

Dynamics of Toxicants

Processes Which Control the Cycling of Toxicants in Galveston Bay

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Aquatic habitats are threatened by urban and industrial inputs of toxicants, such as heavy metals. One of the most toxic heavy metals is Pb, which shows neurotoxic effects to a large number of organisms even at very low concentrations (e.g., Thomas and Juedes, 1985 and Bengtsson et al., 1985). Supported by the Seagrant program, we are currently investigating the processes which control the cycling of the toxicant lead in Galveston Bay's waters. By focusing our current research on a single critical toxic contaminant of estuarine habitats, lead, we will provide the first reliable assessment of its inputs, its sources and rates of exchange between environmental compartments, as well as the processes controlling its fate, in Galveston Bay. Furthermore, quantitative information on removal rates and residence times of Pb and other heavy metals in the bay water will be obtained through the use of its natural radioactive isotope, ^{210}Pb . With support by the Texas Chemical Council, and the Texas Institute of Oceanography, we have previously investigated some of the processes which control the partitioning of the trace metals Pb, Cu, Zn, and Ag between the solution and the particulate phases in six Texas estuaries, including Galveston Bay (Benoit and Santschi, 1991; Benoit et al., 1992; and Morse et al., 1992). For Galveston Bay, the behavior of these trace metals, which are present at nanomolar and picomolar concentrations, was then compared to the geochemically significant metals Fe, Mn, and Al (Benoit et al., 1992). Based on this experience, we attempt here to summarize our conclusions related to the major processes that control the cycling of these metals in Galveston Bay.

There is a need in Texas for reliable data on the equilibrium and dynamic aspects of the behavior of heavy metals in estuarine waters. Currently, very little such data is available. The previous databases of the Texas Water Commission on trace metal concentrations in the water column are suspect due to contamination problems during sampling or analysis, as is readily admitted by its staff members. The ultraclean procedures employed in our lab have recently motivated some state agencies to change their sampling and analytical procedures for trace element analysis. It is, therefore, important that funding of small-scale studies which investigate the scientific facts on heavy metal cycling in Galveston Bay are continued. Any assessment of the effects of anthropogenic changes to the bay will be greatly enhanced by a better understanding of the speciation of heavy metals, relevant for any assessment of their bioavailability. Such studies also provide a critical link between basic science and Bay management. Our data should, therefore, be useful to policy makers when addressing issues such as regulating point source discharges or the effects of dredging. Beyond these local needs, results from our research should be of considerable value in the national and world-wide efforts to understand the coupling of the cycling of trace metals with that of organic carbon.

Estuaries are an especially critical component in the global geochemical cycle of trace metals, since most of the metals carried by rivers from the continents are trapped in or near estuaries (e.g., Santschi et al., 1984). Furthermore, recent studies using radioactive analogs suggest that estuaries may even act as a net sink for marine metals (e.g., Olsen et al., 1989).

Our main estuarine study site is Galveston Bay, not only because of its proximity, but also because the processes encountered in its waters and sediments are typical of other Texas estuaries (e.g., Armstrong, 1982). It is also the second largest, the most highly developed, and the most commercially important of these estuaries.

Our preliminary research on trace element and nutrient cycling in Galveston Bay suggest the importance of:

- 1) colloids to the removal to the sediments for a number of minor and trace elements, including Pb, Zn, Fe, and Al; and
- 2) sediment remobilization to inputs of Pb, Si, and possibly other trace elements, to the water column of the bay during certain parts of the year (Santschi, 1992).

Benthic remobilization of Pb was also suggested by recent study of trace metal cycling in the Brazos River Estuary (Keeney-Kennicutt and Presley, 1986). Therefore, the discussions in this paper will focus on the importance of: 1) benthic remobilization as an additional input to the waters of the bay and 2) colloidal uptake facilitating removal of trace elements to the sediments. Investigations into processes which control the self-cleansing capacity of the bay's waters, and the potential for long-term remobilization from previously contaminated sediments, need to be firmly rooted in basic science.

Physical, Chemical, and Biological Processes Controlling Trace Element Cycling in Estuaries

A concurrent study of the coupling of physical, chemical, geological, and biological processes is required to understand the spatial and temporal distribution of toxicants in Galveston Bay. Conceptually, the linkage of the water column processes to those occurring in the sediments can be achieved by coupling a well-mixed water column to a bioturbated surface sediment layer, both separated by a diffusive boundary layer (Figure 1). Such a conceptual model can easily be expanded to a kinetic multibox model using kinetic constants determined in the laboratory and in the field (e.g., Nyffeler et al., 1986). Such a model system can be of great help to predict the sensitivity of a particular toxicant to the various processes controlling its behavior in the water column and the time scales of its overall or net removal to the sediments, either by computer or by ecosystem simulation. Both allow an estimation of the flux of the same toxicant from the sediments to the water column, should the conditions become favorable for such a remobilization (e.g., for a model ecosystem, see Hunt and Smith, 1983; and Santschi et al., 1987).

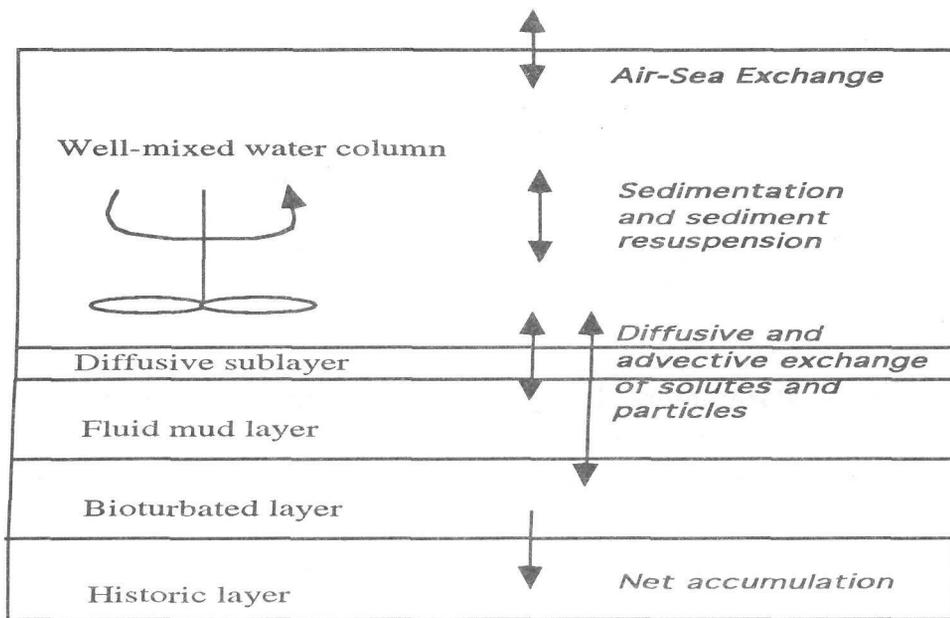


Figure 1. Processes in the Water Column

The sensitivity of a particular trace element to be removed to the sediments, and subsequently to be remobilized to the water column, is dependent on a combination of processes, represented by master variables, which represent some or all of the following dynamical processes.

1) Chemodynamics — Parameters defining the kinetics of physico-chemical exchange processes such as air-water, particle-water, colloid-particle, and sediment-water exchange, as well as the thermodynamic and kinetic parameters describing the chemical composition of particle, colloid, and water phases. Analytical and numerical models linking trace element adsorption onto particles to particle-particle interactions such as coagulation reactions, have previously been described (e.g., Honeyman and Santschi, 1989). As a consequence of the presence of colloidal forms of trace elements in the filter-passing fraction, equilibrium partitioning coefficients and kinetic parameters describing removal rates can become dependent on particle concentration (e.g., Honeyman and Santschi, 1988, 1990, 1992; Benoit et al., 1992; and Baskaran et al., 1992).

2) Hydrodynamics — Parameters defining the energy, momentum, and mass transfer processes of:

a) the water medium — Examples include transfer velocities, $\langle K \rangle$, for advective and diffusive transfer across the sediment-water interface. Diffusive transfer can be described by D/z , with D = molecular diffusion coefficient, and z = diffusive boundary layer thicknesses (z), where z can be related to friction velocities (u^*) or current velocities at 1 m above the sediment bed (U_{100}) (Santschi et al., 1991). Additional parameters include depth scales in water and sediments, and energy dissipation rates

(ϵ) controlling shear coagulation rates in the water column and near the sediment-water interface in the fluid mud or "fluff"-layer.

b) the particle medium: Parameters defining the velocity, mixing rates, and fluxes of particles. *Examples include rates of sediment resuspension (F_R), and benthic mixing (D_B), as well as particle settling (V_p).* A numerical model, which links particle dynamics in the water and sediments to chemical processes, has previously been described by Nyffeler et al. (1986), and further applied to various trace elements by Santschi et al. (1987) and Li et al. (1989).

3) Biodynamics — Parameters defining the input and quality of organic carbon production, kinetics of microbiologically or photochemically mediated degradation, and transformation rates, which often are also a function of temperature and salinity. Examples include primary production, remineralization, biodegradation, and photooxidation rates.

4) Sediment dynamics — Parameters characterizing sediment diagenesis include organic C, Fe, Mn, clay, carbonate and sulfide mineral contents and accumulation rates, grain size distribution, and pore water concentrations of geochemically relevant redox species such as O_2 , nitrogen species, MnO_2/Mn^{2+} , $FeOOH/Fe^{2+}$, SO_4^{2-}/HS^- , CO_2/CH_4 in the sediments (e.g., Santschi et al., 1990).

Chemical Speciation of Heavy Metals: Importance of Colloidal Forms

Heavy metals are among the most toxic, ubiquitous, and persistent contaminants in estuarine habitats. Toxicity to aquatic organisms, bioavailability, and the potential for bioaccumulation and food chain transfer critically depends on the predominant chemical species in the water, in particular, on the truly dissolved forms (e.g., Sunda et al., 1978). Very little is known about reliable concentrations of trace metals in estuarine waters (e.g., Cutter, 1991), and even less about their speciation, especially about their colloidal forms (e.g., Landing and Lewis, 1991). Even though it has been known for many years that the fate of a major fraction of many trace metals in estuarine and coastal waters is intimately tied to the fate of colloidal organic and inorganic matter (Sigleo and Helz, 1981), very little work has been carried out since that time to establish the colloidal association of trace metals.

It is not feasible at this time to predict the behavior and speciation of trace metals in Texas estuaries and their impact on aquatic ecosystems based on studies that have been carried out elsewhere. Unlike many other estuaries, Texas estuaries are subtropical and shallow, with high levels of suspended particulate matter and dissolved organic carbon. Almost no work has been done to establish reliable concentrations of trace metals in Texas estuaries. As a matter of fact, the results of Benoit and Santschi (1991) and Benoit et al. (1992) are the only reliable determinations of Cu, Zn, Pb, and Ag levels in Galveston Bay waters using state-of-the-art, class-100 clean techniques. For Ag, there is

only one other reliable determination in estuarine waters of the U.S., that of Smith and Flegal (1992).

Concentrations in Galveston Bay waters of some selected trace metals are typically of the order of 0.5-1 $\mu\text{g/l}$ of Zn and Cu, 0.01 to 0.1 $\mu\text{g/l}$ of Pb, and 0.1 to 10 ng/l of Ag (Table 1, and Figure 2, Oktay-Marshall and Wen, unpublished results). These values are considerably higher than those in Gulf waters, but orders of magnitude lower than values published by various government agencies for the same waters, as these agencies generally have not applied ultra-clean techniques during all stages of the sampling and analytical protocols. These values are only slightly higher than open ocean values (Bruland, 1983), but attest to our ability to measure low levels of trace metals in the ocean. Our unpublished results from the Gulf of Mexico indicate values of 0.005 $\mu\text{g/L}$ of Pb, 0.04 $\mu\text{g/L}$ of Zn, 0.01 $\mu\text{g/L}$ of Cu, and 0.2 ng/L of Ag in the deep waters (500 m depth) off the shelf/slope break (Figure 3, Wen and Oktay-Marshall, unpublished results).

Table 1. Range of Concentrations.

Element or Component*	Me _D ($\mu\text{G/L}$)		Me _C ($\mu\text{G/L}$)	
	Mean \pm S.D. (Range)	N	Mean S.D. (Range)	N
C _c	--		1700 \pm 500 (2,700 - 1,080)	3
COC	--		900 \pm 200 (1,300 - 500)	3
DOC	5600 \pm 1000 (7100-3800)	3	--	3
Al	26.4 \pm 21.6 (2-80)	20	53 \pm 47 (150-3)	3
Fe	20 \pm 8.3 (1-155)	19	23 \pm 20 (60-7)	3
Si	2000 \pm 500 (1000-3000)	11	150 \pm 130 (288 - 20)	2
Pb	0.07 \pm 0.03 (0.02-0.13)	19	0.075 \pm 0.001 (0.074-0.076)	2
Zn	1.7 \pm 1.1 (0.6-4.5)	17	0.685 \pm 0.015 (0.67-0.70)	2
Cu	0.86 \pm 0.33 (0.13-1.4)	21	0.035 \pm 0.005 (0.03-0.04)	3
Mn	8.7 \pm 15.4 (0.1-55)	16	0.2 \pm 0.1 (0.3-0.1)	3

Note: (from Benoit et al., 1992). Range of concentrations of dissolved and colloidal organic carbon (DOC, COC), colloidal matter (C_c), trace metals, Me_C, associated with colloids $\geq 10,000$ amu and ≤ 0.4 μm in size, based on AAS and ICP-MS analysis of colloids, collected from Galveston Bay, as compared to the "dissolved" concentrations of metal ions passing a 0.45 μm filter, Me_D (N = number of samples).

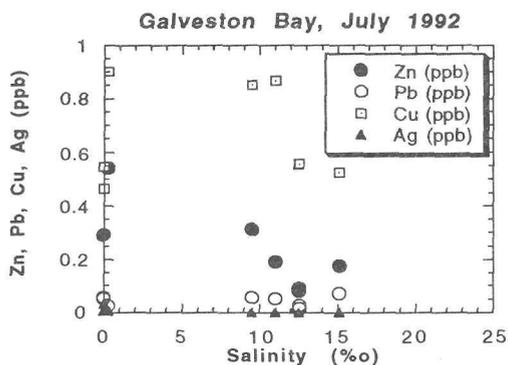


Figure 2.

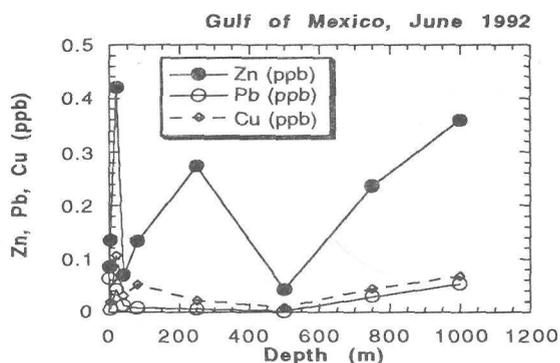


Figure 3.

Our preliminary results from the analysis of a few 200 liter water samples from Galveston Bay (Benoit et al., 1992) show that large fractions of Pb, Zn, Al, and Fe passing a $0.45 \mu\text{m}$ filter, (conventionally called the "dissolved" fraction) are actually associated with colloids rich in organic carbon, allowing for drastically different physicochemical and biological properties and pathways of these metals than their truly dissolved ionic forms (Table 1).

The most promising approaches to study the interactions between the carbon and trace metal cycling are those which combine geochemical with isotopic techniques (e.g., Santschi, 1988; Sigleo and Means, 1990). While the trace metal concentrations in the colloids are a representation of their sorptive and interactive capacities, their radionuclide (e.g., ^{14}C , ^{234}Th , ^7Be , and ^{210}Pb) concentrations are a reflection of their overall dynamic behavior, allowing the calculation of mean ages and turnover times of trace elements and colloids (Baskaran and Santschi, 1992; and Baskaran et al., 1992). Our studies of radioactive isotopes used to investigate the self-cleansing capacity of Galveston Bay water showed that significant fractions of ^{234}Th , ^{210}Pb , and ^7Be were also found to be associated with colloids (Baskaran and Santschi, 1992).

It is currently accepted that biological uptake and effects, bioavailability, and bioaccumulation, are related to the concentration of the free metal ion (e.g., Sunda et al., 1978). Water-soluble chelating agents may reduce uptake by organisms and toxicity of heavy metals (e.g., Furness and Rainbow, 1990, and references therein). However, lipid-soluble complexing agents can increase the bioavailability of heavy metals to marine invertebrates (e.g., Furness and Rainbow, 1990 and Connell, 1990, and references therein). Estuarine colloids are partially hydrophylic and partially lipophylic, and, therefore, it is not at all clear how colloidal associations of heavy metal ions may affect their biological fate and effects (e.g., Sigleo and Means, 1990). Since very little is known about heavy metal ion associations with natural colloids, our measurements should help to predict fate and effects of these potentially toxic heavy metals.

Acknowledgements

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Pollutant Metal Removal and Release Via Reactions with Sedimentary Pyrite in Galveston Bay

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The chemically "reactive" fraction of trace metals commonly undergoes diagenesis in sediments. Trace metals associated with metal oxides, organic matter, and other reactive components can be released to pore waters as these components are broken down by a variety of biologic and chemical processes. The trace metals may then be transported to overlying waters or be reprecipitated as discrete phases or as coprecipitates with other authigenic minerals.

In anoxic sediments, authigenic iron sulfide minerals have long been considered a potential major sink for many trace metals. Although the coprecipitation of trace metals with authigenic iron sulfides has been recognized as a probable significant aspect of trace metal diagenesis, no satisfactory method for quantifying trace metal concentrations in sedimentary iron sulfide minerals has been available. Consequently, progress has been limited in understanding the diagenetic behavior of trace metals in anoxic sediments.

A new method was developed for the quantitative extraction of trace metals from sedimentary pyrite (FeS_2), in order to overcome this limitation. (Pyrite is usually by far the dominant iron sulfide mineral phase.) The method is derived from the Lord technique for determination of pyrite-iron. It involves the sequential leaching of sediments using 1M HCl (reactive fraction), 10M HF (silicate fraction) and concentrated HNO_3 (pyrite fraction). Extensive tests of the method were carried out to insure its selectivity. Using this technique it is possible to follow the transition of trace metals from reactive phases to pyrite. A useful way to do so is to compare the degree of trace metal pyritization (DTMP) with the degree of iron pyritization (DOP).

Having developed this analytic technique, pyritization of reactive metals in a variety anoxic marine sediments was investigated to determine the factors controlling this process. Results indicate that all the trace metals considered increased in DTMP with increasing DOP. However, As, Hg, and Mo exhibited a rapid and almost complete incorporation into the pyrite fraction ($\text{DTMP} \gg \text{DOP}$). Several transition metals displayed a close to equal relationship between DTMP and DOP. Class B metals were also incorporated into the pyrite phase, but had DTMPs substantially less than DOP. These results indicate that transition and class B metals exhibit distinctly different behavior that is characteristic of the chemical attributes of these different classes of trace elements. High DOP and DTMP values are not a characteristic restricted to Fe-poor, H_2S -rich sediments. A close to quantitative transition of several metals, from the reactive phase to pyrite, was observed with

depth in sediments with moderate to high DOP values. This generally occurs within the top 10 to 30 cm of depth in shallow water sediments.

Subsequent studies focused on Galveston Bay, Texas, where there is considerable concern about metal contamination and its impact on the biota. Near-surface (top 10 cm) sediments were studied. It was found that, even within this near-interfacial zone, a major fraction of reactive Cu, Mo, As, and Hg are incorporated into authigenic pyrite. Commonly used methods for determining reactive trace metal concentrations would not have included metals in the pyrite fraction and in many cases would have missed over 90% of metals such as Hg. In order to be coprecipitated with the pyrite, the metals must have initially been in a reactive fraction.

Because trace metals associated with pyrite have not traditionally been considered as part of the reactive fraction, that may have a potential for bioavailability. The question arises as to whether such pyrite-associated trace metals remain in the reactive fraction. The answer to this question depends on whether or not authigenic pyrite can be dissolved once it forms in the sediment. Numerous studies indicate that authigenic pyrite can be dissolved within the sediment. This is accomplished primarily by oxidative processes associated with bioturbation of pyrite into the oxic near-interfacial zone of sediments and by seasonal vertical migration of redox conditions within sediments.

Another mechanism that can result in the oxidative dissolution of pyrite is the resuspension of sediments into the overlying water. This may occur in shallow water areas by natural processes such as storms and through human activities conducted on the oxidation kinetics of sedimentary pyrite, which initially anoxic sediments containing pyrite were suspended in seawater that was kept in equilibrium with the atmosphere. During a initial reaction period the oxidation of pyrite is relatively rapid, with 3% to 18% oxidation occurring in one day. Calculations indicate that this initial rapid rate of oxidation is the result of the oxidation of very fine (<0.3 mm) pyrite.

Experimental measurements of metal release from pyrite, in initially anoxic sediments, from Galveston Bay, exposed to oxic seawater, indicate that a major portion (20% to over 90%) of the pyritized metals can be released in a day or less. The As, Cu, and Hg released from pyrite usually exceeds the concentration of their reactive fraction determined by traditional methods. Consequently, the pyritization-depyritization of trace metals is an important process in controlling the bioavailability of many important trace metals.

In summary, authigenic pyrite is a major sink for Hg, As, Mo, and transition metals in anoxic sediments. However, trace metals associated with pyrite near the sediment-water interface should be considered as part of the pool of reactive trace metals that may be available for incorporation into organisms. This pool of metals is of especially great importance in Galveston Bay due the extensive dredging and bottom trawling that occur in the Bay and resuspend anoxic sediments.

Indicators of Trace Metal Pollution in Galveston Bay

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Sediments and organisms are usually more reliable and more convenient media for trace metal analysis than is water. Even polluted bay and estuarine water is very low in trace metal concentration, making it difficult to analyze reliably. Furthermore, concentrations in water are subject to rapid changes with changing metal inputs. Sediments and organisms have higher metal concentrations and they integrate values over time so less frequent sampling is needed.

Oyster (*Crassostrea virginica*) and other bivalves have been used as "sentinel" organisms for assessing the pollution status of marine water bodies for almost twenty years. For example, Goldberg et al (1983) report data for a USEPA funded "Mussel Watch" program conducted in 1976-78, and the current NOAA-funded "National Status and Trends Program" (NS&T) is an outgrowth and extension of the "Mussel Watch" concept. Bivalves are widely recognized as being responsive to changes in pollution levels in the environment, good accumulators of pollutants, widely distributed along coasts, and easy to collect and analyze. Sediments also respond to changes in pollutant trace metal inputs because most pollutant metals are particle reactive; that is, they readily attach to particles which can then sink to the bottom and become part of the sediments.

Oysters have been collected at six different sites in Galveston Bay (GB) since 1986 as part of NS&T. Each site is on an identifiable oyster reef and, for the first 5 years, twenty oysters were taken from each of three stations, the stations being 100 and 500 m apart. Currently only one station is sampled at each site. Each site has been sampled once each year, except two of the sites were not sampled the first two years. The twenty oysters from each station are combined and analyzed as a single sample each year. In most cases, stations are located hundreds of meters to many kilometers away from any obvious point sources of pollutant inputs in an attempt to characterize large areas of GB, rather than to identify specific point sources of pollutant input. Similar NS&T sampling is conducted in all other major bays and estuaries along the U.S. Gulf of Mexico coastline. The program allows different bays to be compared and pollutant concentration changes with time at a given bay to be documented.

Data obtained by atomic adsorption spectrophotometry (AAS) after acid digestion of oysters from the first four years of NS&T have been reported (Presley et al., 1990, 1991). The samples were analyzed for Ag, As, Cd, Cr, Cu, Fe, Hg, Mn, Ni, Pb, Se, Si, Sn, and Zn. Flame AAS was used for Cu, Fe, and Zn, which exhibit high concentrations in oysters, cold vapor AAS for mercury, and graphite furnace AAS for the remaining elements. Blanks and reference materials were analyzed with the samples. Precision and accuracy of the data was estimated to be $\pm 10\%$.

Trace metal concentrations found in oysters collected along the entire Gulf of Mexico coastline during the first four years of NS&T were generally similar to those reported in oysters taken from non-contaminated water in other parts of the world (Presley et al., 1990). Only a few sites showed obvious trace metal pollution and these were restricted geographically such that nearby sites were usually unaffected. Abnormally high or low values at a site did, however, usually repeat year after year suggesting local control. Abnormal sites for most metals were just as likely to be visibly pristine as to be highly industrialized.

Presley et al. (1991) reported that the oysters collected in Galveston Bay during the first four years of NS&T were similar in trace metal content to those collected elsewhere along the Gulf coastline, i.e., there was no indication of generalized trace metal pollution in GB. The average Ag, Cd, Cr, Fe, Mn, and Pb in GB oysters differed by 10% or less from the Gulf-wide average. Copper was 13% higher in GB, while Ni was 15% higher, and Se 16% higher. Zinc, however, was 43% higher. Furthermore, the highest Zn levels were found along the industrialized west side of Galveston Bay.

The four year NS&T sampling and analysis of oysters from the Gulf Coast discussed by Presley et al. (1990, 1991) has been continued for three more years with at least an additional three years planned. The basic patterns in concentration variability seen earlier have not changed significantly. With few exceptions, Galveston Bay oysters continue to be about average in trace metal content when compared to oysters from other bays along the Gulf Coast. Furthermore, oysters from near the entrance to the inland part of the Houston Ship Channel and from the industrialized western shoreline have about the same metal content as those from pristine areas of East and West Bays.

In non-funded student research designed to further investigate the relationship between trace metal concentrations in oysters and proximity of industry, samples were taken at twelve sites at the end of June and at the end of September, 1992. At most sites, 10-30 individual oysters were taken. They were collected, handled, and analyzed as described previously (Presley et al., 1990). No oysters were collected in extreme northern Galveston Bay, but shoreline samples were taken near Eagle Point and the highly industrialized areas of Texas City. Samples were also taken in central GB along the open-water part of the Houston Ship Channel and from East and West Bays.

Most trace metal concentrations were lower in oysters collected in September, 1992, than those collected at the same locations in June, 1992. In many cases, the decrease was by a factor of two and was, thus, larger than most site to site differences in the bay. It is very unlikely that this change was caused by human activity because there is little correlation between metal concentrations in oysters and proximity to population or industry, and even Fe concentrations in the oysters changed by up to a factor of two. Rather, the change in trace metal concentration must be related to some physiological change in the oysters. In order to minimize such changes, oysters are always collected in December for NS&T. The September, 1992, data is similar to the six-year average NS&T data, so perhaps oysters change in metal content less during fall and winter.

Silver concentrations are above the Gulf-wide average in several GB samples, but with no clear relationship to proximity to industry. Very high Ag concentrations were found in oysters collected at Confederate Reef in years V and VI (1990-1991) of NS&T, but not in previous years. A site on Deer Island near Confederate Reef was sampled for the 1992 student work. Oysters from it were somewhat higher than average in Ag content, but no more so than those from other sites in Galveston Bay. It is possible that human activity is responsible for the silver and zinc enrichments but no specific causative activity can be identified. In any case, the enrichments are not high enough to harm the oysters or human health.

Based on the discussion above and other data from our laboratory, oysters seem to integrate trace metal concentrations in the surrounding environment for one to two months. For a longer integration period sediments can be analyzed. As part of the unpublished student work reported here, sediments were collected at nineteen locations throughout Galveston Bay, including Morgan's Point and other locations along the industrialized northern and western shoreline, as well as locations far back into East Bay well away from industry. The sediment was sieved to separate the <63 μm grain size fraction, which was analyzed along with an aliquot of the unsieved bulk sample. Analysis was by AAS after both a partial leach of the sample with 0.5 N HCl and complete dissolution using HNO₃-HCl-HF. Results showed the sediment to be generally constant in trace metal concentration from place to place when the <63 μm size fractions were compared and to be similar to sediment from other Texas bays which were analyzed for NS&T. Average concentrations of metals in the <63 μm fraction of Galveston Bay sediments and the percentage of that metal leachable with 0.5 N HCl are shown in Table 1, along with average values for other Texas bays (normalized to 100% <63 μm grain size). Data from Morse et al. (in press) on another set of sediment samples taken from throughout GB confirms the relative constancy of trace metal concentrations. The most notable exception to sediment trace metal constancy found in the present work was a sample taken near the end of the Texas City Dike. It had <0.5% fine material but that fine material was enriched in several metals. Based on other data from this laboratory, it may well be that the fine fraction of very sandy sediment is easily enriched in trace metals from human activity.

Table 1. Average concentrations of trace metals in the <63 μm size fraction of Galveston Bay and other Texas bay sediments.

		Fe (%)	Ag (ppm)	As (ppm)	Cd (ppm)	Cu (ppm)	Ni (ppm)	Pb (ppm)	Zn (ppm)
GB	Avg	2.9	0.164	8.21	0.157	28.7	23.9	24.5	98.8
GB	S.D.	0.7	0.040	1.57	0.106	15.5	4.4	4.6	22.7
GB	Leach (%)	15	61	19	76	56	21	68	33
TX	Avg.	2.12	0.156	7.91	0.253	15.1	17.7	24.5	85.3
TX	S.D.	0.83	0.055	3.27	0.171	3.5	4.2	6.0	25.2

Several species of finfish (flounder, drum, trout, catfish, etc.) as well as blue crabs and oysters were collected from Galveston Bay in May-September, 1990, for the Galveston Bay National Estuary Program (GBNEP) (Brooks, 1992). These were analyzed for trace metals in our laboratory following procedures used for NS&T (Presley et al., 1990). The samples came from near Morgan's point, Eagle point, Hannah Reef and Carancahua Reef; thus, from areas of contrasting proximity to population centers and industry. In spite of the contrasts between the collection sites, no clear differences were found in trace metal concentrations in the organisms. Furthermore, the GB organisms were *similar in trace metal content to organisms from non-polluted bays elsewhere.*

Oysters are better accumulators of trace metals and, being attached to the sediment, should better characterize a site than the other organisms collected for GBNEP. The GBNEP oysters were generally similar in trace metal content to NS&T oysters from GB, but somewhat lower in Zn concentration. Zinc was also less clearly related to population and industry than in NS&T.

GB fish flesh was much lower in trace metals than oyster flesh and while isolated high values were found, concentrations were generally similar to those found in non-contaminated bays elsewhere. Trace metals in fish showed no relationship to population or industry. Fish livers proved to have much higher concentrations of trace metals (except for Hg) than fish flesh and high variability, but again no clear relationship to population or industry. Blue crabs from GB were generally intermediate in trace metal concentration between fish flesh and oyster flesh with similar high variability and lack of correlation with population or industry. The GBNEP data, therefore, gave no indication of trace metal pollution in Galveston Bay.

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