. Sites in these two strata were toxic at both the 50% and 25% dilution levels. Only one site each in strata 3 and 4 showed toxic response; no toxic response was noted in stratum 5. The undiluted porewater samples from Station SLE-19, located in the Middle Estuary stratum, showed exceptionally high toxicity; fertilization success in five replicates varied between zero and 1 percent. Similar responses were noted at 50% and 25% dilution of porewater, 3 percent and 7 percent, respectively (Figure 3-5).

As is often the case with results from this test, atypical toxicity was noted at four sites, i.e., samples were found to be more toxic in diluted porewater. For example, at site SLE-22 the mean fertilization was 95 percent at 100 percent porewater, but it was only 55 percent at the 25 percent of water quality adjusted porewater sample. Dilution water blanks run with the test showed no evidence of toxicity associated with the dilution water itself. Other factors, such as a change in pH, may have caused such atypical responses, but their potential effects were not investigated. For this report, toxic responses in diluted samples were not considered when the undiluted sample was not toxic.

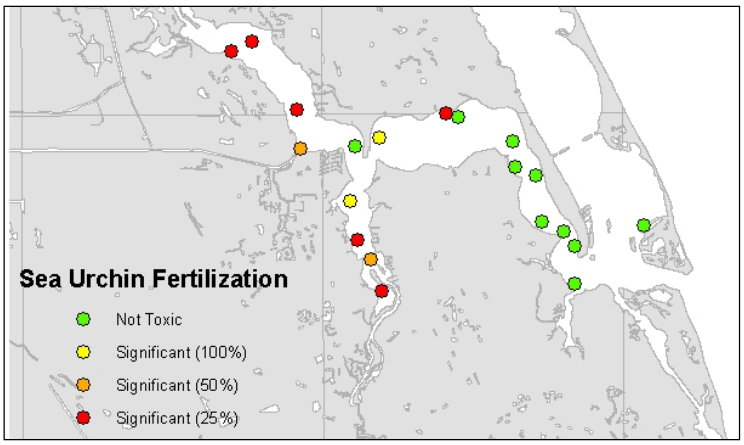

Figure 3-5. Sea urchin fertilization toxicity test.

3.2.4 P450 Human Reporter Gene System

The HRGS test is used to infer the presence of organic compounds that bind to the Ah (aryl hydrocarbon) receptor and induce the CYP1A1 enzyme locus on the vertebrate chromosome. This particular test utilizes a human hepatoma cell line stably transfected with a plasmid containing the firefly luciferase gene downstream of human CYP1A1 promoter sequences. Under appropriate test conditions, when these cells are exposed to CYP1A1-inducing compounds, luciferase is produced and can be measured with a luminometer. Chemical contaminants that are known to induce the CYP1A1 enzyme include certain polycyclic aromatic hydrocarbons, polychlorinated biphenyls, dioxins and furans. Differences in the ability of the P450 enzyme system to metabolize chlorinated and non-chlorinated compounds allow for differentiation between these classes of compounds in environmental samples.

The HRGS response from exposure to St. Lucie sediment extract was generally very low, with a mean value of 8.4 µg B[a]P Eq/g for the 15 randomly selected sites, and an upper confidence limit of 11.8 µg B[a]P Eq/g (Figure 3-6). Although there is no critical threshold value to denote toxicity of sediment from this test, statistical analyses of NOAA's nationwide HRGS data (1309 samples) from 19 coastal regions suggested that induction values at or below 11 µg B[a]P Eq/g represent background concentrations in coastal bays and estuaries; at 32 µg B[a]P Eq/g there is the potential for toxicological impacts on biota (Anderson et al., 2005). At 60 µg B[a]P Eq/g or above, degradation of benthic faunal communities has been observed (Fairey et al., 1996). The HRGS values in St. Lucie Estuary are similar to those observed in Biscayne Bay in 1996 (mean value of 8.2 µg B[a]P Eq/g) but much lower than in estuaries known to be heavily polluted with polycyclic aromatic hydrocarbons, polychlorinated biphenyls, dioxins and furans. In those estuaries HRGS induction values have often exceeded 100 µg B[a]P Eq/g and occasionally exceeding 1,000 µg B[a]P Eq/g (Anderson et al., 2005, Hartwell et al., 2001, Hartwell and Hameedi, 2006).

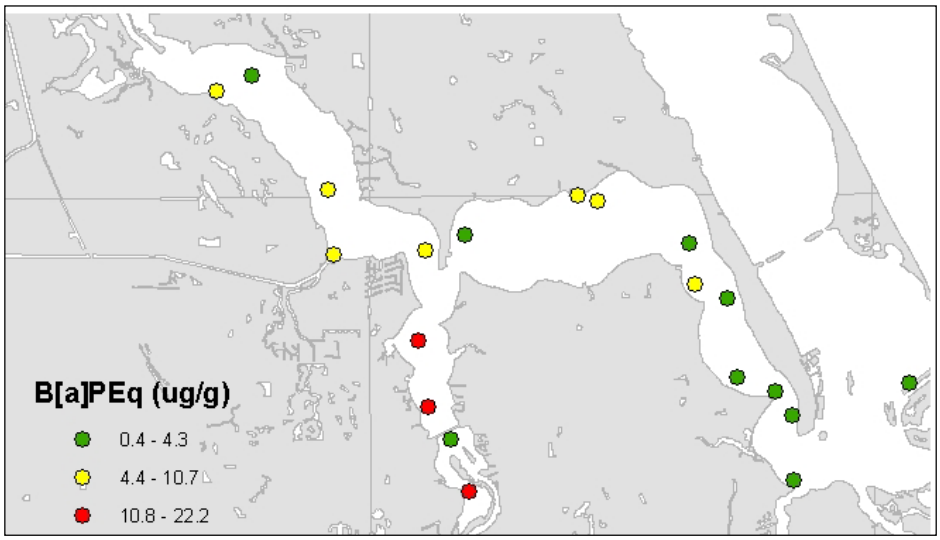

Figure 3-6. P450 HRGS reported in benzo[a]pyrene equivalent units. Higher values were found in the South Fork and convergence zone where the highest PAH concentrations exist.

3.3 Benthic characterization

3.3.1 Habitat characterization

Sediment data for the 30 stratified random sampling sites are given in Figure 3-7. Sediment composition varied considerably and ranged from 98% sand at Station 24 to 87% silt+ clay at Station 18 (Figure 3-7). Muck sediments defined by clay/silt > 60%, TOC > 6%, and water content > 75 % were found in the North Fork, Convergence Zone and South Forks (Figure 3-7). Salinities varied from 17 ppt at Station 1 to 38.1 ppt at Station 28 (Figure 3-8).

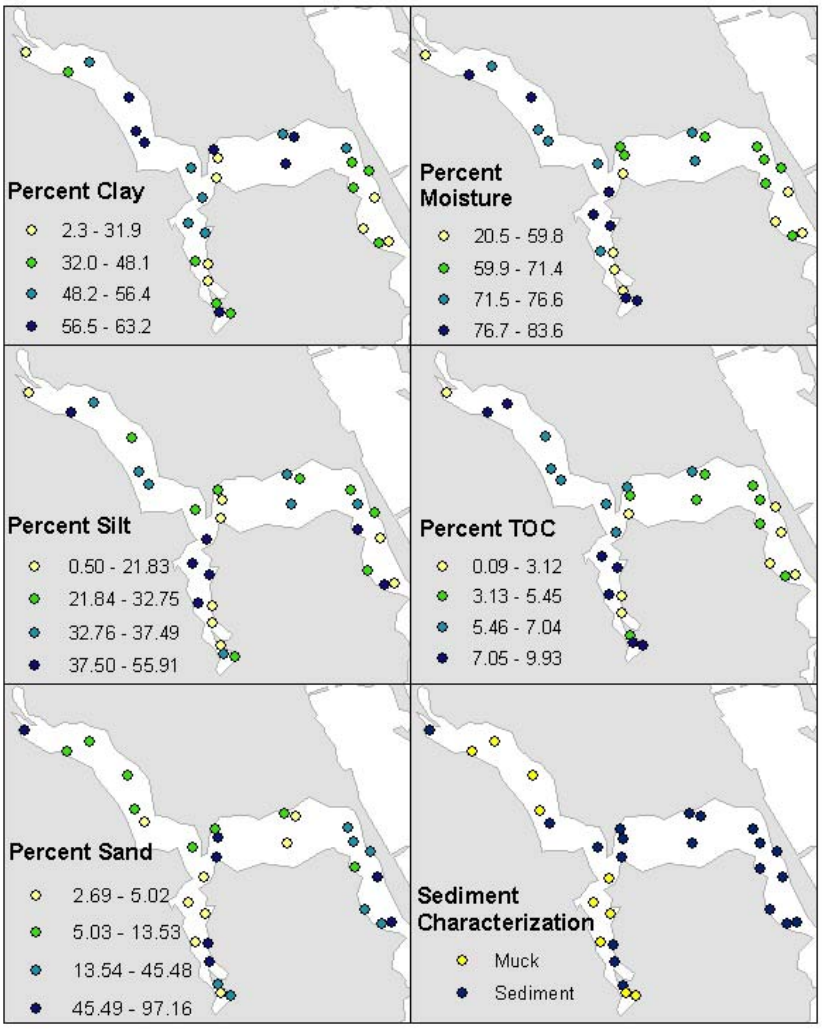

Figure 3-7. Sediment characteristics for the thirty stratified random sites.

3.3.2 Benthic community characterization

A total of 7,429 organisms, representing 148 taxa, were identified from the 30 St. Lucie sites. Polychaetes were the most numerous organisms present representing 48.9% of the total assemblage, followed in abundance by bivalves (27.2%) and malacostracans (17.8%). Polychaetes represented 41.9% of the total number of taxa followed by malacostracans (25.0%) and bivalves (17.6%). The number of taxa per station ranged from 3 to 54. The number of organisms per station ranged from 4 to 1,224.

The dominant taxa collected were the polychaete, *Sternaspis scutata*, the bivalve, *Mulinia lateralis*, the malacostracans, *Ampelisca abdita*, and the polychaete, *Mediomastus* (LPIL) representing 23.4%, 21.7%, 13.7%, and 10.8% % of the total individuals collected. The polychaete, *Glycinde solitaria*, was the most widely distributed taxon being found at 83% of the stations.

Taxa richness varied and ranged from 2.0 (± 1.0) to 53. Taxa richness was positively correlated with the percent gravel + sand in the sediments (Spearman's Rho = 0.748, P > Rho = < 0.0001) and inversely correlated with the percent silt + clay (Spearman's Rho = -0.746, P > Rho = <0.0001) and salinity (Spearman's Rho = -0.382, P > Rho = 0.0449). Station mean densities exhibited considerable variation ranging from 66.7 organisms/m^2 (± 52.0) to 12,275 organisms/m^2 (Figure 3-9). Densities were also positively correlated with the percent gravel + sand in the sediments (Spearman's Rho = 0.466, P > Rho = 0.0095) and inversely correlated with the percent silt + clay (Spearman's Rho = -0.462, P > Rho = 0.0102) and salinity (Spearman's Rho = -0.380, P > Rho = 0.0458).

Taxa diversity (H') ranged from 0.64 to 2.98 (Figure 3-9). Taxa evenness (J') ranged from 0.33 to 1.00.

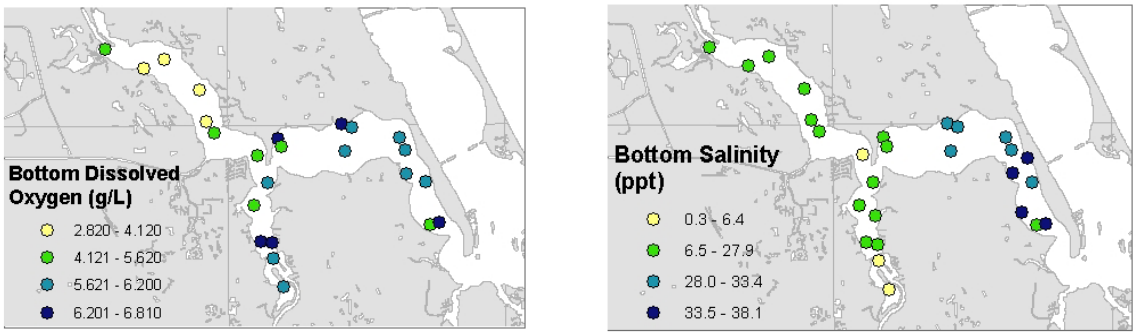

Figure 3-8. Bottom water dissolved oxygen and salinity concentrations.

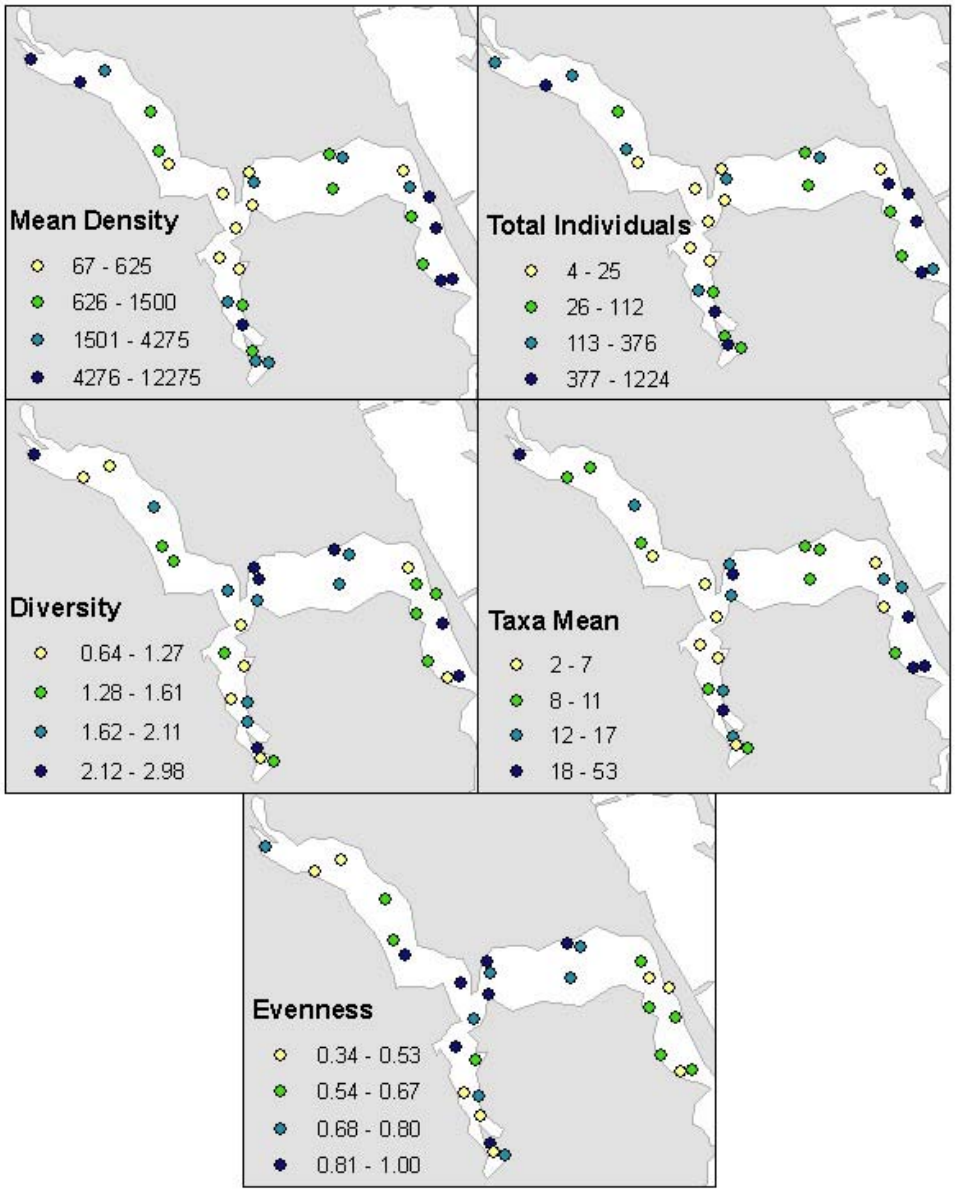

Figure 3-9. Benthic macroinfauna characterization.

4.0 Discussion

This study has characterized the St. Lucie Estuary using the Sediment Quality Triad (SQT) approach (Chapman et al., 1987). It has documented the levels of toxic chemicals, incidence of toxicity when test organisms or cells were exposed to bulk sediment, sediment pore water, and organic extract of sediment, and the distribution and community structure of benthic macrofauna. This approach has previously been used in assessing the

condition of several estuaries and coastal bays (Long et al., 1996, Long, 2000). All data produced under this study have undergone quality assurance procedures and are available on line (http://www8.nos.noaa.gov/cit/nsandt/download/bi_monitoring.aspx). It is anticipated that the study data, including those reported in ancillary studies, will be further analyzed and evaluated in years to come.

4.1 Contaminant Levels

The overall level of chemical contaminants in the estuary was found to be low. Contaminants that generally occur in high levels in urbanized estuaries, such as polycyclic aromatic hydrocarbons (PAHs), cadmium, and polychlorinated biphenyls (PCBs), were in appreciably low levels in the St. Lucie Estuary sediments. For example, the highest observed average concentration of PAHs, 1,064 ng/g (parts per billion) in the Convergence Zone stratum, is much lower than the effect-based threshold values of 4,020 ng/g (ER-L) and 1,684 ng/g (TEL). Low levels of PAHs in the estuary sediments were also reflected in the HRGS bioassay responses, many of which were below the "background" levels for estuaries (Anderson et al., 2005).

A major exception to the generally low levels of chemical contaminants was copper. Its concentration was high, notably in the North Fork stratum where its mean value was 75 µg/g (parts per million) with occasional values exceeding 100 µg/g. In relatively pristine coastal environments, copper levels in sediment are 15µg/g or less.

As is the case with most chemical contaminants, the U.S. Environmental Protection Agency has no sediment quality criteria for copper. However, effects-based benchmarks have been used as screening levels in assessing environmental conditions, such as the sediment quality guidelines developed using a nation-wide database (Long et al., 1995) and data specific to Florida coastal waters (MacDonald et al., 1996). In addition to the difference in the geographical scope of the data used to derive the threshold values, the two approaches are also different in terms of data inclusion and derivation of the benchmarks. Specifically, the derivation of the Florida guidelines, unlike the national guidelines, also included chemical concentrations observed or predicted to be associated with no observed effects (no effects data). The threshold effects level (TEL) in these guidelines represents the geometric mean of the 15th percentile in the effects data set and the 50th percentile in the "no effects" data set. The probable effects level (PEL) is the geometric mean of the 50th percentile of the effects data set and the 85th percentile in the "no effects" data set. Thus PEL represents the upper limit of the range of sediment contaminant concentrations dominated by no observed effects. For copper, this value is defined as 19µg/g; the PEL value for copper is 108µg/g.

The most likely means of copper input to the estuary is canals and tributaries that drain the coastal watershed, notably citrus groves and vegetable fields. In the citrus orchards and row crops within the estuary's watershed, copper based fungicides are used extensively (Leslie, 1992). The publicly-owned treatment plans (POTWs) represent a significant point source of copper and other contaminants. Other land-based sources of copper may include algaecides (used in swimming pools, ponds and lakes), roof runoff,

vehicle brake pads and soil erosion (Arnold, 2005). Leaching of copper from antifouling paint on boat hulls probably also contributes a significant amount to the estuary.

4.2 Spatial Extent of Sediment Toxicity

The battery of toxicity tests of different sensitivities and with different assessment end-points (mortality, impaired fertilization, and cellular response to contaminant exposure), coupled with randomly selected sampling sites, provide an adequate means to infer the spatial extent of sediment toxicity and compare data from different study areas. Sediment toxicity data are weighted to the areas of the sampling strata and toxicity responses at individual sampling sites (Long, 2000).

Reflective of low levels of contaminants in the estuary sediments, toxicity was not pervasive (Table 4-1) Given the critical value for declaring a sediment sample as toxic, no site in the St. Lucie Estuary was toxic based on amphipod mortality for the two species tested. However, toxicity was widespread when juvenile clams were exposed to the sediment. This suggests greater sensitivity of the species involved and its juvenile life stages to contaminant exposure. A nationwide database for the juvenile clam bioassay does not exist; it is therefore not possible to provide the St. Lucie Estuary data in a nationwide context.

However, in nearly all cases there was greater incidence of toxicity in samples from the North Fork and South Fork and much less at the mouth of the estuary and in nearby parts of the Indian River Lagoon and or in Jupiter Inlet, the reference area for assessing fish disease and abnormalities (Browder et al., 2006)

Table 4-1. Spatial extent of sediment toxicity, expressed as percent of study area(s). The nationwide values for amphipod and porewater toxicity are from Long (2000), and for HRGS from Anderson et al. (2005). A similar estimate based on testing of the juvenile clam does not exist.

				Sediment extract HRGS
	-			-0-
				7

The porewater testing to determine impaired fertilization in sea urchin eggs showed that 58 percent of the studied area was toxic. This fraction is considerably higher than for the nationwide data, i.e., 25 percent (Long, 2000). However, that estimate included data encompassing large areas in relatively uncontaminated waterbodies, such as northern Puget Sound, which showed 5 percent of the study area to be contaminated. Previous studies have shown that several shallow, urbanized estuaries indicated pervasive toxicity based on the results of this test, for example, San Pedro Bay (98%). In terms of a regional

context, Tampa Bay (84%), Biscayne Bay (47%) and Apalachicola Bay (34%) had fairly high spatial extent of toxicity (Long, 2000).

4.3 Sediment Quality Triad

The Sediment Quality Triad (SQT) data produced in this study includes data on more than 80 different chemicals, outcomes of multiple toxicity tests, and metrics to describe the distribution and abundance of nearly 150 taxa. Data from such a large number of environmental measures are inherently highly variable and often lack concordance. The SQT data are best suited for interpretation as "weight of evidence" to infer conclusions about the degradation of sediment quality. In a qualitative sense, general observations on the levels of chemical contaminants, positive toxicity results, and apparently altered benthic community structure may be interpreted as noted in Table 4-2.

Table 4-2. Qualitative interpretation of the SQT data

			Strong evidence of contaminant-induced degradation
			Strong evidence against contaminant-induced degradation
			Contaminants are not bioavailable
			Unmeasured contaminants or conditions causing toxicity
			Benthic community alteration not due to contaminants

Another, more effective way of communicating the information from the SQT data is to portray data in the form of triaxial plots (Figure 4-1) as outlined by Hameedi and Pait (2004). The approach includes the following steps:

1. Transform data for consistency and order in scales, i.e., lower values to indicate degraded conditions (e.g., low survival in a toxicity test; inverse of the HRGS response data; etc.);
2. Scale data to a range, e.g., from 0-99;
3. Determine average scores for aggregates of the three legs of the triad;
4. Identify 20th and 80th percentiles of the scaled values and plot on map
5. Make tri-axial plots of the scores for different strata or sites (smaller triangle indicates relatively degraded conditions); and
6. Estimate areas of the triangles to quantify the overall condition in a stratum (or at sites)

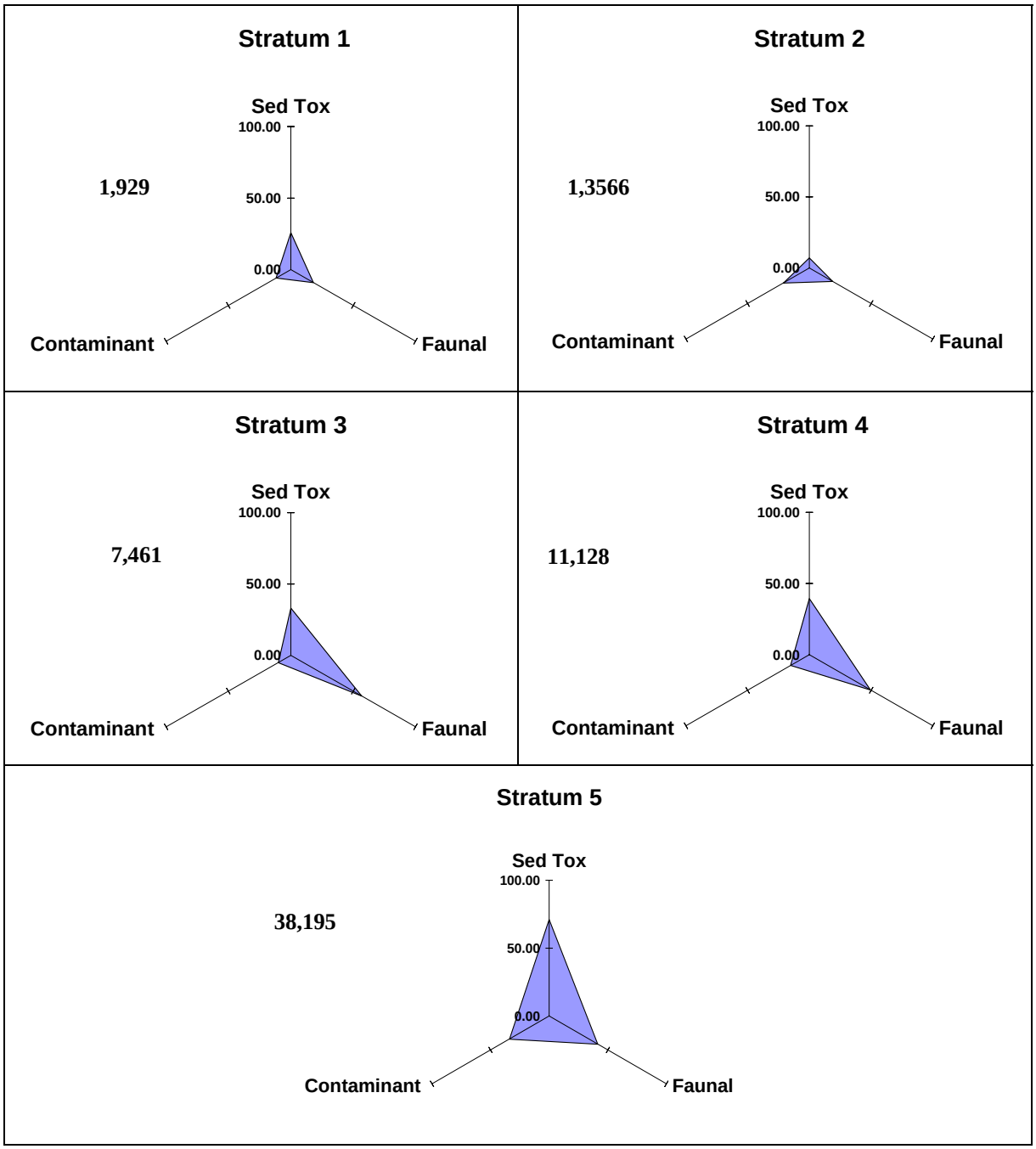

Figure 4-1. Schematic presentation of the Sediment Quality Triad data for five sampling strata in the St. Lucie Estuary. Numbers in bold type represent the volume of the triangles in arbitrary units (smallest area indicates the most degraded stratum).

The results of the SQT index indicate that strata 1 and 2 were relatively highly degraded. In addition to the obvious advantage of reducing large amounts of data into readily perceptible images and arbitrary numerical values, such an SQT index is desirable for the following reasons:

- It relies on observed or derived values, not on their statistics
- It does not use guidelines or "trigger values" most of which have conceptual or data limitations (O'Connor and Paul, 2000)
- It is transparent in composition; i.e., disaggregation of its components is possible for detailed examination using other techniques (cf. multiple regression, similarity analyses or exploratory analyses)

As is the case with all such indices, the SQT index is designed as a product for management use; it is not necessarily an ecological acumen. As such, it readily provides an assessment of relative sediment quality in sub-areas (or geographical strata) in a study region. The index is being further developed to integrate broad regional or national datasets and to further elucidate mathematical relationships among the triad components.

4.4 Copper as a Contaminant of Concern

Published data on background levels of toxic chemicals in the St. Lucie Estuary and adjacent coastal waters are extremely rare. In sediment samples collected during 1978-80 at a few sites in the Indian River Lagoon, copper concentrations were generally very low, ranging from less than 1 to 15 ppm although levels in the Eau Gallie Harbor sediment were much higher, approaching 200 ppm (Trefrey et al., 1983). Haunert (1988) described levels of copper in the St. Lucie Estuary showing a range of concentrations with high values in the North Fork and its tributaries. Differences in sampling strategy and analytical techniques aside, comparison of data from Haunert (1988) and this study indicates a substantive, as much as two-fold, increase in copper concentration in the sediment over approximately a 20-year period. Data are not available to make similar comparisons for other toxic chemicals.

As noted, sediment quality criteria for copper do not exist. The observed concentration of copper reported in this study and a very few other studies in the region approach or exceed the threshold for probable adverse effects on the biota. Further, analysis of data from copper-spiked sediment bioassays using the juvenile clam *Mercenaria mercenaria* resulted in a LC_{50} value [concentration killing 50% of the exposed organisms] of 125 µg/g and the "lowest observed effects concentration (LOEC)" and the "no observable effects concentration (NOEC)" of 31 and 16 µg/g, respectively (Dr. Mike Fulton, NOAA, unpublished data). These parameters are well within the range of reported values of copper in the St. Lucie Estuary. The NOEC value is quite comparable to the TEL value (19 µg/g) and the LC_{50} value to the PEL value (108 µg/g) for Florida coastal waters.

A further indication of copper as a contaminant of concern in the estuary comes from a logistic regression model, which is based on nationwide observations of amphipod mortality when exposed to sediment from estuarine and marine sediment (Field et al., 2002). According to the model, there is 52% probability for sediment to be toxic at a

copper concentration of 100 µg/g; for copper concentration of 10 µg/g about 5 percent of the sediment would be toxic (i.e., result in amphipod mortality).

The U.S. Environmental Protection Agency's National Recommended Water Quality Criteria (EPA, 2004) lists 4.8 µg/L as criterion for maximum concentration (CMC) in saltwater, and 3.1 µg/L as criterion for continuous concentration (CCC), implying acute and chronic values, respectively. There are no State of Florida water quality criteria for copper; only "red flag" values are noted to evaluate observed concentrations under different monitoring programs. The saltwater values are derived from a limited number of toxicity tests involving copper (as copper sulfate). There are only a few reports documenting the effects of copper on the hatching success or growth of crustaceans and other animals at concentrations that are generally found in coastal bays and lakes. Most recently, Brix et al. (2006) documented that hatching in the brine shrimp *Artemia fanciscana* was greatly impaired at a copper concentration of 5µg/L, and that copper had a stronger effect on hatching than cadmium and zinc.

Further, it is well-recognized that copper toxicity in saltwater is large due to divalent copper, Cu (II) aq, which may represent a very small fraction of the total copper and may be strongly affected by the presence of dissolved organic carbon (for example, under eutrophic conditions). Still, the criteria are expressed in terms of the dissolved metal in the water column and, as such, are highly conservative values. This study did not measure copper concentration in the water. However, it is important to acknowledge the amount of dissolved and suspended particulate matter-borne copper that is delivered to the estuary from numerous coastal tributaries, including those receiving effluents or seepage from citrus groves, vegetable fields and suburban communities.

4.5 Evidence from Biomarkers

A preliminary study for determining sub-lethal effects of copper exposure to oysters in the estuary was based on examining a suite of biomarkers. Several sensitive reliable biomarkers have been used to determine if there is evidence of significant contaminant exposures that have exceeded detoxification or compensatory mechanisms and may be adversely affecting physiological and biochemical functions (Ringwood et al., 1999). This study used lysosomal destabilization, glutathione concentrations (GSH), and lipid peroxidation levels (LPx) as indicators of stress on oysters deployed in the field for about a four-month period. The data indicated high bioaccumulation of copper in the oyster tissues amounting to 373 to 505 µg/g (dry weight) of copper, which far exceeded the 50th percentile (150 µg/g) and 85th percentile (360 µg/g) of NOAA's nationwide data for oysters (Mussel Watch data). Although the data were quite limited, overall results from these three tests suggested that copper levels in the St. Lucie Estuary are sufficiently high to contribute to chronic sub-lethal stress on oysters or other benthic organisms and could be affecting normal physiological functions such as reproduction (Ringwood, 2005).

4.6 Simulation Modeling

Since the majority of data under this study are based on static observations and laboratory tests, they are insufficient in detail for describing the dynamic behavior of copper in the

estuary. To address this limitation, a heavy metal model has been formulated with a capability to examine the transport and fate of copper in St. Lucie Estuary and south Indian River Lagoon (Applied Environmental Engineering, 2006a). This model is built upon the Environmental Fluid Dynamics Computer (EFDC) hydrodynamic model that has previously been used to simulate effects of high fresh water inflow events in the St. Lucie Estuary and other coastal waters in the region. The model has been transferred to natural resources management agencies in Florida for refinement with new data acquisition recommended by the modelers (Applied Environmental Engineering, 2006b). The model will help resource managers in gaining a better understanding of the ecological processes and water quality issues in this highly altered and complex estuarine system.

Analyses conducted by the modelers (Applied Environmental Engineering, 2006a) suggest that sediment copper concentrations not only are high presently, as described, in substantial parts of the estuary, but they have also increased markedly over the past two decades. Future increases will depend upon the future of various land use activities, particularly agriculture, and the success in developing and instilling Best Management Practices (BMPs), as well as retrofit and restoration efforts undertaken under the St. Lucie Issue Team Initiative and the Comprehensive Everglades Restoration Project.

4.7 Next Steps

Despite substantive scientific evidence that points to copper as a contaminant of concern in the St. Lucie Estuary, it is debatable as to how much the observed levels of copper in the estuary are adversely affecting the flow of products (such as fish) or services (such as recreation) of the estuary. With the limited scope of the present study, it is also uncertain whether the copper levels may imply signals of change in relationships among ecological attributes that have affected or would affect delivery of products and services in the future. For example, it is unknown at present whether or not the observed copper concentrations are responsible for the declines in the extent and productivity of oyster beds in the estuary or could be a contributing factor to reduced catches of 8 out of 14 fish species as noted in NOAA Fisheries records (Browder, 2005) or the reported >10% prevalence of fish with externally visible abnormalities (Browder et al., 2006)

If copper is recognized as a contaminant of concern in the estuary, it would require both source control and total maximum daily load (TMDL) estimates to assure continued propagation of fish and wildlife and other beneficial uses of the estuary. This would require data on source attribution of copper and identification of major sources that could be controlled through regulations or incentives. The simulation model produced as part of this study could be used as a tool to determine the estuary response in terms of copper loading and controls.

However, it is also important to consider management of the estuary as a whole and in light of different stressors, both natural and anthropogenic. A basis for such an approach exists in the form of multi-disciplinary studies that have been funded by SLRIT since 2000. A logical next step in the study of the St. Lucie Estuary is to synthesize information from the SLRIT-funded studies, preferably using the “integrated assessment” (IA)

protocols that focus on management needs of scientific data, transfer of appropriate tools and technologies to the decision-makers, and determination of future research priorities. Accordingly, the IA consists of the following four elements:

a) Characterization of the status and trends of ecosystem conditions and stressors;
b) Description of the ecological and economic consequences of change for each of the selected stressors;
c) Prediction of outcomes of management options for the most important stressors; and
d) Offering guidance for implementing a range of potential actions.

The geographical scope of this study did not include the Indian River Lagoon, which is located adjacent to the estuary and shares the estuary's watershed. Both waterbodies have been impacted by unnaturally high freshwater discharges and pollution from both point and non-point sources. Both are subject to restoration efforts under the Comprehensive Everglades Restoration Plan; however, the important question of the impact of toxic chemicals on the biota and performance measures to determine improvements in coastal contamination from extensive restoration efforts in South Florida are not presently contemplated. This remains an important missing element in setting up the much needed and reasonably prudent targets for habitat restoration and water quality improvements.

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