

*Final*

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**MCALLISTER POINT LANDFILL  
MARINE ECOLOGICAL RISK ASSESSMENT REPORT:  
TECHNICAL REPORT AND APPENDICES A-D**

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**NAVAL EDUCATION AND TRAINING CENTER  
NEWPORT, RHODE ISLAND**

Prepared For:  
Department of the Navy, Northern Division  
Naval Facilities Engineering Command  
10 Industrial Highway, Mail Stop No. 82  
Lester, Pennsylvania 19113-20

Prepared By:  
Science Applications International Corporation  
165 Dean Knauss Drive  
Narragansett, RI 02882

and

The University of Rhode Island  
Graduate School of Oceanography  
Narragansett, RI 02882

Under Contract To:  
Brown and Root Environmental  
55 Jonspin Road  
Wilmington, MA 01887  
Contract Task Order No. 197  
March 1997

**VOLUME I of II**

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## **1.0. EXECUTIVE SUMMARY**

This report summarizes the results of a Marine Ecological Risk Assessment (ERA) conducted for the McAllister Point Landfill at the Naval Education and Training Center (NETC) - Newport RI. The U.S. EPA's ERA framework and applicable EPA Region I guidance were used to generate and interpret the data required to complete this risk assessment. The objectives of this study were as follows:

- Assess ecological risks to the offshore environments of McAllister Point and Narragansett Bay from chemical stressors associated with the McAllister Point Landfill;
- Develop information sufficient to make informed risk management decisions regarding site-specific remedial options; and
- Support communication to the public of the nature and extent of ecological risks associated with McAllister Point Landfill.

This ERA builds upon and incorporates findings of previous ERA and RI/FS studies at McAllister Point, performed by TRC and Battelle Ocean Sciences, and specifically addresses three data gaps remaining from these earlier studies. These data gaps were:

- Incomplete assessment of the chemical exposure, including toxicity, to biological populations in surficial sediments adjacent to the landfill site;
- Incomplete assessment of the potential impacts of contaminants from the landfill to near shore subtidal areas and adjacent embayments to the south and west of McAllister Point;
- Limited scope of assessment of ecological risks to endemic populations in Narragansett Bay including organism condition and community structure.

The following sections present and discuss the findings of this assessment, including Problem Formulation, Sampling Summary, Site Characterization, Exposure and Ecological Effects Assessments, Characterization of Ecological Risks, and Risk Summary and Conclusions.

### **1.1. PROBLEM FORMULATION**

Proposed Contaminants of Concern (CoCs) for this study were found to include pesticides (except aldrin), butyltins, PCBs, and metals on the target analyte list. They were identified by the comparison of the chemical concentrations in sediment to NOAA Effects Range - Low benchmarks, which represent thresholds of potential biological effects for these media.

If appropriate sediment benchmarks did not exist, concentrations in surface sediments were compared to reference stations. Compounds measured at the site which exceeded either the benchmark or reference concentration (as appropriate) were included as a CoC. In addition, any target analyte for which the frequency of detection exceeded 5% was also included as a CoC.

### **1.2. SAMPLING SUMMARY**

Sampling locations in the McAllister Point Landfill area were selected to fill data gaps from earlier studies, which primarily included sediment and tissue chemistry sampling in 1993 by Battelle Ocean Sciences (BOS) reported in TRC (1994). The URI/SAIC Phase I study in 1994 included 15 stations to characterize the embayment area located primarily to the south of the landfill. In Phase II (1995), an additional 17

stations focused specifically on two primary habitats: the nearshore intertidal (NSB), and the subtidal, offshore habitats of the McAllister Point Landfill (MCL). All Phase II samples were characterized for CoC concentrations and toxicity. In addition, Phase II stations were assessed for community structure and biota condition. A Phase III data collection activity was undertaken by SAIC/URI in 1996 to assess possible changes in chemical exposure conditions due to a sediment erosion event resulting from the landfill revetment construction occurring after Phase II sampling was completed. The Phase III investigation included primarily sediment chemistry and toxicity on surface and selected core samples from the McAllister Point intertidal (NSB) and subtidal (MCL) sampling locations.

### **1.3. SITE CHARACTERIZATION**

The primary objectives of the site characterization are to identify the types, spatial extent and processes affecting marine and estuarine habitats that are present in and around McAllister Point Landfill, as well as the species and biological communities that may be exposed to site-related contaminants.

The McAllister Point Landfill is approximately 11.5 acres and is situated between Defense Highway and Narragansett Bay in the central portion of the NETC facility (Figure 1.3-1). During 1995-1996, the surface of McAllister Point Landfill was covered with an impermeable cap, and the landfill slope facing Narragansett Bay was covered with a stone revetment. A debris removal effort was also undertaken in the intertidal zone as part of the cap construction.

A variety of habitat types were observed to exist around McAllister Point, ranging from the upland and landfill areas, to rocky intertidal, fringing rock terrace and intertidal sand beach. A side-scan and sub-bottom sonar and sediment core survey of subtidal

habitats near shore suggested that the substrate is sand/cobble with macroalgal cover and abundant rocks; while subtidal depositional sand regions occur offshore.

An analysis of shoreline change in the immediate vicinity of McAllister Point from aerial photos taken between 1951 and 1975 reveal the period of the landfill seaward transgression to present conditions (1992). Some fluctuation in shoreline position was noted in the southern landfill intertidal over time, which is consistent with the expected cycles of beach deposition/erosion in the area. This process appears unrelated to the erosion event observed for the middle landfill intertidal area.

Shoreline observations during revetment construction suggested that erosion of surface sediment had occurred during winter storms, resulting in the exposure of underlying fill material. After the revetment construction was completed, the additional "Phase III" sampling, discussed above, was initiated to assess the extent of changes in environmental conditions that may affect the present ERA investigation. Results of chemistry and toxicity testing are incorporated into the present ERA. A topography survey of the area determined that up to 1.72 vertical feet of surficial sediment had eroded seaward of the revetment between Stations NSB-2 and NSB-5. Borings taken seaward of the stone revetment found landfill material at up to nine feet in thickness, with identifiable fill being observed up to 50 feet from shore.

The cause of the erosion event was attributed to the instability of the shoreline caused by construction and the increase in erosional energy at the foot of the landfill caused by large waves reflecting off the revetment and onto the intertidal zone. The sediment eroded from the shoreline to the north of McAllister Point is presumed to be permanently lost to deep water, whereas the sediment lost from areas to the east and south of McAllister Point appear to be located in an offshore bar.

## 1.4. EXPOSURE ASSESSMENT

Exposure assessment in the McAllister Point Landfill investigation addresses the spatial distribution and concentration of contaminants in bottom sediments and biological tissues, as well as the probable fate and transport mechanisms by which contaminants from the McAllister Point landfill reach the receptors of concern (bivalves, lobster, fish, birds).

### 1.4.1. Sediment Contaminants

*Sediment Geotechnical Characteristics.* The sediments near McAllister Point Landfill and within Jamestown Cranston Cove are very coarse grained and are generally characterized by very high sand content. They are less likely to retain contaminants than those from offshore stations, where sediments are composed of clay and fine silt, because the organic fraction of the sediments, which tends to bind the CoCs, is low. Sediments obtained from subtidal cores near the McAllister Point Landfill reveal that grain size rapidly increases with depth. The deepest samples show a layer of either coarse gravel, or highly weathered rock. These data suggest it is generally unlikely that significant reservoirs of contaminants exist deep beneath surface sediments (> 40 cm) in the vicinity of McAllister Point Landfill.

*Sediment Organic Contaminants.* Sediment samples collected at several intertidal (NSB) stations during Phase I and Phase II investigations were found to contain PCBs at concentrations exceeding the NOAA upper (i.e. less conservative) adverse effects benchmark (Effects Range - Median; ER-M) for PCBs (180 ng/g), while most of the offshore stations exceeded the lower (Effects Range Low; ER-L) guideline. For total PAHs, about one-third of the intertidal stations exceeded the ER-L guideline, but none exceeded the ER-M guideline. In contrast to the above organics, the pesticide

p,p'-DDE, and the antifoulant, tributyltin (TBT) were uniformly low and only slightly above benchmarks.

In comparison, the mid-depth and deep reference stations (JCC-M1 and JCC-D1) were below the ER-L PCB criteria, although Station JCC-S1 (22 ng/g) was very close to that level. The concentration of all butyltins and p,p'-DDE were also low as found for the McAllister Point area stations. The mixtures of individual PCB congeners and PAH analytes in sediments near McAllister Point suggest various types of source materials contribute to the contamination, including combustion products from used motor oil, creosote/coal tar, and asphalt. Only two stations, NSB-3 and NSB-7, showed qualitative evidence of unweathered petroleum hydrocarbons, probably originating from diesel and/or bunker fuel in the landfill. In contrast, PAHs found at Station S2B were highly weathered, and clearly different from the mixture of PAHs found at the landfill. Organic contaminants in sediment cores generally decreased with depth to near background at 46 to 65 cm.

*Inorganic Contaminants.* As with organic contaminants, trace metal concentrations were also compared with benchmarks for biological effects. Trace metal concentrations for most of the area offshore and south of McAllister Point Landfill were below ER-L guidelines. SEM/AVS studies (a method used to measure bioavailability of metals to biota) predicted that the divalent metals (Cu, Cd, Pb, Ni and Zn) are potentially bioavailable at some subtidal stations; the significance of this find is discussed further in Section 1.6.

Highest metal concentrations were found in the relatively coarse-grained sediments of the intertidal stations (NSB-1 to NSB-7). Here, copper, lead, mercury, nickel, and zinc exceeded ER-M guidelines while cadmium and chromium were found to exceed ER-L guidelines. The spatial pattern of metal distribution and concentrations observed in surface sediments were found to be similar between the present study and

the 1994 TRC/BOS study and strongly implicates McAllister Point Landfill as the dominant source of these metals within the study area.

*Comparison of pre- and post-erosion sediment concentrations.* The purpose of the comparison between Phase II (pre-erosion) and Phase III (post-erosion) results is to assess whether sediment erosion (discussed in Section 3.1) from the nearshore environment of McAllister Point Landfill had increased possible CoC exposure to aquatic biota. Total PCBs and Total PAHs measured in surface sediments and sediment cores during Phase III were compared to chemical concentrations found at the same stations prior to the erosion event. For PCBs, stations with significant increases (RPD  $\geq$  +30%) and values above the ER-M included intertidal surface sediments from Stations NSB-4, NSB-5 and NSB-7 and both surficial and subsurface (0-18 cm) sediment at offshore Station MCL-12. Increased concentrations to levels above the ER-L were observed for PCBs at Stations NSB-1 and NSB-2, and for PAHs at Stations NSB-6 and MCL-12 (surface and core). The distribution of individual PAH components were generally similar, both within and between most stations.

Metal concentrations analyzed during Phase III were higher than metal concentrations determined during Phases I and II for several metals at stations in the study area. Concentrations of three metals (copper, lead, and zinc), which exhibited the greatest degree of change; surface sediment concentrations of copper increased to levels above the ER-M at Stations NSB-2, NSB-3 and NSB-4. Concentrations of lead also exhibited elevations to levels above the ER-M at Stations NSB-2, NSB-4, and NSB-5, while similar increases in zinc were noted at Stations NSB-2, NSB-4, NSB-5, NSB-7, MCL-10 and MCL-14. Offshore, concentrations of copper and zinc increased to levels above the ER-L guidelines at Stations MCL-10 and MCL-12, respectively. The observed increase in surficial sediment concentrations, particularly at Stations NSB-2, NSB-3, NSB-4, and NSB-5, indicate that erosion at the McAllister Point Landfill has

increased potential adverse exposure from contaminated sediments with respect to trace metals.

Direct comparisons for core data for the nearshore stations was not possible since pre-erosion cores were not collected in this area. In general, Stations NSB-2 through NSB-4 had higher concentrations of PCBs and PAHs in the core samples (0-18 cm section) relative to surface sediments (0-2 cm). Metals in intertidal core samples were generally comparable to surface sediment concentrations. A notable exception may include increased levels of zinc at Stations NSB-2 and NSB-4, but reduced Zn levels at other stations. With respect to core samples, somewhat elevated metal concentrations to levels above the ER-L were observed for Stations MCL-10 and MCL -12 in relation to the pre-erosion surface sediment concentration.

#### 1.4.2. Tissue Residues

*Organic contaminants.* A total of thirty-eight tissue samples were analyzed for organic contaminants including: mussels (*Mytilus edulis*) and hard clams (*Mercenaria mercenaria*), with and without gut sediment removed (depurated and non-depurated, respectively); cunner (*Tautoglabrus adspersus*); and two tissue types (muscle and hepatopancreas) from lobster (*Homarus americanus*). Spatial patterns observed for tissue residues of blue mussels (which characterize bivalve exposure to CoCs in intertidal areas) differed depending on contaminant class. Total PCBs in blue mussels tissue residues were highest at Station NSB-3 (>1800 ng/g); p,p'-DDE residues were highest at Stations NSB-6 and NSB-7 (735 ng/g); TBT was highest at Stations NSB-1 and NSB-3 (>35 ng/g); and Total PAHs were highest at Station NSB-1 (71,550 ng/g). For Total PCBs, p,p'-DDE, and Total PAHs, the depurated mussel values were about 70 to 90 percent of the non-depurated levels, indicating that sediment in the gut of the organism contributed about 10-30 percent of the total chemical load of the animal. In

contrast, higher (7 to 40%) TBT values were observed in depurated mussels than in the non-depurated samples.

Concentrations of PCBs, PAHs, and p,p'-DDE contaminants in the hard clams (sampled in the present study to characterize infaunal conditions of the subtidal environment) were more spatially uniform among stations than mussels, and were also about 5 times lower in residue concentrations. TBT concentrations were two-fold higher in hard clams than in mussels. From depuration studies, sediments in the guts of hard clams contributed 5 to 10 percent of PCBs, p,p'-DDE, and total PAHs contained in the animal. Some of this difference between mussels and clams may be due to higher excretion rates of sediment by mussels. As with mussels, TBT values were higher in depurated hard clams than in the non-depurated samples.

Organic contaminants in cunner (*Tautogolabrus adspersus*) indicated that no single station had elevated values for all components. Total PAHs were five- to thirty-fold lower than bivalve residues, partly because fish can better metabolize, and thus more easily excrete, PAHs than bivalves. TBT values in the fish were also five- to ten-fold lower than that observed for bivalves, whereas, total PCBs and p,p'-DDE levels in fish tissues were many times higher than in bivalves (2-20 times and 4-12 times, respectively), possibly reflecting either differences in lipid content (fat) or gut sediment content.

Concentrations of contaminants in lobster (chosen to characterize subtidal epibenthic invertebrates) were approximately two times higher at Stations MCL-13 and MCL-14 as compared to other subtidal stations in the northern landfill region. Hepatopancreas (a liver-type organ) concentrations were 20 times higher for total PCBs and p,p'-DDE, and 4 times higher for total PAHs. Organic contaminants are lipid-soluble and therefore should be expected to be higher in the hepatopancreas relative to the muscle because of the associated differences in the tissue lipid content.

*Inorganic contaminants.* Samples of blue mussels, hard clams, and lobster muscle and hepatopancreas (discussed above) were also analyzed for trace metals. The distributions of arsenic, iron, chromium, zinc, manganese, and nickel did not appear to exhibit either spatial- or species-dependent patterns. In addition, little difference was observed between metal content of non-depurated and depurated bivalves, except for lead and aluminum, where a significant proportion is lost by sediment gut purging. For lobster, cadmium, copper, and silver were about ten-fold higher in the hepatopancreas than in muscle samples. The concentration of mercury is highest in lobster muscle, possibly reflecting biological magnification at higher trophic levels.

#### 1.4.3. Sediment Fecal Pollution Indicators

Sediments collected from the vicinity of McAllister Point were analyzed for fecal pollution indicator bacteria (total and fecal coliforms, *Escheria coli*, *C. perfringens*) to assess the sanitary quality of this marine environment and to indicate potential sewage-related anthropogenic inputs to this area. All sediments contained one or more of the fecal pollution indicators, and the relative densities suggested the influence of fresh sources of sewage or animal waste-related contamination in the study area, but fecal pollution indicator values decreased with proximity to the middle landfill region where concentrations of contaminants of concern were highest. Hence it is possible that other potential transport pathways for CoCs other than the McAllister Point Landfill may be present in the study area, but such sources are relatively unimportant for the areas where highest landfill-related contamination was observed.

## 1.5. ECOLOGICAL ASSESSMENT

Ecological effects are quantified by determining the relationships between exposure patterns and resulting responses of ecological systems, as determined from measurement endpoints identified during Problem Formulation. Site-specific evaluations of toxicity bulk surface sediments using the 10-day amphipod (*Ampelisca abdita*) mortality test and porewater/elutriates using the sea urchin (*Arbacia punctulata*) fertilization and larval development tests were performed to assess adverse effects of landfill-related contaminants. In addition, field-based assessments including community structure analyses and biota condition were conducted.

### 1.5.1. Toxicity Evaluations

Spatial trends in amphipod survival exposed to sediments from Stations S2B, and NSB-1 through NSB-7 indicate significant toxicity for intertidal areas near the landfill, particularly near the middle landfill. In contrast, no significant toxicity to amphipods was observed for any offshore stations. Toxicity of interstitial waters (porewaters) to sea urchins was apparent at three middle landfill stations (NSB-3, NSB-4, NSB-5) and S2B, a nearby subtidal station. Toxicity was also observed at one offshore location (D3), but, similarly-located stations in Phase II (MCL-15, MCL-16) did not exhibit toxicity. Two additional subtidal stations near the landfill (S2B and M1) which exhibited high toxicity (<50% fertilization success) were located near the intertidal stations discussed above. Toxicity was observed for one additional station (S3) but this response may be due in part to ammonia present in the sample.

*Comparison of pre- and post-erosion results.* Results of amphipod survival in bulk sediments and sea urchin fertilization in sediment elutriates collected during the Phase III McAllister Point Landfill study area resampling event were compared to toxicity results from Phase II testing. Stations for which sediment toxicity increased, as

measured by toxicity to amphipods, included primarily Stations NSB-2 and NSB-4 where survival declined to less than 10%. In contrast, post-erosion toxicity was lower than that found for pre-erosion conditions at Stations S2B and NSB-6, while toxicity was unchanged at the remaining stations.

Toxicity, as measured by sea urchin successful fertilization in 100% sediment elutriate preparations from Phase III sediment samples was, with the exception of Station NSB-4, greater than that observed in porewater testing during Phase II. Results for the sea urchin larval development test, also conducted on sediment elutriates, reflected a broader range of toxicity, but a comparable rank order sensitivity as observed for the fertilization test. Better separation in apparent toxicity for the fertilization test was observed for the 50% elutriate exposure, where in this case, Stations NSB-2 and NSB-5 exhibited fertilization success less than 5%, while at other stations the response improved to greater than 70%.

The cause of the uniformly low success of sea urchin fertilization in Phase III sediment elutriates in contrast to generally high amphipod survival in the parent bulk sediment samples was evaluated further and found to be at least partially attributable to interference caused by the suspended particulates present in sediment elutriates, perhaps limiting the mobility of the sperm or penetrability of the egg.

The combined results of the Phase III toxicity tests suggest high toxicity (unchanged and/or increased from Phase II) at Stations NSB-2, NSB-4 and NSB-5, while possible increased toxicity was apparent for Station MCL-12 and, to a lesser extent Stations MCL-10 and NSB-3. The remaining stations suggest little evidence of increased toxicity as a result of the erosion event.

### 1.5.2. Field Effects Evaluations

Sampling included infauna and epifauna for benthic community structure analysis and large bivalves for assessment of condition.

*Benthic Community Structure.* Benthic organisms at intertidal and subtidal stations adjacent to the McAllister Point Landfill were sampled and identified in order to assess potential environmental stress on macroinvertebrate populations. At intertidal stations (NSB-1 to NSB-7), single samples were taken in sediment with and without embedded clusters of blue mussels (*Mytilus edulis*), referred to as "sediment" and "mussel bed" samples, respectively. At subtidal stations, only sediment samples were taken.

To discriminate between degraded and reference sites, benthic data were evaluated for species diversity, total abundance and the relative abundance of opportunistic, pollution-tolerant species, including the capitellid polychaete, *Capitella capitata*, certain spionid polychaetes (*Marenzelleria viridis*, *Pygospio elegans*, *Streblospio benedicti*) and the oligochaete, *Pelosclex benedeni*.

The shore between Stations NSB-1 and NSB-4 is steep with a narrow intertidal zone and no obvious fresh water discharge. In contrast, the shoreline between Stations NSB-5 and NSB-7 has a low slope, areas of standing water and fresh water discharge and finer grained sediments. For mussel samples, there was no change in epifaunal species number or abundance between high and low areas. For both mussel and sediment samples, however, the change from high to low slope (Stations NSB-4 to NSB-5) was correlated with increased density of the pollution-tolerant spionid polychaetes (*M. viridis*, *N. succinea*, *P. elegans*, *S. benedicti*) but also juvenile soft shell clams (*Mya arenaria*) and hard clams (*Mercenaria mercenaria*). *C. capitata* was also notably more abundant in sediment in the middle landfill area Stations NSB-5 and

NSB-6, relative to northern stations. The oligochaete *P. benedeni* was of relatively uniform abundance across the intertidal stations, but became the dominant oligochaete at Stations NSB-5 and NSB-6.

The possible effect of slope on habitat as an alternate explanation to landfill impacts resulting in observed increases in pollution tolerant forms cannot be discounted. Lacking a proper reference site for this environment, landfill-related impacts on benthic community structure in the intertidal appear possible, but are difficult to evaluate conclusively.

Subtidal samples for benthic community analysis consisted of McAllister Point Landfill study area Stations MCL-8 through MCL-12 and MCL-14, and the Jamestown Cranston Cove reference area (JCC stations). No spatial patterns of increased abundance of pollution-tolerant forms were observed among the MCL stations, and the diversity of species at subtidal stations near McAllister Point landfill was high relative to previous subtidal surveys in lower Narragansett Bay, perhaps due to the hard substrate at MCL stations which provided a more diverse habitat.

*Bivalve condition indices.* Bivalve condition indices, based on allometric relationships of length, tissue weight, and shell weight, were used to indicate the ecophysiological status of animals as a function of spatial proximity to the landfill. No statistically significant differences ( $P_f = 0.05$ ) in indices for blue mussels were observed. However, a tendency for reduction of the tissue weight/length ratio was noticed at Stations NSB-1 through NSB-7 (particularly for Station NSB-2 and NSB-7), relative to stations away from the landfill (Stations NSB-8 through NSB-11). Other indicators did not exhibit discernable trends.

## 1.6. RISK CHARACTERIZATION

Risk Characterization phase for the McAllister Point ERA includes the evaluation of the Exposure Assessment and Effects Assessment Weights of Evidence (WoE).

The five principal WoE of Exposure Assessment include:

- Comparisons of sediment concentrations with ER-L and ER-M benchmarks;
- Comparisons of porewater concentrations with Water Quality Criteria;
- Assessment of divalent metal (Simultaneously Extracted Metal) bioavailability;
- Sediment fecal pollution indicator concentrations; and
- Evaluation of tissue CoC concentrations at the site relative to the reference location (Tissue Concentration Ratios);

while the WoE for Effects Assessment are:

- Evaluation of toxicity and comparison of these results with CoC sediment and pore-water concentrations;
- Analysis of CoC concentration versus effects measurements; and
- Analysis of CoC bioaccumulation in fish, bivalves and lobster, and related potential impacts on the avian predators due to the ingestion of aquatic biota.

Each WoE also has multiple supporting indicators, such as analyte-specific Hazard Quotients for sediments and porewater, TCR values for each of the aquatic receptors (mussels, clams, lobster, fish), amphipod and sea urchin toxicity, etc.. These indicators are intended to increase the certainty of the assessment with regard to the presumption of adverse exposure or effects conditions. A final, but critical element of

the risk characterization is an analysis of uncertainties associated with the above interpretations.

The individual indicators within each Weight of Evidence (WoE) were interpreted and summarized using semi-quantitative ranking schemes so as to allow the synthesis of the overall probability of adverse Exposure/Effects (E/E) indicated for each of the primary weights of evidence. Comparability of ranking strategies for the synthesis of indicators within the various WoE was deemed necessary in order to provide a consistent evaluation of exposure/effects (E/E) data. Thus, for the majority of WoE, the quantity and nature of indicator data permitted the development and application of the following E/E ranking strategy, as follows:

<i>Baseline Adverse E/E Probability (-):</i>	Baseline (-) ranking for all indicators, or low (+) ranking observed for only one indicator;
<i>Low Adverse E/E Probability (+):</i>	Low (+) ranking observed for two or more indicators, or intermediate (++) ranking for only one indicator;
<i>Intermediate Adverse E/E Probability (++):</i>	Intermediate (++) ranking observed for two or more indicators, or high (+++) ranking for one indicator;
<i>High Adverse E/E Probability (+++):</i>	Intermediate (++) or greater ranking observed for two or more indicators, <u>and</u> high (+++) ranking for one or more indicators.

For the two WoE where this approach was not deemed appropriate (SEM bioavailability and benthic community data), the overall rank was taken as the maximum of the indicator-specific values.

The above ranking strategy is intended to characterize the extent and pervasiveness of CoC-related exposure or effects. For the exposure WoE, for example, the extent to which CoC concentrations in various matrices (sediment, porewater, tissue) exceed benchmarks, and how often this exposure or effect was observed among the individual WoEs. It should be noted that the above rankings for exposure-based WoE did not consider exposure-response relationships; this information was incorporated into the effects-based WoE evaluation. In addition, this type of ranking scheme is intended only as a qualitative tool. The ranking approach is based on best professional judgement, since the "true" ecological risk of, for example, benchmark exceedence or observed toxicity, is not presently known. Hence, the risk manager is encouraged to keep in mind the nature of the risk ranking approach when evaluating the general outcome of the risk assessment.

As an additional step in the summarization of exposure and effects WoE, Ecological Exposure Zones (EEZs) were delineated based on an understanding of the general hydrographic, bathymetric and habitat characteristics of the area, as well as trends in spatial distribution and composition of contaminants found in sediments and tissues, the distribution of effects, and the proximity among sampling stations in the study area. Figure 1.6-1 shows the eight EEZs that have been identified for the McAllister Point ERA, containing stations from four sampling events (TRC/BOS, Phase I, Phase II and Phase III investigations). The CoC-related characteristics of each zone are discussed in the appropriate WoE section. A brief description of the natural characteristics of these zones is included below:

*Zone 1: Landfill Intertidal North.* A steep-sloping intertidal, this zone includes Station NSB-1. This station was selected to characterize the northern extent of intertidal environment assessed for potential landfill-related impacts. This zone contains habitat for primarily epifaunal macroinvertebrate communities and blue mussels growing on and between large boulders. Small fish, including cunner, are

likely to occupy the habitat. Shore birds including the herring gull and great blue heron may feed upon the epibenthic communities and fish located in this area. This zone faces west-northwest and is the most highly exposed zone in the study area to both winter storm conditions and swell resulting from summer sea breezes. Gomes Brook drains into Narragansett Bay to the north of this zone. The substrate is rocky/sandy sediment as typical of a high energy intertidal environment. Some sediment erosion was observed toward the southern end of this zone as assessed during the Phase III investigation.

*Zone 2: Landfill Intertidal Middle.* This west/southwest-facing zone includes the intertidal habitat Stations NSB-2, NSB-3, NSB-4, and NSB-5. This area has a habitat generally comparable to that of Zone 1. This zone also has the greatest degree of visible refuse and sediment staining. This area represents the furthest point of extension of the landfill into Narragansett Bay. The substrate is rocky/sandy sediment as typical of a high energy intertidal environment. As in Zone 1, this region provides habitat for mussels and small fish which may be consumed by shore birds such as the gull or heron. This zone was also the region where, prior to the Phase III investigation, the greatest degree of sediment erosion was observed to have occurred.

*Zone 3: Landfill Intertidal South.* This zone includes Stations NSB-6 and NSB-7, and comprises a shallow-sloping rock/pebble beach environment with relatively low surface relief, lacking larger rocks found in zones to the north. This zone is southwest-facing and is moderately exposed to wave action during summer sea breeze conditions but is shielded by the landfill from northeasterly storms, a condition which has allowed the development of a sand/pebble beach which may vary in extent depending upon seasonal cycles of sand migration. The area appears influenced by creek drainage from a culvert located southeast of NSB-7. Sparse eel grass has been observed to the southwest of NSB-7. These stations were selected to characterize the southern extent of the intertidal environment and associated potential for landfill-related impacts. As with

the northern and middle intertidal zones, this region provides habitat for macrobenthos, mussels and small fish which may be consumed by shore birds.

*Zone 3A.* This isolated area is located approximately 50 meters offshore of Zone 3, and includes Stations S2B, S2C, M1, MCL-12, and TRC Station OS-28. This area has been given separate designation due to unique chemical and toxicological conditions observed during the present and previous studies (discussed below). A sand bar was observed to have formed in this zone after the sediment erosion event. This transitional habitat from shallow to deep water would be expected to contain macrobenthos, as well as mussels and small fish where hard substrate is available, but may also be frequented by more mobile fish species such as winter flounder. Water depths exceed 3 m, limiting availability of prey to avian predators.

*Zone 4: Landfill Subtidal - Nearfield.* This area includes Stations MCL-8 to MCL-11, which define an area of approximately 50 m wide which runs the length of the landfill immediately off shore of the intertidal Stations NSB-1 through NSB-4. This habitat is characterized as silty sand and supports relatively sparse populations of hard clams, but abundant lobsters; sidescan imagery of the area reveals that larger boulders are also present in this area. Winter flounder may also range into this area, feeding on macrobenthos, although the primary habitat for this species is expected to be the deeper offshore flats which better support its primary food resource (nereid worms).

*Zone 5: Landfill Subtidal - Farfield.* This area defines the subtidal environment offshore of McAllister Point Landfill seaward of Zone 4, and includes TRC/BOS Stations OS-22 through OS-27. Hard clams were collected by TRC (1994) from this area. Numerous floats for lobster traps are visible in the area, suggesting suitable habitat for this species. Winter flounder would be expected to occupy this region. Maps of regional geology and a side scan sonar survey of the area suggest sand and silty sand bottom with boulders.

*Zone 6: "Southern Depositional Area".* This region extends from the Coddington Cove breakwall south of the site to the north to Zones 3, 3A, and 5, as defined above, and extends seaward from the intertidal zone adjacent to NETC properties to offshore areas of approximately 12 m depth. This area was sampled primarily in Phase I to determine whether sediment potentially originating from McAllister Point may have been deposited there. Stations sampled in this zone include Stations S1 through S4, M2 and M3, D1 through D3, MCL-13 through MCL-16, OS-30A, OS-30B, and TRC/BOS Stations OS-29 and OS-30. Sidescan of the area shows relatively featureless relief of sediment characterized as silty sand, except for a deltoid-shaped region of disturbed sediment extending away from the Coddington Cove breakwall. This habitat is expected to contain macrobenthic communities, hard clams and lobsters.

*Zone 7: Reference.* Includes reference stations at Jamestown Cranston Cove (JCC), including shallow (<3 m; JCC-S1), mid-depth (3-5 m; JCC-M1) and deeper water (>10 m; JCC-D1) stations. The area receives freshwater input from Carr Creek on Conanicut Island and has viable eelgrass beds nearshore. The nearshore macrobenthos resembles that offshore of McAllister Point (Zone 4), while the deep reference station contain macroinvertebrate species typical of shelf communities. Sparse numbers of hard clams are apparent; lobsters also appear to occupy the area based on commercial trap deployments.

#### 1.6.1. Exposure Assessment Summary

*Comparison of CoC Concentrations with Criteria and Standards.* Concentrations of contaminants of concern (CoC) were compared against effects-based screening benchmarks for sediments (NOAA ER-L and ER-M values) whereas porewater concentrations were compared against EPA Water Quality Criteria. For each matrix, Hazard Quotients were calculated as the measured concentration at the station divided by the benchmark concentrations.

Site concentrations of sediment organics including Total PAHs, Total PCBs, the pesticide p,p'-DDE relative to ER-L and ER-M guidelines generally exhibited similar spatial and temporal trends. Total PAH concentrations exceeded the ER-L (4,022 ng/g) at TRC/BOS Stations OS-22, OS-25 through OS-30, at Phase I/II Stations NSB-3, NSB-4, NSB-5, NSB-7, MCL-8, MCL-10, MCL-16, S2B, and M1, at Phase III Stations NSB-6-R, MCL-12-R, and S2C (stations shown on Figure 1.6-1), and at reference Station JCC-D1. Total PCBs generally exceeded the ER-L (22.7 ng/g) throughout the study area. Total PCB concentrations exceeded the ER-M (180 ng/g) by more than two-fold at Phase II Stations NSB-3 through NSB-7, Phase III Stations NSB-3-R through NSB-7-R, and between one- and two-fold at Phase II Stations MCL-10 and MCL-11, and Phase III Stations S2C and MCL-12-R. Tributyltin concentrations did not exceed the U.S. EPA suggested lower effects benchmark (20 ng Sn/gat 2% TOC) for any stations measured (U.S. EPA, 1997). Similarly, p,p'-DDE concentrations slightly exceeded the ER-L (2.2 ng/g) at Phase I/II Stations NSB-3 through NSB-7. Neither Total PAHs or p,p'-DDE exceeded the ER-M guidelines (40,000 and 22 ng/g, respectively).

Concentrations of metals relative to ER-L and ER-M guidelines generally reflect the trends observed for the organic CoCs. the most impacted areas are in Zones 2, 3, and 3A (stations shown on Figure 1.6-1), particularly for Cu, Pb, Hg, Ni, Ag, and Zn. Comparisons against ER-M values suggest particularly high adverse exposure due to copper at Phase III Stations NSB-2 -R and NSB-4-R (HQ>28), as well as Zn at Phase III Station NSB-4-R (HQ>16).

ER-L Hazard Quotients discussed above for PAH and metal contaminants were summed by CoC class to derive Hazard Indices (HI). Metal HIs showed that the highest values were at stations in Zone 2, particularly Phase III Stations NSB-2-R and NSB-4-R (HI=387 and HI=372, respectively). Generally, the HI for PAHs indicated greatest exposure for stations in Zone 3A, particularly for TRC/BOS Station OS-28

(HI=90), Phase II Station S2B (HI=92), and Phase III Stations MCL-12-R and S2C (HI=63 and HI=112, respectively). Phase III Station NSB-6-R (HI=85) also exhibited PAH exposure equivalent to that observed for Zone 3A stations.

A comparison of sediment organic contaminant concentrations between pre-erosion (Phase I/II) and post-erosion (Phase III) sediment Hazard Quotients was performed to assess whether nearshore sediment erosion related to landfill revetment construction had increased possible CoC exposure to aquatic biota. For Total PAHs, notable increases of relative to the ER-L benchmark were observed for Phase III over Phase II for Stations NSB-6 and MCL-12, while reductions in Total PAH were noted for Stations NSB-3 and S2B (Figure 1.6-1). Total PCBs, when compared to the ER-M benchmark, exhibited increased potential for impacts at Stations and NSB-5). The pesticide p,p'-DDE or tributyltin was not measured in Phase III sediment samples due to relatively low concentrations observed in previous studies.

A similar comparison between previous studies and Phase III sediment HQs for metals relative to the ER-L benchmark revealed increases for mercury at Station NSB-4 and cadmium at Stations NSB-3 and NSB-4. Increased HQs relative to the ER-M were observed for nickel, copper, and zinc at Station NSB-4 and for copper, silver and zinc at Station NSB-2.

Overall sediment Hazard Quotient rankings for selected CoCs were developed for each station based on NOAA ER-L and ER-M guidelines (Long *et al.*, 1995) as follows:

- “-” CoC concentration does not exceed the ER-L value (ER-L HQ<1);
- “+” CoC concentration equals or exceeds the ER-L value (ER-L HQ≥1);

- “++” CoC concentration equals or exceeds the ER-M value (ER-M HQ $\geq$ 1); and
- “+++” CoC concentration exceeds the ER-M value by two-fold or greater (ER-M HQ $\geq$ 2).

*Ecological Exposure Zone-based* exposure rankings were performed as described at the beginning of Section 1.6. In general, HQs are greatest for stations in Zone 2, with somewhat lower HQs for stations in Zones 3 and 3A (Table 1.6-1). With some exceptions for individual sampling stations, remaining Zones 1, 4, 5, and 6, as well as reference Zone 7, generally exhibit comparatively lower CoC exposure conditions.

*Porewater Hazard Quotients.* Concentration of CoCs measured in sediment porewater (metals only) were evaluated against EPA saltwater acute and chronic criteria concentrations. Copper porewater concentrations exceeded the saltwater chronic criteria by two fold at intertidal stations NSB-1, NSB-2, NSB-5 and NSB-7, but not at subtidal stations. Zinc exceeded saltwater acute at Stations NSB-4 and NSB-5 (particularly NSB-5), while nickel exceeded saltwater chronic criteria all intertidal stations except NSB-6. In contrast, the metals generally did not exceed criteria at subtidal stations (Table 1.6-1), except for mercury which exceeded the chronic criteria at all stations including the reference station JCC-D1. Hazard Indices for the above metals suggest highest risk (HI > 15) at Stations NSB-2, NSB-5 and NSB-7, with NSB-1, NSB-3 and NSB-4 demonstrating about two-thirds less risk (HI > 5).

Overall porewater Hazard Quotient rankings for metal CoCs measured in porewater were developed based on EPA Water Quality Criteria - Saltwater Chronic (WQC-SC) and Water Quality Criteria - Saltwater Acute (WQC-SA) criteria (EPA, 1989) as follows:

- “-” = CoC concentration does not exceed the WQC-SC value (WQC-SC  $HQ < 1$ );
- “+” = CoC concentration equals or exceeds the WQC-SC value (WQC-SC  $HQ \geq 1$ );
- “++” = CoC concentration equals or exceeds the WQC-SA value (WQC-SA  $HQ \geq 1$ ); and
- “+++” = CoC concentration exceeds the WQC-SA value by two-fold or greater (WQC-SC  $HQ > 2$ ).

*Ecological Exposure Zone-based* exposure rankings were performed as described at the beginning of Section 1.6. In general, adverse porewater exposure conditions were high for Zone 2, whereas low adverse exposure conditions were observed for Zones 1, 4, and 6 (Table 1.6-1). No apparent risk was observed for Zone 3A and Reference Zone 7.

*SEM Bioavailability.* Simultaneously Extracted Metals (SEM) bioavailability is a measure of the simultaneous and cumulative impact of 5 divalent metals (Cu, Cr, Pb, Ni and Zn) on sediment toxicity. The concentration of SEM is operationally defined by the chemical extraction procedure, which is less robust in comparison to conventional (e.g. strong acid) sediment digestion methods. Because acid volatile sulfides in sediments form stable bonds with metals under anoxic conditions, toxicity of metals is limited when the molar concentration of AVS exceeds that of SEM.

Because sulfides are easily oxidized to sulfates which do not bind metals, the interpretation of metal bioavailability must consider possible scenarios which may control AVS concentrations, including seasonality, but also sample handling and processing artifacts. To aid in this interpretation, three measures of SEM bioavailability in sediments, including SEM concentration, SEM/AVS, and SEM-AVS were assessed. An SEM/AVS ratio of 1.0 has been recommended as a threshold value of potential

metal bioavailability; a value of 0.5 has been conservatively adopted for this ERA to allow for seasonal variation in AVS concentration. An SEM-AVS concentration of 5  $\mu\text{mol/g}$  dry weight was used as this value has been adopted as an approximate threshold for toxicity to amphipods by the EPA National Sediment Quality Inventory. Total SEM concentration was also adopted for this investigation assuming that all AVS could be lost from the sediment.

SEM concentration divided by AVS concentration (SEM/AVS) results revealed several stations with ratios greater than 0.5, including all stations in Zones 1 and 2, most stations in Zone 6 (with the exception of Stations OS-29, OS-30, and S4), Stations NSB-7, M1, and MCL-8, and reference Stations JCC-D1 and JCC-M1. An SEM-AVS concentration of 5  $\mu\text{mol/g}$  dry weight was exceeded for Stations NSB-1, NSB-2, NSB-4, NSB-5 and NSB-7. Finally, SEM concentrations would exceed the SEM-AVS threshold value of 5  $\mu\text{mol/g}$  at Stations NSB-1 through NSB-5, NSB-7 and M3, assuming a total absence of AVS in the sediment.

SEM bioavailability rankings for each of the metrics, as well as an overall SEM ranking, have been applied, as follows:

Indicator-specific ranking:

- SEM concentration:  $\leq 5 \mu\text{mol/g} = \text{"-"}, >5 \mu\text{mol/g} = \text{"+"}$ ;
- SEM/AVS:  $\leq 0.5 = \text{"-"}, >0.5 = \text{"+"}$ ;
- SEM-AVS:  $\leq 5 \mu\text{mol/g} = \text{"-"}, >5 \mu\text{mol/g} = \text{"+"}$

Overall SEM Exposure Ranking:

- "-" = no observed exposure for any indicator;
- "+" = exposure observed in one indicator only;
- "++" = exposure observed in two indicators; and
- "+++" = exposure observed in all indicators.

Based on the above ranking strategy, greatest bioavailability and potential toxicity due to SEM bioavailability are observed in Zones 1, 2, and somewhat reduced bioavailability was observed in Zone 3 (Table 1.6-1). Additionally, low but possible SEM-related toxicity was observed for Zone 6 and reference Zone 7.

*Sediment fecal pollution indicators.* Fecal pollution indicators were measured in sediments as an indicator of potential sewage-related contaminant transport pathways affecting the study area. Evidence of high fecal pollution was observed for intertidal Zone 1 while the lowest concentrations were observed for Zone 3 (Table 1.6-1). Intermediate fecal indicator concentrations were observed for remaining Zones 2, 3A, 4, and 6. Data were not available for Zone 5 and Zone 7. Overall, the data suggest that potential sewage-related pathways for contaminant exposure exist in the study area, perhaps coming from the North, but the trend does not explain the occurrence of the highest CoC exposure conditions observed for Zone 2 and Zone 3.

*Tissue Concentration Ratios.* This section evaluates tissue residues in target species as indicators of CoC-related exposure. CoC exposure was assessed by comparison of site tissue residue concentrations with reference tissue residue concentrations (Tissue Concentration Ratios).

Site vs. Reference Tissue Concentration Ratios (TCRs) were employed to evaluate the potential significance of CoC tissue residues in target species. The analysis involves the comparison of receptor- and analyte-specific tissue body burdens from the McAllister Point Landfill study area stations against the corresponding data for the Jamestown Cranston Cove reference location. Comparisons of site tissue concentrations against reference stations were made only for the same species and analytes. For this analysis, species- and analyte-specific data collected from the reference stations were numerically averaged to yield a single best estimate for the reference-based value. For organics data, tissue concentrations were normalized to

the lipid content of the organism. Site and reference values below the Method Detection Limit (MDL) were not used to calculate TCRs in this analysis.

In the present study, the availability of some species limited the biomass of tissue available for chemical analysis. To augment the data, reference station metals data for hard clams, as well as organic and metal data for mussels, were employed as reported in TRC (1994). The TRC reference station was located on the shore of Conanicut Island station just north of the reference station used in this study (Jamestown Cranston Cove). Data for mummichog fish collected the Jamestown Potter Cove reference location for the Derecktor Shipyard Marine Ecological Risk Assessment (SAIC and URI, 1996) were used as a surrogate for cunner.

For PAHs, the results generally suggest the highest enrichment in lobsters collected from Zone 6, and in blue mussels collected from Zone 3. Metals were elevated in lobsters collected from stations in Zones 4 and 6 (particularly copper), and hard clams collected from Zone 5.

The overall TCR rankings for organic contaminants and metals in target receptors from the McAllister Point Landfill study area as evaluated as follows:

- “-” indicates  $TCR \leq 1$ ;
- “+” indicates  $TCR > 1$ ;
- “++” indicates  $TCR > 10$ ; and
- “+++” indicates  $TCR > 40$ ,

these results are further summarized by zone in Table 1.6-1 using the E/E ranking strategy discussed in the beginning of Section 1.6.

Overall, TCRs indicated that the greatest CoC exposure to target species occurred in Zones 3, 4, and 6 (Table 1.6-1). PAHs and metals were the primary CoC classes of concern, though possible impacts were observed for stations in Zones 2 and 3 due to p,p'-DDE and TBT.

#### 1.6.2. Effects Assessment Summary.

##### *Analysis of Bioaccumulation and Trophic Transfer to Avian Predators.*

Relationships between contaminant exposure and tissue residue concentration for the target species were evaluated through exposure-residue relationships as well as Biota-Sediment Accumulation Factors (BSAFs) for organics, and Bioaccumulation Factors (BAFs) for metals. Linear regression analysis did not generally indicate statistically significant correlations between clam and mussel tissue and sediment CoC concentrations. For BSAF analyses, the data indicate that there was considerable overlap in central tendency about the median BSAF value for a specific contaminant or group of contaminants for all species. BSAF values obtained for bivalves, lobster and fish were similar to those in previous studies. The data suggest a single exposure pathway model appears most appropriate to predict ultimate fate of organic contaminants (i.e. tissues) for the target receptors of concern.

The overall pattern of BAFs for metals was found to fall into four groups relative to the propensity for accumulation into tissues: 1) High (As, Ag, Cd); 2) Intermediate (Zn, Hg, Cu); 3) Low (Ni, Mn, Cr); and 4) Very Low (Pb, Fe, Al). The highly bioavailable metals (As, Ag, Cd) are relatively mobile in the aquatic environment. Because highest BAFs for subtidal species, the result suggest that remobilization of these metals via resuspension or ingestion of offshore sediments is the probable exposure route to target receptors.

The least bioavailable metals (Pb, Fe, Al) have high particle affinities and are not easily digestible, a fact consistent with the observation that the largest differences between depurated and non-depurated bivalve tissues were observed for these metals. Highest BAFs for these metals were identified in mussels inhabiting the intertidal zone suggesting that these metals are not being transported in great quantities far from the source.

Metals exhibiting comparable BAFs among intertidal and subtidal species (Zn, Hg, Cu and, to a lesser extent Ni, Mn and Cr) are those most likely affected by a variety of processes ranging from dissolved-particulate partitioning to internal metabolic regulation. Hence active transport processes are occurring at McAllister Point Landfill which tend to minimize exposure gradients, resulting in comparable exposures for intertidal and subtidal biota.

The potential for adverse effects to avian aquatic predators from the ingestion of contaminated food within the McAllister Point Landfill study area was assessed by comparison of prey Exposure Point Concentrations (EPC) and prey-associated CoC dosage (Dose) to appropriate Toxicity Reference Values (TRV-EPC and TRV-Dose, respectively) using a Hazard Quotient approach as follows:

- 1)  $HQ-EPC = \text{prey EPC} / \text{TRV-EPC}$ ; and
- 2)  $HQ-Dose = \text{prey Dose} / \text{TRV-Dose}$ .

In the above equations, TRV-EPC and TRV-Dose benchmarks are defined as the concentration/ingestion rate of CoCs in prey (mg CoC/kg prey; dry weight) which would result in CoC uptake by the avian predator in an amount *equivalent* to the No Observable Adverse Effects Level. When CoC uptake exceeds the benchmark (i.e.  $HQ > 1$ ), a potential for adverse effects on the receptor is presumed to exist.

For this ERA, a combination of the two approaches (TRV-EPC and TRV-Dose) was implemented to allow better characterization of possible adverse effects due to the trophic transfer of CoCs from prey to the avian aquatic predators. The receptor-exposure pathway scenarios evaluated for herring gull and great blue heron included cunner, blue mussels, lobsters (hepatopancreas and muscle), and hard clams as prey items. In reality, herring gulls and great blue herons are not likely to feed on all of the aforementioned species, but consumption of these prey species by avian aquatic predators has been modeled as part of a comprehensive and conservative approach in the assessment of potential risk, assuming that these prey species are surrogates for other organisms which might be part of the diet of gulls or herons.

As discussed previously, a sediment erosion event at the toe of the McAllister Point Landfill heightened concern about potential change in CoC exposure concentrations. For the avian predator assessment, the possible impacts of this event were assessed through prediction of CoC concentrations in mussels and fish that may occupy the area of erosion and therefore would be the most likely food source for herons and gulls. CoC concentrations for organics and metals were predicted from BSAF and BAF relationships, discussed above.

Apparent adverse effects to great blue heron generally followed that of the herring gull; HQ-EPCs and HQ-Dose values generally suggest that greatest adverse effects for both birds would result from ingestion of copper, lead, zinc and PCBs in cunner and blue mussels at Stations NSB-2-R and NSB-4-R. Lesser impacts were predicted for ingestion of chromium in hard clams at Stations OS-22 through OS-25.

The ranking of HQ-Dose and HQ-EPC data indicating potential adverse effects on avian aquatic predators from prey consumption was as follows:

- “-” =  $HQ \leq 1$ ;
- “+” =  $HQ > 1$ ;
- “++” =  $HQ > 10$ ; and
- “+++” =  $HQ > 20$ .

These results are further summarized by zone in Table 1.6-2 using the E/E ranking strategy discussed in the beginning of Section 1.6.

Table 1.6-2 also provides an overall summary of adverse effects ranking by zone. Results show that CoCs in prey occupying Zone 2 appear to provide the most important CoC-avian receptor pathway of potential concern.

*Toxicity versus CoC Concentrations.* This section evaluates the relationship between CoC sediment and porewater contamination and toxicity to the amphipod (Phase I/II/III studies) and the sea urchin (Phase I/II studies) for co-located samples as available. Although similar analyses of Phase III sea urchin fertilization and larval development success data were not conducted because corresponding chemistry data on sediment elutriates were not available, the data are evaluated here as part of a overall weight of evidence for identifying station locations posing CoC exposure conditions of concern. As a framework for evaluation of the data, the sea urchin endpoint is considered relatively more sensitive to metals than organics, because short exposure times limit the bioavailability of larger organic molecules. In contrast, the amphipod test, being a 10-day exposure, is expected to adequately reflect the bioavailability of both metal and organic CoCs.

Exposure-response relationships between SEM-related measures of divalent metal bioavailability and amphipod survival suggest that some combination of Cu, Zn, Ni, Cd and Pb may be causing toxicity in amphipods at Stations NSB-1, NSB-4 and NSB-5. Although Hg was included in the summation, concentrations of this metal were low (<0.1 uMol/g) and thus would appear to have contributed little, if any, toxicity to the sample. Patterns observed for amphipod toxicity vs. Total PAHs, p,p'-DDE and tributyltins were less suggestive of exposure-response relationships. Possible interactions included PCB-related toxicity at Phase II/III Stations NSB-4 and NSB-5.

As with the amphipod results discussed above, sea urchin fertilization responses were used to evaluate porewater toxicity. Responses were compared only to metal CoC concentrations, since organics analysis of pore waters were not possible due to volume limitations. Results from fertilization success - SEM/AVS comparisons indicate that observed toxicity at Stations NSB-4 and NSB-5 are most likely related to metal exposure (Table 1.6-2). Similar comparisons of SEM metal concentration vs. toxicity also suggest Station NSB-3 might also exert toxicity if AVS concentrations were low.

Comparisons between sea urchin fertilization success and concentration of organic contaminants in McAllister Point sediments did not generally suggest exposure-response relationships. Only Station S2B had reduced fertilization success which could be related to PAH exposure. Similarly, high PCBs and p,p'-DDE at Stations NSB-4 and NSB-5 were correlated with reduced urchin fertilization, but these results are better explained by the metals-related responses. Reduced sea urchin fertilization was also observed at a number of additional stations (JCC-D1, S3, S2B, M1, JCC-S1 and D3; Table 1.6-2) where exposure-response relationships related to metal toxicity were not observed. For two of these stations (S3 and JCC-S1), unionized ammonia concentrations were high and close to the LC<sub>50</sub> value for this compound. Both stations were located near eelgrass beds with organically rich sediments that could provide the source of the ammonia.

The Phase III sea urchin larval development tests with sediment elutriates prepared from core samples revealed high toxicity at Stations NSB-2, intermediate toxicity at Stations NSB-4 and NSB-5, and MCL-12, low toxicity at Station MCL-10, and no toxicity at Stations NSB-3 and NSB-6. The responses agreed well with sea urchin fertilization results for the same samples at the 50% elutriate concentration. Relatively higher toxicity was observed for sea urchin fertilization at the 100% elutriate concentration. This discrepancy was evaluated and found to be partially attributable to suspended particulates in the sample which affected fertilization but not larval development.

The results of amphipod and sea urchin results, when considered collectively, support the conclusion that metals are primarily responsible for observed toxicity at the McAllister Point Landfill Stations NSB-1, NSB-3, NSB-4 and NSB-5. The high toxicity observed at NSB-2 could not be attributed to metals from evaluation of exposure-response relationships. Only porewater Zn concentrations were correlated with observed toxicity in both sea urchin and amphipod tests at Stations NSB-4 and NSB-5. When porewater metals concentrations are expressed in Chronic Water Quality Criteria (CWQC) units, the analysis suggests that Zn, Ni and possibly Cu are above criteria and may contribute to the observed toxicity at Stations NSB-4 and NSB-5. Additional stations exhibiting toxicity for both tests included S2B in which toxicity appears potentially related to PAH concentration. CoC concentrations at remaining stations exhibiting toxicity did exceed ER-L criteria, including Stations JCC-D1 (PAHs), M1 (PAHs, PCBs and Pb), and D3 (Ni) but the findings were not supported by exposure-response relationships.

A summary of the amphipod whole sediment exposures (Phase II and Phase III), sea urchin porewater exposures (Phase II) and sea urchin elutriate exposures (Phase III) are presented in Table 1.6-2. These WoE indicators were interpreted as follows:

- "-" = no significant toxicity relative to control;
- "\*" = statistically significant toxicity relative to control;
- "\*+" = "\*" and low toxicity;
- "\*\*++" = "\*" intermediate toxicity; and
- "\*\*+++ " = "\*" high toxicity.

Overall, laboratory toxicity results indicate the greatest likelihood of adverse CoC exposure in Zone 2, and to a somewhat lesser extent, Zone 3A (Table 1.6-2), suggesting that CoCs are both bioavailable and toxic in these locations. It was noted for Zone 3A, however, that impacts on one of the three endpoints (amphipod survival) were not generally observed. Reduced effects were observed for the remaining zones where toxicity at some stations was occasionally observed, including the reference location (Zone 7).

*Field Effects Measurements.* Visual shoreline observations indicate significant physical habitat disruption due to the presence of solid waste, particularly at Stations NSB-3 and NSB-4. Particles of ash, metal, glass, and iron-stained sediment were very obvious at Stations NSB-3, NSB-4 and NSB-5. In addition, Station NSB-5 was near the open face of the landfill disposal area (opened during capping) and to a seep which may contain waste leachate. CoC concentrations were especially elevated in this area.

Regressions of biotic condition indices for bivalves versus CoC sediment concentrations resulted in non-significant correlations, or in correlations which would otherwise contra-indicate adverse exposure effects (i.e. positive correlations).

Community structure studies included the subtidal zone sediments and both sediments and mussel clumps in the intertidal zone. In both intertidal and subtidal samples, a variety of taxa and life forms existed to indicate that despite physical and chemical stressors, the habitat was stable enough to allow specialized forms to co-

occur. Total species number was not consistently related to changes in habitat, although enhanced abundance of pollution tolerant species (*Capitella capitata*, spionids) was observed at the landfill Station NSB-5 and NSB-6. The increase in relative abundance of the pollution tolerant species at Stations NSB-5 and NSB-6 suggests possible landfill-related effects on benthic community composition.

However, the change in slope from the northern intertidal stations to the middle landfill area may also explain the relative increases in pollution tolerant species at NSB-5 and NSB-6. Low slope areas, such as the middle landfill area, tend to favor organic matter deposition. The fact that pollution tolerant forms are at a competitive advantage in organically enriched sediments may perhaps explain the increase in relative abundance observed at Stations NSB-5 and NSB-6. With the possible exceptions of Ni and Ag, there were no concurrent increases in CoC concentrations from Station NSB-4 to Station NSB-5 which would otherwise account for these trends. Dissimilarly, benthic community structure at subtidal stations was consistent with that found for reference stations. Hence, with the exception of possible impacts at intertidal Stations NSB-5 and NSB-6, changes in benthic community structure due to a landfill effect were not apparent.

Bivalve tissue collected in the McAllister Point study area were analyzed for fecal pollution indicator bacteria to assess the sanitary quality of the marine environment, as well to identify potential transport pathways for CoCs which might adversely impact growth. The station-specific ranking presented in Table 1.6-2 was based on the relative concentrations of total coliforms, fecal coliforms, fecal streptococci and *Clostridium perfringens*.

Overall, the data suggest that potential sewage-related pathways for CoC exposure exist in the study area, but do not explain the occurrence of the highest CoC exposure conditions observed for Zone 2 and Zone 3. Evidence of high fecal pollution

in intertidal Zone 1 and subtidal Zone 3A, while lesser fecal concentrations were observed for Zones 2 and 3 (Table 1.6-2). Relatively low fecal indicator concentrations were observed for Zones 4 and 6 as well as Reference Zone 7.

The overall ranking of field effects indicators employed the E/E ranking strategy discussed in the beginning of Section 1.6; a summary of adverse field effects ranking by zone is presented in Table 1.6-2. Intermediate probability of adverse field effects is apparent for Zones 1, 2 and 3A, low effects are evident for Zones 3 and 7, no effects are presumed for Zones 4 and 6. No data were available to evaluate field effects for Zone 5.

### 1.6.3. Risk Synthesis

The individual Exposure and Effects WoE underlying indicator measures were discussed in the previous sections and summarized in Table 1.6-1 and Table 1.6-2, respectively. As a framework for discussion of risks for various areas of the McAllister Point Landfill study area, the following definitions of ecological risks has been developed for the McAllister Point Landfill Marine ERA:

*Baseline* risk is defined as the probability of adverse exposure and/or ecological effects equivalent to that from contamination and other environmental conditions not associated with the site.

A *Low* probability of ecological risks suggests possible, but minimal impacts based on some of the exposure or effects-based weights of evidence, while impacts are undetectable by the majority of exposure and effects-based weights of evidence. Conditions of low risk probability typically lack demonstrable exposure-response relationships.

An *Intermediate* probability of ecological risk occurs for site conditions falling between high and low probabilities of risk. As such, the intermediate risk probability condition is typically characterized by multiple exposure or effects weights of evidence suggesting that measurable exposure or effects, but not both, are occurring at the site. Typically, quantitative exposure-response relationships are lacking. Intermediate risk probability may also be indicated if the spatial extent of apparent impact is highly localized (e.g., a single station), or if the impact occurs for periods of very limited duration.

Conditions indicating *High* probability of ecological risk occurs when numerous weights of evidence suggest pronounced contaminant exposure and effects, the spatial extent of apparent impact is great, the impact is likely to be persistent over long periods of time, and the available data support demonstrable exposure-response relationships.

As can be seen in the above definitions, a key element to the interpretation of ecological risk in this assessment is the extent to which adverse exposure and effects occur concurrently. Where such concurrence exists, there is strong evidence for a completed exposure pathway between the CoCs and the receptors of concern.

An overall evaluation of exposure and effects WoE is needed to facilitate the risk characterization, just as WoE-specific indicator data were evaluated to determine and carry forward information about each WoE into the summaries of exposure and effects data in Tables 1.6-1 and 1.6-2. The following approach was used to maintain overall consistency with the evaluation method used for the primary WoE:

*Baseline Adverse E/E Probability (B):*

Baseline (-) ranking for all indicators, or low (+) ranking observed for only one indicator;

<i>Low Adverse E/E Probability (L):</i>	Low (+) ranking observed for two or more indicators, or intermediate (++) ranking for only one indicator;
<i>Intermediate Adverse E/E Probability (I):</i>	Intermediate (++) ranking observed for two or more indicators, or high (++++) ranking for one indicator;
<i>High Adverse E/E Probability (H):</i>	Intermediate (++) or greater ranking observed for two or more indicators.

Following the derivation of overall exposure and effects ranking for each zone by the above criteria, the joint probability of exposure and effects is used to presume the probability of risk for each exposure zone, as follows:

- *Baseline Risk:* No greater than Baseline (B) ranking for Exposure and Effects WoE summaries;
- *Low Risk:* No greater than Low (L) ranking for Exposure and Effects WoE summaries;
- *Intermediate Risk:* Intermediate (I) ranking for both Exposure and Effects WoE summaries, or High (H) ranking for one WoE summary and no greater than Low (L) ranking for the other WoE summary; and
- *High Risk:* High (H) risk ranking for one Exposure and Effects WoE summary and Intermediate (I) or High (H) ranking for the other WoE summary.

As discussed previously for the individual WoE ranking, this approach is based on best professional judgement and the risk manager is encouraged to evaluate alternative ranking approaches as it might relate to the general outcome of the risk assessment.

Eight EEZs were identified for the McAllister Point ERA, including: 1) Landfill Intertidal North; 2) Landfill Intertidal Middle; 3) Landfill Intertidal South; 4) Zone 3A; 5) Landfill Subtidal - Nearfield; 6) Landfill Subtidal - Farfield; 7) "Southern Depositional Area"; and 8) the Reference Site. Each of these zones appears to provide a potentially unique habitat for target species, as well as considerable differences in CoC exposure, effects and risks, as discussed below:

*Zone 1: Landfill Intertidal North EEZ.* The exposure and effects WoE summary suggest a high adverse exposure condition but a low adverse effects probability (Table 1.6-3). CoC concentrations in sediment and porewater for Zone 1 stations did not generally exceed sediment benchmarks. In addition, exposure-response relationships between toxicity measures and CoC concentrations were not generally observed although in one instance, SEM metals were elevated and was shown to exhibit exposure-response relationships explaining observed toxicity in *Ampelisca*. Exposure-response relationships were not observed based on comparisons with the sea urchin fertilization test, nor were macrobenthic community structure responses discernable. There was indication of recent sources of fecal pollution in sediments and of the area, possibly originating from Gomes Brook which discharges north of the landfill (or from shorebirds inhabiting the intertidal), such that alternate CoC sources are possibly impacting this area. Low enrichment of CoCs in aquatic biota were evident, but this did not pose a risk to avian predators consuming these organisms. The sediment erosion event did not appear to increase CoC bioavailability for this zone.

Based on the above data, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, blue mussels and fish living in Zone 1 is presumed to be intermediate.

*Zone 2: Landfill Intertidal Middle.* The exposure and effects WoE summary suggest a high adverse exposure condition and an intermediate adverse effects

probability (Table 1.6-3). Sediment-based Hazard Quotients reveal high CoC concentrations in this zone, particularly for PCBs and metals. SEM metals are high, and measured pore water copper and zinc concentrations exceeded the corresponding EPA Acute Water Quality Criteria for these metals. In general, sediment and tissue fecal pollution indicators did not indicate any significant contribution of alternate pollution sources to the area. Results of mussel tissue concentration comparisons of site vs. reference confirm CoC bioavailability of most metals (particularly lead at Station NSB-3) while similar comparisons for organics did not show evidence of enrichment. Avian predators were at high risk from consumption of prey in this zone. Clear, unambiguous exposure-response relationships between high SEM metals and high amphipod toxicity were observed. Porewater concentrations for zinc were more than twice the Water Quality Acute Criteria at Station NSB-5. An increased number of pollution-tolerant species were apparent at Station NSB-5 relative to northern zones; although this trend may be in part related to a habitat change between Stations NSB-4 and NSB-5 which would favor these macrobenthos. Bivalves had elevated tissue residues, which translated into intermediate risks to avian predators. The sediment erosion event resulted in increased CoC bioavailability in this zone.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, blue mussels and fish living in Zone 2 is presumed to be high.

*Zone 3: Landfill Intertidal South.* As with Zone 1, the exposure and effects WoE summary suggest a high adverse exposure condition but a low adverse effects probability (Table 1.6-3). Sediment-based Hazard Quotients generally high adverse exposure conditions, but the associated porewater concentrations for metals were only occasionally above Saltwater Chronic values. SEM bioavailability was high at one of two zone stations (NSB-7) but baseline at the other station (NSB-6), hence intermediate overall. Sediment and tissue fecal pollution indicators did suggest recent sewage-

related contaminants and thus the possible contribution of alternate pollution sources to the area. Tissue Concentration Ratios (TCRs) were high for mussels tissues relative to the reference location but were low for cunner. Avian predators were generally not at risk from consumption of biota inhabiting this zone. Slight toxicity to *Ampelisca* was observed at both stations sampled, but generally no toxicity was observed in porewater or elutriate tests with *Arbacia*. No effects on mussel condition were noted, but benthic community indicators did suggest species shifts in favor of pollution-tolerant forms at NSB-6, but this could be due to the availability of finer-grained sediments.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, blue mussels and fish living in Zone 3 is presumed to be intermediate.

**Zone 3A.** For this zone, the exposure and effects WoE summary suggest both intermediate adverse exposure and adverse effects probabilities (Table 1.6-3). Sediment Hazard Quotients suggest CoCs for two of eight sampling events had exceeded the ER-M by greater than two-fold, two were greater than ER-M and four were greater than the ER-L benchmark. Porewater metals at the one sampled station did not generally exceed criteria, and SEM metals were typically not bioavailable. Some indication of recent fecal pollution to the area was evident, but the limited data for bivalve TCRs suggest CoCs are not being concentrated in tissues to levels greatly above the reference condition. Accordingly, risk to avian predators was low for this zone. Benthic community analyses conducted at one station in this zone (MCL-12) did not suggest adverse effects. Toxicity to *Ampelisca* was not generally apparent, but there were indications of CoC toxicity to *Arbacia* fertilization. In this case, however, there exists uncertainty because of a lack of definitive exposure-response relationships for the porewater test (where matching CoC-toxicity data were available) and possible sediment interference for the elutriate test. Still, the toxicity results, overall, suggest the probability of adverse CoC exposure, although the magnitude of this exposure is

unclear. Also, the limited geographical extent and substrate character (e.g. hard pebble/shell cover) indicates reduced potential for widespread exposure or CoC remobilization to target receptors in the area.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, hard clams, lobster and fish living in Zone 3A is presumed to be intermediate.

*Zone 4: Landfill Subtidal - Nearfield.* The exposure WoE summary for this zone suggests intermediate adverse exposure conditions but a baseline adverse effects probability (Table 1.6-3). Sediment concentrations for stations in this zone exceeded ER-L benchmarks, but did not generally exceed ER-M benchmarks. Porewater concentrations were generally below criteria values, and SEM metals were not typically bioavailable. Some indication of possible alternate CoC sources were suggested from levels of fecal indicators in sediment and tissue residues. CoCs in tissue residues (particularly copper in lobster hepatopancreas) were high relative to reference values. Avian predators were not generally observed to be at risk from prey consumption in this zone. Toxicity was generally not apparent, and no indication of altered benthic community structure could be discerned.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, hard clams, lobster and fish living in Zone 4 is presumed to be intermediate.

*Zone 5: Landfill Subtidal - Farfield.* The exposure WoE summary for this zone suggests a low probability of adverse exposure and a baseline adverse effects probability (Table 1.6-3). Data available for evaluation of risk for this zone consisted entirely of sediment and tissue data collected by TRC (1994). Sediment Hazard Quotients generally exceeded ER-L values. Tissue data for hard clams were slightly

elevated relative to reference. SEM metals were not bioavailable. Avian predators were observed to be at low risk from ingestion of prey in this zone.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, hard clams, lobster and fish living in Zone 5 is presumed to be low.

*Zone 6: "Southern Depositional Area".* The exposure WoE summary for this zone suggests an intermediate probability of adverse exposure but a low adverse effects probability (Table 1.6-3). Extensive sampling in this region occurred during Phase I with largely confirmatory sampling during Phases II and III. Stations in this zone exhibited CoC concentrations which generally exceeded ER-L values. Porewater metals were not generally above WQC criteria, but SEM metals were generally bioavailable. Low levels of fecal pollution indicators were observed in sediments and biota. Tissue concentrations of CoCs in lobster at two sampled locations were highly elevated relative to reference, while hard clams were also enriched in CoCs, but to a lesser extent. However, the nature of CoCs were such that risks to avian predators consuming these biota were low. There did exist evidence of high toxicity to sea urchins during porewater fertilization tests for Station D3, sampled in Phase 1, but this observation was not confirmed in repeat sampling during Phase II. Also, definitive exposure-response relationships were not observed, partly because the observed toxicity was generally not high.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, hard clams, lobster and fish living in Zone 6 is presumed to be intermediate.

*Zone 7: Reference.* The exposure WoE summary for this zone suggests a baseline probability of adverse exposure and a low adverse effects probability

(Table 1.6-3). CoC concentrations were generally below sediment benchmarks, and porewater metals were typically below criteria. Toxicity was observed for *Arbacia*; however, high un-ionized ammonia concentrations due to decomposition of organic matter contained in the eelgrass habitat appear responsible. Sediment fecal pollution indicators suggest recent sources of contamination at the deep station, possibly originating from Carr Creek on Conanicut Island. Macrobenthos species numbers and abundance were low relative to landfill zones. Tissue residues were also low, and associated impacts on avian predators from consumption of reference location mussels, hard clams and fish were also low.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, hard clams, lobster and fish living in the reference zone is presumed to be low.

## 1.7. RISK UNCERTAINTY

The conclusions drawn in this assessment are based on an extensive database of sediment chemistry, biological indicators, and toxicity evaluations, with broad spatial and temporal coverage. The present study provides multiple weights of evidence for assessment of impacts in the vicinity of McAllister Point Landfill. Because a number of conservative indicators are used (e.g. ER-Ls), the estimates of risk are more likely to be overestimates rather than underestimates of true risks. The present study was conducted under a comprehensive Work/Quality Assurance Plan, and data validation has been performed and found to meet the study requirements. Potential errors in the study design and protocols were minimized through peer review and evaluation. Data collection activities were reasonably complete. Thus, it is concluded that overall uncertainty with regard to the accuracy of risk estimations is acceptable.

Figure 1.3-1. Results of Phase III investigations of landfill extent for the McAllister Point Landfill study area.

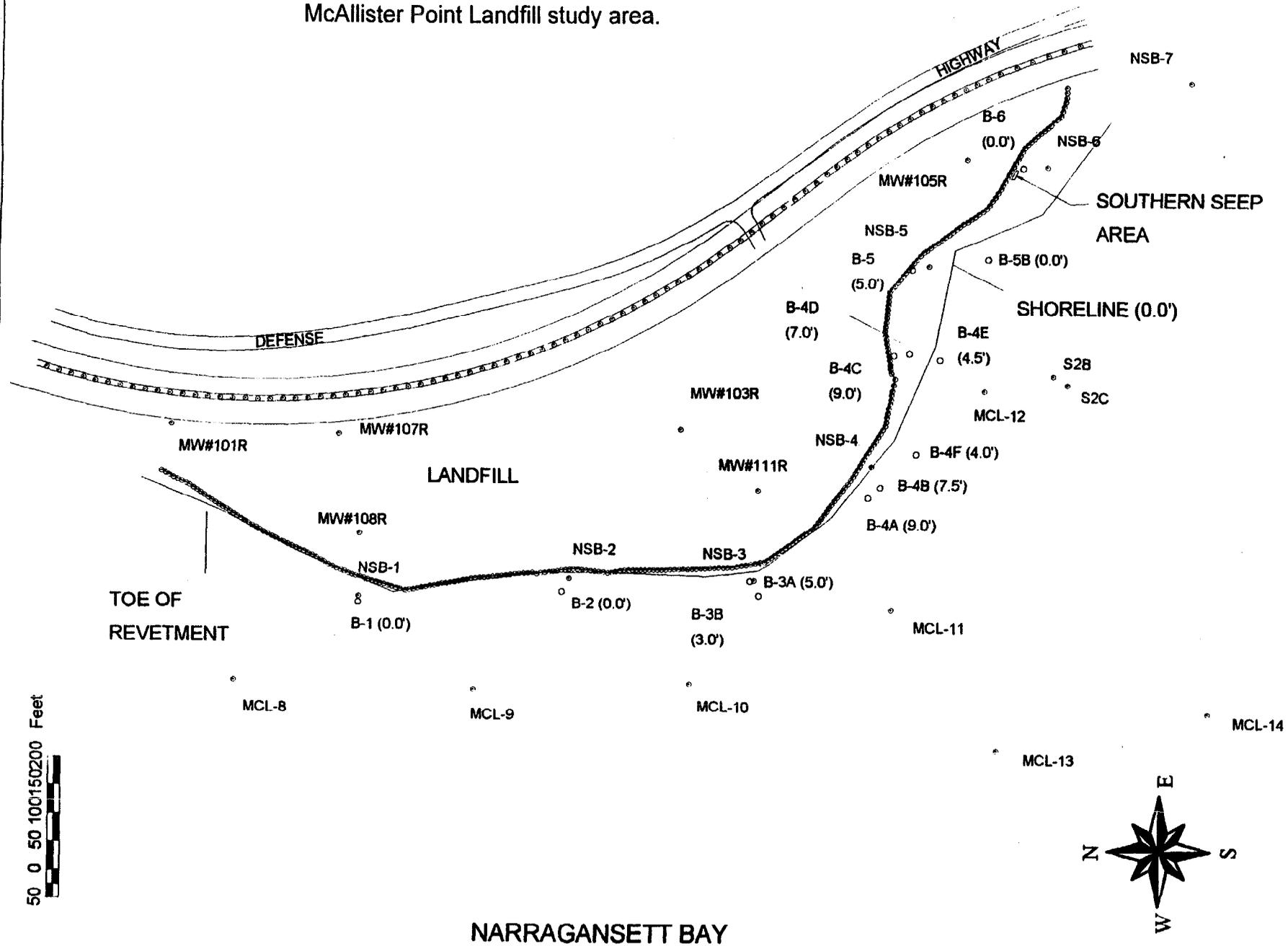


Figure 1.6-1. Ecological Exposure Zones (EEZs) for the McAllister Point Landfill Marine Ecological Risk Assessment.

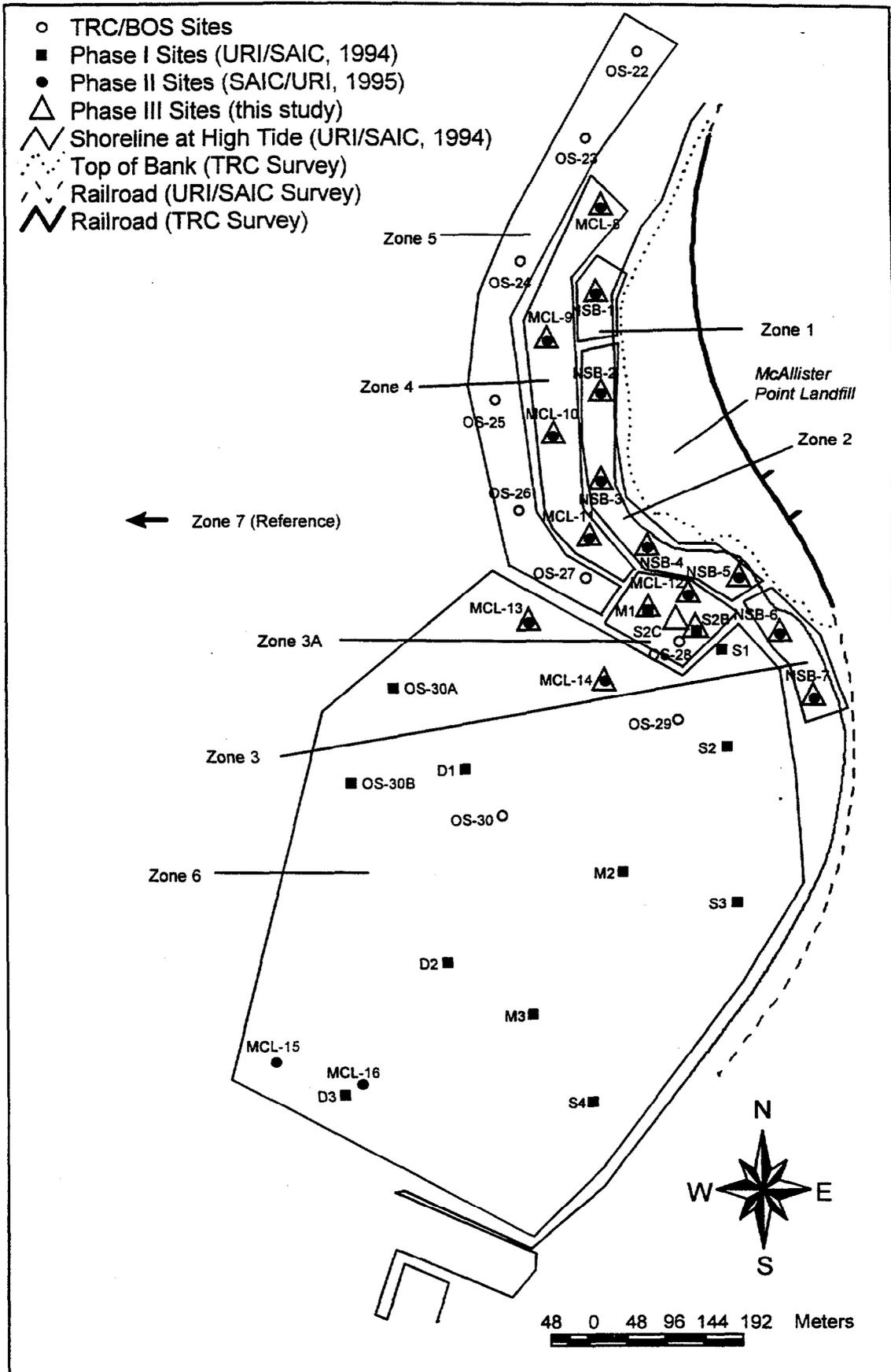


Table 1.6-1. Summary of Exposure-based Weights of Evidence for the McAllister Point Landfill Marine Ecological Risk Assessment.<sup>1</sup>

Zone <sup>5</sup>	Station	Sediment Hazard Quotients <sup>2</sup>		Porewater Hazard Quotients <sup>3</sup>				SEM/AVS <sup>4</sup>		Fecal Pollution Indicators <sup>5</sup>		Tissue Concentration Ratios <sup>6</sup>						
		Sediment Hazard Quotients	Zone Exposure Ranking <sup>7</sup>	Copper <sup>3A</sup>	Zinc	Mercury	Nickel	Zone Exposure Ranking <sup>7</sup>	SEM Bioavailability	Zone Exposure Ranking <sup>7</sup>	Sediment Fecal Pollution Indicator Ranking	Zone Exposure Ranking <sup>7</sup>	Blue Mussel	Hard Clam	Lobster Hepatopancreas	Lobster Muscle	Crab	Zone Exposure Ranking <sup>7</sup>
1	NSB-1	+	+	++	-	+	+	+	+++	+++	+++	++					+	+
	NSB-1-R	+	+	++	-	+	+	+	+++	+++	+++	++					+	+
2	NSB-2	+++	+++	++	-	+	+	+	+++	+++	++	++						++
	NSB-2-R	+++	+++	++	-	+	+	+	+++	+++	++	++						++
	NSB-2-FD-R	+++	+++	++	-	+	+	+	+++	+++	++	++						++
	NSB-3	+++	+++	++	-	+	+	+	+++	+++	++	++						++
	NSB-3-R	+++	+++	++	-	+	+	+	+++	+++	++	++						++
	NSB-4	+++	+++	++	-	+	+	+	+++	+++	++	++						++
3	NSB-4-R	+++	+++	++	-	+	+	+	+++	+++	++	++						++
	NSB-5	+++	+++	++	+++	+	+	+	+++	+++	++	++						++
	NSB-5-R	+++	+++	++	+++	+	+	+	+++	+++	++	++						++
	NSB-6	+++	+++	+	-	+	-	+	-	++	+	++						+++
	NSB-6-R	++	++	+	-	+	-	+	-	++	+	++						+++
3A	NSB-7	++	++	++	-	+	+	+	+++	+	+++							+++
	NSB-7-R	+++	+++	++	-	+	+	+	+++	+++	++	++						+++
	OS-28†	++	+++								++		+					+
	S2B	+++	+++															
	S2B-R	-	-															
	S2C	+++	+++															
	MCL-12	+	+	-	-	+	-	+	-	++								
	MCL-12-R	++	++	-	-	+	-	+	-	++								
4	M1	+	+						+				++					
	M1-R	+	+						+				++					
	MCL-8	++	+	-	-	+	-	+	-		++							+++
	MCL-8-R	+	+	-	-	+	-	+	-		++							+++
	MCL-9	+	+	-	-	+	-	+	-				++	+++	++			
	MCL-9-R	+	+	-	-	+	-	+	-				++	+++	++			
	MCL-10	+	+	-	-	+	-	+	-				++	+++	++			
	MCL-10-R	+	+	-	-	+	-	+	-				++	+++	++			
5	MCL-11	+	+	-	-	+	-	+	-	++								
	MCL-11-R	-	-	-	-	+	-	+	-									
	OS-22†	+	+															+
	OS-23†	++	++															+
	OS-24†	-	-															+
	OS-25†	+	+															+
6	OS-26†	+	+															+
	OS-27†	+	+															+
	MCL-13	+	+	-	-	+	-	+	+	++	++	++	++	++	+++			+++
	MCL-13-R	-	-	-	-	+	-	+	+	++	++	++	++	+++	+			+++
	MCL-14	-	-	-	-	+	-	+	+	++	++	++	++	+++	+			+++
	MCL-14-R	++	++	-	-	+	-	+	+	++	++	++	++	+++	+			+++
	MCL-15	+	+	-	-	+	-	+	+	++	++	++	++	+++	+			+++
	MCL-16	+	+	-	-	+	-	+	+	++	++	++	++	+++	+			+++
	OS-29†	+	+															+
	OS-30†	+	+															+
	OS-30A	-	-															+
	OS-30B	-	-															+
	D1	+	+															+
	D2	+	+															+
	D3	+	+															+
	M2	+	+															+
M3	+	+															+	
S1	-	-															+	
S2	-	-															+	
S3	-	-															+	
S4	+	+															+	
7	JCC-D1	-	-	-	-	+	-	-	+									
	JCC-M1	-	-	-	-	+	-	-	+									
	JCC-S1	-	-	-	-	+	-	-	+									

† - TRC (1994) data.

SEM = Simultaneously Extractable Metals; AVS = Acid Volatile Sulfides; WQC = EPA Water Quality Criteria.

1 - Phase I = Samples collected by URI/SAIC (1994); Phase II = SAIC/URI (1995); Phase III = resampling (1996); TRC = TRC (1994).

2 - Sediment Hazard Quotient analyte-specific rankings: see Table 6.1-1.

3 - Porewater Hazard Quotients: Analytes included for which WQC are available and CoCs were above detection. See Appendix A-2-3.

Rankings: < WQC-Chronic = -; WQC-Saltwater Chronic to Saltwater Acute = "+"; > WQC-Saltwater Acute = "++"; "+++ = > 2 x WQC-Saltwater Acute.

3A - No WQC-SA value for copper: "-" = < WQC-SC; "+" = > WQC-SC; "++" = > 2 X WQC-SC.

4 - SEM Bioavailability Ranking (see Table 6.1-2).

5 - Sediment Fecal Pollution Indicator Ranking: see Table 4.2-4.

6 - Site vs. Reference Tissue Concentration Ratios (TCRs; Table 6.2-1).

7 - Zone Exposure Ranking: "+++ = intermediate (++) or higher exposure observed for two or more indicators, one of which indicates high (+++) exposure;

"++ = intermediate (++) exposure observed for two or more indicators or high (+++) exposure for one indicator; "+" = low (+) exposure observed for two or more

indicators or intermediate (++) exposure for one indicator; "-" = low (+) exposure observed for only one indicator or no exposure for all indicators. See text

in Section 6.0-2. Exposure rankings for stations for which only one indicator observation was available are equal to the indicator observation ranking.

Table 1.6-2. Summary of Effects-based Weights of Evidence for the McAllister Point Landfill Marine ERA.<sup>1</sup>

Zone	Station	Laboratory Toxicity <sup>2</sup>				Field Effects Indicators <sup>3</sup>						Avian Predators <sup>4</sup>				
		Ampelisca Survival <sup>2A</sup>	Arbacia Fertilization <sup>2B</sup>	Arbacia Development <sup>2C</sup>	Zone Effects Ranking <sup>5</sup>	Benthic Community Structure <sup>3B</sup>					Benthic Community Structure Ranking <sup>3C</sup>	Tissue Fecal Pollution Indicator Ranking <sup>3D</sup>	Zone Effects Ranking <sup>5</sup>	Herring Gull	Great Blue Heron	Zone Effects Ranking <sup>5</sup>
						Bivalve Condition Index <sup>3A</sup>	Number of species	Number of Individuals	<i>C. capitata</i> <sup>8</sup>	<i>Spionidae</i> <sup>8</sup>						
1	NSB-1 NSB-1-R	++	.	.	+	.	.	.	.	.	.	.	++	.	.	.
2	NSB-2	.	.	.	+++	.	.	.	.	.	.	.	++	+	.	+
	NSB-2-R	+++	+++	+++		.	.	.	.	.	.			+++	+++	+++
	NSB-2-FD-R	.	.	.		.	.	.	.	.	.			+	+	+
	NSB-3	+	+	.		.	.	.	.	.	.	++		+	+	+
	NSB-3-R	.	+++	.		.	.	.	.	.	.			+	+	+
	NSB-4	+++	+++	.		.	.	.	.	.	.			+	+	+
	NSB-4-R	+++	+++	++		.	.	.	.	.	.			+++	+++	++
3	NSB-5	+++	++	.		.	.	.	.	.	+	++		+	+	+
	NSB-5-R	+++	+++	+++		.	.	.	.	.	+	++		+	+	+
	NSB-6	+	.	.	+	.	.	.	.	.	+	+		.	.	.
	NSB-6-R	.	.	.		.	.	.	.	.	.			.	.	.
3A	NSB-7	+	.	.		.	.	.	.	.	.	++		.	.	.
	NSB-7-R	+	.	.		.	.	.	.	.	.			.	.	.
	OS-28†	+	++	.	+++	.	.	.	.	.	.		++	+	+	+
	S2B	.	.	.		.	.	.	.	.	.			.	.	.
	S2B-R	.	.	.		.	.	.	.	.	.			.	.	.
4	S2C	.	.	.		.	.	.	.	.	.			.	.	.
	MCL-12	.	.	.		.	.	.	.	.	.	+++		+	+	+
	MCL-12-R	.	+++	+++		.	.	.	.	.	.			.	.	.
	M1	.	++	.		.	.	.	.	.	.			.	.	.
	M1-R	.	.	.		.	.	.	.	.	.			.	.	.
	MCL-8	.	.	.	+	.	.	.	.	.	.			.	.	.
	MCL-8-R	.	.	.		.	.	.	.	.	.			.	.	.
5	MCL-9	.	.	.		.	.	.	.	.	.			+	+	+
	MCL-9-R	.	.	.		.	.	.	.	.	.			+	+	+
	MCL-10	.	.	.		.	.	.	.	.	.			+	+	+
	MCL-10-R	.	++	+		.	.	.	.	.	.			+	+	+
	MCL-11	.	.	.		.	.	.	.	.	.	+		.	.	.
	MCL-11-R	.	.	.		.	.	.	.	.	.			.	.	.
6	OS-22†	.	.	.		.	.	.	.	.	.			.	.	.
	OS-23†	.	.	.		.	.	.	.	.	.			+	+	+
	OS-24†	.	.	.		.	.	.	.	.	.			+	+	+
	OS-25†	.	.	.		.	.	.	.	.	.			+	+	+
	OS-26†	.	.	.		.	.	.	.	.	.			+	+	+
	OS-27†	.	.	.		.	.	.	.	.	.			+	+	+
	MCL-13	.	.	.	++	.	.	.	.	.	.			+	+	+
	MCL-13-R	.	.	.		.	.	.	.	.	.			+	+	+
	MCL-14	.	.	.		.	.	.	.	.	.			+	+	+
	MCL-14-R	.	.	.		.	.	.	.	.	.			+	+	+
7	MCL-15	.	.	.		.	.	.	.	.	.			.	.	.
	MCL-16	.	.	.		.	.	.	.	.	.			.	.	.
	OS-29†	.	.	.		.	.	.	.	.	.	+		.	.	.
	OS-30†	.	.	.		.	.	.	.	.	.			.	.	.
	OS-30A	.	.	.		.	.	.	.	.	.			.	.	.
	OS-30B	.	.	.		.	.	.	.	.	.			.	.	.
	D1	.	.	.		.	.	.	.	.	.			.	.	.
	D2	.	.	.		.	.	.	.	.	.			.	.	.
	D3	.	.	.		.	.	.	.	.	.			.	.	.
	M2	.	.	.		.	.	.	.	.	.			.	.	.
	M3	.	.	.		.	.	.	.	.	.			.	.	.
	S1	.	.	.		.	.	.	.	.	.			.	.	.
	S2	.	.	.		.	.	.	.	.	.			.	.	.
S3	.	.	.		.	.	.	.	.	.			.	.	.	
S4	.	.	.		.	.	.	.	.	.			.	.	.	
7	JCC-D1	.	+	.	++	.	.	.	.	.	.		+	+	+	+
	JCC-M1	.	.	.		.	.	.	.	.	.			.	.	.
	JCC-S1	.	+++	.		.	.	.	.	.	.			.	.	.

† - TRC (1994) data.

1 - Phase I = Samples collected by URI/SAIC (1994); Phase II = SAIC/URI (1995); Phase III = resampling (1996); TRC = TRC (1994).

2 - Laboratory Toxicity rankings: 2A - see Table 5.2-1 (Phase I/II amphipod survival), 2B - see Table 5.2-2 (Phase I/II sea urchin fertilization), and 2C - see Table 5.2-6 (Phase III amphipod survival, sea urchin fertilization, and sea urchin larval development).

3 - Field Effects Indicators: 3A - see Figures 6.5-1 and 6.5-2 (condition indices); 3B - Tables 5.3-1 and 5.3-2, Figure 6.5-3 and Figure 6.5-4 (benthic community structure); 3C - Benthic Community Structure Ranking - see text in Section 6.0-2; 3D - Table 5.3-3 (Tissue Fecal Pollution Indicators).

4 - Toxicity Reference Value Hazard Quotient (TRV-HQ); see Table 6.3-4.

5 - Zone Effects Ranking: "+++" = intermediate (++) or higher effects observed for two or more indicators, one of which indicates high (++) effects; "++" = intermediate (++) effects observed for two or more indicators or high (++) effects for one indicator; "+" = low (+) effects observed for two or more indicators or intermediate (++) effects for one indicator; "." = low (+) effects observed for only one indicator or no effects for all indicators. See text in Section 6.0-2. Effects rankings for stations for which only one indicator observation was available are equal to the indicator observation ranking.

Table 1.6-3. Overall Summary of Exposure and Effects-based Weights of Evidence and Characterization of Risk for the McAllister Point Landfill Marine Ecological Risk Assessment.

Zone	WEIGHTS OF EVIDENCE										Overall Risk Probability Ranking <sup>10</sup>
	EXPOSURE						EFFECTS				
	Sediment Hazard Quotients <sup>1</sup>	Porewater Hazard Quotients <sup>2</sup>	SEM and AVS <sup>3</sup>	Fecal Pollution Indicators <sup>4</sup>	Tissue Conc. Ratio <sup>5</sup>	Rank <sup>9</sup>	Laboratory Toxicity <sup>6</sup>	Field Effects <sup>7</sup>	Avian Predators <sup>8</sup>	Rank <sup>9</sup>	
1	+	+	+++	+++	+	H	+	++	-	L	Intermediate
2	+++	+++	+++	++	++	H	+++	++	+++	H	High
3	+++	+	++	+	+++	H	+	+	-	L	Intermediate
3A	+++	-	-	++	+	I	+++	++	+	I	Intermediate
4	+	+	-	++	+++	I	+	-	+	L	Intermediate
5	+	-	-	-	+	L	-	-	+	B	Low
6	+	+	+	++	+++	I	++	-	+	L	Intermediate
7	-	-	+	-	-	B	++	+	+	L	Low

1- Sediment Hazard Quotient Zone Exposure Ranking: see Table 6.6-1.

2- Porewater Hazard Quotient Zone Exposure Ranking: see Table 6.6-1.

3- SEM and AVS Zone Exposure Ranking: see Table 6.6-1.

4- Sediment Fecal Pollution Indicators Zone Exposure Ranking: see Table 6.6-1.

5- Tissue Concentration Ratios Zone Exposure Ranking: see Table 6.6-1.

6- Laboratory Toxicity Zone Effects Ranking: see Table 6.6-2.

7- Field Effects Ranking: Based on results of Condition Index, Benthic Community Structure, and Tissue Fecal Pollution Indicators: see Table 6.6-2.

8- Avian Predator Zone Effects Ranking: see Table 6.6-2.

9- Overall Zone Exposure/Effects (E/E) Risk Probability Ranking (see text Section 7.1):

B = Baseline Risk; L = Low Risk Probability; I = Intermediate Risk Probability; H = High Risk Probability.

B = Low (+) E/E ranking observed for only one indicator or baseline E/E ranking observed for all indicators;

L = Intermediate (++) E/E ranking observed for only one indicator or low (+) E/E ranking observed for two or more indicators;

I = High (+++) E/E ranking observed for only one indicator or intermediate (++) E/E ranking observed for two or more indicators;

H = High (+++) E/E ranking observed for two or more indicators;

10- Overall Zone Risk Probability Ranking (see text Section 7.1):

Baseline = No greater than Baseline (B) ranking for E/E WoE summaries;

Low = No greater than Low (L) ranking for E/E WoE summaries;

Intermediate = Intermediate (I) ranking for both E/E WoE summaries, or High (H) ranking for one WoE and no greater than Low (L) ranking for the other WoE summary;

High = High (H) ranking for one WoE summary and Intermediate (I) or greater ranking for the other WoE summary.

## 1.8. CONCLUSIONS AND RECOMMENDATIONS

Based on the results of the Marine Ecological Risk Assessment for McAllister Point Landfill, the following conclusions and recommendations are put forth for consideration in risk management:

- In the assessment of ecological risks, landfill-related Contaminants of Concern (CoCs) for the middle intertidal landfill area (Zone 2) were determined to pose a high probability of ecological risk to aquatic species (bivalves, lobster and fish) inhabiting this zone. The principal CoCs responsible for this risk were PCBs, PAHs and metals (copper, lead, mercury, nickel and zinc). Seabirds (herring gull and great blue heron) were conservatively estimated to be at high risk due to potential ingestion of CoCs in mussels from this zone. Based on the extent of adverse exposure and effects and demonstrable exposure-response relationships which were observed, the assigned degree of risk is considered unacceptable from an ecological perspective, and thus this area should receive highest priority in the risk management decision process.
- An intermediate probability of ecological risks was assigned to the northern intertidal landfill area (Zone 1), the southern intertidal landfill area (Zone 3), the southern nearshore subtidal area (Zone 3A), the offshore subtidal area (Zone 4), and the "Southern Depositional Area" (Zone 6). In general, the same aquatic receptors and CoCs as observed for high risk stations were of concern, but at lower levels. Seabirds were generally at low risk due to potential ingestion of CoCs in prey from these zones. There existed considerable differences among the individual Weights of Evidence (WoE) within each of these zones with regard to the relative contribution of exposure versus effects indicators, and hence the overall likelihood of risk. Given the apparent indications of adverse exposure or effects (but not both) and a lack of clear exposure-response relationships, the overall risk for these zones is considered acceptable from an ecological

perspective. However, the associated uncertainty of presumed risk for these zones is sufficiently high so as to merit their inclusion as areas of consideration for risk management.

- A low probability of ecological risks was assigned to the remaining Zone 5 and reference Zone 7. Although the data for these stations suggest possible adverse exposure or effects, CoC concentrations were generally low and definitive exposure-response relationships were not observed. Based on these observations, the observed risks for these zones are considered acceptable from an ecological perspective, and relatively low priority should be given to these locations in the risk management decision process.
- A baseline probability of risk was not assigned to any of the zones investigated in this ERA, although Zone 7 was baseline for the overall exposure WoE and thus can be considered generally unaffected by chemical contamination.

Note to reviewers: Comparable text would also be included in the Section 7.0; "Summary and Conclusions".

## 2.0. INTRODUCTION

This report describes the results of a marine ecological risk assessment conducted for the McAllister Point Landfill, which is part of the Naval Education and Training Center (NETC) - Newport, RI. The McAllister Point Landfill at NETC is in the lower East Passage of Narragansett Bay. In November 1989, NETC (including McAllister Point Landfill) was added to the National Priorities List (NPL) of abandoned or uncontrolled hazardous waste sites. A Federal Facilities Agreement (FFA) was signed by the Navy, U.S. EPA, and the State of Rhode Island in March 1992. On September 27, 1993, a Record of Decision (ROD) was signed, which mandated capping of the McAllister Point Landfill as a source control remedy, conducting a study of leachate generation, fate and transport, and performing an ecological risk assessment study (TRC, 1994). This decision was based in part on results of previous assessment of risks to ecological receptors in Narragansett Bay as a result of landfill leachate contaminants of concern (CoCs) (identified as those chemicals which were detected in landfill ground water) under current (uncapped) and remediated (capped) scenarios (Menzie-Cura and Associates, Inc. 1993, as Appendix G in TRC, 1994).

NETC must comply with requirements specified under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), the National Contingency Plan (NCP), and Rhode Island State Statutes. The Federal regulations mandate assessment of the risk of hazardous waste disposal sites on human health and the environment, and identification of appropriate cleanup levels. In 1994, Hallibuton NUS contracted the University of Rhode Island and Science Applications International Corporation to conduct a site-specific offshore ecological investigation and to prepare an offshore ecological risk assessment for McAllister Point Landfill. The purpose of this report is to communicate the results of the assessment of ecological

risks to Narragansett Bay systems posed by the contaminants associated with the landfill.

## 2.1. BACKGROUND

The ERA described in this report has been prepared following the Work/Quality Assurance Project Plan (W/QAPjP) for Narragansett Bay Ecorisk and Monitoring for Navy Sites, referred to herein as the "Master Work Plan", and the site-specific W/QAPjP for McAllister Point included as Addendum A of the Master Plan (URI/SAIC, 1995). This assessment does not consider terrestrial, freshwater wetland, or human health risks associated with the site; separate reports have been prepared to address these issues (TRC, 1994). Rather, this assessment focuses on the impacts of landfill-related contaminants on intertidal and subtidal habitats offshore of McAllister Point and greater Narragansett Bay.

The Master Work Plan and the McAllister Point Addendum collectively provide a thorough description of the approaches and methodologies utilized to conduct the ERA for McAllister Point. The scope of this report is to present the results of the ERA and includes an overview of the sampling and analysis activities conducted in support of the ERA. Complete descriptions of sampling and analytical methodologies are provided in the Work Plan; any deviations from the plan are noted where appropriate in this report and in the QA/QC Appendix C.

This ERA report follows the organization suggested in Eco Update (U.S. EPA, 1991) with appropriate elements from the EPA Region I Supplemental Risk Assessment Guidance for the Superfund Program (U.S. EPA, 1989) and Risk Assessment Guidance for Superfund, Volume II Environmental Evaluation Manual (U.S. EPA, 1989). These guidance documents recommend a "weight of evidence" approach to assess potential

ecological risks. The approach should be based on evaluation of contaminant analytical data relative to environmental benchmarks, direct field observations, and selected field and laboratory studies from the scientific literature.

To guarantee that the required activities are conducted to meet these objectives, the ERA was conducted following general guidance provided by the U.S. EPA (1989, 1992b) and incorporated input provided by U.S. EPA Region I, the State of Rhode Island, and Natural Resource Trustees, representatives of which jointly constitute the Narragansett Bay Ecorisk Advisory Group. The scope of this ERA report includes:

1. Problem Formulation. This involves determining the nature and extent of contamination of offshore (intertidal and subtidal) media associated with landfill sources. Specifically, this activity involves identification of contaminated media, identification of contaminants of concern (CoCs), evaluation of the spatial extent of contamination, identification of the ecological receptors potentially at risk from CoCs, and identification of appropriate assessment and measurement endpoints. The information generated during the Problem Formulation is integrated into a conceptual model, which identifies the possible exposure scenarios and mechanisms of ecological impact associated with the CoCs.
2. Exposure and Ecological Effects Assessments. These assessments include collection of information to quantify chemical exposures and observed or predicted ecological effects resulting from exposure. The Exposure Assessment involves quantification or estimation of the concentrations of CoCs in environmental media in the exposure pathways from source to ecological receptors. The Ecological Effects Assessment involves a combination of toxicological literature review, *in situ* characterizations of the status of receptor species, toxicity evaluations of

exposure media, and modeling exercises to predict the occurrence of adverse ecological impact. Site-specific Exposure and Ecological Effects Assessment activities are determined based on the conceptual model developed during Problem Formulation.

3. Characterization of Ecological Risks. Risk characterization is an integration of the results of the Exposure and Ecological Effects Assessments. This represents a weight of evidence approach involving analysis of CoC concentrations *versus* observations of adverse effects, analysis of CoC bioaccumulation, comparisons of toxicity evaluations with observed ecological effects, comparisons of exposure point concentrations with established standards and criteria for offshore media, comparisons of exposure point concentrations with published information regarding the toxicity of CoCs, and qualitative comparisons of apparent adverse impacts with conditions at reference stations. The results of these analyses are summarized together with information obtained during each study to characterize ecological risks associated with the McAllister Point Landfill.
  
4. Communication of Study Results. Communication of the study objectives, methods, and findings of the ERA is provided in a format which supports informed risk management decisions for the site. Results of weights of evidence are assembled into a risk summary table in order to further facilitate the communication of potential ecological risks in support of risk management decisions.

Based on these guidelines, this ERA presents background information integrated with contemporary data to develop the Problem Formulation (Section 3), Exposure and Ecological Effects Assessments (Sections 4 and 5, respectively), Risk Characterization

(Section 6), Summary and Conclusions (Section 7), References (Section 8), Appendices including original data for Exposure and Effects Assessments (Appendices A and B, respectively), QA/QC Summary Information (Appendix C), and Maps of CoC Concentrations in Sediment and Biota (Appendix D).

## **2.2. PURPOSE, SCOPE, AND OBJECTIVES**

The purpose of this report is to describe the information that was collected to evaluate risks to marine ecological receptors around McAllister Point and greater Narragansett Bay, from contaminants associated with McAllister Point Landfill. The general approach taken in this investigation follows that described in the main body of the Master Work Plan (URI/SAIC, 1995).

The U.S. EPA's ERA Framework and applicable EPA Region I guidance (U.S. EPA, 1989) were used to generate and interpret the data required to complete this risk assessment. The objectives of this ERA are as follows:

- Assess ecological risks to the offshore environments of McAllister Point and Narragansett Bay from chemical stressors associated with the McAllister Point Landfill;
- Develop information sufficient to support risk management decisions regarding site-specific remedial options; and
- Support communication to the public of the nature and extent of ecological risks associated with McAllister Point Landfill.

This ERA builds upon and incorporates findings of previous ERA and RI/FS studies at McAllister Point, and specifically addresses three data gaps remaining from these earlier studies. These data gaps are as follows:

- Need to better assess the chemical exposure to biological populations in surficial sediments adjacent to the landfill site;
- Need to determine the potential migration of contaminants from the landfill to adjacent embayments to the south and west of McAllister Point;
- Need to expand the investigation of ecological risks to endemic populations in Narragansett Bay to include toxicity assessments, organism condition and community structure.

The following sections present and discuss the data requirements and data products of the McAllister Point ERA, including Problem Formulation, Exposure and Ecological Effects Assessments, and Characterization of Ecological Risks.

### **3.0. PROBLEM FORMULATION**

Five principal activities have been conducted in support of the Problem Formulation component for the McAllister Point Landfill ERA:

- Site Description, including characterization of the nature and extent of contamination of offshore media associated with McAllister Point Landfill (Section 3.1);
- Specification of assessment and measurement endpoints (Section 3.2);
- Identification of contaminants of concern (CoCs, Section 3.3);
- Identification of the ecological receptors potentially at risk from site-related CoCs (Section 3.4); and
- Development of a site-specific conceptual model of marine ecological risks associated with the McAllister Point Landfill (Section 3.5).

A summary of sampling and analysis activities related to the ERA effort is also provided (Section 3.6).

### **3.1. SITE CHARACTERIZATION**

The primary objectives of the site characterization are to identify the types, spatial extent and processes affecting marine and estuarine habitats that are present in and around McAllister Point Landfill, as well as the species and biological communities that may be exposed to site-related contaminants. In Section 3.1.1., a complete site history and description of the McAllister Point Landfill is summarized, primarily based on the Phase I Remedial Investigation Report (TRC Environmental Corporation, 1992).

Section 3.1.2. provides a description of the habitats and benthic communities in the area. Section 3.1.3. presents information on the hydrology and geology of the McAllister Point Landfill intertidal zone and adjacent subtidal areas. Finally, Section 3.1.4. presents the results of several contaminant distribution surveys (exclusive of the present ERA investigation). These data are used as background information in the remainder of Problem Formulation.

### 3.1.1. Characterization of McAllister Point Landfill

The location of the McAllister Point Landfill study area and the reference location in Narragansett Bay, RI is shown in Figure 3.1-1. The NETC occupies approximately 1,374 acres on the west shore of Aquidneck Island in Narragansett Bay. The NETC is spread out along nearly six miles of the shoreline in the towns of Newport, Middletown and Portsmouth, Rhode Island.

The McAllister Point Landfill is approximately 11.5 acres and is bounded by Defense Highway and adjacent railway to the east and a stone revetment at the edge of Narragansett Bay in the central portion of the NETC facility (Figure 3.1-2). Prior to capping (discussed below), the site was mounded in its central to north-central portion and was flat at the northern and southern ends and was vegetated with grass, weeds and some small trees. A small, lightly wooded area was present at the northern end of the mounded area. A more mature wooded area was present near the northeastern edge of the site between the railroad tracks and Defense Highway.

The landfill reportedly was in operation for approximately twenty years from 1955 to the mid-1970's. Historically, the landfill received barrels containing liquid wastes, including paints and oils, and at least two transformers containing PCBs, along with domestic wastes. A waste incinerator operated at the site from the late 1950's into the early 1970's, with the incinerator ash residue disposed of on site. On portions of the

site, wastes were placed directly on top of the bedrock. The depth of waste materials at the landfill varies from 3 to 27 feet. The landfill reportedly was closed in 1973 with a three-foot-thick soil cap. This cap varied in thickness from 0 to 4 feet and is discontinuous over the site.

The earliest surveys of environmental contamination at McAllister Point Landfill included the Initial Assessment Study (Envirodyne Engineers, 1983) and the Confirmation Study (Loureiro Engineering Associates, 1986). The studies found that samples of landfill cap material and leachate from springs contained metals, cyanide, phenol, and some organic contaminants. Groundwater samples also exhibited elevated levels of metals.

Based on the above evidence for site-related contamination, onshore Remedial Investigations (RI) were initiated to further characterize the landfill and included ambient air and radiological surveys, soil gas surveys, geophysical surveys, surface soil sampling, test borings, test pits, ground water monitoring well installation and sampling, and leachate spring sampling. Details of these varied investigations are contained in two reports (TRC, 1992; TRC, 1994a). The RI found organic and inorganic pollutants in soil and groundwater at concentrations which exceed established criteria (Tables 3.1-1 and 3.1-2, respectively).

In general, the greatest amount of soil and fill contamination was located in the central portion of the landfill area of the site where significantly elevated concentrations of Volatile Organic Compounds (VOCs), Polycyclic Aromatic Hydrocarbons (PAHs) and metals were detected in subsurface soils. Significant metals contamination was also detected in the ground-water in the central area of the landfill, where large amounts of trash and debris were also apparent.

In the north-central area of the site where an incinerator reportedly once operated, samples of soil/fill mixed with trash/debris indicated high levels of Semi-Volatile Organic Compounds (SVOCs) and metals, and low but measurable levels of dioxins and furans. The ground water in this area also has elevated levels of metals and phthalate contamination.

In the southern portion of the site the fill consisted primarily of construction- and demolition-like debris materials. Elevated SVOCs, metals and petroleum-related contamination levels were found in the fill while Volatile organic compounds (VOCs) and PolyChlorinated Biphenyls (PCBs) were detected in the ground water. Some of the metals (aluminum, iron, lead, manganese, and nickel) were also detected in off-site background soil and off-site upgradient ground water, indicating that their presence at the site may be at least in part reflective of background conditions.

Construction of the landfill cap was initiated during 1995-1996. During the construction of the stone revetment, the visible debris was removed from the shoreline of the landfill, and placed on top of the landfill to be covered later. This debris included concrete, asphalt, scrap metal, bricks and other landfill-type debris. Large items were moved using excavation equipment and trucks, and smaller items were hand-picked and carried to the top of the landfill in trucks. After completion of the revetment, the shoreline consisted of sand, gravel, and cobbles.

Preliminary observations at the site in April 1996 indicated noticeable changes in the intertidal zone including loss of sand from the northern section of the landfill shoreline, and replacement by a "shingle" beach. At the central section of the shoreline, the sand and gravel was also absent, and landfill debris, consisting of wire, metal, concrete, asphalt, glass and other material was visible at low tide. In addition, fill material was visible off-shore of the stone revetment. These observations suggested

that erosion of surface sediment had occurred during winter storms, resulting in the exposure of underlying fill material.

After the revetment construction was completed, additional "Phase III" sampling was initiated to assess the extent of changes in environmental conditions that may affect the present ERA investigation. Studies included intertidal/subtidal measurements of geology (subtidal topography and sediment stratigraphy), chemistry and toxicity testing. Results of chemistry and toxicity testing are incorporated into the present ERA and are discussed in Sections 4 and 5. Full details of the Phase III investigation are provided in the *Technical Memorandum for Phase III Investigations* (Brown and Root, 1996).

### 3.1.2. McAllister Point Habitat Characterization

Two primary investigations of natural habitat and ecological communities in the vicinity of McAllister Point which have been conducted include the bay-wide Narragansett Bay Project studies conducted by Applied Science Associates (French *et al.*, 1992) and the site-specific habitat assessment conducted by TRC/BOS (1994) in support of the Phase I Remedial Investigation for McAllister Point Landfill.

#### 3.1.2.1. Narragansett Bay Project Studies

In 1991, the Narragansett Bay Project contracted ASA to map habitats and natural resources in and around Narragansett Bay. Aerial photo data collected in April 1988 were obtained, interpreted, and translated into ArcInfo Geographic Information System (GIS) format (French *et al.*, 1992).

A variety of habitat types were observed to exist around McAllister Point Landfill, ranging from the upland and landfill areas, to the rocky intertidal in the northern and

central areas and the sand/cobble beach to the south (Figure 3.1-3). North of the site, the intertidal habitat is fringing rock terrace, and habitats offshore of this intertidal are macroalgal. An intertidal sand beach lies south of the artificial supratidal shoreline of McAllister Point, with subtidal depositional sand regions lying offshore. Beyond this area lie small areas of dynamic subtidal sand and fringing sand flats, with an intertidal gravel beach along the southernmost shoreline of the study area. Inland lie small areas of estuarine emergent wetland, surrounded by shrub swamp.

#### 3.1.2.1. McAllister Point Area Habitat Investigations

As part of the investigation, TRC (1994) reviewed available data and performed additional sampling to characterize habitats in the vicinity of McAllister Point. In the sections below, information contained in the TRC (1994) report on threatened and endangered species, terrestrial and nearshore habitat surveys, benthic infauna surveys and finfish surveys are discussed.

*Review of Threatened and Endangered Species.* According to RIDEM's National Heritage Program (RIDEM, 1994), rare plants or animals or ecologically significant natural communities are not present in the vicinity of McAllister Point Landfill. In addition, RIDEM conducted an endangered species survey of several Navy facilities including NETC, Newport in 1989 (RIDEM, 1989). At that time, the potential for any rare species to occur at the NETC was extremely low because of heavy development throughout much of the area. Based on this information, RIDEM concluded that threatened or endangered species are not likely to be of concern at McAllister Point Landfill.

*Terrestrial and Nearshore Habitat Surveys.* Menzie-Cura & Associates, Inc. conducted qualitative reconnaissance surveys on July 23, 1993, and May 9, 1994, to identify habitats and associated wildlife in the vicinity of the McAllister Point Landfill

(TRC, 1994). During the surveys, observations were made on major flora in wetland and upland areas, including bird, amphibian, and mammal sightings or physical evidence of their presence (e.g., nesting sites, tracks).

At the southwestern end of the McAllister Point shoreline, the substrate above mean high water is a mix of sand, gravel, stone, shells, broken glass and asphalt. Some of this area is covered by dune vegetation: beach pea, beach rose, seaside goldenrod, and American beach grass. The wrack line on the beach consists of several algal species such as rockweed (*Fucus*), eel grass (*Zostera*), and allochthonous debris. Within the wrack were numerous beach fleas, whelk (*Busycon*) egg cases, green crab carapaces and signs of several other marine invertebrates that may inhabit the nearshore waters. Below mean high water, the substrate is rock and cobble with rockweed, hollow green weed, and some barnacles (*Balanus*), periwinkles (*Littorina*) and blue mussels (*Mytilus edulis*) attached. At high tide, the beach can be only 10 feet in width while at low tide it may be as much as 50 feet in width in areas. Birds observed in the bay included a pair of Canada geese adjacent to the site, and several marine birds (double-crested cormorant, great black-backed gull, and herring gull).

*Benthic Infauna Survey.* Benthic samples were collected or observations were made on benthic infauna (organisms that live within the sediment) at stations offshore of McAllister Point Landfill and also Jamestown Cranston Cove reference locations (Figure 3.1-1).

Benthic habitats off of the McAllister Point Landfill exhibited an onshore-offshore zonation. Nearest to the shore, the sediment was reported to consist of a sandy bottom which probably reflects the higher wave energy (TRC, 1994). The sediments became increasing more silty with distance off shore (100 to 250 meters). At 100 to 250 meters offshore, the sediment surface had a layer of slipper shells (*Crepidula fornicata*). There appeared to be an in-shore band of primarily empty shells and an offshore band of live

*Crepidula*. Observations of scuba divers revealed that the *Crepidula* was being preyed upon by broad bands of starfish and that the starfish bands occurred along the margins separating the “dead” *Crepidula* beds from the “live” *Crepidula* beds. Surface layers of sediment were oxidized throughout the sampling area. Oxidized sediments are relatively rich in oxygen, ferric oxides, nitrates, and nitrites, and support the bulk of the benthic animals.

Sediments in the Cranston Cove reference area consisted of silty sand (TRC, 1994). Shallow water areas of Cranston Cove supported eelgrass beds (*Zostera*), and beyond the eelgrass beds the sediments exhibited a layer of *Crepidula* shells. In deeper water (15 m), the sediments also consisted of silty sand. Reference stations exhibited some of the highest densities and species richness values. No obvious relationships of species density or diversity were observed with respect to concentrations of COCs in the sediment.

Epifauna and epiflora surveys at the Jamestown Cranston Cove (JCC) reference location revealed, overall, 14 species of algae, 1 seed plant, and 32 species of fauna, including 6 fish species (TRC, 1994). Similar numbers of species were observed at stations offshore of McAllister Point. For both sites, the epifloral community was primarily composed of red and green algae. The red algae, *Argardiella tenera* and *Polysiphonia* spp., and the green algae, *Ulva lactuca* and *Codium fragile*, were the most frequently observed species. *Zostera marina* (eelgrass), a seed plant, was found only at the nearshore end of the transect near the JCC reference location.

Seven phyla were represented in the epifauna of both the reference and site stations; these included sponges, corals, mollusks, segmented worms, arthropods (jointed-leg animals including the crustaceans), starfish, and chordates (tunicates and fish). The most frequently observed epifaunal species were the mollusk, *Crepidula fornicata* (slipper shell), a terebellid worm, the crustaceans, *Pagurus longicarpus* (long-

clawed hermit crab) and *Libinia emarginata* (common spider crab), the starfish, *Asterias forbesi*, and the bottom fish, *Pseudopleuronectes americanus* (winter flounder).

Lobsters (*Homarus americanus*) were observed only at the McAllister Point Landfill station which had significant rocky substrate. Based on these limited data, TRC (1994) concluded that there were no obvious differences between the communities observed at the landfill site versus those observed at the reference sites which could be related to proximity to the McAllister Point Landfill. However, contaminant concentration effects were not adequately accounted for. Thus, the survey results do not present worst-case exposure scenarios and thus may underestimate the maximum potential impacts associated with landfill-related exposures.

*Finfish Survey.* The Rhode Island Department of Environmental Management (RIDEM, 1993) conducted a finfish survey, in which species likely to be present in areas of Narragansett Bay near McAllister Point Landfill are identified. This study included monthly sampling for target species. The RIDEM has identified the winter flounder as a species commonly found in Narragansett Bay.

### 3.1.3. Hydrology/Marine Geology of Nearshore McAllister Point

Knowledge of the hydrological and geological characteristics of the intertidal and subtidal habitats of Narragansett Bay adjacent of McAllister Point Landfill is required to adequately understand the processes governing the transport and fate of site-related contaminants to Narragansett Bay receptors. In the sections below, groundwater flow, shoreline stability, shoreline change, regional sediment lithology, localized sediment lithology and stratigraphy, and subsurface landfill topography/vertical extent are discussed.

*Groundwater flow.* The site elevation is approximately 15 to 35 feet above mean low water level. The general site topography slopes in an east to west direction. The

western edge of the site drops steeply to the shoreline. Ground water flow direction at the site is also from east to west, toward Narragansett Bay. During periods of heavy rain, pooled water forms in a small depression in the north-central portion of the site. At low tide, springs have been observed discharging from the bottom of the landfill bank into the Bay. A leachate fate and transport analysis conducted for McAllister Point landfill (TRC, 1994a) provides an estimate of about 4.67 m<sup>3</sup> of groundwater per tidal cycle.

*Shore line stability.* The location of McAllister Landfill suggests that the intertidal zone is highly exposed to storms, and as with other beach systems, may undergo seasonal changes in onshore and offshore sediment transport. The southern landfill coastline is expected to follow the general storm beach cycle depicted in Figure 3.1-4. In calm weather, a berm may form as shown in Figure 3.1-4A, but after a major storm, the berm may be eroded and moved to offshore bars (Figure 3.1-4B). In the intervals between storms, normal shore processes are expected to move the sand stored in offshore bars back onto the beach in the form of swash bars (Figure 3.1-4C). After a long storm-free interval (Figure 3.1-4D), the beach will return to the general mature beach profile (Figure 3.1-4A).

The coastline of the northern McAllister Point study area differs from somewhat the mature beach discussed above. The northern coastline north of McAllister Point is very rocky, lacks a dune zone, and has a steep drop off to deep water. The steep slope, being characteristic of the coastline north of McAllister Point, prevents material that is eroded during stormy weather from being re-deposited during calm weather. Thus, eroded material is most likely lost forever to deeper water.

Natural cycles of beach erosion occur on several time scales. A time series of beach profile volume from Charlestown Beach, a typical Rhode Island beach, is shown in Figure 3.1-5. The typical seasonal cycle in beach erosion is for the berm to erode

during large fall and winter storms and to be rebuilt in relatively calm weather during the spring and summer months. Cycles that span several years are strongly related to the frequency of large storms. Beach profile volume increases during calm intervals, and decreases during stormy intervals. In addition, the potential rise of global sea level due to global warming may cause a long-term trend of decreasing beach profile volume at most beaches. This long-term trend is not typically evident in the relatively short time series such as shown in Figure 3.1-5.

*Shore line change analysis.* Shoreline change measurements (French *et al.*, 1992) for the interval 1938-1988 indicate that the shoreline in the vicinity of McAllister Point increased significantly in extent during the interval 1938-1975 (with a maximum increase of 2.7 m/year) when the landfill was active. The shoreline was stable (i.e., changed less than 0.1 m/year) during the interval 1975-1988, after landfill operations had ceased.

As part of the present investigation, more detailed analysis of shoreline change in the immediate vicinity of McAllister Point was conducted to determine whether localized areas were being eroded. Aerial photo data was obtained and analyzed covering the period from 1938 to 1992. Results of the analysis show substantial seaward transgression between 1951 and 1975, relative to its approximate present day position (Figure 3.1-6). Indications of transgression or regression were not evident from 1980 to 1992, indicating that, within the resolution of the analysis ( $\pm 5$  m), significant erosion of the landfill toe had not occurred during that period.

*Regional sediment lithology.* The lithography of surficial sediments in the lower East Passage of Narragansett Bay are illustrated in Figure 3.1-7. Nearshore sediments in the McAllister Point Landfill study area are generally characterized as silty sand, while sediments offshore of McAllister Point are a mixture of sand, silt, and gravel. Similarly, nearshore sediments to the reference location at Jamestown Cranston Cove

are silty sand. Offshore sediments northeast of Jamestown Cranston Cove are characterized as sandy silt, while offshore sediments east and southeast of Cranston Cove are silty sand. Sediments in the interior region of the east passage are generally clayey silt north of McAllister Point, changing to sand-silt-gravel, and finally silty sand approaching southern Conanicut and Aquidneck Islands.

*Localized sediment lithology.* A side-scan survey was conducted by URI in 1994 in order to further clarify the localized sediment lithography offshore of McAllister Point. A map produced from a mosaic of sidescan images is presented in Figure 3.1-8. The spatial distribution of several distinct sediment types is indicated by the gray scale shading of the map. The lighter shading and "speckled" character of the shallow water zone in the study area indicates the presence of sand and abundant rocks. The darker shading with increased water depth indicates the presence of finer sediment. A distinct north-south oriented tongue of finer sediment (darker shading) is a major feature in the study area. Sediment farther offshore is coarser than the tongue of finer material and is "pock-marked", exhibiting circular deformities on the sediment surface. The "pock-marks" are often interpreted as areas where methane gas escapes from the sediments. A final distinctive feature is the washboard pattern of the pile of sediment located just to the north of the Coddington Cove breakwater. This feature is likely to be man-made and is probably related to either construction of the breakwater, or to dredging and dumping of material from Coddington Cove.

*Sediment stratigraphy.* The characteristic layering of sediment types with depth in the sediment, or sediment stratigraphy, in the McAllister Point Landfill study area and a reference location (Jamestown Cranston Cove) were inferred from lithological descriptions of sediment cores and measurement of magnetic susceptibility (MS). The MS measurements were performed because previous studies (e.g. Corbin, 1988) have found that MS is an indicator of land-derived soils (and accordingly, fine-grained sediments) because of their naturally high concentrations of certain metals (e.g. iron

and aluminum). Conversely, low MS occurs in sediments dominated by gravels and sands, and thus indicates low loading of soil-derived materials.

Nine piston cores were collected from the McAllister Point Landfill area, ranging in length from 37-73 cm. All of the piston cores exhibited both an increase in magnetic susceptibility and a finer-grained sediment in the upper section, followed by a decrease in these values to the surface and terminated in gravel layers (Figure 3.1-9). The thickness of the upper surface unit was generally 25 cm or less throughout the study area. The maximum thickness of fine-grained sediment is approximately 35 cm at Station S2B.

*Characterization of subtidal landfill extent.* Subtidal topography was measured to compare elevations of specific points of the shoreline under current conditions to the elevations of those points prior to revetment. The results of the topography survey were compared to the baseline topography survey performed by TRC Environmental Consultants in 1994 which established the mean low water mark for the area (Figure 3.1-10). This comparison determined that up to 1.72 vertical feet of surficial sediment had eroded seaward of the revetment between Stations NSB-2 and NSB-5.

Thirteen borings were performed seaward of the stone revetment to determine the presence and thickness of the fill material and to evaluate other subsurface conditions. Landfill material was found up to nine feet thick seaward of the bottom of the stone revetment in the area of NSB-2 to NSB-5 with identifiable fill being observed up to 50 feet from shore (Figure 3.1-10).

The instability of the shoreline after construction of the revetment can be readily explained by an increase in erosional energy at the foot of the landfill caused by large waves reflecting off the revetment and onto the intertidal zone without impediment from large boulders and other stabilizing materials that were previously present. The

sediment eroded from the shoreline to the north of McAllister Point is probably permanently lost to deep water, whereas the sediment lost from areas to the east and south of McAllister Point appear to be located in an offshore bar. Some of this sediment may be redeposited back onto the beach during calm intervals.

#### 3.1.4. Contaminant Distribution Surveys

Data from prior chemical contamination surveys are discussed in this section as part of the Problem Formulation, which included the selection of Contaminants of Concern (CoCs). Surveys of chemical contamination prior to the present ERA Investigation conducted in the vicinity of McAllister Point Landfill include onshore studies by the U.S. Army Corps of Engineers (USACE), and U.S. Navy-sponsored offshore investigations by TRC Environmental Corporation and Battelle Ocean Sciences (TRC/BOS). Data was also collected in 1994 by the University of Rhode Island Graduate School of Oceanography and Science Applications International Corporation (Quinn *et al.*, 1994); for the purposes of this report, these data are considered part of the present ERA investigation conducted in 1995-1996, and are discussed as appropriate in Section 3.6 and Sections 4 through 6.

##### 3.1.4.1. USACE Investigations

A preliminary screening assessment of metal contamination was performed along the McAllister Point shoreline by the U.S. Army Corps of Engineers during 1987 (USACE, 1988). Maximum concentrations of metals found at the site sediment sampling stations are summarized in Table 3.1-3. All metals measured were found to exceed sediment benchmarks.

#### 3.1.4.2. TRC/BOS Investigations

The RI offshore investigations conducted by TRC/BOS (TRC, 1994) included sediment and biota sampling in support of a marine ecological risk assessment. The findings of the chemical sampling program are presented in TRC, 1994. Sampling stations included nearshore (NS) and offshore (OS) stations in the immediate vicinity of the landfill (Figure 3.1-11). In the nearshore environment, sample compositing in groups of three stations (e.g. NS-1/2/3, NS-4/5/6 etc.) were performed with sediment and bivalve collections, preventing detection of maximum concentrations. However, the available data provide some indication of the nature and magnitude of sediment and biota contamination.

*Sediments.* Maximum concentrations of metals, PAHs, two SVOCs and PCBs found for nearshore samples are summarized in Table 3.1-3. All metals, PCBs, and most PAHs measured were found to exceed sediment benchmarks.

Results of sediment metal contaminant analyses from the TRC/BOS offshore (OS) stations (TRC, 1994a) are presented in Table 3.1-4. Results indicate that concentrations of Cu, Hg, Ni and Pb in sediments from stations located west of McAllister Point slightly exceed the ER-L guidelines (Long *et al.*, 1995). The one notable exception was observed for Ni at Station OS-23, where the concentrations exceeded the benchmark by about five-fold.

Results of sediment organic contaminant analyses from the TRC/BOS investigation are presented in Table 3.1-5. All stations exceeded the ER-L guideline of 22.7 ng/g for Total PCBs (Long *et al.*, 1995); with the highest concentrations observed at Station OS-26 (81.4 ng/g) and OS-28 (73.9 ng/g). All stations except OS-23 and OS-24 exceed the ER-L guideline of 4,022 ng/g for Total

PAHs (Long *et al.*, 1995). The highest concentrations were observed in sediments from Stations OS-28 (44,054 ng/g) and OS-30 (16,042 ng/g).

*Biota.* The TRC/BOS investigation also included sampling of mussels in the near shore and hard clams offshore from a subset of stations occupied for sediment collections. As with sediment analyses, compositing of mussel samples was performed across stations for NS samples, preventing detection of maximum concentrations, but is summarized here to provide some indication of the nature and extent of tissue contamination (Table 3.1-6). Because established criteria for evaluation of ecological effects of tissue residues are not available, hence the primary basis of assessment is the comparison of site concentrations against reference concentrations. Results show that for all measured contaminants, site-related tissue residues exceed the average concentrations determined for reference locations.

Results of similar analyses of clam samples obtained from offshore stations is shown in Table 3.1-7. All PAHs but pyrene, and all metals and PCBs were elevated for at least one station relative to reference.

### **3.2. ASSESSMENT AND MEASUREMENT ENDPOINTS**

The Master Work Plan includes a target analyte list which was developed in response to the regulatory requirements of the RI/FS for NETC Newport and NCBC Davisville, and through recognition of a number of potential chemical stressors associated with past disposal practices and other Naval operations (Table 3.2-1). The list was based on those chemical contaminants detected during previous offshore (i.e., Quinn *et al.*, 1994; Battelle, 1994) and on-shore investigations (e.g., TRC, 1994), and includes both metals (Hg, Ni, Zn, Cu, Cd, Cr, Pb, and Hg ) and organic compounds (PAHs, PCBs, butyltins, and organochlorine pesticides (OCPs)). This list also includes

several other metals (aluminum, iron, manganese, and nickel) which were detected in the site soil and ground water samples. TRC (1994) reported that these common elements were also detected in off-site background soil and off-site upgradient ground water, indicating that the presence of these metals at the site may be at least in part reflective of background conditions.

The list also reflects current understanding of those chemicals which are both of toxicological importance and persistent in estuarine systems. It encompasses selected potentially toxic chemicals which may serve as indicators of human activity (although for different uses, e.g. PCBs) and whose discharge into the environment has been enhanced through industrialization (NOAA, 1991).

In keeping with the requirements of the RI/FS process, and based on the potential ecological effects of the chemical stressors (identified above), a suite of assessment and measurement endpoints were identified as important in the ecological risk assessment. As indicated in Table 3.2-2, these focus on the vitality of pelagic, epibenthic, and infaunal communities in Narragansett Bay which are expected to occur in the vicinity of the McAllister Point Landfill.

Several measurement endpoints are also identified in Table 3.2-2 as indicators of potential impacts on the assessment endpoint/receptors (Table 3.2-2).

The measurement endpoints are used to evaluate the occurrence of, or potential for, adverse ecological effects, while exposure point measurements were employed to evaluate exposure conditions. As shown in Table 3.2-2, these exposure point measurements include chemistry-measurements made in environmental media (water, sediment, pore water, and biota), as well as geochemical attributes of exposure media which may influence the availability of contaminants to receptors. The measurement endpoints were selected based on their relevance to:

- The assessment endpoint and receptors of concern, their relevance to expected modes of action, and effects of CoCs;
- Determination of adverse ecological effects;
- Availability of practical methods for their evaluation; and
- Their usefulness in extrapolating to other endpoints.

Most of these measurement endpoints have been used in other studies, and have proven to be informative indicators of ecological status in marine and estuarine systems with respect to the CoCs identified as important in this assessment. Many serve a dual purpose in that they provide information relevant to two or more assessment endpoints.

The exposure point measurements include fecal pollution indicators, which are microbial organisms whose abundance is measured as the concentration of the organism per unit of matrix (e.g., no./ml, no./g wet tissue). These organisms are released into the environment via discharges of human and/or animal feces, or improperly treated sewage effluent (Cabelli, 1978). As such, fecal indicators reflect potential contaminant migration pathways and other indirect stresses caused by co-mingled contaminants in waste streams, and/or other undesirable ecological changes associated with fecal pollution (e.g. nutrient-induced sediment organic enrichment and anoxia, and altered ecological function due to shifts in species composition).

Benthic community data is used to provide an indication of not only chemical stress but also physical stress due to disturbance caused by anthropogenic material present in the nearshore habitat zones of the landfill. The type and abundance of species present which are of known pollution tolerances and substrate preference can

be used to distinguish landfill effects. These measurement endpoints will be used as an additional weight-of-evidence in the effects assessment component of the risk characterization summary.

The protocols and methods used to evaluate measurement endpoints and exposure point measurements are discussed further in Section 4.0.

### **3.3. CONTAMINANTS OF CONCERN**

Proposed Contaminants of Concern (CoCs) have been identified for this investigation using a rationale which links the source (McAllister Point Landfill) to potential marine receptors around McAllister Point Landfill and Narragansett Bay through plausible exposure pathways. In this approach, frequency of detection, range of concentration, and elevation relative to minimum effects benchmarks and, for metals only, reference concentrations are evaluated for chemicals detected in offshore sediments.

Benchmarks are numerical criteria or guidelines which establish chemical concentrations presumed to be protective of biological systems. For derivation of CoCs in this ERA, site sediment concentrations are of primary consideration as sediments are the major reservoir for CoC constituents. Nationally recognized benchmarks for sediments include the Apparent Effects Threshold (AET; U.S. EPA, 1989a), Effects Range-Low and Effects Range-Median (Long *et al.*, 1995), and Equilibrium Partitioning-based Aquatic Life criteria (EqP-AL; U.S. EPA 1989b, Adams, Kimerle and Barnett, 1992). The AET approach uses data from matched chemistry and biological effects measures, and is the concentration of a selected chemical above which statistically significant biological effects are expected to occur (U.S. EPA, 1989a). Effects Range-Low (ER-L) and Effects Range-Median (ER-M) are benchmarks representing the 10th

and 50th percentiles, respectively, of ranked chemical concentrations (predicted or measured) at which biological effects were observed. The Equilibrium Criteria-Aquatic Life Approach (Adams, *et al.*, 1992) predicts effects in porewater for non-ionic organic contaminants based on the water quality benchmark, accounting for partitioning between dissolved and particulate phases. Each benchmark has advantages and disadvantages as well as differing degrees of applicability for various chemical groups.

For this ERA, the lowest of the matrix-specific benchmarks was used as the screening value for the particular compound (Table 3.3-1). Note that in most cases, the NOAA ER-L is the minimum benchmark value. For chemical constituents lacking sediment criteria, reference sediment concentrations were used as the basis of comparison. Data used for the evaluation include sediment chemistry results obtained from 46 stations obtained from TRC (16 station groups; Figure 3.1-11 as well as URI/SAIC Phase I (13 stations) and Phase II (17) investigations (discussed in Section 3.6, below).

Results of this screening process for the development of the marine sediment CoC list is summarized in Table 3.3-2. Frequency of detection was calculated as the total site samples analyzed, representing samples with detected concentrations. The range of concentrations reported for site data excludes non-detected values. One-half of Sample Quantitation Limits were substituted for non-detects when calculating mean of site and reference station data. The 95% upper confidence limit was calculated according to standard statistical procedures (Snedecor and Cochran, 1984), assuming a one-tailed distribution (i.e. only data exceeding the upper 95% confidence limit are of interest). Where the 95% UCL was greater than the site maximum concentration, the maximum concentration was used to screen against benchmark or reference data. For organic contaminants lacking benchmarks, site concentrations were compared against reference concentrations.

Thirteen of twenty-six PAH analytes (including calculated sums for low and high molecular weights and total PAHs) were found to exceed benchmarks where comparisons were possible. Three PAHs (acenaphthylene, benzo(b+k)fluoranthene and naphthalene) and two pesticides (aldrin and hexachlorobenzene) did not exceed benchmarks. For metals, maximum concentrations in bulk sediments exceeded background concentrations. For organics lacking benchmarks, all analytes exceeded background concentrations. Frequency of detection was > 5% for all analytes except aldrin. Because aldrin was detected only once, the 95% UCL could not be calculated; therefore, the site maximum concentration was used to screen against benchmark or background.

As a result of this screening process, all target analytes except for aldrin, were included as CoCs for the ERA. The analytical results used in this ERA were consistent with those of earlier studies (e.g., TRC, 1994) with respect to the specific classes of compounds which are elevated in the marine sediments adjacent to McAllister Point (see Section 3.1). It should be noted that this list is conservative in that the screening procedure involved maximum contaminant concentrations and conservative benchmark concentrations. Final consideration of CoCs for offshore exposure media will be made following completion of the Exposure Assessment (see Section 4.0 of this report).

#### **3.4. RECEPTORS OF CONCERN**

Identification of ecological systems/species/receptors of concern (hereafter collectively termed "receptors of concern") involved evaluations of the importance of the receptor to the ecology of McAllister Point and Narragansett Bay, its sensitivity to stressors associated with the site, and its aesthetic, recreational, and commercial importance as a natural resource of Narragansett Bay. The site characterization for McAllister Point identified a number of estuarine systems and habitat types (Section

3.1.4). The nature of chemical stressors originating from the McAllister Point Landfill suggests that several ecological receptors are potentially at risk, including:

- Nearshore habitats directly adjacent to landfill areas;
- Pelagic communities, including plankton and fish;
- Infaunal benthic communities in sediment depositional areas;
- Soft- and hard-bottom epibenthic communities; and
- Commercially, recreational, and/or aesthetically important natural resource species.

Added to this list are ecological systems involving critical habitats, such as eelgrass beds, bird rookeries, and unique spawning areas. Although French *et al.* (1992) provide a bay-wide perspective of habitat types, the lack of information concerning critical habitats in immediate association with the landfill site at McAllister Point represents a data gap which is addressed in this study.

The identification of estuarine systems and habitats potentially at risk from the McAllister Point Landfill provides a natural progression to the selection of target receptors of concern for this ecological risk assessment (Table 3.4-1). The rationale for selection of these receptors includes:

- **Blue mussel (*Mytilus edulis*)** - This species is a locally abundant and ecologically important filter-feeding bivalve found in intertidal and subtidal habitats. It is an important food source for birds, fish, starfish, and occasionally humans. Blue mussels are surrogates for epibenthic species in the intertidal environment, where they are potentially exposed to water-borne and particulate-bound contaminants. Blue mussels may also serve as surrogate species for pelagic species when collected from mid-upper water column (i.e. deployment on mooring floats).

- Cunner (*Tautoglabrus adspersus*) - This species is a locally abundant and ecologically important estuarine fish which feed opportunistically upon both animals and plants, and has limited home range due to territorial behaviors. It is an important food source for birds and other fish, and is a surrogate for other pelagic fish species potentially exposed to water-borne and bulk sediment contaminants.
- Winter flounder (*Pseudopleuronectes americanus*) - This species is locally abundant, as well as an ecologically and economically important fish species. It feeds upon benthic organisms and has wide ranges of exposure range due to migratory behavior. It is an important food source for birds, other fish, and humans. Flounder represent demersal fish species potentially exposed to water-borne and bulk sediment contaminants. Toxicity exposure information for fish, except for direct contact exposure, is scarce. This species was observed inshore near McAllister Point Landfill (RIDEM, 1993).
- Lobster (*Homarus americanus*) - This species is locally abundant, and an ecologically and economically important subtidal crustacean which feeds opportunistically as a scavenger. It is an important food source for fish and humans. The lobster represents an epibenthic species potentially exposed to water-borne and bulk sediment contaminants.
- Hard clam (*Mercenaria mercenaria/Pitar morrhuana*) - These morphologically and ecologically similar subtidal bivalve filter feeders are locally abundant, and are ecologically and economically important. They are important food sources for birds and occasionally humans. Hard shell clams represent infaunal species potentially exposed to bulk sediment and pore water contaminants.
- Benthic community - The benthic community (including sponges, corals, mollusks, segmented worms, arthropods (including crustaceans), starfish, and chordates (tunicates and fish)), is an ecologically important, potentially rich assemblage of species with numerous life histories and feeding strategies. It is an important food source for birds, fish, and benthic and epibenthic invertebrates. The benthic community is potentially exposed to contaminants in bulk sediments, pore water, and the water column.

- Great Blue Heron (*Ardea herodias*) / Herring Gull (*Larus argentatus*) - These species are local avian aquatic predators which feed upon invertebrates and fish. The heron represents a carnivore in the food web and is representative of other principally piscivorous, wading shorebirds (e.g., snowy egret, *Egretta thula*) that may occur on site. This species is important to both the local aquatic ecology and the larger ecosystem. Herring gulls are common to the area and display an omnivorous feeding habit.

Many of these receptors are important resource species for Narragansett Bay, but also they can be considered surrogate receptors for larger groups of species. For instance, the hard clam, *Mercenaria mercenaria*, is an important commercial species for Rhode Island, as well as an indicator species for infaunal bivalves in general. However, as discussed below, not all of these species occurred at all of the sampling stations. For example, nearshore highly weathered habitats associated with the intertidal zone adjacent to the landfill are unsuitable for hard clams.

Stressors introduced to the bay may indirectly affect avian receptors. For example, bivalves contaminated with chemicals may be consumed by shorebirds, resulting in direct or indirect biological effects. For this reason, avian target receptors of concern are also included in Table 3.4-1.

### 3.5. CONCEPTUAL MODELS

Conceptual models are developed to provide a framework for hypotheses concerning how a given stressor might cause ecological impacts on receptors of concern (U.S. EPA, 1992b). Four models have been developed for this assessment using a tiered strategy where models in earlier tiers are more general and inherently carry greater uncertainty, to the more complex Fourth Tier models which have greater complexity and certainty for the specific pathways being evaluated. In the process of

further refinement of models in subsequent tiers, hypotheses are retained or rejected based on existing knowledge of contaminants and receptors of concern.

The initial three tiers describe stressor origin, transport, fate, and effects at different spatial and temporal scales: 1) the general north to south gradient of chemical contamination in Narragansett Bay, 2) initial release and transport of site-specific CoCs to the bay from the McAllister Point Landfill and other NETC sites, and 3) longer-term transport, fate, and effects of those CoCs. The Fourth Tier models include specific receptors and stressors as identified in Section 3.3 and 3.4, respectively.

#### 3.5.1. First Tier Model

The First Tier of the conceptual model describes the general down-bay, higher-to-lower gradient in stressor concentration described earlier (Figure 3.5-1). Although many sources contribute to this gradient, and local sources may influence specific stressor concentrations anywhere in Narragansett Bay, this model suggests that contaminant concentrations in the immediate vicinity of Navy disposal areas should be evaluated within the context of the lower Bay so that extent and significance of Navy disposal areas on the ecology of the Bay can be determined. It is assumed in this model that there are no ecologically significant stressors which are more concentrated in the south than in the north. As a result of this model evaluation, a reference station located opposite to McAllister Point which occupies a similar down-bay environment is appropriate to identify baseline ecological conditions without landfill-related influences.

#### 3.5.2. Second Tier Model

The Second Tier of the conceptual model describes the local release of constituents from the landfill site at McAllister Point and other NETC sites into greater Narragansett Bay (Figure 3.5-2). The first hypothesis framed by this model is that

CoCs are being transported from land-based sources to adjacent coves and Narragansett Bay, predominately *via* surface and ground water (including seeps) routes, although transport of chemical pollutants bound to soil and dust particles also may occur.

The geographical configuration of McAllister Point is such that it is exposed to the main flow of tidal currents in the bay, and hence reduces sediment deposition immediately adjacent to the landfill. Accordingly, the circulation results in a hydrographic continuity between the landfill intertidal environment and greater Narragansett Bay. Areas to the south of McAllister Point may experience a longer residence time due to more restricted circulation and thus sediments may have higher concentrations of CoCs than those released directly into greater Narragansett Bay from locations further north along the landfill. Thus, a localized, steep gradient in contaminant concentrations would be expected, with the highest levels occurring in areas immediately adjacent to the landfill.

The hypothesis that there are alternate, ecologically significant transport pathways for Navy-related stressors associated with the landfill other than those discussed above is rejected based on extensive study of surface water and groundwater transport pathways (TRC, 1994a).

### 3.5.3. Third Tier Model

The Third Tier of the model describes details of the aquatic behavior of contaminants hypothesized to exert ecological effects within the McAllister Point/Narragansett Bay system (Figure 3.5-3). The model arrows indicate that the short-term behavior of contaminants in the water column depends on their solubility, degradation rates, and sorption to particulate matter. Bound contaminants may be transported with the current in association with particles, but may also settle to the

bottom in localized depositional areas, such as that suspected to the southwest of the landfill. Individual molecules may remain in a dissolved state or will adsorb and desorb in a dynamic fashion, maintaining an apparent equilibrium relative to sorption state. Dissolved contaminants are transported to other parts of the estuary by prevailing current patterns.

Once on the bottom, local currents may result in bedload transport of sediment, resulting in a further redistribution of the contaminants. Subsequent deposition of uncontaminated particles may bury earlier settling particles, and eventually remove them from contact with ecological systems. Chemical-specific partitioning dynamics will occur in the sediments and interstitial (pore) waters in response to the geochemical conditions (e.g., redox potential) of those sediments. Contaminants may be available to biological systems in the water column, pore water, and surficial sediments, resulting in direct toxicological effects and/or biological uptake and transfer through food chains.

Based on this generalized conceptual model, ecosystems potentially at risk are hypothesized to include nearshore habitats, pelagic, benthic, and epibenthic communities, and natural resource species. In addition, stressor partitioning dynamics suggest that risks to receptors should be highest in nearshore/intertidal areas adjacent to the landfill site, and that the assessment should focus on CoCs associated with depositional sediments. Stressors which conform to this model of contaminant behavior include metals, organic contaminants such as PAHs, PCBs, butyltins, and OCPs.

#### 3.5.4. Fourth Tier Models

The description of stressor dynamics suggests risks to the aforementioned systems to be highest in areas adjacent to McAllister Point Landfill. Although risks to other ecological systems present in the Narragansett Bay area cannot be dismissed,

this conceptual model focuses the assessment on ecosystems considered to be directly influenced by depositional sediments near the landfill.

The initial three tiers describe the origin, transport and fate of stressors at different spatial and temporal scales. To complete the model, receptors and stressors specific to the McAllister Point Landfill are added in the fourth and final tier, which describes exposure pathways (from source to receptor) hypothesized for the site.

The Fourth Tier conceptual models describe hypothesized exposure pathways relating CoCs to the receptors of concern identified in Table 3.4-1. These models were developed for receptors by ecological habit (pelagic, epibenthic, infaunal and avian aquatic predator), and their respective exposure pathways are illustrated in Figure 3.5-4 through Figure 3.5-7. Measurement endpoints directly evaluating the effects of CoCs on avian aquatic species are not included in this study. However, an evaluation of the potential impacts to this species group from ingestion of prey organisms (mussels and cunner) hypothesized to be part of the exposure pathways to the predator is characterized through measurement of the spatial distribution and residue concentration of the food source. Hence, relevant issues for this trophic group with regard to the ERA framework are addressed from this perspective.

Illustrated in Figures 3.5-4 through 3.5-7 are the routes of CoC transport from terrestrial sources, through intermediate sources (runoff, soils), to the proximal source of exposure, and to receptors. These proximal sources become the exposure points in the Exposure Assessment (Section 4.0). Also illustrated are the measurement endpoints which will be evaluated in the Ecological Effects Assessment (Section 5.0).

### 3.6. SAMPLING AND ANALYSIS SUMMARY

This section describes the sources and types of data used for the Ecological Risk Assessment. Sample locations and summary of samples collected along with the rationale for selection is summarized in Section 3.6.1. A brief description of sampling and analysis protocols is included in Section 3.6.2. Detailed sample inventory and analysis information is presented in Appendix A-1-3.

#### 3.6.1. Sample Location and Collection Summary

Data used for the assessment include selected (i.e. non-composited) data from TRC (1994) as well as the URI/SAIC Phase I, Phase II and Phase III investigations. A summary of data used in the present ERA is presented in Table 3.6-1.

*TRC/BOS Data.* As discussed in Section 3.1, nearshore sediment and biota samples were deemed unsuitable for incorporation into the present ERA because of a compositing strategy which would tend to obscure true CoC distribution and concentration patterns. As a result, only data from offshore (OS) stations were used (Figure 3.1-11), consisting of bulk sediment, Simultaneously Extractable Metals/Acid Volatile Sulfides (SEM/AVS), and tissue residue chemistry for hard shell clams. The apparent data gap for nearshore samples was address in the URI/SAIC Phase II sampling, discussed below.

*URI/SAIC Phase I Data.* In August 1994, scientists from the Graduate School of Oceanography, University of Rhode Island, and Science Applications International Corporation, Narragansett, Rhode Island, collected surface sediment samples from stations offshore and south of McAllister Point, NETC (Figure 3.6-1) and three reference stations from Cranston Cove off Conanicut Island in Narragansett Bay (Figure 3.1-1). The purpose of this investigation was to characterize the embayment

area located primarily to the south of the landfill, referred to as the Southern Depositional Area (SDA) for magnitude and extent of chemical contamination which may be landfill-related. These samples were analyzed for inorganic contaminants, organic contaminants and toxicity. Results of this study are presented in conjunction with Phase II results in Sections 4 and 5.

*URI/SAIC Phase II Data.* The purpose of the Phase II data collection and analysis activities was to fill the data gaps in the information base as discussed in Section 3.1. Sampling was needed to acquire chemistry and toxicity data for surficial sediments in the area adjacent to the landfill, to obtain similar data for offshore areas to the south and west of the site, and to gather biological data to assess the condition of potentially impacted receptors. Measurements of organic and metal contaminant concentrations in sediment and organisms, and studies of porewater metal concentrations and SEM/AVS ratios, were performed in conjunction with toxicity, biological condition, and community analysis studies to assess the potential impact of the landfill on the biota.

Phase II station locations are shown in Figure 3.6-2. Collections of surface sediments were completed at all 17 stations from approximately mid-March through June 1, 1995. Both grab samples and deep core samples were collected at offshore Stations MCL-8 through MCL-15 and at Station JCC-M1 (the reference station at Jamestown Cranston Cove, at approximately 5 m water depth). An additional core sample was taken at Station S2B, sampled previously in the URI/SAIC Phase I investigation. Sediment samples were collected from a series of intertidal nearshore (NSB), and subtidal offshore (MCL) locations off of the McAllister Point Landfill, and at three reference sites along a depth-transect in-Jamestown Cranston Cove (JCC). Stations NSB-1 through NSB-7 correspond to the nearshore zone of the McAllister Point Landfill.

Stations MCL-8 through MCL-12 were positioned to fill the data gap between the nearshore stations and the TRC/BOS offshore stations. Stations MCL-13 and MCL-14 were positioned further offshore of Stations MCL-11 and MCL-12 to fill a spatial data gap between sites earlier found to have contamination. Stations MCL-15 and MCL-16 are located to the south of McAllister Point near the Coddington Cove breakwall to further investigate potential anthropogenic material as identified in side-scan sonar data.

The reference site was located in Cranston Cove on Jamestown Island (JCC), due west of McAllister Point (Figure 3.1-1). This site is at approximately the same latitude as the landfill along the north-south Narragansett Bay contamination gradient, and thus is an appropriate reference site for assessing baseline contaminant impacts in the absence of NETC activities.

*URI/SAIC Phase III Data.* Phase III data collection activities included resampling of the intertidal and nearshore subtidal stations for post-erosion conditions as determined by surficial chemical contaminants and associated toxicological effects (Figure 3.6.3). Measurements of organic and metal contaminant concentrations in sediment, and studies of elutriates metal concentrations and SEM/AVS ratios, were performed in conjunction with toxicity studies to assess the potential impact of post-revetment erosion on the biota. Upon identification of several stations noting increased chemical concentrations, additional core samples were collected and analyzed for sediment chemistry (PCBs, PAHs, metals) and associated toxicological effects (elutriate toxicity to sea urchin fertilization and larval development).

### 3.6.2. Sediment and Biota Sampling and Analysis Protocols

In the sections that follow, a brief discussion is presented on collection and analysis methods for chemical, geotechnical and biological endpoints. A complete

description of the methods and QA/QC procedures is contained in the Master Work Plan (URI/SAIC, 1995). Detailed QA/QC information is presented in Appendix C.

#### 3.6.2.1. Sample Collection Methods

*Sampling platform.* Sampling was done from three research vessels as well as from shore. For relatively shallow stations (< 3 meters of water), a 7-meter pontoon boat and a 6-meter support motorboat owned by the URI Graduate School of Oceanography was used for sampling. For deeper stations, the 20 meter URI Ocean Engineering Department research vessel CT-1 (Fred Pease, Captain) was used for sampling. The research vessels were moored at the Navy facilities in Coddington Cove when not being used for sampling activities.

*Sediment Collections.* The surficial sediment (upper 0–6 cm and 0–2 cm for intertidal (NSB) and offshore (MCL) stations, respectively) was collected with titanium scoops to obtain chemical and toxicological data on the most represent recently deposited materials. Subsurface sediments (from >10 cm in depth) were also collected by piston coring at subtidal offshore stations to enable evaluation of the contaminant distribution in subsurface sediment layers. The offshore stations surficial sediment was obtained from an undisturbed Van Veen grab sample was collected using a clean titanium scoop. The surface material was composited in a 12-liter, pre-cleaned polyethylene bucket, stirred with a titanium stirrer for approximately 30 seconds, and then subsampled into pre-cleaned containers for organic and inorganic chemistry, SEM/AVS and toxicity studies. Approximately 2-3 grab samples were required to obtain a 3-liter composite sample. For nearshore samples, intertidal sediments were collected within about 1 hr of low tide by scooping about 0.5 m<sup>2</sup> area from a location above the low tide mark.

After each sample, the collection scoop was rinsed with deionized water, 1:1 nitric acid, and methanol with a final rinse of deionized water between grabs. Field rinsate blanks of the scoop water were collected and analyzed. The grab sampler was washed-down with seawater between stations. Samples were stored on ice during collection and at -20°C upon return to the laboratory.

Piston cores were used to take deep ( $\approx 1$  m) cores. A standard piston corer, also known as the biological corer, was used. The cores were transported in the vertical position on ice to the lab for storage at 4°C until logging and sectioning. Sectioning was completed within the required holding limits and sectioned sediment samples were stored at -20°C until chemical analysis. (It was not possible to obtain a core at Station MCL-8 because the sediments of the site consist of  $\sim 10$  cm or less of sand and gravel overlying or interspersed between gravel and large rocks). In general, two depths per core sample were analyzed, such that the complete analysis suite consists of three vertical measurements (surface + 2 depths), from which vertical contamination gradients were discerned.

*Biota Collections.* Biota sampling for chemical analysis was conducted at all surface sediment sampling stations, and spatially coincided with sediment sampling to the maximum possible extent. Target species at the nearshore stations were the blue mussels and fish while at offshore stations, hard clams and lobster, were collected as identified in Table 3.6-1. Nearshore bivalves were removed from hard substrates, scrubbed free of sediment, and placed whole into polyethylene bags inside of coolers. Upon return from the field, subsamples for chemistry and condition analyses were taken and frozen at -20°C as described in the Master Work Plan. A subset of bivalves were depurated (held in clear seawater for 24 hours) prior to freezing in order to assess the significance of gut contents on tissue residues. For blue mussel collections, depurations were completed for samples collected from NSB-1 and NSB-3.

Depurations for hard clams were performed for organisms obtained at Stations MCL-10, MCL-11 and MCL-12.

Fish samples were successfully collected using minnow traps deployed at Stations NSB-1, NSB-3, NSB-4, and NSB-6 (Table 3.6-1). No fish were available at the reference location; data for this endpoint were taken from a mummichog fish collection at Jamestown Potter Cove (SAIC/URI, 1996). Fish samples were placed whole into polyethylene bags inside of coolers. Upon return from the field, subsamples for chemistry and condition analyses were taken and frozen at -20°C as described in the Master Work Plan. Lobsters were collected using lobster traps baited with locally caught fish deployed at Stations MCL-9, MCL-10, MCL-13 and MCL-14, and at the deep reference station (JCC-D1). Upon collection, the lobsters were placed whole into polyethylene bags inside of coolers and returned to the laboratory for processing as described in the section below. This latter group of samples required a sampling effort over an extended period (3 weeks).

#### 3.6.3.2. Sediment, Tissue, and Porewater Chemical Analyses

*Sediments.* The concentrations of selected metals, PCB congeners, pesticides, PAHs and butyltins were determined from surface and core sediment samples following NOAA Status and Trends procedures as prescribed in the Work Plan (refer to Table 3-2 of Master Work Plan). In addition, the concentrations in these sediments of Simultaneously Extracted Metals (SEM) and Acid Volatile Sulfides (AVS) were determined.

*Tissues.* Tissue analytes included the same suite as determined in sediments. Shell and exoskeletal material were not analyzed for any species. Bivalve and fish tissue were frozen whole after collection and analyzed whole. Samples of bivalves from the collection were selected at random and tissues excised at the organic or inorganic

lab depending on the analysis. Lobster specimens were resected alive, immediately following euthanasia, to obtain separate tissue groups (muscle, hepatopancreas). In addition, the lipid content of the biota tissue was determined and used in bioaccumulation factor calculations.

*Porewater.* Interstitial (pore) water metals were measured in surface sediment samples utilizing the vacuum extraction method of Winger and Lasier (1991). Duplicate sample preparations were prepared for pore water toxicity and metals analyses. Approximately 100 ml of pore water was obtained, within a 24 h period after collection, from sediment held at 4°C. Total and unionized ammonia concentrations were determined for the samples to support interpretation of toxicity testing results (Hampton et al., 1977, Bower and Holm-Hansen, 1980).

*Grain Size/Total Organic Content.* Percentages of sand, silt, and clay in sediment samples were determined for each station. Samples were first treated with dilute acid for removal of carbonates and organics, and then sieved using the Elzone Model 180XY particle size analyzer. Estimation of the organic carbon content was accomplished by first drying a sediment sample, combusting the sample for 1 hour at 550°C, and then determining the weight lost on ignition at 550°C. Multiplication of the weight lost by ignition at 550°C by the factor 0.44 provided an estimate of the organic carbon content of the sediment sample.

*Condition Indices.* Condition indices (CI) were evaluated for all bivalve species, and fish were inspected for external evidence of pathological damage (fin rot, gill lesions, etc.). In addition, non-chemistry sites for blue mussel collections at Stations NSB-8 through NSB-11 were sampled to obtain a more complete spatial picture of distribution, abundance and condition for this species. Methods for CI determinations followed procedures recommended by Lawrence and Scott (1982).

*Benthic Community Structure Analyses.* Quantitative samples for benthic community structure analysis were taken at the 16 stations at McAllister Point and at three reference stations at the Jamestown Cranston Cove Reference Site. Measured parameters included species richness and dominance and the number of opportunistic forms present. Identifications were carried out to the species level. Sampling and counting techniques closely followed those used in the EPA Environmental Monitoring and Assessment Program and in the benthic infauna survey of McAllister Point carried out by Menzie-Cura & Associates in August 1993 (TRC, 1994). At each location, two 400 cm<sup>2</sup> Van Veen grab samples were obtained and sieved to 0.5 mm. Organisms were removed, identified and counted. Additional box core samples were obtained at Stations NSB-1 through NSB-7 and MCL-8 through MCL-14, as well as the three reference locations (JCC-S1, JCC-M1, and JCC-D1), and used for benthic infaunal analysis.

*Fecal Pollution Indicators.* Total and fecal coliforms (including *E. coli*), fecal streptococci and enterococci, as well as *Clostridium perfringens* spores, were enumerated in marine animal tissues and sediments using the most probable number method (U.S. EPA, 1995).

*Toxicity Testing.* All surface grab samples were evaluated for bulk sediment and pore water toxicity using the amphipod (*Ampelisca abdita*) 10-day acute test and the sea urchin (*Arbacia punctulata*) fertilization test, respectively. Elutriate toxicity analyses were performed on selected samples. The elutriate was prepared as a 1:4 dilution of whole sediment followed by centrifugation (USACE/EPA, 1992). A complete description of these test methods are contained in *Technical Memorandum for Phase III Investigations* (Brown and Root, 1996).

Figure 3.1-1. Location of McAllister Point Landfill study area and reference locations in Narragansett Bay, RI.

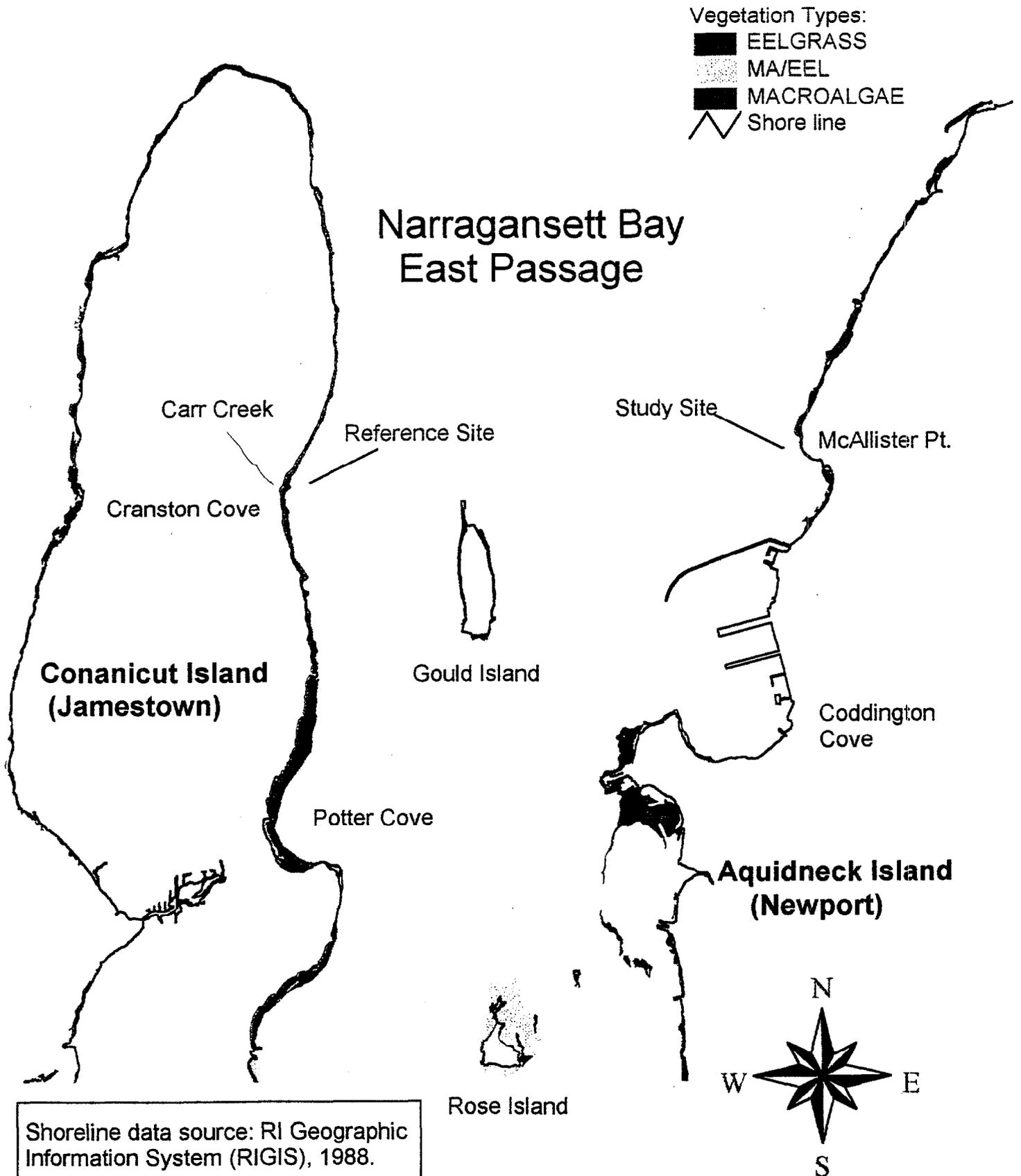


Figure 3.1-2. Location of McAllister Point Landfill study area and adjacent shoreline of Narragansett Bay, RI.

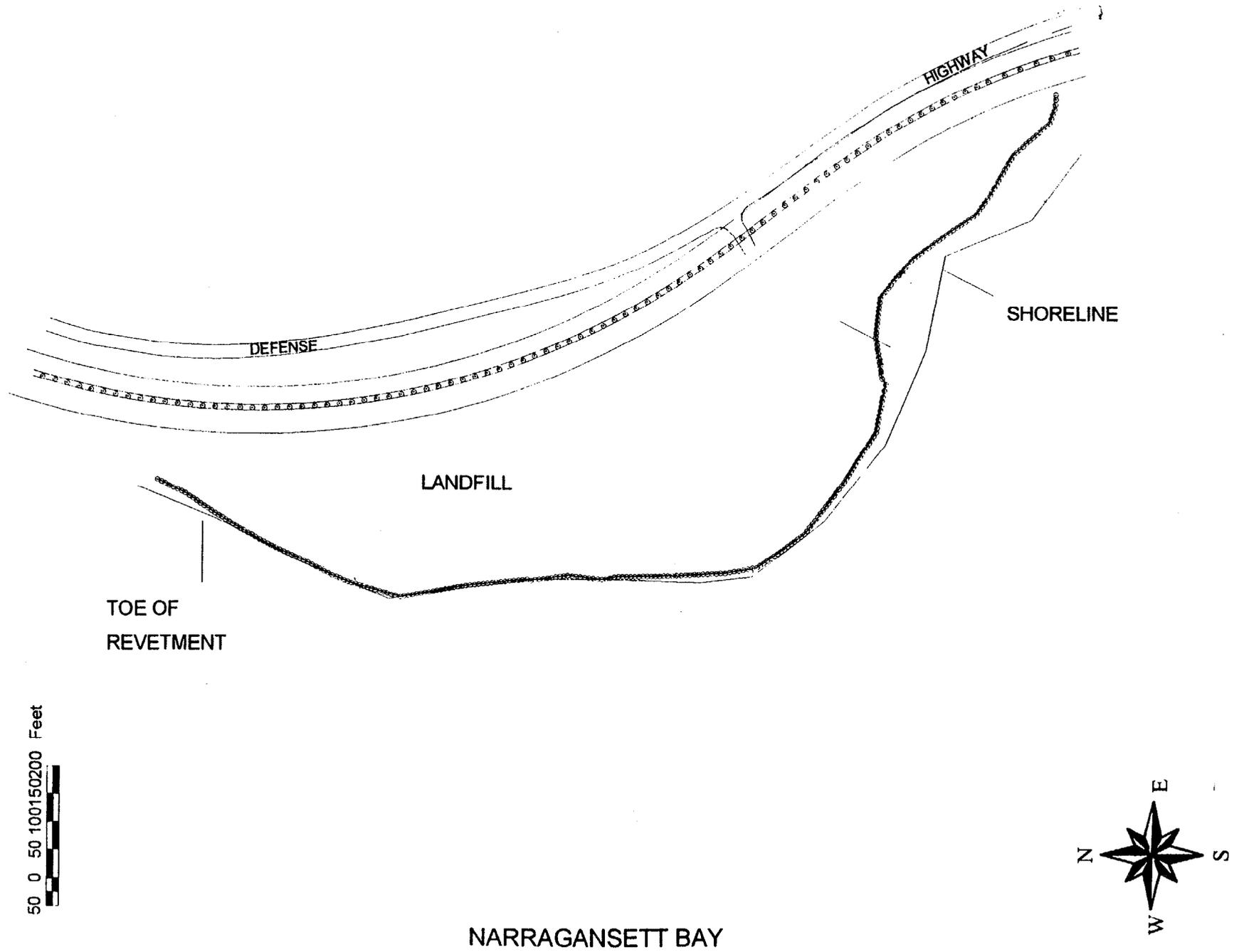


Figure 3.1-3. Habitats in the vicinity of McAllister Point Landfill, Newport, RI

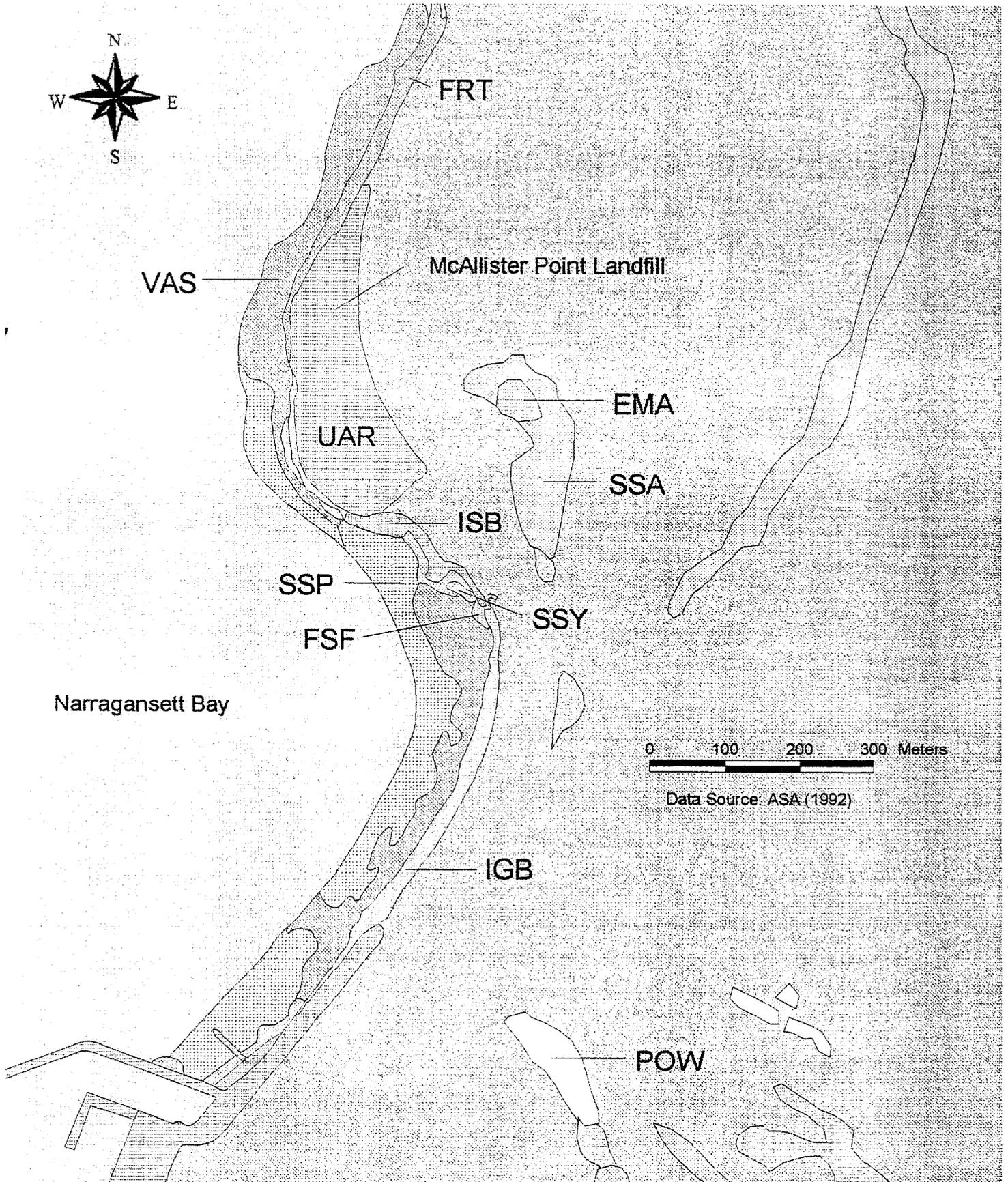
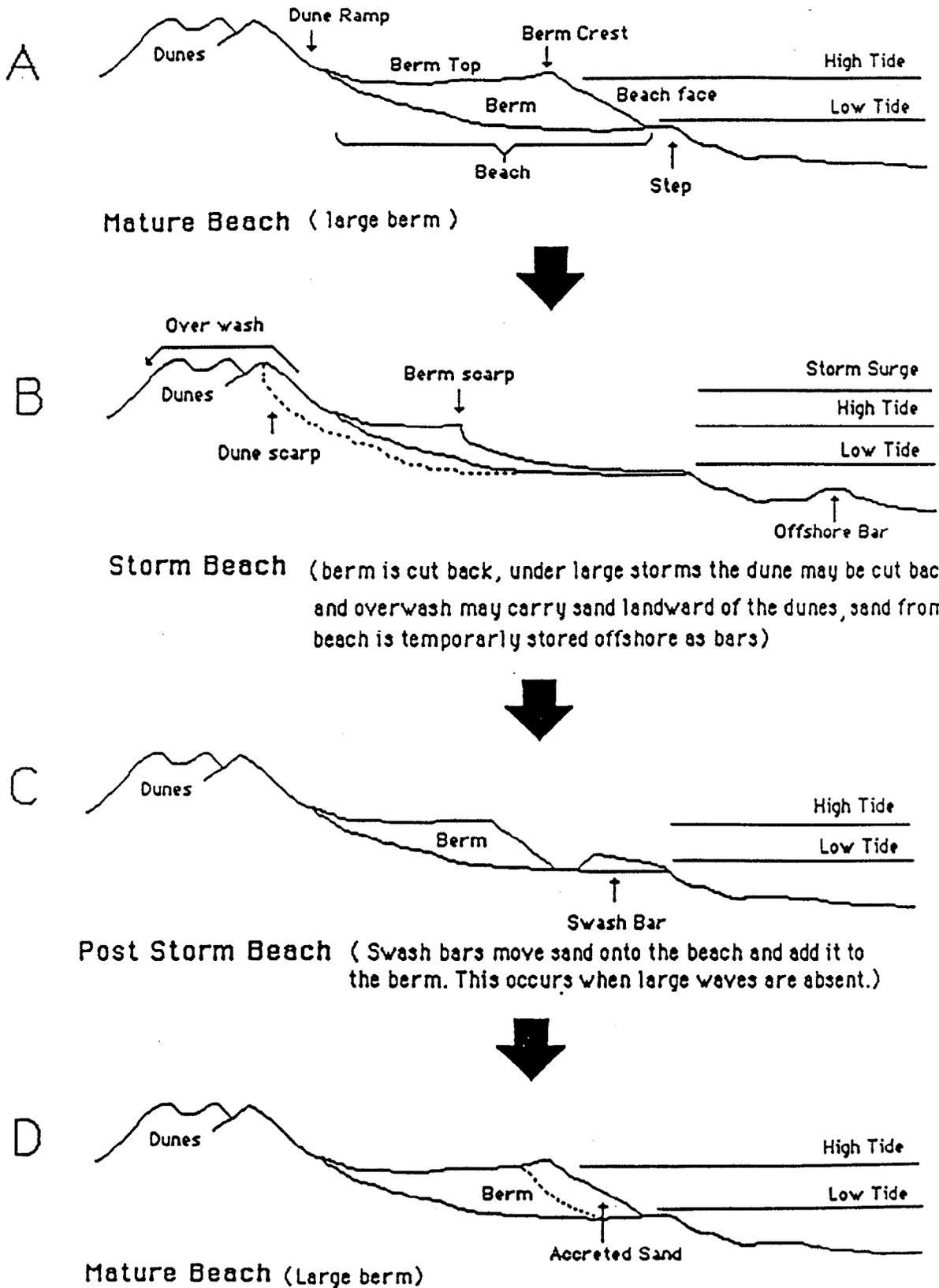


Figure 3.1-3 (continued). Habitat coding for Figure 3.1-3.

<u>Habitat Code</u>	<u>Description</u>
EMA	Estuarine emergent wetland, marsh/wet meadow
FRT	Fringing rock terrace
FSF	Fringing sand flat
IGB	Intertidal gravel beach
ISB	Intertidal sand beach
POW	Palustrine open water
SSA	Scrub-shrub wetland: Shrub swamp
SSP	Subtidal sand (depositional)
SSY	Subtidal sand (dynamic)
UAR	Supratidal artificial
VAS	Macroalgal

Figure 3.1-4. General cycle of sand movement and associated topographic changes for storm-exposed beaches in the temperate coastal zone (Peck, 1989).



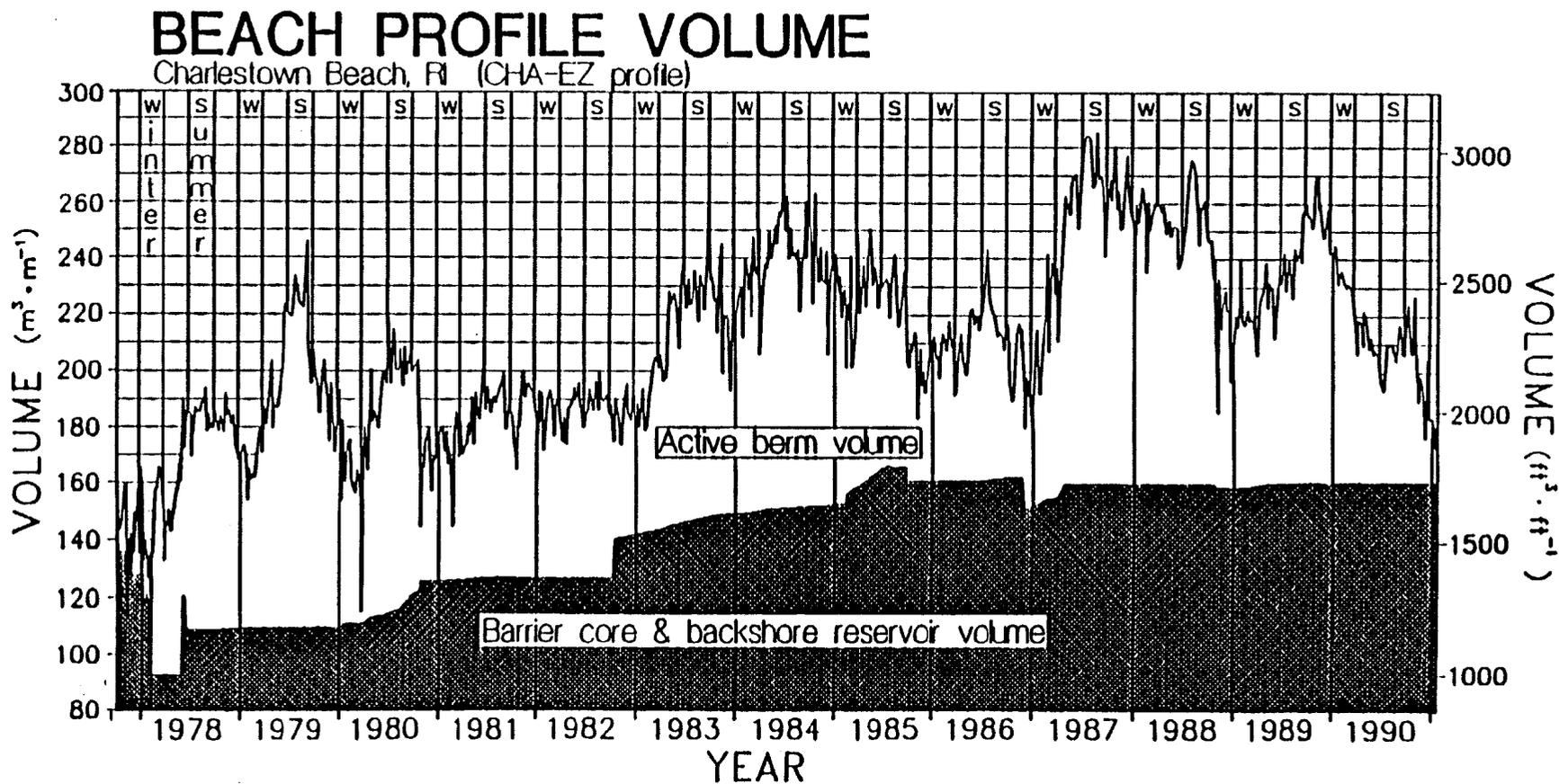


Figure 3.1-5. Plot of beach profile volume at Charlestown Beach, Rhode Island.  
 Source: French *et al.*, 1992.

Figure 3.1-6. Shoreline change over time at McAllister Point.

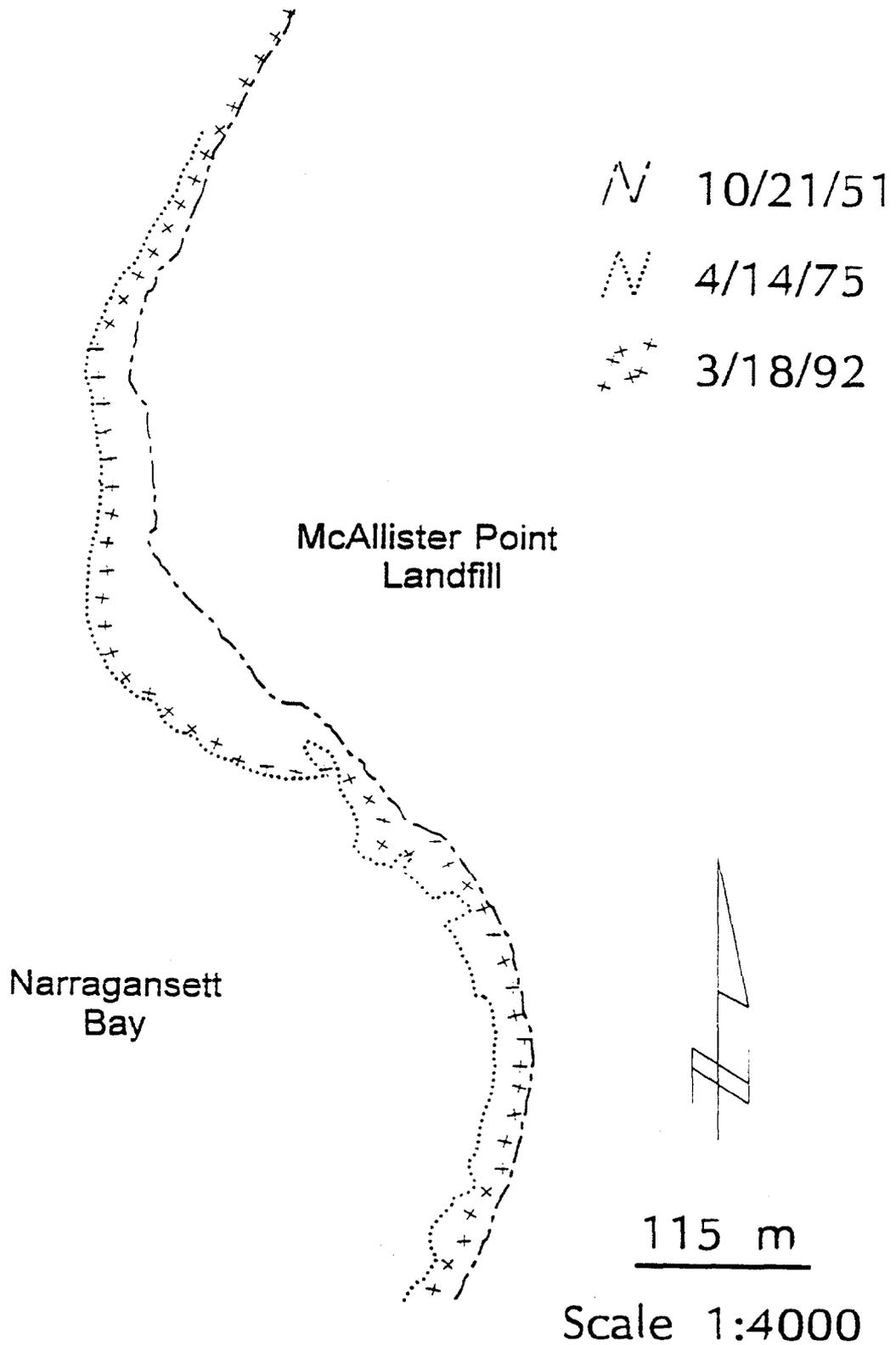
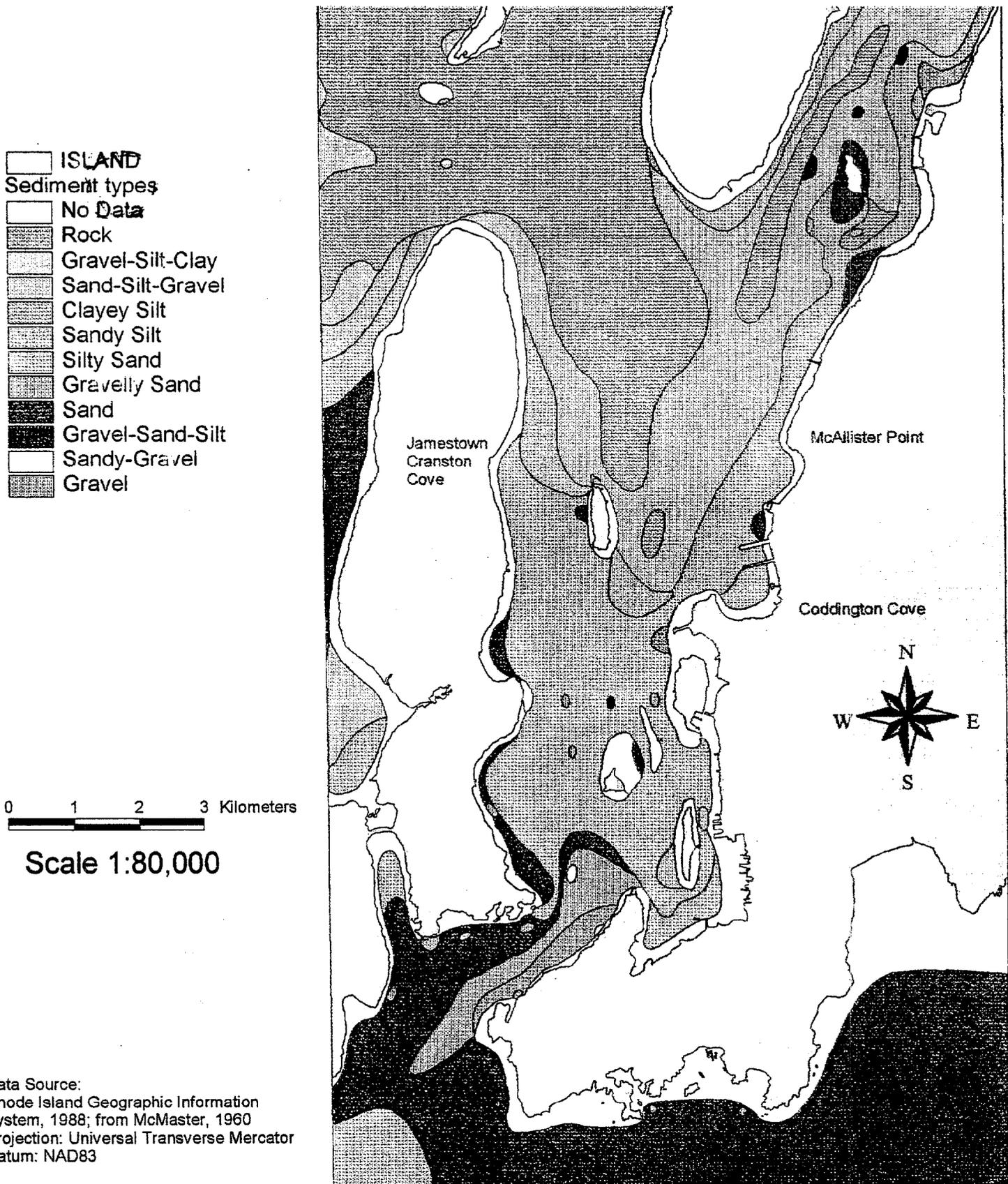


Figure 3.1-7. Surficial Sediment Types of the Lower East Passage, Narragansett Bay, RI



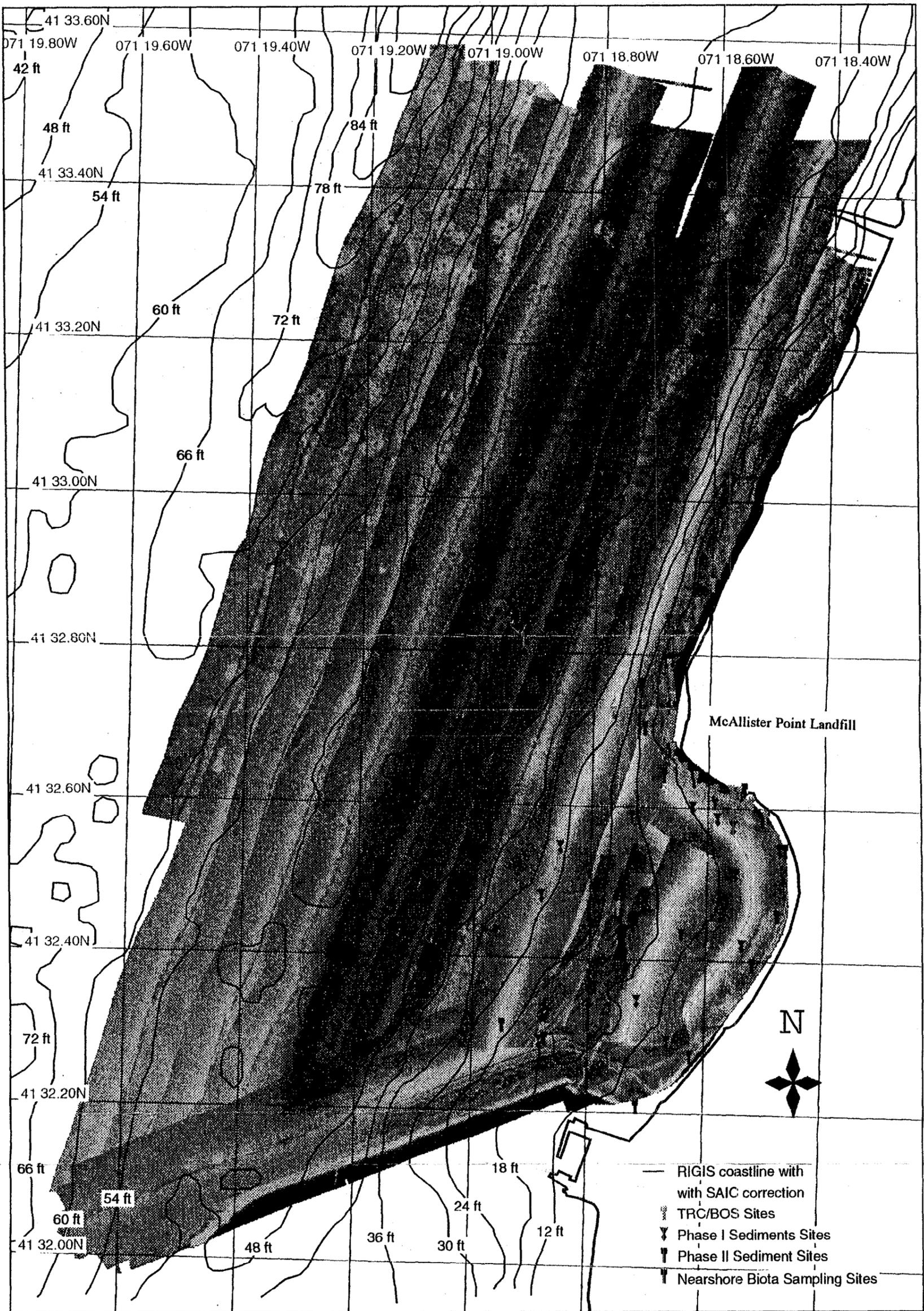


Figure 3.1-8. Sidescan sonar mosaic image of the sediment-water interface at McAllister Point.

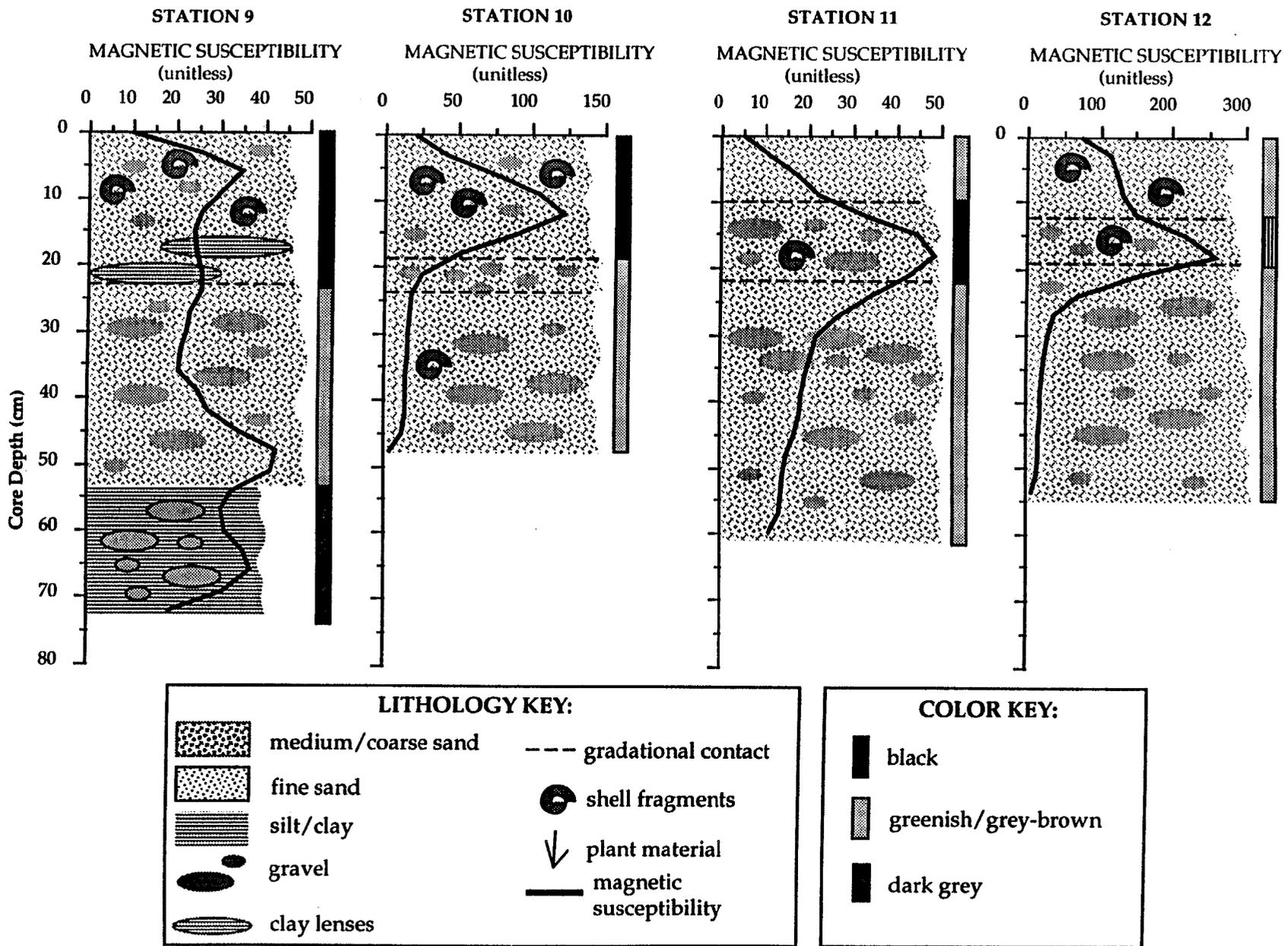


Figure 3.1-9. Results of magnetic susceptibility logging and characterization of sediment cores near McAllister Point.

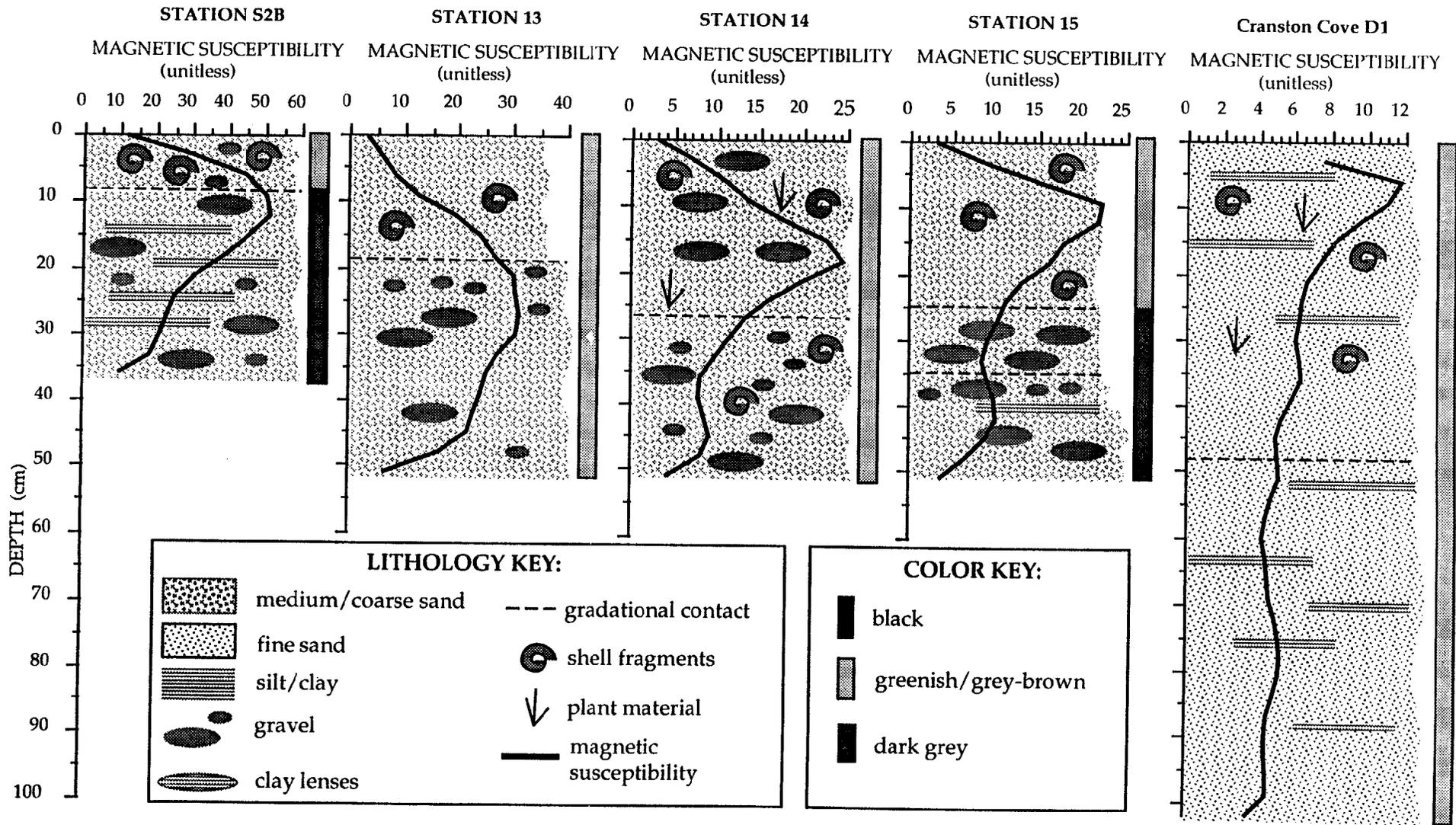


Figure 3.1-9 (continued). Results of magnetic susceptibility logging and characterization of sediment cores near McAllister Point

Figure 3.1-10. Results of Phase III investigations of landfill extent for the McAllister Point Landfill study area.

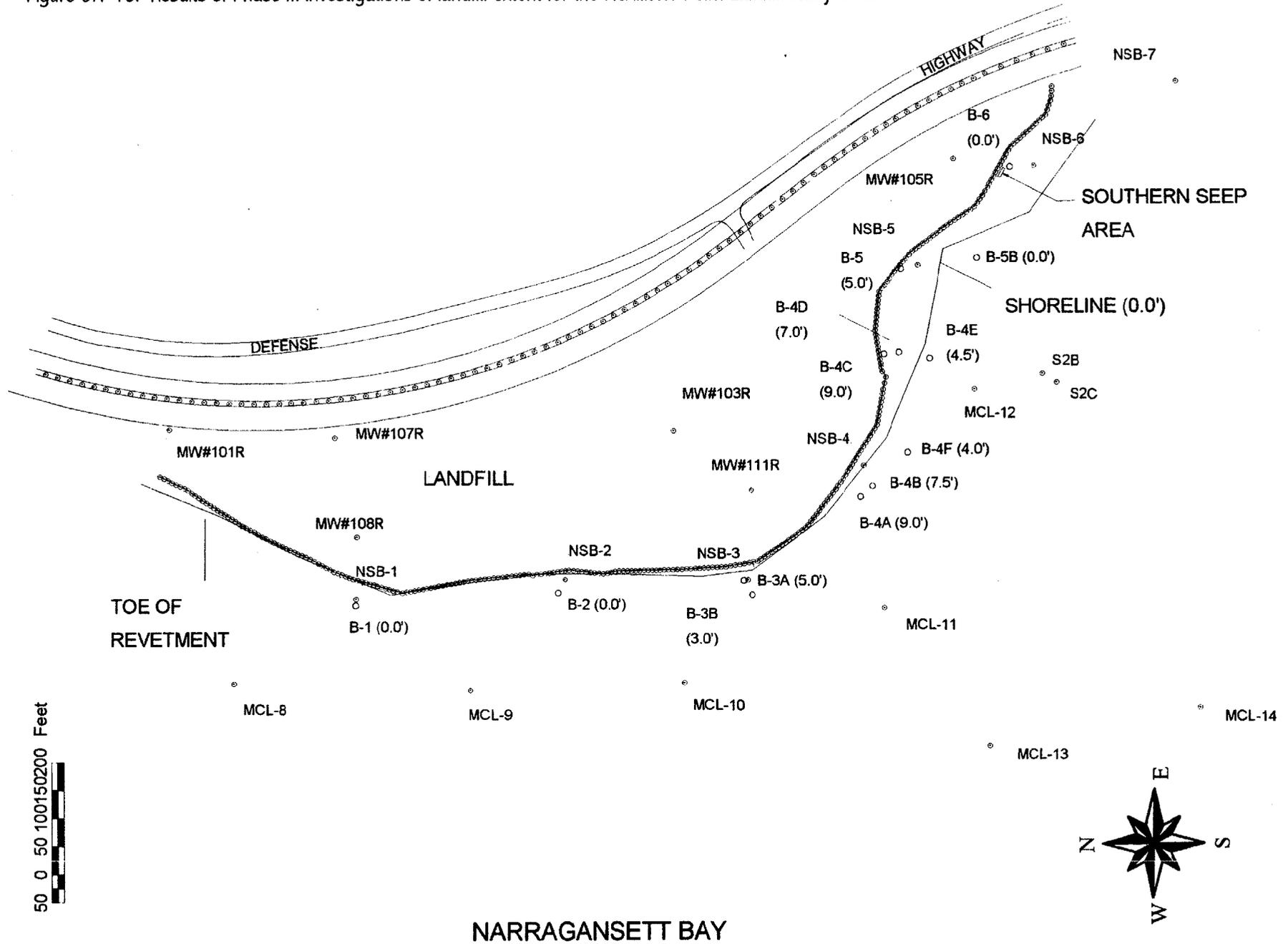
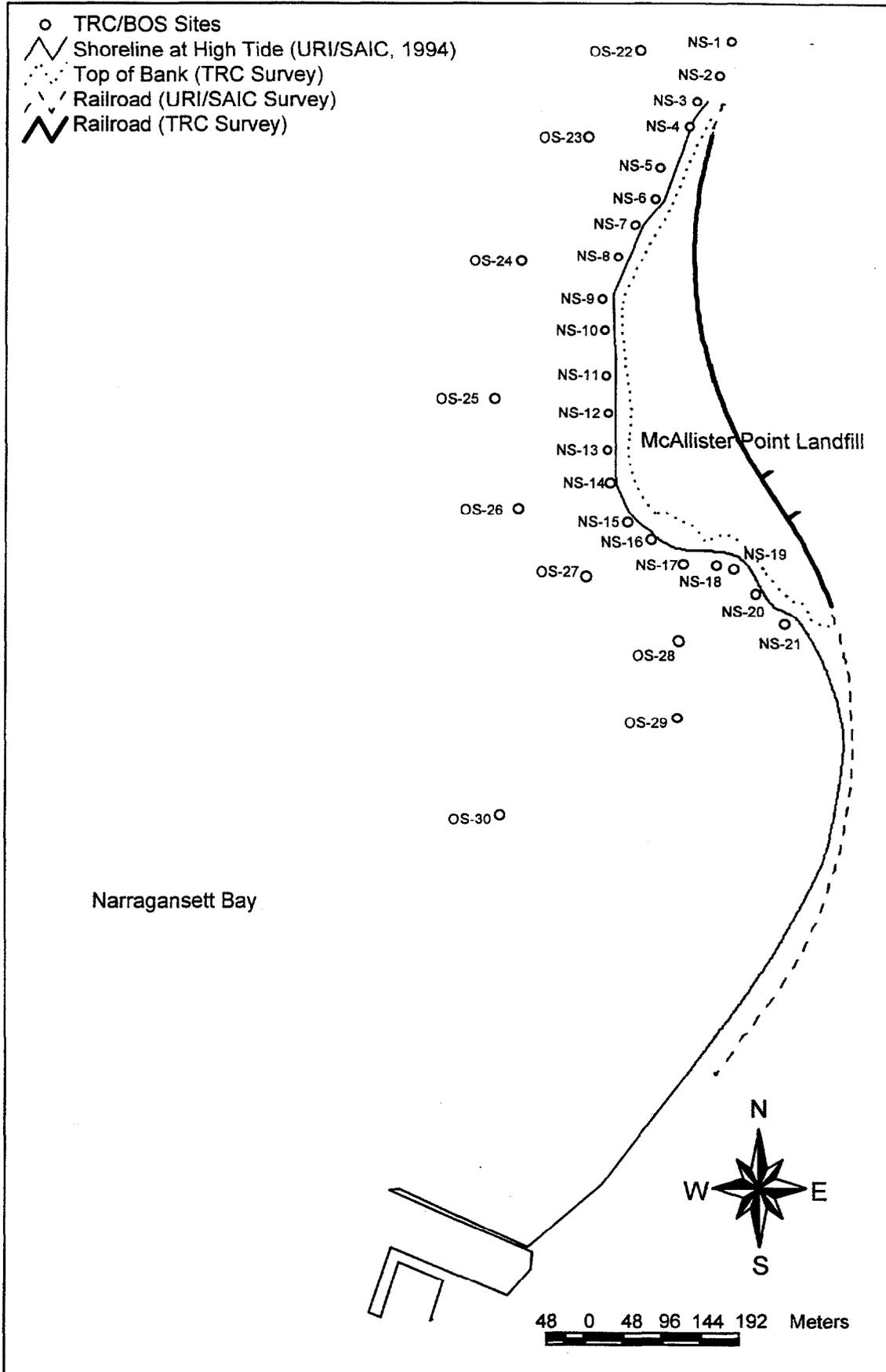


Figure 3.1-11. TRC/BOS sampling stations in the McAllister Point Landfill area.



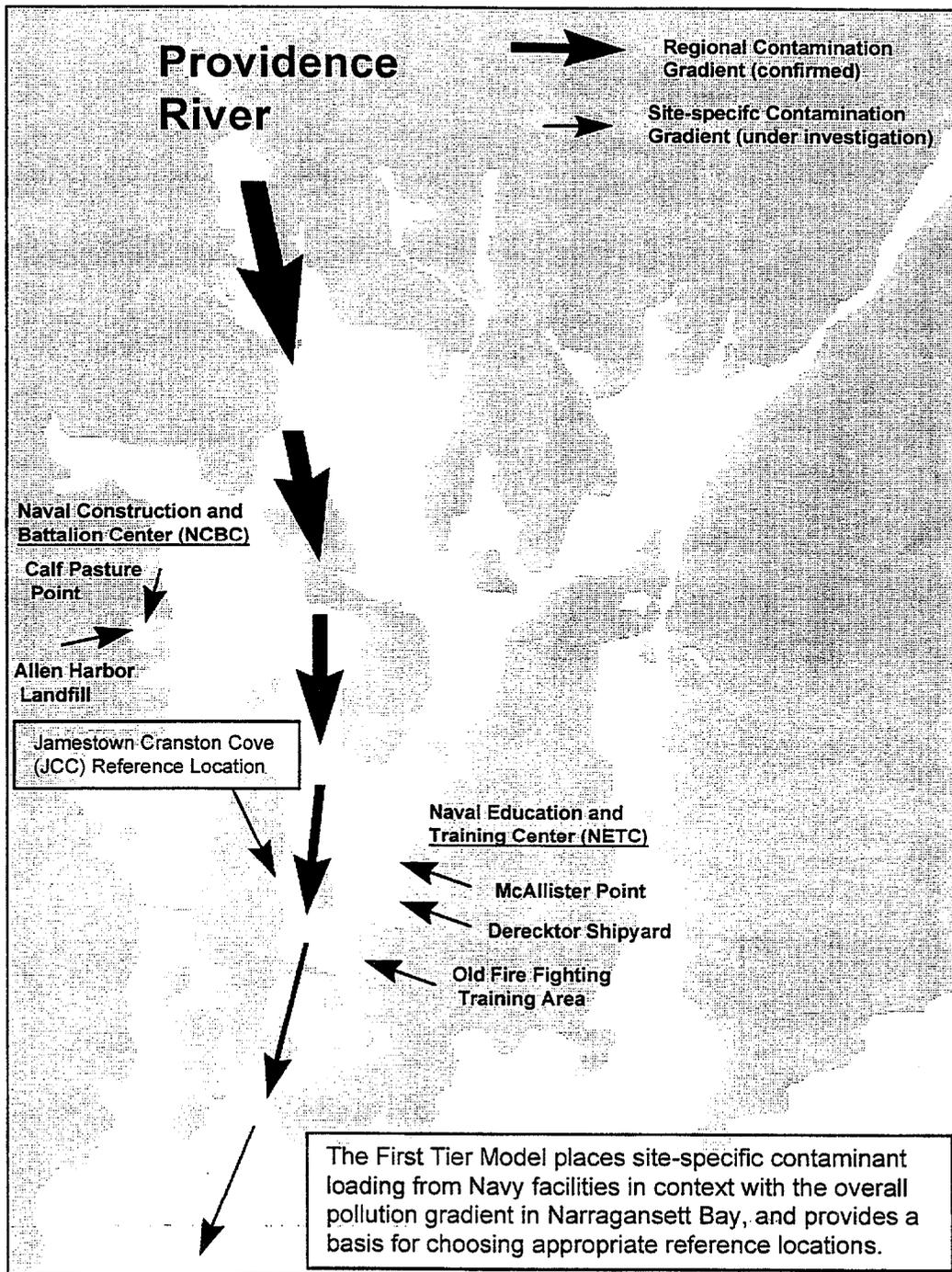


Figure 3.5-1. First Tier conceptual model for contaminant transport in Narragansett Bay.

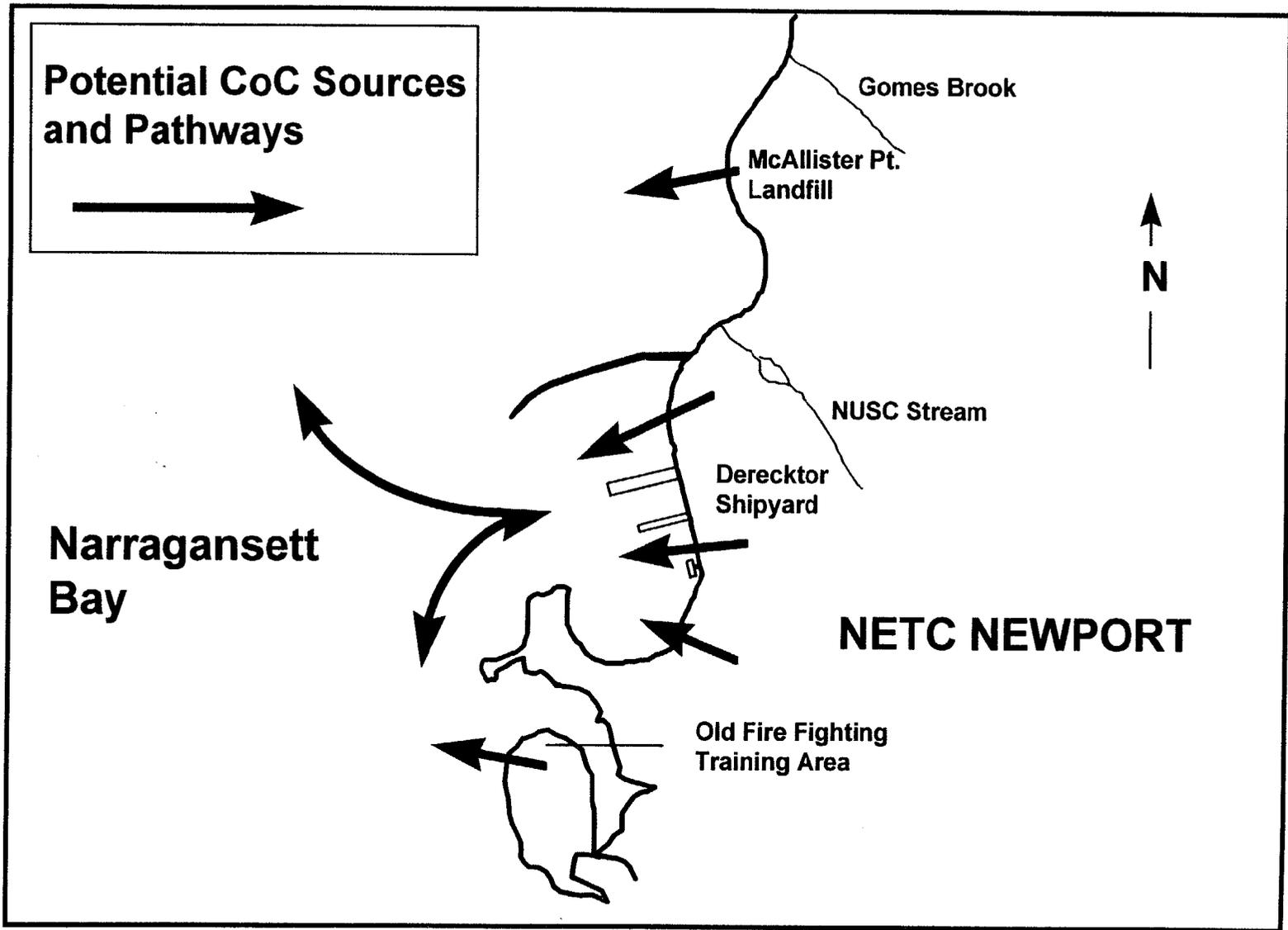


Figure 3.5-2. Second Tier conceptual model of contaminant transport for McAllister Point Landfill.



# PELAGIC RECEPTORS

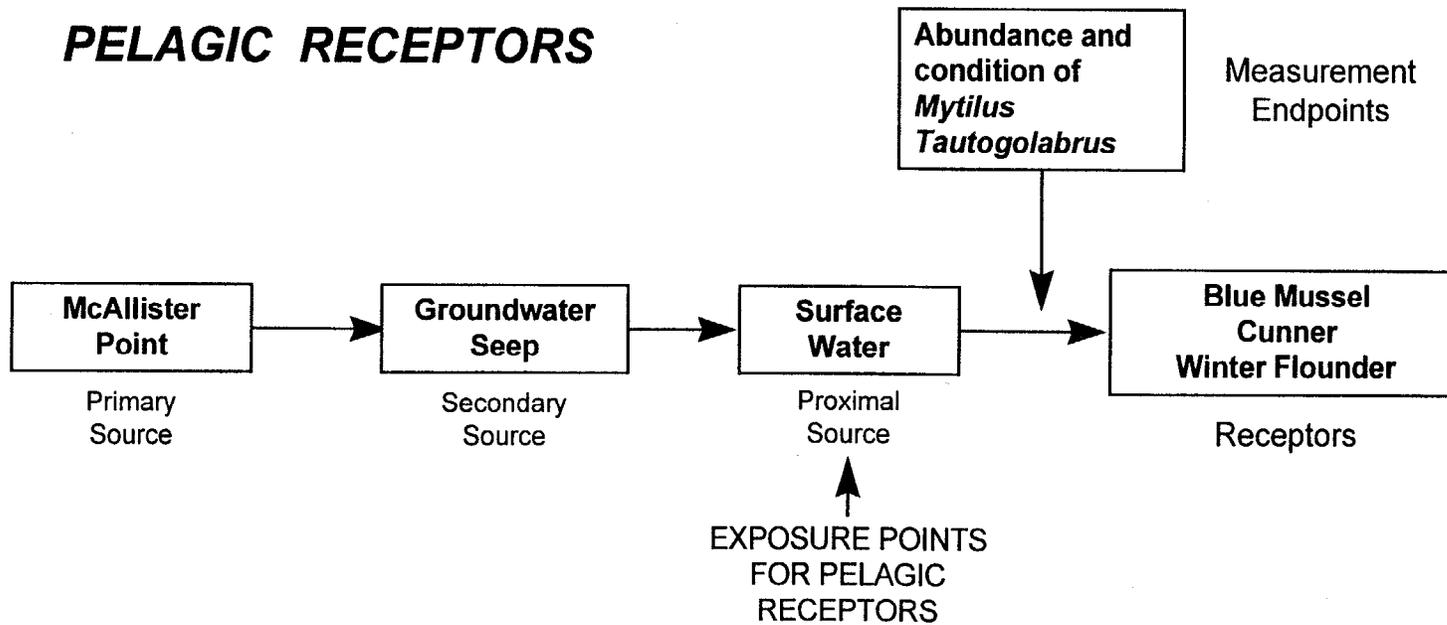


Figure 3.5-4. Fourth Tier conceptual model of contaminant transport for McAllister Point Landfill: Exposure pathway to pelagic receptors.

## EPIBENTHIC RECEPTORS

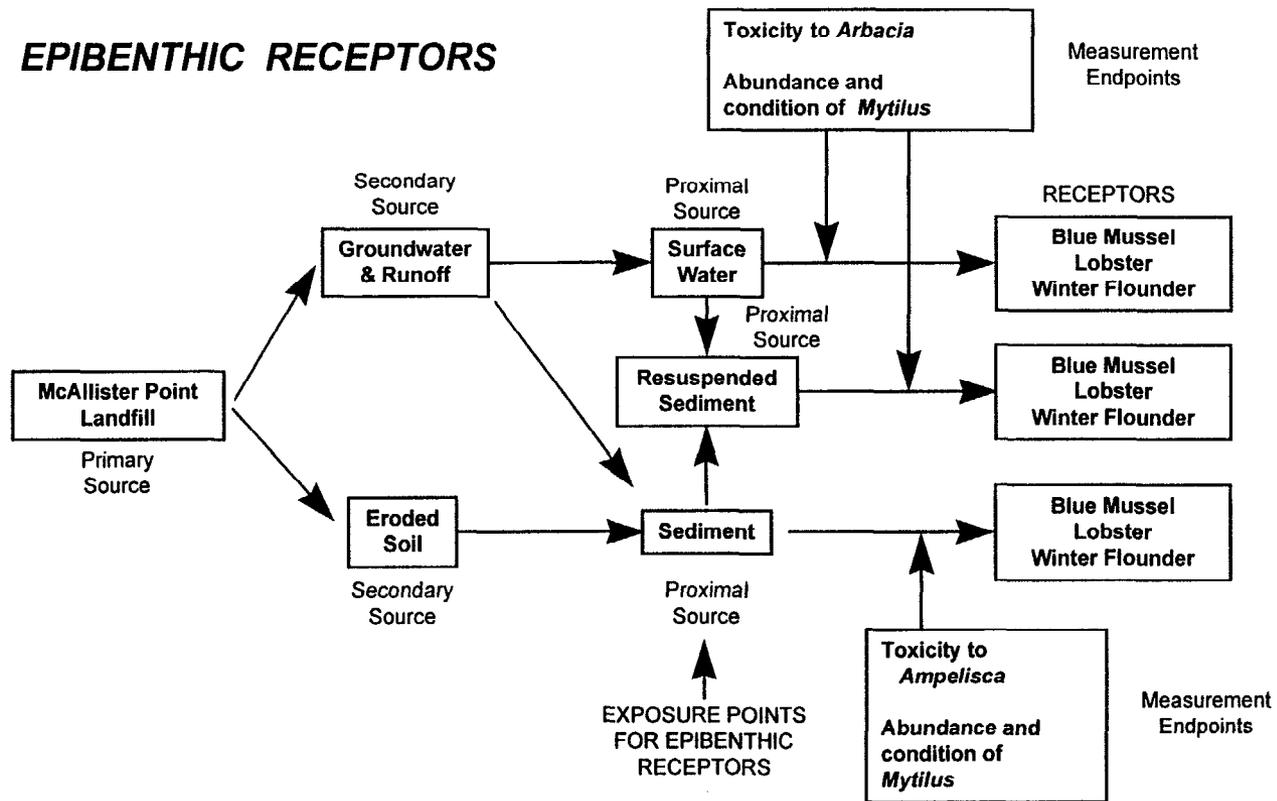


Figure 3.5-5. Fourth Tier conceptual model of contaminant transport for McAllister Point Landfill: Exposure pathway to epibenthic receptors.

# INFAUNAL RECEPTORS

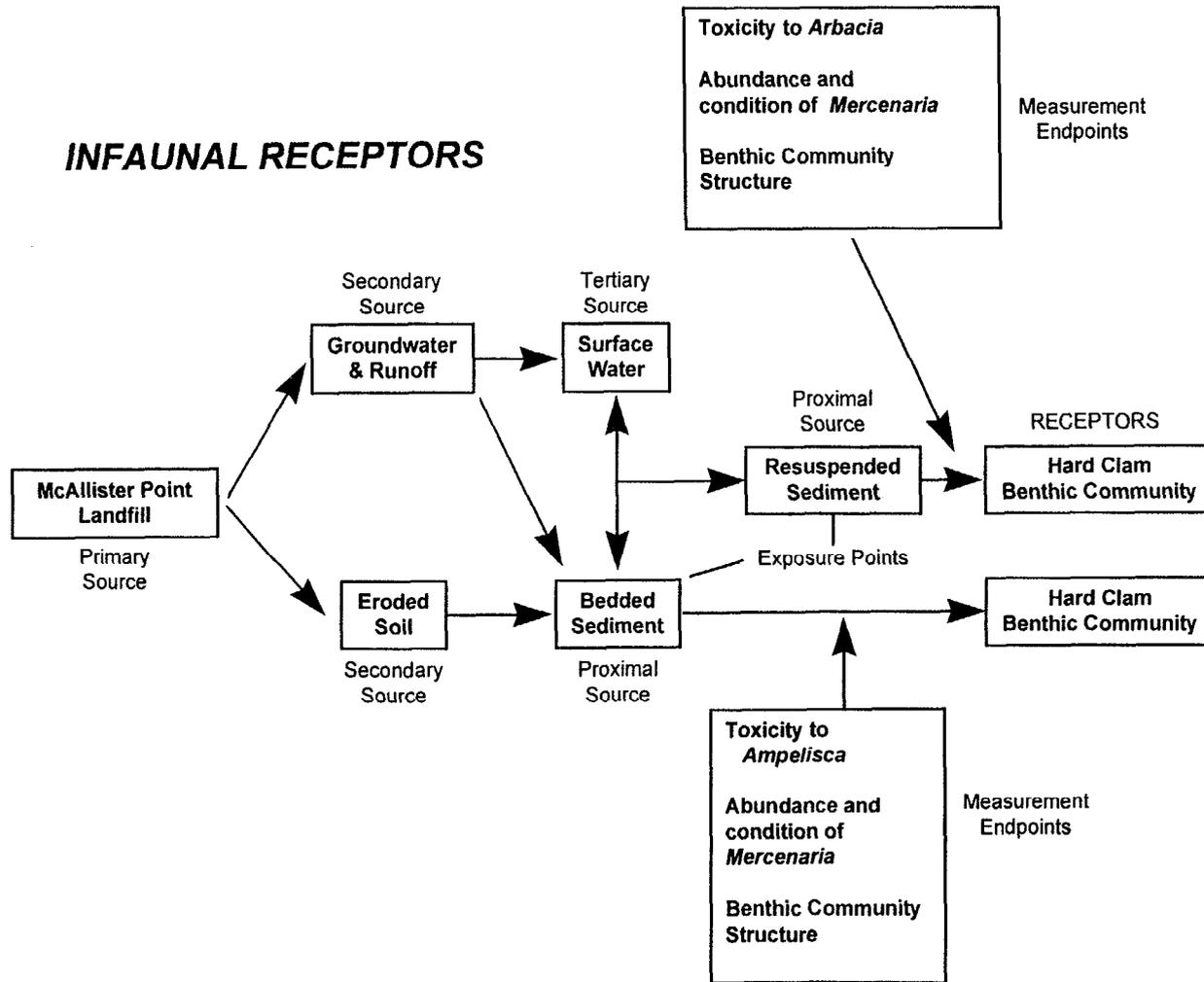


Figure 3.5-6. Fourth Tier conceptual model of contaminant transport for McAllister Point Landfill: Exposure pathway to infaunal receptors.

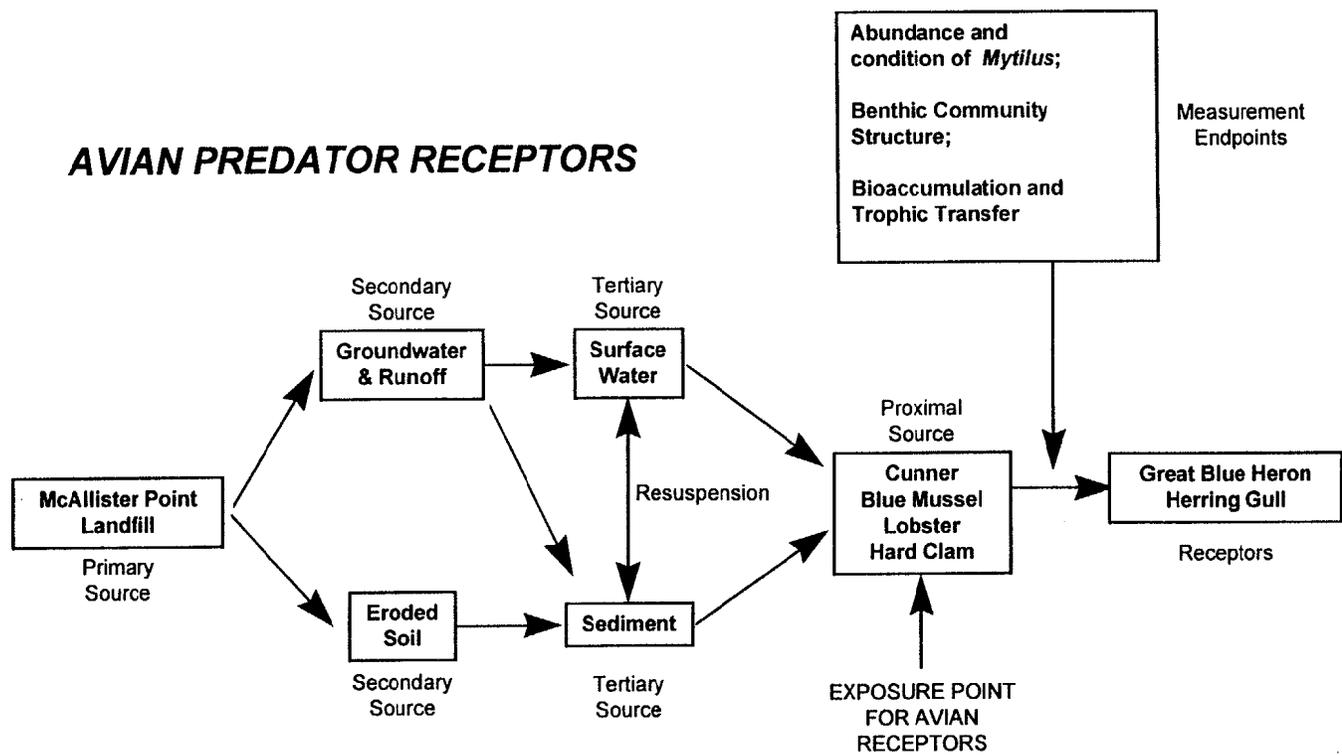


Figure 3.5-7. Fourth Tier conceptual model of contaminant transport for McAllister Point Landfill: Exposure pathway to avian receptors.

Figure 3.6-1. URI/SAIC Phase I sampling stations for the McAllister Point Landfill Marine ERA.

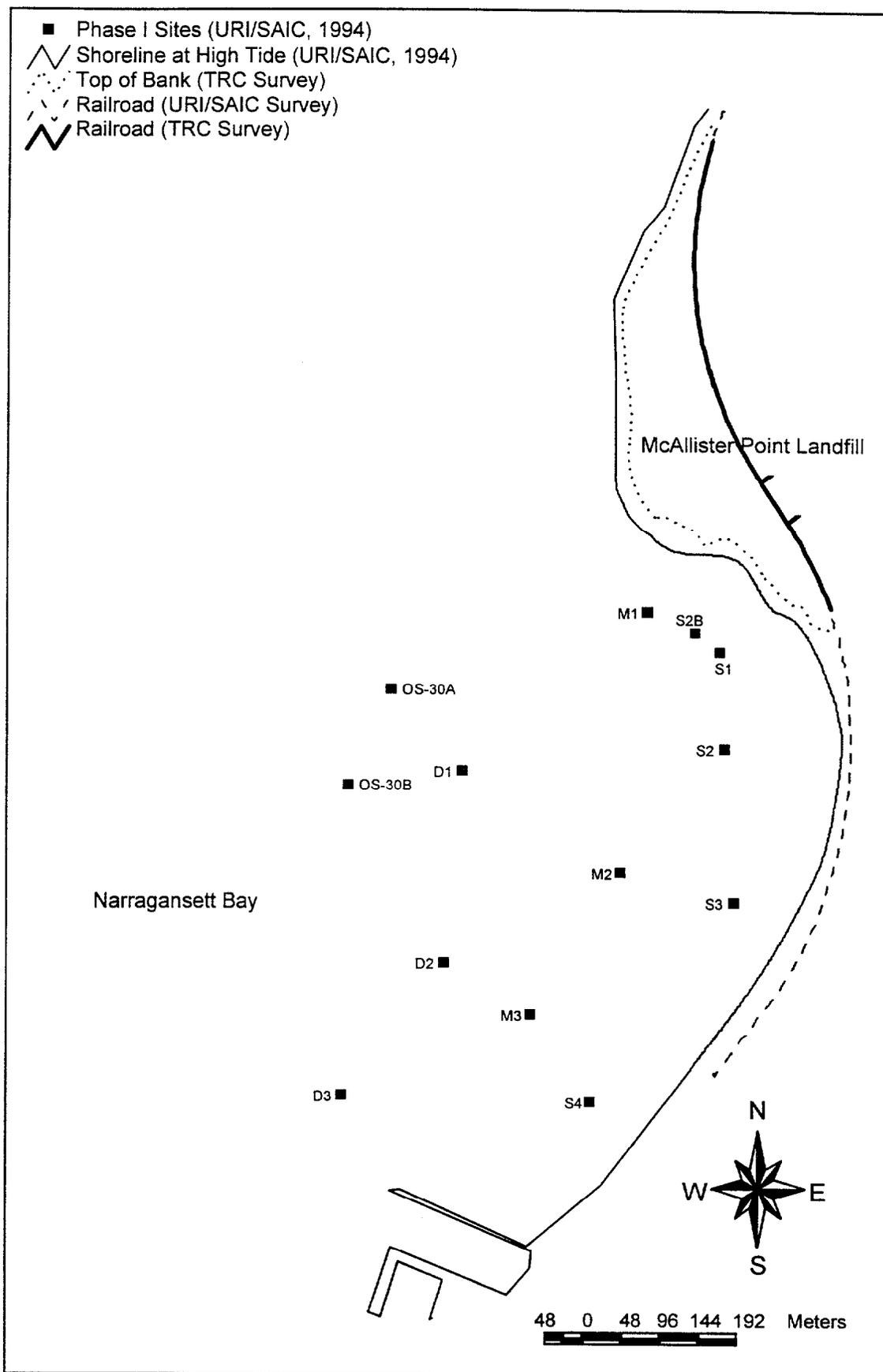


Figure 3.6-2. SAIC/URI Phase II sampling locations for the McAllister Point Landfill Ecological Risk Assessment.

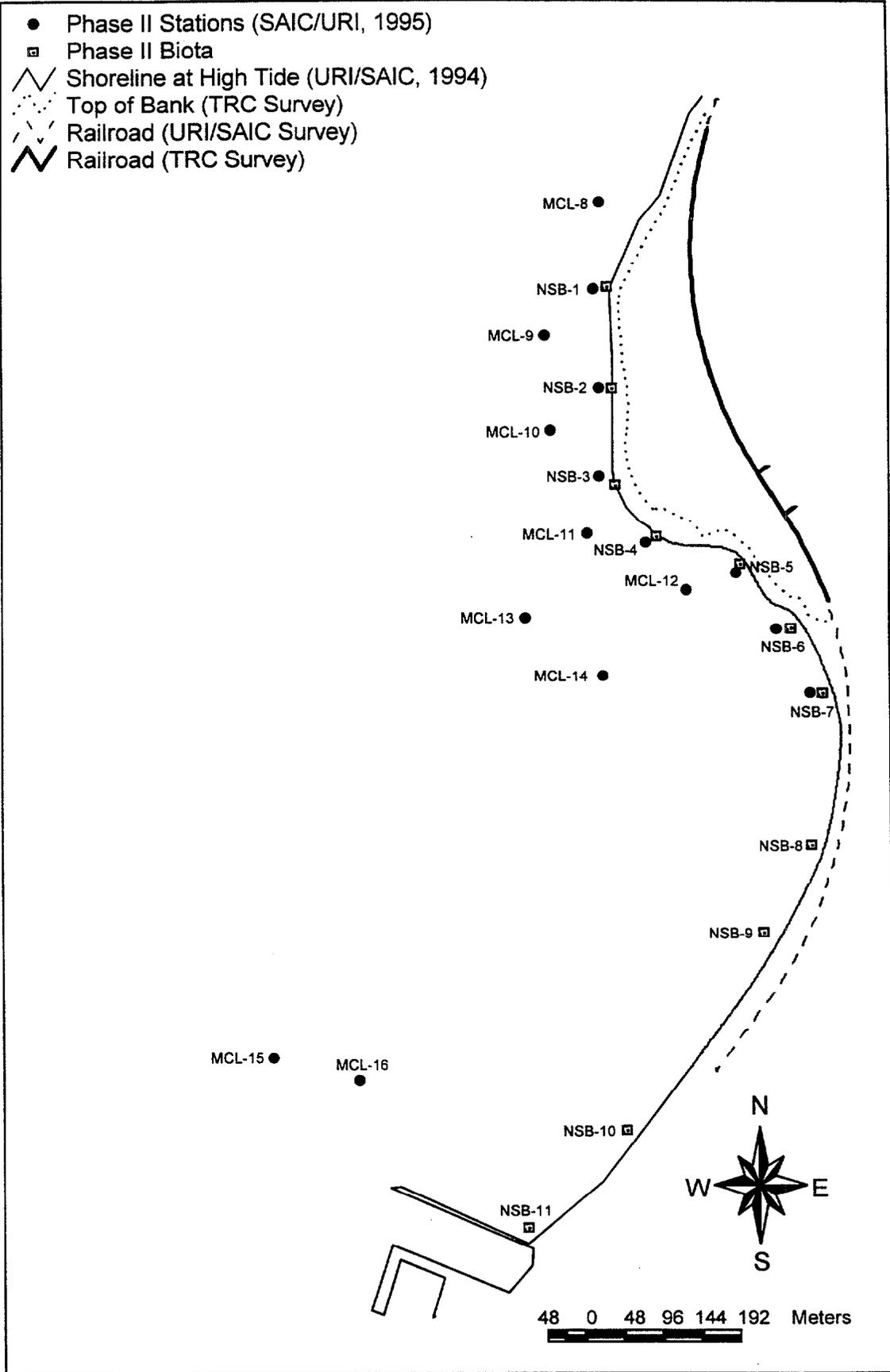


Figure 3.6-3. SAIC/URI Phase III resampling locations for the McAllister Point Landfill Ecological Risk Assessment.

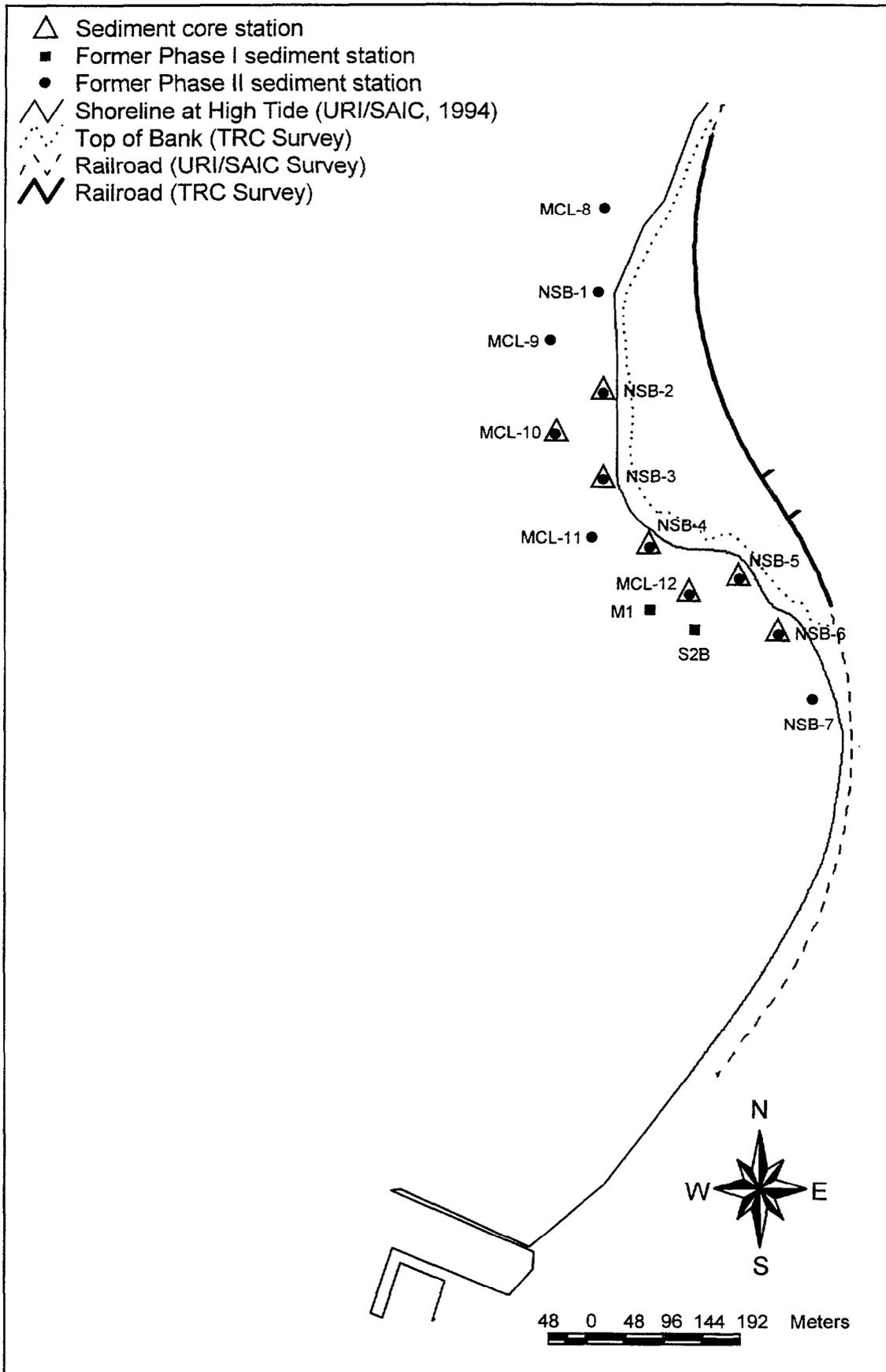


Table 3.1-1. McAllister Point Landfill soil contaminant concentrations.

Contaminant	Maximum Concentration <sup>1</sup>	Proposed NJ ECRA Guideline <sup>2</sup>	Exceeds Guideline?	Federal Action Limit <sup>3</sup>	Exceeds FAL?
Total VOCs	40,900	1,000	Yes		
Total PAHs	344,040	10,000	Yes		
Total PCBs	350 <sup>a</sup>	1,000	No	90	Yes
antimony	73.9	10	Yes	30	Yes
arsenic	24.1	20	Yes	80	No
beryllium	0.57	1	No	0.2	Yes
copper	145	170	No		
mercury	1.9	1	Yes	20	No
zinc	377	350	Yes		

1 - TRC Environmental Corporation (1994c); ( $\mu\text{g}/\text{Kg}$  for organics,  $\text{mg}/\text{Kg}$  for metals)

2 - New Jersey Environmental Clean-up Responsibility Act (ECRA) Guidance Values, ( $\mu\text{g}/\text{Kg}$  for organics,  $\text{mg}/\text{Kg}$  for metals)

3 - Federal Register (55 FR 30865, 27 July 1990), ( $\mu\text{g}/\text{Kg}$  for organics,  $\text{mg}/\text{Kg}$  for metals).

a - as Arochlor 1254

Table 3.1-2. McAllister Point Landfill groundwater contaminant concentrations.

Contaminant	Maximum Concentration <sup>1</sup> (µg/L)	AWQC Chronic (µg/L) <sup>2</sup>	Exceeds AWQC?
phenathrene <sup>4</sup>	23	4.6	Yes
PCB (1254)	1.8	0.03	Yes
cyanide <sup>3</sup>	54.8	1	Yes
silver <sup>3,5</sup>	25	2.3	Yes
cadmium <sup>3,5</sup>	28	9.3	Yes
copper <sup>3,5</sup>	1,730	597	Yes
mercury <sup>3,5</sup>	4.5	0.025	Yes
nickel <sup>3,5</sup>	386	8.3	Yes
lead <sup>3,5</sup>	4,060	8.5	Yes
zinc <sup>3,5</sup>	6,800	86	Yes

1 - TRC Environmental Corporation (1994a)

2 - acute AWQC used for cyanide and copper due to absence of chronic value

3 - total metal

4 - U.S. EPA (1993f)

5 - U.S. EPA (1985)

Table 3.1-3. Selected contaminant concentrations in sediments in the nearshore environment of the McAllister Point Landfill and comparisons to sediment-based criteria.

Group	Analyte	Benchmark <sup>2</sup>	U.S. Army Corps of Engineers (1988)		TRC (1994) <sup>1</sup>	
			Maximum Location Value	Exceed Benchmark?	Maximum Location Value	Exceed Benchmark?
Metals (µg/g)	Arsenic	8.2			240.0	Yes
	Cadmium	1.2	12.0	Yes	3.3	Yes
	Chromium	81	2190	Yes	359	Yes
	Copper	34	25000	Yes	1140	Yes
	Lead	46.7	4410.0	Yes	12900.0	Yes
	Mercury	0.15			10.70	Yes
	Nickel	20.9	1340.0	Yes	344.0	Yes
	Silver	1.0			3.0	Yes
	Zinc	150	2440	Yes	1580	Yes
PAHs (ng/g)	Acenaphthene	16			307	Yes
	Acenaphthylene	44			38	No
	Anthracene	85.3			797.6	Yes
	Benzo(a)anthracene	261			1521	Yes
	Benzo(a)pyrene	430			1298	Yes
	Benzo(b)fluoranthene				1717	NA
	Benzo(e)pyrene				852	NA
	Benzo(g,h,i)perylene				686	NA
	Benzo(k)fluoranthene				667	NA
	Biphenyl				77	NA
	Chrysene	384			1553	Yes
	Dibenz(a,h)anthracene	63.4			189.6	Yes
	Fluoranthene	600			3119	Yes
	Fluorene	19			463	Yes
	High Molecular PAHs	1700			10118	NA
	Indeno(1,2,3-cd)pyrene				787	NA
	Low Molecular PAHs	552			4356	NA
	Naphthalene	160			147	No
	Perylene				368	NA
Phenanthrene	240			2669	Yes	
Pyrene	665			2437	Yes	
Total PAHs	4022			44054	Yes	
SVOCs (ng/g)	dibenzofuran				217	NA
	dibenzothiophene				188	NA
PCBs (ng/g)	Total PCBs	22.7			598.0	Yes

NA = Benchmark Not Available

1 - Compositd nearshore stations, individual offshore stations

2 - Benchmark = NOAA Effects Range - Low (ER-L; Long *et al.*, 1995)

Table 3.1-4. Total concentration ( $\mu\text{g/g}$  dry wt) of metals in offshore sediments from the TRC/BOS survey of the McAllister Point Landfill study area.

Station		Ag	Cd	Cr	Cu	Hg	Ni	Pb	Zn
<i>Benchmark</i> <sup>1</sup>		1.00	1.20	81.0	34.0	0.15	20.9	46.7	150
TRC/BOS Stations <sup>2</sup>	OS-22	0.52	0.17	48.6	32.4	0.17	16.0	56.1	107
	OS-23	0.41	0.15	54.9	27.9	0.21	123	49.6	108
	OS-24	0.41	0.09	48.6	23.5	0.13	20.1	37.8	96.0
	OS-25	0.69	0.16	68.2	37.9	0.24	22.3	51.2	121
	OS-26	0.53	0.15	66.6	30.8	0.17	25.1	44.7	114
	OS-27	0.32	0.15	49.6	24.8	0.12	26.3	41.1	102
	OS-28	0.83	0.24	66.8	37.2	0.16	36.5	49.1	139
	OS-29	0.57	0.14	48.7	24.9	0.13	28.5	40.7	106
	OS-30	0.66	0.19	52.9	36.8	0.20	20.6	48.9	124

1 - NOAA Effects Range Low (ER-L) sediment criteria (Long *et al.*, 1995). Shaded values indicate ER-L guideline exceeded.  
 2 - TRC (1994).

Table 3.1-5. Total concentration of organic contaminants in offshore sediments from the TRC/BOS survey of the McAllister Point Landfill study area.

Station		PAHs <sup>1</sup> (ng/g)	PCBs <sup>2</sup> (ng/g)	Tributyltin <sup>3</sup> (ng Sn/g)
<i>Benchmark</i> <sup>4</sup>		4022	22.7	NA
TRC/BOS Stations <sup>5</sup>	OS-22	4887	33.9	NA
	OS-23	3982	32.1	NA
	OS-24	2953	24.1	NA
	OS-25	6250	69.6	NA
	OS-26	4217	81.4	NA
	OS-27	6484	32.8	NA
	OS-28	44054	73.9	NA
	OS-29	5824	37.6	NA
	OS-30	16042	56.3	NA

Shaded value indicates ER-L guideline exceeded (Long *et al.*, 1995).

1 - PAHs: Sum of 24 PAHs.

2 - PCBs: Sum of congeners x 2.

3 - concentration of tin (Sn) in organic matrix.

4 - Long *et al.*, 1995.

5 - TRC (1994).

NA = not available.

Table 3.1-6. Tissue residue concentrations in blue mussels from the TRC/BOS survey of the nearshore environment of the McAllister Point Landfill study area.<sup>1</sup>

Indigenous Blue Mussels<sup>2</sup>

CLASS	ANALYTE	Range of Site Concentration <sup>a</sup>		Mean Site Concentration <sup>b</sup>	Site 95% Upper Confidence Limit	Mean of Reference Locations Concentration <sup>b</sup>	95% UCL or Max. Concentration <sup>c</sup>	
		Minimum	Maximum				Exceeds Reference?	
MET	Ag	0.078	0.292	0.14	0.29	0.07		YES
	As	13.9	18.4	16.56	19.08	16.67	+	YES
	Cd	0.898	1.14	1.02	1.19	0.91	+	YES
	Cr	3.72	37.5	15.03	34.65	6.54		YES
	Cu	5.25	12.1	7.76	11.63	5.73		YES
	Hg	0.095	0.25	0.14	0.23	0.11		YES
	Ni	4.33	18.6	9.41	18.42	5.21		YES
	Pb	1.51	8.9	3.50	7.73	1.55		YES
	Zn	93.1	135	110.22	134.45	94.53		YES
PAH	acenaphthene	2.38	3.76	2.87	3.68	1.23		YES
	acenaphthylene	2.4	4.12	3.45	4.46	2.26	+	YES
	anthracene	5.1	7.23	6.03	7.22	2.74		YES
	benzo[a]anthracene	11.5	21.82	14.40	20.36	7.96		YES
	benzo[a]pyrene	4.8	9.67	6.30	9.24	5.59		YES
	benzo[b]fluoranthene	21.38	30.81	24.03	29.38	19.56		YES
	benzo[e]pyrene	20.02	25.97	22.55	26.87	16.54	+	YES
	benzo[g,h,i]perylene	6.37	9.12	8.26	10.07	8.40	+	YES
	benzo[k]fluoranthene	5.14	9.48	6.69	8.91	5.49		YES
	biphenyl	8.73	14.56	12.74	17.05	11.85	+	YES
	chrysene	14.05	26.8	17.83	24.72	9.12		YES
	dibenz[a,h]anthracene	1.11	1.75	1.34	1.71	1.27		YES
	dibenzofuran	5.36	6.62	6.09	6.97	5.02	+	YES
	dibenzothiophene	1.76	2.92	2.21	2.96	1.46	+	YES
	fluoranthene	57.93	95.86	71.57	95.44	30.65		YES
	fluorene	3.29	5.01	4.50	5.57	2.38	+	YES
	indeno[1,2,3-c,d]pyrene	4.94	7.09	5.85	7.20	5.42	+	YES
	naphthalene	37.36	69.16	57.96	79.46	55.80	+	YES
	perylene	4.19	13.64	9.27	16.07	5.26	+	YES
	phenanthrene	24.17	30.44	27.35	31.23	15.76	+	YES
pyrene	45.92	78.62	56.32	76.98	27.99		YES	
	Sum PAHs	336.88	421.34	367.62	412.20	241.73		YES
PCB	PCB Sum of Congeners x 2	382.12	2002.9	937.82	1843.67	380.72		YES

Notes 1 - Data from TRC (1994)

2 - Concentration units (dry wt.): Metals (MET) - µg/g; PAHs, PCBs - ng/g

a - The range of concentrations reported for site data excludes non-detected values.

b - 1/2 Sample Quantification Limits substituted for non-detects when calculating mean of site and reference data.

c - If 95% UCL is greater than the Maximum Concentration, as indicated with a "+", then Maximum Concentration is used to screen against background.

Table 3.1-7. Tissue residue concentrations in hard clams from the TRC/BOS survey of the nearshore environment of the McAllister Point Landfill study area.<sup>1</sup>

HARD CLAMS<sup>2</sup>

CLASS	ANALYTE	Range of Site Concentration <sup>a</sup>		Mean Site Concentration <sup>b</sup>	Site 95% Upper Confidence Limit	Mean of Reference Locations Concentration <sup>b</sup>	95% UCL or Max. Concentration <sup>c</sup>	
		Minimum	Maximum				Exceeds Reference?	
MET	Ag	0.683	1.96	0.99	1.73	0.87		YES
	As	13.7	16	14.61	16.38	13.66	+	YES
	Cd	0.441	1.77	1.10	2.08	1.02	+	YES
	Cr	37.1	102	42.99	88.35	35.55		YES
	Cu	7.9	22.8	12.69	21.71	10.55		YES
	Hg	0.082	0.182	0.11	0.18	0.13		YES
	Ni	14.4	49.6	29.04	50.96	25.43	+	YES
	Pb	3.71	16.7	8.89	19.80	9.30	+	YES
	Zn	113	167	132.42	173.33	139.75	+	YES
PAH	acenaphthene	1.9	5.47	2.93	4.91	1.74		YES
	acenaphthylene	2.4	4.36	3.05	4.10	2.11		YES
	anthracene	3.44	9.93	4.91	8.30	2.20		YES
	benz[a]anthracene	16.57	50.97	22.94	41.52	12.56		YES
	benzo[a]pyrene	10.87	37.29	16.08	30.97	10.09		YES
	benzo[b]fluoranthene	28.6	82.24	37.50	66.09	24.85		YES
	benzo[e]pyrene	21.5	51.24	28.80	47.11	20.39		YES
	benzo[g,h,i]perylene	11.6	44.31	22.48	43.74	18.50		YES
	benzo[k]fluoranthene	8.93	29.78	12.97	24.35	7.96		YES
	biphenyl	10.86	23.48	17.81	24.85	16.04	+	YES
	chrysene	14.16	37.99	21.14	35.74	13.90		YES
	dibenz[a,h]anthracene	1.88	6.29	3.02	5.62	2.31		YES
	dibenzofuran	5.22	9.17	7.20	9.82	6.08	+	YES
	dibenzothiophene	2.04	3.86	2.57	3.66	1.72		YES
	fluoranthene	39.7	86.22	52.96	81.09	27.44		YES
	fluorene	2.7	6.71	3.57	5.81	1.99		YES
	indeno[1,2,3-c,d]pyrene	9.38	32.94	14.57	27.54	10.35		YES
	naphthalene	61.58	123.39	96.23	130.60	90.08	+	YES
	perylene	4.52	17.22	7.55	14.79	7.40		YES
phenanthrene	19.81	51.86	28.91	48.99	18.23		YES	
pyrene	34.23	76.12	47.36	70.83	105.65		NO	
	Sum PAHs	331.43	784.24	454.52	701.29	401.57		YES
	PCB Sum of Congeners x	124.73	124.73	124.73		52.97	+	YES

Notes 1 - Data from TRC (1994)

2 - Concentration units (dry wt.): Metals (MET) - µg/g; PAHs, PCBs - ng/g

a - The range of concentrations reported for site data excludes non-detected values.

b - 1/2 Sample Quantification Limits substituted for non-detects when calculating mean of site and reference data.

c - If 95% UCL is greater than the Maximum Concentration, as indicated with a "+", then Maximum Concentration is used to screen against background.

Table 3.2-1. Target analytes for chemical characterization for the McAllister Point Landfill Marine Ecological Risk Assessment.

Analyte	Sample matrix	Target method detection limits <sup>a</sup>
<b>Polycyclic Aromatic Hydrocarbons (PAHs)</b>		
	sediment	5 ng/g
	biota	10 ng/g
naphthalene		
2-methylnaphthalene		
1-methylnaphthalene		
biphenyl		
2,6-dimethylnaphthalene		
acenaphthylene		
acenaphthene		
1,6,7-trimethylnaphthalene		
fluorene		
phenanthrene		
anthracene		
1-methylphenanthrene		
fluoranthene		
pyrene		
benz[a] anthracene		
chrysene		
benzo [b] fluoranthene		
benzo [k] fluoranthene		
benzo [e] pyrene		
benzo [a] pyrene		
perylene		
indeno [1,2,3-cd] pyrene		
dibenz [a,h] anthracene		
benzo [ghi] perylene		
<b>Organo-Chlorine Pesticides (OCPs)</b>		
	sediment	1 ng/g
	biota	2 ng/g
Aldrin		
hexachlorobenzene		
Mirex		
o,p' - DDE		
p,p' - DDE		

Table 3.2-1 (continued). Target analytes for chemical characterization for the McAllister Point Landfill Marine Ecological Risk Assessment.

Analyte	Sample matrix	Target method detection limits <sup>a</sup>
<b>Polychlorinated Biphenyl (PCB) Congeners</b>		
	sediment	1 ng/g
	biota	2 ng/g
8 (2 4') <sup>b</sup>	126 (3 3'4 4'5)	
18 (2 2'5)	128 (2 2'3 3'4 4')	
28 (2 4 4')	138 (2 2'3 4 4'5)	
29 (2 4 5)	153 (2 2'4 4'5 5')	
44 (2 2'3 5')	154 (2 2'4 4'5 6')	
50 (2 2'4 6)	170 (2 2'3 3'4 4'5)	
52 (2 2'5 5')	180 (2 2'3 4 4'5 5')	
66 (2 3'4 4')	187 (2 2'3 4'5 5'6)	
77 (3 3'4 4')	188 (2 2'3 4'5 6 6')	
87 (2 2'3 4 5')	195 (2 2'3 3'4 4'5 6)	
101 (2 2'3 5 5')	200 (2 2' 3 3' 4 5 6 6')	
104 (2 2'4 6 6')	206 (2 2'3 3'4 4'5 5'6)	
105 (2 3 3'4 4')	209 (2 2'3 3'4 4'5 5'6 6')	
118 (2 3'4 4'5)		
<b>Major elements</b>		
aluminum	sediment	0.18 µg/g
	water	75.0 µg/L
	biota	0.18 µg/g
iron	sediment	0.5 µg/g
	water	20.0 µg/L
	biota	0.5 µg/g
manganese	sediment	0.01 µg/g
	water	0.50 µg/L
	biota	0.01 µg/g

Table 3.2-1 (continued). Target analytes for chemical characterization for the McAllister Point Landfill Marine Ecological Risk Assessment.

Analyte	Sample matrix	Target method detection limits <sup>a</sup>
<b>Trace elements</b>		
copper	sediment	0.01-0.7 µg/g
nickel	water	0.5-3.0 µg/L
chromium	biota	0.01-0.7 µg/g
lead		
silver		
cadmium	sediment	0.05 µg/g
	water	0.20 µg/L
	biota	0.005 µg/g
zinc	sediment	0.003 µg/g
	water	0.10 µg/L
	biota	0.003 µg/g
arsenic	sediment	0.08 µg/g
	water	3.0 µg/L
	biota	0.08 µg/g
mercury	sediment	0.125 µg/g
	water	0.10 µg/L
	biota	0.125 µg/g
<b>Butyltins</b>	sediment	1.0 ng Sn/g
	biota	1.0 ng Sn/g
monobutyltin		
dibutyltin		
tributyltin		

<sup>a</sup> Sediments and tissues measured on a dry weight basis.

<sup>b</sup> congener number (position of chlorines)

**Table 3.2-2. Assessment and measurement endpoints for the McAllister Point Landfill Marine Ecological Risk Assessment.**

<b>Assessment Endpoint/Receptor</b>	<b>Measurement Endpoint</b>
<b>Vitality of Pelagic Community:</b> Blue Mussel Cunner Winter Flounder	Fecal Pollution Indicator Residues in Mussels Tissue Residues Sediment Elutriate Toxicity
<b>Vitality of Epibenthic Community:</b> Blue Mussel Lobster Benthic Community	Indigenous Blue Mussel Condition, Tissue Residues Lobster Tissue Residues Community Structure of Mussel Beds Elutriate Toxicity to Sea Urchin Gametes Bulk Sediment Chemistry
<b>Vitality of Infaunal Community:</b> Hard Clams Benthic Community	Hard Clam Condition and Tissue Residues Porewater Toxicity to Sea Urchin Gametes Bulk Sediment Toxicity to Amphipods Infaunal Benthic Community Structure Ammonia, Grain Size, Organic Carbon Sediment Chemistry, SEM Bioavailability Sediment Fecal Pollution Indicator Concentration

Table 3.3-1. Sediment benchmarks for target analytes and derived Screening Values for the McAllister Point Landfill Marine Ecological Risk Assessment.

		Sediment Benchmark <sup>1</sup>					
Group	Target Analyte <sup>2</sup>	AET	AL	ER-L	ER-M	SQC	SV
Metals	Arsenic	57		8.2	70		8.2
	Cadmium	5.1		1.2	9.6		1.2
	Chromium	260		81.0	370		81.0
	Copper	390		34.0	270		34.0
	Lead	450		46.7	218		46.7
	Mercury	0.41		0.15	0.71		0.15
	Nickel	140		20.9	51.6		20.9
	Silver	6.1		1.0	3.7		1.0
	Zinc	410		150	410		150
	PAHs	1,6,7-Trimethylnaphthalene					
1-Methylnaphthalene							NA
1-Methylphenanthrene							NA
2,6-Dimethylnaphthalene							NA
2-Methylnaphthalene				70.0	670		70.0
Acenaphthene		500	1300	16.0	500	1300	16.0
Acenaphthylene		1300	71000	44.0	640		44.0
Anthracene		960	580	85.3	1100		85.3
Benzo(a)anthracene		1300	4000	261	1600		261
Benzo(a)pyrene		1600	73000	430	1600		430
Benzo(b,j,k)fluoranthene		3200	3800				3200
Benzo(e)pyrene							NA
Benzo(g,h,i)perylene							NA
Biphenyl							NA
Chrysene			409000	384	2800		384
Dibenz(a,h)anthracene				63.4	260		63.4
Fluoranthene		1700	6200	600	5100	6200	600
Fluorene		540	2000	19.0	540		19.0
High Molecular Weight PAHs				1700	9600		1700
Indeno(1,2,3-cd)pyrene							NA
Low Molecular Weight PAHs				552	3160		552
Naphthalene		2100	11000	160	2100		160
Perylene							NA
Phenanthrene	1500	1800	240	1500	1800	240	
Pyrene	2600	97000	665	2600		665	
Total PAHs			4022	44792		4022	
PCBs	PCB Sum of Congeners x 2			22.7	180		22.7
Pesticides	Aldrin		2.0				2.0
	Hexachlorobenzene	22.0	6000				22.0
	Mirex						NA
	o,p'-DDE						NA
	p,p'-DDE <sup>3</sup>			2.2	27.0		2.2
TBT	Dibutyltin						NA
	Monobutyltin						NA
	Tetraibutyltin						NA
	Tributyltin <sup>4</sup>			20.0	40.0		20.0

1 - Benchmark units (dry wt): Metals (MET) - µg/g; PAHs, PCBs, Pesticides - ng/g; Butyltins (TBT) ng Sn/g.

2 - Analytes measured by Quinn *et al.* (1994) and in present study.

3 - ER-M Benchmark for p,p'-DDE assumed to be the same as for o,p'-DDE.

4 - Benchmark for tributyltin assuming 2% TOC taken from U.S. EPA, 1997.

AET = Apparent Effects Threshold (PTI Environmental Services, 1988).

AL = Equilibrium Partitioning- Aquatic Life (based on 1 % TOC) ( U.S. EPA, 1989b, Adams, Kimerle and Barnett, 1992).

ER-L = NOAA Effects Range-Low (Long *et al.*, 1995).

ER-M = NOAA Effects Range-Median (Long *et al.*, 1995).

SQC = EPA Sediment Quality Criteria (U.S. EPA, 1993a,b,c).

SV = Minimum of Benchmarks.

NA = Benchmark not available.

Table 3.3-2. Sediment Data Summary and Selection of Contaminants of Concern (CoC) for the McAllister Point Landfill Marine Ecological Risk Assessment. <sup>1</sup>

SEDIMENT<sup>2</sup>

Class	Analyte	Frequency of Detection at Site			Range of Site Concentration <sup>a</sup>		Mean Site Concentration <sup>b</sup>	Site 95% Upper Confidence Limit	Mean Reference Concentration <sup>b</sup>	Minimum Benchmark <sup>c</sup>	95% UCL or Max. Concentration <sup>d</sup>		Frequency of Detection > 5%	Is Target Analyte a CoC?
		# Detects	# Samples	%	Minimum	Maximum					Exceeds Minimu Benchmark?	Exceeds Reference?		
MET	Arsenic	34	46	74%	0.65	42.4	7.5	26.0	0.53	8.2	YES	YES	YES	YES
	Cadmium	46	46	100%	0.05	2.59	0.33	1.21	0.11	1.2	YES	YES	YES	YES
	Chromium	46	46	100%	15.8	195.0	52.4	99.7	31.6	81	YES	YES	YES	YES
	Copper	46	46	100%	1.9	1298.8	87.4	504.2	3.13	34	YES	YES	YES	YES
	Lead	46	46	100%	12.5	595.0	74.2	267.3	23.8	46.7	YES	YES	YES	YES
	Mercury	46	46	100%	0.02	1.22	0.18	0.63	0.05	0.15	YES	YES	YES	YES
	Nickel	46	46	100%	10.6	168.8	36.2	89.0	17.1	20.9	YES	YES	YES	YES
	Silver	46	46	100%	0.01	8.50	0.51	2.62	0.12	1.00	YES	YES	YES	YES
	Zinc	46	46	100%	35.2	2590.0	207.9	904.8	49.6	150	YES	YES	YES	YES
PAH	1,6,7-Trimethylnaphthalene	44	46	96%	0.09	67.4	8.0	29.3	26.6	NA	NA	YES	YES	YES
	1-Methylnaphthalene	37	46	80%	0.00	235.0	20.7	87.4	10.5	NA	NA	YES	YES	YES
	1-Methylphenanthrene	41	46	89%	0.00	602.6	64.0	263.3	181.9	NA	NA	YES	YES	YES
	2,6-Dimethylnaphthalene	45	46	98%	0.25	125.7	15.1	60.2	33.4	NA	NA	YES	YES	YES
	2-Methylnaphthalene	37	46	80%	0.00	220.1	22.6	93.4	10.5	70	YES	-	YES	YES
	Acenaphthene	46	46	100%	0.17	548.4	43.5	199.9	21.3	16	YES	-	YES	YES
	Acenaphthylene	46	46	100%	0.38	56.1	16.5	42.9	71.4	44	NO	-	YES	YES
	Anthracene	45	46	98%	0.00	1260.0	158.2	589.3	461.1	85.3	YES	-	YES	YES
	Benzo(a)anthracene	46	46	100%	1.14	1700.0	236.1	852.9	684.6	261	YES	-	YES	YES
	Benzo(a)pyrene	46	46	100%	1.08	1630.0	217.5	757.4	564.9	430	YES	-	YES	YES
	Benzo(b+k)fluoranthene	46	46	100%	5.63	2930.0	416.9	1423.5	893.6	3200	NO	-	YES	YES
	Benzo(e)pyrene	46	46	100%	2.75	1060.1	160.4	535.5	319.7	NA	NA	YES	YES	YES
	Benzo(g,h,i)perylene	46	46	100%	1.21	685.3	117.6	359.1	335.4	NA	NA	YES	YES	YES
	Biphenyl	45	46	98%	0.23	71.8	10.2	34.6	1.6	NA	NA	YES	YES	YES
	Chrysene	46	46	100%	1.05	1970.0	265.8	938.6	511.3	384	YES	-	YES	YES
	Dibenz(a,h)anthracene	44	46	96%	0.11	262.5	36.3	128.0	62.0	NA	NA	YES	YES	YES
	Fluoranthene	46	46	100%	3.28	3720.0	558.9	1893.3	1677.6	600	YES	-	YES	YES
	Fluorene	44	46	96%	0.09	759.4	57.8	264.3	129.7	19	YES	-	YES	YES
	High Molecular Weight PAHs	46	46	100%	10.07	11773.2	1793.8	5958.6	4973.1	1700	YES	-	YES	YES
	Indeno(1,2,3-cd)pyrene	45	46	98%	0.25	762.4	110.7	365.9	314.8	NA	NA	YES	YES	YES
	Low Molecular Weight PAHs	46	46	100%	5.49	6926.4	735.8	2889.7	1577.8	552	YES	-	YES	YES
	Naphthalene	46	46	100%	0.40	287.7	38.6	141.9	63.4	160	NO	-	YES	YES
	Perylene	46	46	100%	0.82	400.5	61.5	197.5	196.9	NA	NA	YES	YES	YES
Phenanthrene	46	46	100%	1.87	3820.0	401.4	1601.1	854.4	240	YES	-	YES	YES	
Pyrene	46	46	100%	2.62	3010.3	480.6	1573.0	1472.8	665	YES	-	YES	YES	
Total PAHs	46	46	100%	30.69	25628.3	3507.0	12008.4	8859.6	4022	YES	-	YES	YES	

1 - Data summary includes URI/SAIC (1994) data and present study

2 - Concentration and benchmark units (dry wt): Metals (MET) - µg/g; PAHs, PCBs, Pesticides (PST) - ng/g; Butyltins (TBT) ng Sn/g.

a - The range of concentrations reported for site data excludes non-detected values.

b - 1/2 Sample Quantitation Limits substituted for non-detects when calculating mean of site and reference station data.

c - Minimum benchmark = NOAA ER-L (Long et al., 1995), except for benzo(b+k)fluoranthene, aldrin and hexachlorobenzene = AET or AL; see Table 3.3-1.

d - If 95% UCL is greater than the Maximum Concentration, as indicated with a "+", then Maximum Concentration is used to screen against benchmark or background.

NA = Benchmark Not Available

- = Site concentrations of organic contaminants were compared to reference concentrations only when no appropriate benchmark was available.

Table 3.3-2 (continued). Sediment Data Summary and Selection of Contaminants of Concern (CoC) for the McAllister Point Landfill Marine Ecological Risk Assessment. <sup>1</sup>

SEDIMENT<sup>2</sup>

Class	Analyte	Frequency of Detection at Site			Range of Site Concentration <sup>a</sup>		Mean Site Concentration <sup>b</sup>	Site 95% Upper Confidence Limit	Mean Reference Concentration <sup>b</sup>	Minimum Benchmark <sup>c</sup>	95% UCL or Max. Concentration <sup>3</sup>		Frequency of Detection > 5%	Is Target Analyte a COC?		
		# Detects	# Samples	%	Minimum	Maximum					Exceeds Minimum Benchmark?	Exceeds Reference?				
PCB	101 (2,2',4,4',6')	45	46	98%	0.08	17.7	3.1	10.1	0.4	NA	NA	YES	YES	YES		
	104 (2,2',4,6,6')	45	46	98%	0.05	1.2	0.3	0.7	0.2	NA	NA	YES	YES	YES		
	105 (2,3,3',4')	44	46	96%	0.14	13.5	1.7	6.4	0.2	NA	NA	YES	YES	YES		
	118 (2,3',4,4',5)	44	46	96%	0.05	22.9	3.3	11.8	0.4	NA	NA	YES	YES	YES		
	126 (3,3',4,4',6)	42	46	91%	0.02	3.5	0.5	1.9	0.1	NA	NA	YES	YES	YES		
	128 (2,2',3,3',4')	43	46	93%	0.06	5.4	0.9	3.2	0.1	NA	NA	YES	YES	YES		
	138 (2,2',3,4,4',5)	46	46	100%	0.05	20.2	3.8	12.1	0.5	NA	NA	YES	YES	YES		
	153 (2,2',4,4',5,5')	46	46	100%	0.06	11.8	2.8	8.2	0.5	NA	NA	YES	YES	YES		
	170 (2,2',3,3',4',5)	39	46	85%	0.16	5.1	1.3	3.2	0.6	NA	NA	YES	YES	YES		
	18 (2,2',5)	42	46	91%	0.06	24.6	2.6	11.7	0.3	NA	NA	YES	YES	YES		
	180 (2,2',3,4,4',5,5')	45	46	98%	0.05	10.5	2.0	5.8	0.5	NA	NA	YES	YES	YES		
	187 (2,2',3,4',5,5',6)	44	46	96%	0.05	7.9	1.3	4.1	0.3	NA	NA	YES	YES	YES		
	188 (2,2',3,4',5,5',6)	46	46	100%	0.03	2.5	0.5	1.6	0.2	NA	NA	YES	YES	YES		
	195 (2,2',3,3',4,4',5,6)	36	46	78%	0.05	1.5	0.4	0.9	0.3	NA	NA	YES	YES	YES		
	200 (2,2',3,3',4,5',6,6')	45	46	98%	0.01	2.5	0.3	1.0	0.1	NA	NA	YES	YES	YES		
	206 (2,2',3,3',4,4',5,5',6)	45	46	98%	0.07	33.6	2.1	10.7	0.4	NA	NA	YES	YES	YES		
	209 (2,2',3,3',4,4',5,5',6,6')	46	46	100%	0.06	8.6	1.0	3.4	0.3	NA	NA	YES	YES	YES		
	28 (2,4,4')	42	46	91%	0.06	18.8	2.9	10.2	0.1	NA	NA	YES	YES	YES		
	29 (2,4,5)	44	46	96%	0.05	1.5	0.2	0.7	0.3	NA	NA	YES	YES	YES		
	44 (2,2',3,5')	40	46	87%	0.06	11.3	1.7	6.3	0.3	NA	NA	YES	YES	YES		
	50 (2,2',4,6)	46	46	100%	0.04	12.9	1.4	6.3	0.1	NA	NA	YES	YES	YES		
	52 (2,2',5,5')	44	46	96%	0.06	14.8	2.2	8.3	0.5	NA	NA	YES	YES	YES		
	66 (2,3',4,4')	39	46	85%	0.11	10.1	2.3	6.6	0.3	NA	NA	YES	YES	YES		
	8 (2,4)	39	46	85%	0.07	9.1	1.7	5.6	0.3	NA	NA	YES	YES	YES		
	87 (2,2',3,4,5')	46	46	100%	0.06	12.1	1.5	5.8	0.1	NA	NA	YES	YES	YES		
	Total PCBs (Sum Congeners)	46	46	100%	3.23	547.7	102.1	324.1	13.8	22.7	YES	-	YES	YES	YES	
	PST	Aldrin	1	46	2%	0.00	0.2	0.0	1.8	0.0	2.0	+	NO	-	NO	NO
		Hexachlorobenzene	27	46	59%	0.00	3.0	0.7	1.1	0.3	22	NO	-	YES	YES	YES
Mirex		30	46	65%	0.00	1.6	0.5	4.4	0.1	NA	NA	YES	YES	YES	YES	
o,p'-DDE		28	46	61%	0.00	8.9	1.2	4.2	0.4	NA	NA	YES	YES	YES	YES	
p,p'-DDE		44	46	96%	0.00	8.5	1.4	4.2	0.4	2.2	YES	-	YES	YES	YES	
TBT	Dibutyltin	26	46	57%	0.00	3.8	0.9	2.4	0.8	NA	NA	YES	YES	YES	YES	
	Monobutyltin	24	46	52%	0.00	8.9	1.1	4.0	0.2	NA	NA	YES	YES	YES	YES	
	Tetra-butyltin	24	46	52%	0.10	3.2	0.3	1.3	0.0	NA	NA	YES	YES	YES	YES	
	Tributyltin	27	46	59%	0.10	8.6	2.4	6.3	3.5	NA	NA	YES	YES	YES	YES	

1 - Data summary includes URI/SAIC (1994) data and present study

2 - Concentration and benchmark units (dry wt): Metals (MET) - µg/g; PAHs, PCBs, Pesticides (PST) - ng/g; Butyltins (TBT) ng Sn/g.

a - The range of concentrations reported for site data excludes non-detected values.

b - 1/2 Sample Quantitation Limits substituted for non-detects when calculating mean of site and reference station data.

c - Minimum benchmark = NOAA ER-L (Long *et al.*, 1995), except for benzo(b+k)fluoranthene, aldrin and hexachlorobenzene = AET or AL; see Table 3.3-1.

d - If 95% UCL is greater than the Maximum Concentration, as indicated with a "+", then Maximum Concentration is used to screen against benchmark or background.

NA = Benchmark Not Available

- = Site concentrations of organic contaminants were compared to reference concentrations only when no appropriate benchmark was available.

Table 3.4-1. Target ecological systems/species/receptors of concern for the McAllister Point Landfill Marine Ecological Risk Assessment.

Habitat	Ecological System/Species/Receptor of Concern
Pelagic	blue mussel ( <i>Mytilus edulis</i> ) cunner ( <i>Tautoglabrus adspersus</i> ) winter flounder ( <i>Pseudopleuronectes americanus</i> )
Epibenthic	blue mussel ( <i>Mytilus edulis</i> ) lobster ( <i>Homarus americanus</i> ) winter flounder ( <i>Pseudopleuronectes americanus</i> ) benthic community in mussel beds
Infaunal	hard clams ( <i>Mercenaria mercenaria</i> , <i>Pitar morrhuana</i> ) infaunal benthic community
Avian Aquatic	great blue heron ( <i>Ardea herodias</i> ) herring gull ( <i>Larus argentatus</i> )

Table 3.6-1. Summary of data for the McAllister Point Landfill Marine Ecological Risk Assessment.

PHASE	STATION	Sediment Chemistry							Tissue Chemistry							Geotechnical						Biological/Toxicological Tests							
		Bulk Sediment			Forewater <sup>1</sup>	SEM and AVS			Mussels		Hard Clams		Lobster		Fish	Grain Size			TOC			Toxicity		Field Biota			Micro		
		S	M	D	S	S	M	D	DEP	ND	DEP	ND	MUS	HPP	CN	S	M	D	S	M	D	ARB	AMP	CI BM	CI HC	DIV	SED	BM	HC
PHASE 1	OS-30A	1				1									1			1			1	1							
PHASE 1	OS-30B	1				1									1			1			1	1							
PHASE 1	D1	1				1									1			1			1	1							
PHASE 1	D2	1				1									1			1			1	1							
PHASE 1	D3	1				1									1			1			1	1							
PHASE 1	M1	1				1									1			1			1	1							
PHASE 1	M2	1				1									1			1			1	1							
PHASE 1	M3	1				1									1			1			1	1							
PHASE 1	S1	1				1									1			1			1	1							
PHASE 1	S2B	1				1									1			1			1	1							
PHASE 1	S3	1				1									1			1			1	1							
PHASE 1	S4	1				1									1			1			1	1							
PHASE 1	JCC-D1	1				1									1			1			1	1							
PHASE 1	JCC-M1	1				1									1			1			1	1							
PHASE 1	JCC-S1	1				1									1			1			1	1							
PHASE 2	NSB-1	1			1	1			1	1					1	1		1			1	1	1		1	1	1	1	1
PHASE 2	NSB-2	1			1	1			1	1					1	1		1			1	1	1	1	1	1	1	1	1
PHASE 2	NSB-3	1			1	1			1	1					1	1		1			1	1	1	1	1	1	1	1	1
PHASE 2	NSB-4	1			1	1			1	1					1	1		1			1	1	1	1	1	1	1	1	1
PHASE 2	NSB-5	1			1	1			1	1					1	1		1			1	1	1	1	1	1	1	1	1
PHASE 2	NSB-6	1			1	1			1	1					1	1		1			1	1	1	1	1	1	1	1	1
PHASE 2	NSB-7	1			1	1			1	1					1	1		1			1	1	1	1	1	1	1	1	1
PHASE 2	NSB-8			1																									
PHASE 2	MCL-8	1			1	1									1			1			1	1	1						
PHASE 2	MCL-9	1	1	1	1	1	1	1				1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
PHASE 2	MCL-10	1	1	1	1	1	1	1			1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
PHASE 2	MCL-11	1	1	1	1	1	1	1			1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
PHASE 2	MCL-12	1	1	1	1	1	1	1			1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
PHASE 2	MCL-13	1	1	1	1	1	1	1				1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
PHASE 2	MCL-14	1	1	1	1	1	1	1				1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
PHASE 2	MCL-15	1	1	1	1	1	1	1							1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
PHASE 2	MCL-16	1			1	1	1	1			1	1			1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
PHASE 2	S2B	1																											
PHASE 2	JCC-S1																												
PHASE 2	JCC-M1	1				1									1						1	1							
PHASE 2	JCC-D1		1	1				1																					
PHASE 2	JPC-1														1														

Table 3.6-1 (continued). Summary of data for the McAllister Point Landfill Marine Ecological Risk Assessment.

PHASE	STATION	Sediment Chemistry							Tissue Chemistry					Geotechnical			Biological/Toxicological Tests												
		Bulk Sediment			Porewater <sup>1</sup>	SEM and AVS			Mussels		Hard Clams		Lobster	Fish	Grain Size			TOC		Toxicity		Field Biota			Micro				
NUM	NUM	S	M	D	S	S	M	D	DEP	ND	DEP	ND	MUS	HPP	CN	S	M	D	S	M	D	ARB	AMP	CI	CI	DIV	SED	BM	HC
PHASE 3	MCL-10-R	1		1												1		1	1			1	1						
PHASE 3	MCL-11-R	1														1			1			1							
PHASE 3	MCL-12-R	1		1												1		1	1			1	1						
PHASE 3	MCL-13-R	1														1			1			1							
PHASE 3	MCL-14-R	1														1			1			1							
PHASE 3	MCL-8-R	1														1			1			1							
PHASE 3	MCL-9-R	1														1			1			1							
PHASE 3	NSB-1-R	1														1			1			1							
PHASE 3	NSB-2-R	1		1												1		1	1			1	1						
PHASE 3	NSB-2FD-R	1														1		1	1										
PHASE 3	NSB-3-R	1		1												1			1										
PHASE 3	NSB-4-R	1		1												1		1	1			1	1						
PHASE 3	NSB-5-R	1		1												1		1	1			1	1						
PHASE 3	NSB-6-R	1		1												1		1	1			1	1						
PHASE 3	NSB-7-R	1														1			1			1							
PHASE 3	M1-R	1														1			1			1							
PHASE 3	S2B-R	1														1			1			1							
PHASE 3	S2C	1														1			1			1							
TRC/BOS <sup>2</sup>	OS-22	1				1						1							1										
TRC/BOS	OS-23	1				1						1							1										
TRC/BOS	OS-24	1				1						1							1										
TRC/BOS	OS-25	1				1						1							1										
TRC/BOS	OS-26	1				1						1							1										
TRC/BOS	OS-27	1				1						1							1										
TRC/BOS	OS-28	1				1						1							1										
TRC/BOS	OS-29	1				1													1										
TRC/BOS	OS-30	1				1													1										
TRC/BOS	JPC-1											1																	
TOTAL	TOTAL	60	8	16	16	41	9	9	2	8	5	17	5	5	5	50	9	17	58	8	8	49	40	8	4	15	8	4	1

OS = Offshore  
 JCC = Jamestown Cranston Cove  
 JPC = Jamestown Potter Cove  
 S = Nearshore  
 M = Mid-depth  
 D = Deep  
 "R" = Phase III resampling station  
 "FD" = field duplicate  
 1 = Metals Only  
 2 = Data from TRC, 1994  
 SEM/AVS = Simultaneously Extractable Metals/Acid Volatile Sulfides

DEP = Depurated  
 ND = Non-Depurated  
 BM & MUSSELS = *Mytilus edulis*  
 HC & HARD CLAM = *Mercenaria mercenaria* and *Pitar morrhuana*  
 LOBSTER = *Homarus americanus*  
 MUS = Lobster muscle  
 HPP = Lobster hepatopancreas  
 CN & FISH = cunner, except Phase II JPC-1 mummichog (*Fundulus* spp.)

GRAIN SIZE = Sediment Grain Size  
 TOC = Sediment Total Organic Carbon  
 AMP = Whole Sediment Amphipod Survival Test with *Ampelisca*  
 ARB = Sediment Porewater Test With *Arbacia*  
 CI = Bivalve Condition Index  
 DIV = Benthic Community Structure  
 Micro = Fecal Pollution Indicators  
 SED = Sediment

## 4.0. EXPOSURE ASSESSMENT

Exposure assessment in the McAllister Point Landfill investigation involves the evaluation of the site-specific conceptual model with respect to hypothesized exposure pathways and includes the direct measurement of exposure point concentrations along these pathways. For this assessment, McAllister Point Landfill is considered to be the primary (but not proximal) source of CoCs in nearshore areas. In addition to direct measurement of chemistry, other exposure measures (identified in Table 3.2-2) are assessed to aid in the interpretation of chemical exposure conditions. Methods and QA/QC considerations and protocols relevant to analytical chemistry are presented in the master Work Plan and in Section 3.6 above (detailed QA/QC information is presented in Appendix C).

Exposure information derived from previous investigations at the site have been evaluated for applicability to this assessment and used as appropriate. Accompanying the description of these data is a discussion of the comparability of the various data sets as well as an evaluation of the uncertainty associated with the exposure analyses.

Exposure Assessment results are described below in three sections: an examination of sources and exposure pathways of CoCs (Section 4.1), estimates of exposure point concentrations and analyses of fate and transport for CoCs (Section 4.2), and an analysis of the uncertainty related to the exposure assessment (Section 4.3).

#### 4.1. SOURCES AND EXPOSURE PATHWAYS OF CoCs

Several exposure pathways are likely to exist from contaminant sources associated with historical activities at McAllister Point Landfill. Early characterization studies of landfill contaminants (discussed in Section 3.1) have concluded that PAHs, PCBs, numerous metals, and the chlorinated pesticide, p,p'-DDE, were present in concentrations that represent significant ecological risk potentials. These results were supported by analyses of soil, ground water, and seep water samples.

Sources and exposure pathways for contaminants from the landfill to the marine environment and associated biota were introduced in Section 3.5 as a series of conceptual models. First Tier exposure pathways are related to the relative magnitude of site-specific sources vs. regional sources. Initial exposure pathways as defined by the Second Tier model are concluded to occur primarily via surface and ground water flows from the landfill. The Third Tier model describes the behavior of dissolved and particle-bound contaminants in the aquatic environment, including transport by and/or association with surface water, sediments, pore water, and biota. Finally, the Fourth Tier model identifies sources and exposure pathways for biological receptors, including: surface water exposures of pelagic organisms such as fish and filter-feeding infauna and epifauna; soil (particle), sediment, and pore water exposures to bottom-dwelling fish, infauna and epifauna; and the potential for fish and invertebrate prey to function as proximal sources and exposure points for upper level predators such as fish-eating birds.

Contaminant exposure routes for aquatic biota can involve exposure through water, sediments, and pore water via partitioning across cell membranes, incidental contact or feeding mode ingestion of sediments (e.g., by bottom deposit-feeding invertebrates), and consumption of contaminated prey. Thus, it is important to identify the behavior and potential effects of CoCs as a key part of the risk assessment. Based

on the general models described above, a more detailed evaluation of exposure pathways can be derived for specific classes of CoCs as related to their chemical and physical behavior, and characteristics such as specific bioaccumulation potentials. The toxicity of CoCs is addressed in this section, as well as in Section 5.1.

Some organic contaminants identified in source samples, including the organochlorinated pesticides (OCPs) such as p,p'-DDE and the polychlorinated biphenyls (PCBs), share similar properties in that they are characterized by relatively low solubilities in water and high solubilities in lipid phases (e.g., many animal tissues). The low water solubilities tend to result in a net transfer of such compounds from aqueous to particulate phases, with subsequent accumulation in sediments and to a lesser degree, pore water (via partitioning; Clayton *et al.*, 1977). Transfer of this type of CoC to organisms living on or in the sediments can occur through direct uptake (e.g., dermal contact or sediment ingestion), through partitioning to interstitial pore water, or through food web transfer. Because of the tendency for these compounds to remain adsorbed to sediments, there should be relatively low dissolved-phase concentrations above the sediments, thereby minimizing direct exposures to pelagic organisms via the water column.

It is notable that respiratory surfaces of water-breathing organisms, such as fish and invertebrates, provide an effective transfer mechanism for these lipid-soluble organic contaminants between the aqueous environment and lipid-rich tissues. Thus, the concentrations of highly lipid-soluble organic contaminants in these organisms may be somewhat controlled by these transfer mechanisms. Consequently, contaminant concentrations in these species may be more dependent on the lipid content as related, for example, to reproductive condition, than on magnification of the chemical within a food web (Clayton *et al.*, 1977). In contrast to water-breathing organisms, air-breathing organisms associated with aquatic environments (e.g., water fowl or aquatic predatory birds) do not have external surfaces that readily facilitate the transfer of lipid-soluble

chemicals between internal lipid and external water phases. Consequently, such chemicals are more susceptible to biomagnification in these species. As noted in Clayton *et al.* (1977), concentrations of contaminants such as PCBs in water-breathing biota from different trophic levels (e.g., zooplankton, herring, and salmon) can be very similar when the values are lipid-normalized. In contrast, concentrations in air-breathing aquatic biota (e.g., birds, seals) can vary widely among species and be considerably higher than in water-breathing biota.

Other organic contaminants, particularly PAHs, also tend to have low water solubilities (solubility decreases with increasing molecular weight) and primarily are found associated with particles and sediments (Pruell and Quinn, 1986). Thus, the principal risk from PAHs would be to bottom-dwelling fish and invertebrates, including filter-feeders that ingest PAH-laden particles. However, in contrast to chlorinated compounds such as PCBs, there appears to be a reduced association of PAHs with lipid-rich tissues (Tracey and Hansen, 1996). Because PAH exposures tend to derive primarily from weathered sources (e.g., combusted fossil fuels), these compounds may be more highly particle-associated and/or bioavailable than would be predicted from this chemical structure (Tracey and Hansen, *ibid*). In addition, marine vertebrates, (e.g., fish) are very capable of metabolizing PAHs. These factors perhaps explain why this compound class is not bioaccumulated to the same extent as other lipophilic organics. The primary effects from PAHs are as carcinogens, particularly at the point of contact, as influenced by the formation of metabolic intermediates.

Metals, such as silver, lead, zinc, arsenic, manganese, mercury, and chromium(+3), all are relatively insoluble in aqueous media and tend to be associated with particles and sediments. Thus, organism exposure pathways are expected to be similar to those noted for the organic contaminants. In contrast, nickel, copper, cadmium, and to a lesser extent, chromium(+6), are relatively soluble and characteristically are associated with dissolved phases. However, various complex

reactions ultimately result in the deposition of copper in bottom sediments; additionally, methylation can result in releases of arsenic from sediments back into the water column. Exposure pathways for dissolved phases primarily would include water column effects to pelagic organisms and filter feeders. Toxicity responses are highest for copper, mercury, silver, chromium(+6), and to a lesser extent zinc, manganese, and arsenic. It is notable that most of the chromium in aquatic environments occurs as chromium(+3), therefore substantially reducing the potential toxicity. Copper toxicity is greatest in fish and invertebrates, but its toxicity is moderated substantially in higher animals due to homeostatic mechanisms that limit adsorption. Mercury is of substantial concern because of high potentials for bioconcentration and magnification of methyl mercury within food webs. Biomagnification of lead (i.e., a progressive increase in concentrations from the source of exposure through the trophic levels) does not appear to occur in aquatic organisms, such that primary consumers such as bivalves will tend to have higher lead in tissues than predatory fish (Paine, 1995). Manganese may be associated with bioaccumulation in fish, but not in higher animals (also due to homeostatic compensating mechanisms). Finally, carcinogenic responses have been documented for several metals including chromium (+6), arsenic, nickel and, potentially, cadmium and lead.

#### **4.2. ESTIMATE OF EXPOSURE POINT CONCENTRATIONS/FATE AND TRANSPORT ANALYSIS**

This section evaluates the spatial distribution and concentration of contaminants in bottom sediments and biological tissues to describe the possible fate and transport of contaminants from the McAllister Point landfill to receptors of concern. Comparative information on a station-specific basis is available mainly from data collected during the Phase I and Phase II URI/SAIC studies (URI/SAIC, 1994; SAIC/URI, 1995). The sections below present data obtained from sediment geotechnical characterization (grain size and total organic carbon, Section 4.2.1), and the analysis of organic

(Section 4.2.2) and inorganic (Section 4.2.3) contaminants in offshore sediments and organisms from the McAllister Point Landfill study area.

During the Phase II investigation, sediment samples were collected from a series of intertidal nearshore (NSB), shallow water nearshore (S), and subtidal offshore (M, D, and MCL) locations off of the McAllister Point Landfill, and at three reference sites along a depth transect in Jamestown Cranston Cove (JCC). All station locations are shown in Figure 3.6-2. The surficial sediment (upper 0–2 cm or 0–6 cm) samples collected at these stations represent recently deposited materials. Subsurface sediments (from >10 cm in depth) were also collected by piston coring at nine of the subtidal offshore stations to enable evaluation of the contaminant distribution in subsurface sediment layers. It was not possible to obtain a core at Station MCL-8 because the sediments of the site consist of ~10 cm or less of sand and gravel overlying or interspersed between gravel and large rocks. Surface sediments at nearshore Stations NSB-1 through NSB-7, offshore Stations MCL-8 through MCL-12 and M1, Station S2B, and a new Station S2C were resampled for Phase III in order to provide comparison of pre- and post-revetment conditions. In addition, core sediments from Stations NSB-2 through NSB-6, MCL-10, and MCL-12 were sampled for the Phase III investigation. Sampling locations are shown in Figure 3.6-3.

#### 4.2.1. Sediment Geotechnical Characterization

*Grain size/Total Organic Carbon.* The sediment proximal to McAllister Point Landfill and within Jamestown Cranston Cove are very coarse grained. In fact, gravel sized material (small pebbles to boulders) is common near McAllister Point (TRC, 1994). The fraction of material finer than gravel (i.e., <5.00 mm) was analyzed in this study because the sand, silt, and clay sediment fractions actually contain the contaminants of concern. In addition, the gravel size fraction of McAllister Point sediments commonly contains metallic debris (nuts, bolts, cables, etc.) that is unlikely to

be bioavailable but would anomalously skew to very high values the concentrations of some metal analytes when a total digestion extraction procedure is used. A conservative approach was selected to determine the maximum concentrations of metals that the biota are likely to be exposed to within the study area.

The grain size results are summarized in Figure 4.2-1. Raw data are presented in Appendix A-1-5. The surface sediments of most stations are generally characterized by very high sand content (>80% sand). Only four stations (MCL-10, MCL-11, MCL-16, OS-30B) in the vicinity of McAllister Point have appreciable clay and/or fine silt (i.e., <15  $\mu\text{m}$  size fraction) composition (Figure 4.2-1). These results are consistent with the general regional surficial stratigraphy (Figure 3.1-7) and side-scan sonar survey results (Figure 3.1-8).

The percent of sand versus depth for the piston cores is shown in Figure 4.2-2. The sediments obtained in piston cores near the McAllister Point Landfill generally coarsen with depth and terminate in a layer of either coarse gravel, or highly weathered rock. Exceptions included the observed stratigraphy at Stations MCL-9 and reference location Jamestown Cranston Cove, where underlying material increased in silt/clay content. Areas of thick accumulation (>1 m) of sand size or finer sediment were not observed in the vicinity of McAllister Point Landfill, and hence the subtidal environment proximal to the landfill are generally characterized as non-depositional.

Total organic carbon (% TOC) content of sediments in the McAllister Point Landfill and Jamestown Cranston Cove study areas is illustrated in Figure 4.2-3. Raw data are presented in Appendix A-1-5. With some exceptions, TOC values were generally <2% throughout the study area. Highest TOC values were observed at Stations S2B (core sample, 3.7%), and Station M1 (4.7%). These stations will be shown to also have high CoC concentrations (Section 4.2.2 and 4.2.3). There was no

readily apparent spatial pattern in TOC within the study areas, and no consistent downcore pattern was observed in the piston core samples.

*Comparison to TRC/BOS (1994) Study.* The TOC values in the present study are generally comparable to that observed by TRC/BOS (1994) (range 0.2–2.7%) within the study area. The grain size distributions determined in both studies are similar despite the differences in methodology. Nearshore stations are observed to have significantly coarser grained sediments than offshore stations in both studies.

#### 4.2.2. Organic Contaminants

##### 4.2.2.1. Sediments

During Phase I/II investigations, a total of 32 surface sediments and 20 core sections were analyzed for 27 polychlorinated biphenyl (PCB) congeners, 23 Polycyclic Aromatic Hydrocarbons (PAHs), 5 Organochlorine Pesticides (OCPs) and 4 Butyltins (BTs; Appendix A-1-1; spatial distribution presented in Appendix D-1). The sum of the 27 PCB congeners times two is the Total PCBs (equivalent to the total Aroclors), and the sum of the 23 PAHs is the Total PAHs. For the OCPs, p,p'-DDE was the predominant pesticide detected, and tributyltin (TBT) was the predominant butyltin (BT) found in the samples. All values are reported on a dry weight basis. The analytes o,p'-DDE and aldrin could not be accurately quantified due to analytical interferences from coeluting congeners. However, the sum of these two OCPs were low relative to the p,p'-DDE concentration. Therefore, the absence of quantitative data for o,p'-DDE and aldrin should not reduce the degree of conservatism in assessing pesticides as a source of adverse impact on biota in the McAllister Point study area.

Figure 4.2-4 presents the concentrations of Total PCBs, Total PAHs, p,p'-DDE and TBT in surface sediments from the McAllister Point. For the Total PCBs, nearshore

Stations NSB-3 through NSB-7 had values exceeding the ER-M guidelines of 180 ng/g for Total PCBs and most of the other stations exceeded the ER-L concentration of 22.7 ng/g (Long *et al.*, 1995). About one-third of the sites exceeded the ER-L concentration of 4,022 ng/g for Total PAHs but none was greater than the ER-M of 44,792 ng/g. Highest Total PAHs were observed at Stations S2B (18,800 ng/g) and NSB-3 (10,000 ng/g).

Figure 4.2-4 also shows the concentration trends for p,p'-DDE and TBT. For p,p'-DDE, four stations exceeded the ER-L value of 2.2 ng/g (Long *et al.*, 1995); the highest concentration was observed at Station NSB-3 (8.5 ng/g). TBT values were generally low, ranging from <1 to 5.3 ng Sn/g. There are no ER values for any of the butyltins; however, U.S. EPA has suggested 20 ng Sn/g dry weight at 2% TOC as a lower range-of-effects concentration (U.S. EPA, 1997). Organic carbon normalized values for Total PCBs and Total PAHs showed similar concentration versus station trends as the sediment-based values and, in addition, indicated that reference Station JCC-D1 had an elevated Total PAHs/OC value (Figure 4.2-5) due to a combination of lower organic carbon in the sediment and moderate PAH concentrations. Additionally, organic carbon (OC) normalized concentrations of p,p'-DDE and TBT showed similar concentration versus station trends as sediment-based values (Figure 4.2-5).

The distributions of individual PCB congeners for Stations S2B, NSB-3 and NSB-5 are illustrated in Figure 4.2-6. These stations were selected because of their elevated concentrations relative to the other stations. For PCBs, the distribution is very similar at all three stations; the major congeners were the 3 to 6 chlorine PCBs: 28, 66, 101, 77/154 (77 could not be confirmed by GS/MS), 118, 153 and 138, probably coming from AR 1254, the major Aroclor formulation in Narragansett Bay surface sediments, as well as smaller amounts of AR 1260 (Latimer *et al.*, 1991; Quinn *et al.*, 1992). Major sources of PCBs to Narragansett Bay include rivers, combined sewer overflow (CSO) / sewage discharges and atmospheric deposition (Latimer and Quinn, 1995).

The major PAH components in sediments from Stations S2B, NSB-3, and NSB-7 were the three- to five-ring pyrogenic compounds phenanthrene, fluoranthene, pyrene, chrysene, and benzo(BJK)fluoranthene (Figure 4.2-7; Pruell & Quinn, 1988; Quinn *et al.*, 1992 and TRC, 1994). Sources of these compounds include combustion products from used motor oil and atmospheric deposition; creosote/coal tar and asphalt from local activities; rivers and land runoff; and sewage effluents and overflows (Latimer and Quinn, 1995). Stations NSB-3 and NSB-7 also showed qualitative evidence of unweathered petroleum hydrocarbons, probably from diesel and/or bunker fuel. This was also reflected in the elevated concentrations of four- and five-ring PAHs.

Concentrations of organic contaminants in sediment cores, as well as surface sediments, from representative stations are presented in Table 4.2-1. In general, the results for the nine stations suggest markedly reduced CoC concentrations relative to surface concentrations below a depth of 35 cm. For example, at Station S2B, the levels of p,p'-DDE and PAHs decreased with depth from relatively high values at the core surface (0-8 cm) to low concentrations at a depth of 30-45 cm. Also, surficial sediment (0-2 cm) CoC concentrations at Station S2B were less than those found for the top layer in the sediment core (0-8 cm), suggesting a subsurface maximum between 2-8 cm. However, Total PCBs and TBT exhibited a subsurface maximum at 8-15 cm. The reference station at Jamestown Cranston Cove (JCC-D1) showed a decrease in concentration with depth for Total PAHs, from 44,600 ng/g at 0-8 cm, the highest level found in the entire study, to 35 ng/g at 45-65 cm. The PAH distribution at this depth was dominated by three- to five-ring pyrogenic components and the qualitative analysis of petroleum hydrocarbons in the sample suggests that possible sources are weathered petroleum product(s) or creosote/coal tar hydrocarbons. The presence of TBT in cores suggests that the deposition occurred after 1960, when the use of this compound in manufacturing began.

*Comparison with Historical Data.* A confirmation study at the McAllister Point Landfill was conducted in the mid 1980s during which leachate springs, ground water, soils, sediments, and blue mussels were analyzed for a variety of chemical contaminants (Loureiro Engineering Associates, 1986). PCBs were present in mussels near background levels but were absent from the sediments. The U.S. Army Corps of Engineers also conducted a survey of sediment (top 10-20 cm) and blue mussels in the vicinity of the landfill in 1988. PCBs were detected above background levels in mussels (0.01 to 0.03 ppm) and sediments (0.01 to 2.03 ppm), and total petroleum hydrocarbons were detected in the sediments at the 30 to 1,100 ppm range (U.S. Army Corps of Engineers, 1988).

*Comparison with TRC/BOS data for McAllister Point.* The TRC/BOS study reported (TRC, 1994) values for 20 of the 27 PCB congeners measured by URI/SAIC. Highest concentrations were found at Stations NS-13/14/15 and NS-16/17/18 (184 to 582 ng/g). In comparison, the highest URI/SAIC values ranged from 368 ng/g to 484 ng/g at Stations NSB-3 to NSB-5, located in approximately the same area as that in which the highest TRC/BOS study concentrations were observed. Thus, there exists generally good agreement between these studies.

Total PAH concentrations (sum of 24 PAHs) for the TRC/BOS study ranged from 43 to 22,100 ng/g dry weight of sediment, with highest levels at Stations OS-28 (21,100 ng/g) and NS-19/20/21 (9,500 ng/g). URI/SAIC values (Total PAHs = sum of 23 PAHs) ranged from 18,800 ng/g at Station S2B (close to OS-28) and 3,020 to 4,170 ng/g at NSB-5 to NSB-6 (close to NS-19/20/21). These results also suggest generally good agreement between these studies.

*Comparison of pre- and post-erosion conditions.* The purpose of the comparison between Phase II (pre-erosion) and Phase III (post-erosion) results is to assess whether sediment erosion (discussed in Section 3.1) from the nearshore environment of

McAllister Point Landfill had increased possible CoC exposure to aquatic biota. Concentrations of Total PCBs and Total PAHs measured in surface sediments and sediment cores during Phase III are presented in Table 4.2-2 and compared to chemical concentrations found at the same stations prior to the erosion event. To facilitate the evaluation, station data for which concentrations increased from 1995 to 1996 by a Relative Percent Difference (RPD) greater than 30% were considered significant and are presented within dark bordered cells in Table 4.2-2. In addition, the elevated concentrations (as defined above) were compared against ER-L and ER-M guidelines (Long *et al.*, 1995; light and dark shading, respectively).

For PCBs, stations with significant increases (RPD  $\geq$  +30%) and values above the ER-M included intertidal surface sediments from Stations NSB-4, NSB-5 and NSB-7 and both surficial and subsurface (0-18 cm) sediment at offshore Station MCL-12 (Table 4.2-2). Increased concentrations to levels above the ER-L were observed for PCBs at Stations NSB-1 and NSB-2, and for PAHs at Stations NSB-6 and MCL-12 (surface and core). The distribution of individual PAH components were generally similar both within and between stations.

Direct comparisons for core data for the nearshore stations was not possible since pre-erosion cores were not collected in this area. In comparison to surface sediments, however, Station NSB-2 showed a significant increase in the concentration of PCBs in the core samples (Table 4.2-2). Stations NSB-2 through NSB-4 also had higher concentrations of PAHs in the core samples relative to surface sediments.

#### 4.2.2.2. Tissue Residues

A total of 38 tissue samples, including blue mussels, hard clams, lobster, and fish, were analyzed for organic contaminants during Phase I/II investigations. Complete

data for organics residues are presented in Appendix A-1-1 (spatial distribution presented in Appendix D-2).

*Blue Mussels.* Figure 4.2-8 compares the concentration of organic contaminants in blue mussels by station from the McAllister Point Landfill intertidal habitat. Also included for the same stations are analyses of mussels which were depurated, i.e., placed in clean seawater in the laboratory for 24 hours prior to freezing in order to remove sediment particles from the guts of the organisms.

Comparisons among samples reveal high values at Stations NSB-3 to NSB-7 for Total PCBs; while NSB-6 and NSB-7 were elevated for p,p'-DDE. Differences in TBT concentrations in mussel tissues among stations were less than two-fold across the intertidal area, while PAH residue concentrations appeared to decrease from Station NSB-1 to NSB-7.

For Total PCBs, p,p'-DDE and Total PAHs, the depurated values were about 70% to 90% of the non-depurated levels. In the case of TBT, the depurated value was 60 to 80% higher than the non-depurated concentration, which is a surprising finding and difficult to explain. Perhaps the mussels were somehow contaminated with TBT during the 24 hour depuration period, or other substances which would have otherwise interfered with TBT analyses were lost during the depuration process.

*Hard clams.* Concentrations of organic contaminants in the hard clam *Mercenaria mercenaria*, collected from subtidal stations off of the McAllister Point Landfill and at the reference location (JCC-S1), are shown in Figure 4.2-9. For PCBs, tissue residues from stations offshore of the central landfill area (Stations MCL-10 to MCL-12) were generally lower than levels found in clams for the northern (MCL-9) and southern (MCL-13, MCL-14) landfill area. (Note that Station MCL-16 is located well away from the landfill and near the Coddington Cove breakwall; Figure 3.1-2).

For p,p'-DDE, clam tissue residues were notably lower at Station MCL-11 (and reference station JCC-S1) than at other stations to the north and south, while for TBT central landfill area stations MCL-10 to MCL-13 were lower than at stations to the north and south. Again, the depurated samples had lower values than non-depurated organisms for Total PCBs, p,p'-DDE and Total PAHs, but here the difference was usually only 5 to 10%. Apparently, the mussels depurate more contaminants than the clams over a 24 hour time period; this may be related to lipid concentration and/or composition, as well as differences in filtering rates of the two bivalves. As with mussels, the depurated clams had higher TBT values (7 to 40%) than the non-depurated samples which could not be readily explained.

*Cunner.* Samples of cunner (*Tautoglabrus adspersus*) were also analyzed and the results indicated that, although all samples had measurable concentrations of the contaminants, there was no one station that had elevated values for all components (Figure 4.2-10). Total PAHs were low relative to mussels and clams, which may be in part explained by the fact that fish can metabolize PAHs to a greater extent than bivalves. TBT values in the fish were also lower than those in the bivalves, whereas Total PCBs and p,p'-DDE levels were higher, reflecting a possible food web biomagnification of these lipophilic contaminants.

*Lobster.* Figure 4.2-11 presents data on the concentration of contaminants in both lobster muscle and hepatopancreas taken from animals collected offshore of McAllister Point and the Jamestown Cranston Cove reference location. Samples of hepatopancreas showed much higher values for Total PCBs, p,p'-DDE and Total PAHs than the muscle tissue, while TBT values did not show this tissue-specific trend. Higher concentrations of lipophilic organic contaminants (e.g., PCBs, p,p'-DDE, and PAHs) should be expected in hepatopancreas relative to muscle because of the difference in lipid content of the tissues; hepatopancreas had 27 to 48% lipid and muscle had 0.8% to 1.2% lipid.

*Comparison with TRC/BOS (1994) results (ng/g dry weight of tissue).* The highest Total PCB values (recalculated here as sum of congeners X 2) from the TRC/BOS study (TRC, 1994) were found in blue mussels from intertidal Stations NS-10/11/12 through NS-19/20/21 (836 to 2110 ng/g) and in clams from subtidal Station OS-22 to OS-26 (132 to 156 ng/g). Corresponding levels from the URI/SAIC study were ~850 to 1840 ng/g at intertidal Stations NSB-1 to NSB-7 (mussels) and 168 to 346 ng/g at subtidal Stations MCL-9 to MCL-11 (hard clams). PAH concentrations for TRC/BOS mussels were 413 to 499 ng/g at Stations NS-1/2/3 through NS-19/20/21, while concentrations in clams were 372 to 919 ng/g at Stations OS-22 through OS-28. In comparison, URI/SAIC values for mussels were 482 to 1530 ng/g at Stations NSB-1 to NSB-7, and clams were 156 to 221 ng/g at MCL-9 through MCL-12. TBT values for TRC/BOS mussels (1994) were non-detected (<18.8 ng/g); they were 14 to 22 ng Sn/g for URI/SAIC mussels from Stations NSB-1 through NSB-7. Pesticides were not measured for the TRC/BOS study.

In summary, the trends in concentration per station location for PCBs and PAHs in sediments, and PCBs and TBTs in mussels and clams are in good agreement between the TRC/BOS investigation and the present URI/SAIC data. However, PAH concentrations for TRC/BOS mussels are at the low end of the range of URI/SAIC values, and concentrations of PAHs in TRC/BOS clams are 2-4 times higher than the URI/SAIC clams. This poor agreement in PAH concentrations of bivalve tissue samples may be due to differences in exact sampling locations and temporal (August, 1993 vs. March to June, 1995) variations in the contaminant sources, uptake and metabolism. Apparently, the PCBs are not as influenced by these variables.

### 4.2.3. Inorganic Contaminants

#### 4.2.3.1. Simultaneously Extracted Metals/Acid Volatile Sulfide

Toxicity of sediment metals is correlated with divalent metal activity in interstitial water. DiToro *et al.* (1990) have shown that metal availability, particularly in anoxic sediments, is controlled by the concentration of insoluble metal sulfides, called Acid Volatile Sulfides (AVS), which act to bind divalent metals (cadmium, copper, mercury, nickel, lead, and zinc). Conceptually, divalent metals bound to AVS are not toxic to sediment biota. Acid treatment releases these reactive sulfides which then can be analyzed as acid volatile sulfides. The relative concentration of metal extracted with the AVS, or Simultaneously Extracted Metals (SEM), from the sediment is used to determine metal bioavailability and potential toxicity. For example, if the SEM/AVS ratio is greater than one ( $>1$ ), then there is more metal available than sulfide and the metal is assumed to be bioavailable and potentially toxic.

For the present study, samples for SEM and AVS data are available for 32 surface stations and 20 sub-surface intervals from sediment cores. Raw data are presented in Appendix A-1-4. The results of one measure of SEM bioavailability are also shown in Figure 4.2-12. For the present evaluation, SEM/AVS ratios greater than one were found at 15 of 32 surface sample stations. For the purpose of calculating the SEM/AVS ratio, AVS concentrations below the detection limit of  $0.1 \mu\text{mol/gram}$  were assumed to be  $0.05 \mu\text{mol}$ . Using a benchmark of SEM/AVS=0.5, the majority of sediment stations are considered to have potential bioavailability of metals (Figure 4.2-12). Surface sediment stations with SEM/AVS $<0.5$  include MCL-9 through MCL-12, NSB-6, S1, and S2B. For core samples, the general pattern observed is that SEM/AVS ratios increase with depth (Figure 4.2-12). Previous studies of AVS concentrations versus depth in Narragansett Bay sediments indicate that concentrations are lower in the surface mixed layer ( $\sim 1\text{-}8 \text{ cm}$ ), increase to a maximum

between ~ 8-25 cm, and then decrease significantly below the zone of maximum concentration (W. Boothman, EPA, personal communication). The zone of maximum concentration of AVS is believed to coincide with the zone of maximum metabolism of sulfate reducing bacteria.

*Comparison with TRC/BOS results.* The SEM/AVS ratios observed for nearshore stations in this study are comparable to those observed by TRC/BOS (1994), where values of 0.5 or greater were observed for SEM/AVS ratios from the nearshore stations, indicating that metals are bioavailable at these stations. In contrast, the SEM/AVS results from this study differ from those of the TRC/BOS study for the offshore stations; ratios higher than 0.5 were observed for many of the offshore stations in this study, whereas no values greater than 0.5 were observed for offshore stations by the TRC/BOS study. In the URI/SAIC study, samples were analyzed from either the interval 0-2 cm or 0-6 cm, whereas in the TRC/BOS study the interval 0-5 cm was analyzed. Given that higher AVS concentrations are usually found in the interval 6-15 cm than in the interval 0-2 or 0-6 cm, lower SEM/AVS ratios would be expected for a composited sample of the interval 0-15 cm than for a surficial (0-6 cm) sediment composite. This difference in methodology between the two studies is the most plausible explanation for the higher SEM/AVS observed for offshore stations in the present investigation.

#### 4.2.3.2. Total Sediment Metals

Summary data for the selected CoC metals in sediment are presented in Figures 4.2-13 through 4.2-24. Raw data are presented in Appendix A-1-2 (spatial distribution presented in Appendix D-1). Spatial patterns in surface sediment metal concentrations vary depending on the probable source of the metal (Appendix D-1-1). Surface sediment concentrations of the metals that are probably of anthropogenic origin, including arsenic (Figure 4.2-13), cadmium (Figure 4.2-14), copper

(Figure 4.2-15), chromium (Figure 4.2-16), lead (Figure 4.2-17), mercury (Figure 4.2-18), nickel (Figure 4.2-19), silver (Figure 4.2-20) and zinc (Figure 4.2-21), are generally found in maximum concentrations in the nearshore stations at the edge of McAllister Point Landfill, whereas metals that are primarily of natural origin including aluminum (Figure 4.2-22), manganese (Figure 4.2-23), and iron (Figure 4.2-24), tend to be much more evenly distributed within the study area.

The maximum metal concentrations of anthropogenic metals are found in relatively coarse-grained sediments (NSB-1 to NSB-7) that mark the boundary between McAllister Point Landfill and Narragansett Bay, whereas finer grained subtidal sediments offshore of McAllister Point Landfill generally have significantly lower concentrations of metals. This spatial distribution of metals of anthropogenic origin indicates that McAllister Point Landfill is the dominant source of these metals within the study area. However, it is noteworthy that high arsenic values are found at Stations MCL-15 and D3 in an area that was identified as an anomalous region by the geophysical survey.

Surface sediment concentrations of arsenic (Figure 4.2-13), cadmium (Figure 4.2-14), and chromium (Figure 4.2-16) only exceed ER-L Guidelines (Long *et al.*, 1995) within the study area, whereas concentrations of copper (Figure 4.2-15), lead (Figure 4.2-17), mercury (Figure 4.2-18), nickel (Figure 4.2-19), silver (Figure 4.2-20), and zinc (Figure 4.2-21) exceed both ER-L and ER-M Guidelines (Long *et al.*, 1995) within the study area.

The down core distribution of metals of anthropogenic origin is complex, but in general maximum concentrations are found in the surface sediments (0-20 cm) and decrease with depth. Arsenic (Figure 4.2-13) is an interesting exception to this pattern, as maximum concentrations are often found at depth.

*Comparison to TRC/BOS results.* The results of this study are very comparable to those obtained by the TRC/BOS study (TRC, 1994). Both studies observed maximum concentrations of metals in the nearshore stations and lower concentrations in offshore stations. In both studies, cadmium and chromium are found to exceed ER-L Guidelines (Long *et al.*, 1995), and at comparable stations, copper, lead, mercury, nickel, and zinc were observed to exceed ER-M Guidelines (Long *et al.*, 1995). Higher maximum arsenic concentrations exceeding the ER-M were found in the TRC/BOS (1994) study, whereas higher maximum silver concentrations exceeding the ER-M were found in this study. The spatial pattern of metal distribution observed in surface sediments is very similar in both studies.

*Comparison of pre- and post-erosion conditions.* As discussed previously, the purpose of the comparison between Phase II (pre-erosion) and Phase III (post-erosion) results was to assess whether the erosional event had increased possible CoC exposure to aquatic biota. Major macroscopic changes, which may relate to changes in sediment metal concentrations observed in the study area during Phase III sampling, included: (1) removal of 1 to 2 feet of sediment from the base of the landfill revetment, (2) exposure of new metal debris at and immediately north of Station NSB-2, and (3) rapid deposition of silty clay at Station S2B.

Metal concentrations analyzed during Phase III were higher than metal concentrations determined during Phases I and II for several metals at stations in the study area. Concentrations of three metals (copper, lead, and zinc), which exhibited the greatest degree of change, are presented in Table 4.2-3. Raw data for all analytes are presented in Appendix A-1-2.

As discussed for organics, stations for which surface sediment concentrations increased from 1995 to 1996 with a relative percent difference (RPD) greater than 30%

are indicated by dark bordered cells; these values are additionally shaded relative to ER-L and ER-M benchmarks.

Surface sediment concentrations of copper were greater than 30% higher than previously measured concentrations and above the ER-M at Stations NSB-2, NSB-3 and NSB-4 (Table 4.2-3). Concentrations of lead also increased by greater than 30% and were above the ER-M at Stations NSB-2, NSB-4, and NSB-5. Similar increases in zinc were noted at Stations NSB-2, NSB-4, NSB-5, NSB-7, MCL-10 and MCL-14. Concentrations of copper and zinc also exceeded ER-L guidelines at Stations MCL-10 and MCL-12, respectively. The observed increase in surficial sediment concentrations, particularly at Stations NSB-2, NSB-3, NSB-4, and NSB-5, indicate that erosion at the McAllister Point Landfill has increased potential adverse exposure from contaminated sediments with respect to trace metals. Furthermore, the increased concentrations of zinc observed at Stations MCL-10 and MCL-14 indicated an offshore area of contaminated sediments not observed in Phase II, possibly resulting from the eroded shoreline.

With respect to core samples, significant increases (RPD >30%) from previously detected concentrations and in relation to ER-L levels are apparent for Stations MCL-10 and MCL-12 (Table 4.2-3). For the intertidal samples, sediment concentrations for core samples are generally comparable to surface sediment concentrations. A notable exception may include high levels of zinc at Stations NSB-2 and NSB-4, but reduced concentrations at other stations. In general, the data suggest somewhat increased exposure at offshore stations, and relatively uniform vertical concentrations at nearshore stations.

#### 4.2.3.3. Inorganic Tissue Residues

Results of tissues of three species, (blue mussel (BM), hard shell clam (HC), and lobster (L)) analyzed for trace metals are shown in Figures 4.2-25 to 4.2-30 (spatial distribution presented in Appendix D-2). Raw data are presented in Appendix A-1-2. In addition, lobster samples were apportioned and analyzed in two components, lobster muscle (LM; which included claw and tail) and hepatopancreas (LHep). Some bivalve samples were analyzed as paired sets; one set (several individuals that were composited) of bivalves from the pair was depurated prior to analysis (DEP), whereas one set of bivalves in the pair was analyzed without depuration. Insufficient sample material was available to conduct analyses of fish.

Little difference was observed between non-depurated and depurated bivalves for most analytes in this study, with the exceptions of lead and aluminum (Figure 4.2-25). Blue mussels and hard clams contained more lead and aluminum than lobsters and a significant proportion of these analytes was lost by depuration of sediment from the organisms. Cadmium, copper, and silver appeared in highest concentrations in the lobster hepatopancreas samples (Figures 4.2-26 and 4.2-27). The mechanism for this concentration is unknown, but accumulation of cadmium and copper may be associated with the higher lipid content in the hepatopancreas samples. The distributions of arsenic, iron, chromium, zinc, manganese, and nickel did not appear to have either a spatial- or species-dependent pattern (Figure 4.2-27 through Figure 4.2-30). The concentration of mercury was highest in lobster muscle (Figure 4.2-29) and may reflect biological concentration at higher trophic levels.

*Comparison to TRC/BOS Results.* A comparison of the results from blue mussel tissue from this study with those reported by TRC (1994) indicates that observed concentrations are similar for all analytes with the exceptions of chromium and nickel. The concentrations, in the blue mussel, observed by TRC/BOS were five times higher

than those observed in the present study. A similar comparison with the data from hard clam tissues revealed the same pattern except that TRC/BOS values are 50 times higher for chromium, and nickel values are 10 times higher than those observed in this study. A general explanation for the observed differences is not evident.

#### 4.2.3.4. Porewater Metal Concentrations

Metal concentrations were determined for porewater samples extracted from surface sediments from a total of 17 stations in the McAllister Point Landfill study area and Jamestown Cranston Cove. Raw data are presented in Appendix A-1-2 (spatial distribution presented in Appendix D-1). Concentrations were below detection for most analytes with the exceptions of copper, zinc, mercury, and nickel (Figures 4.2-31 and 4.2-32). Copper was found to exceed the EPA Ambient Water Quality Criterion - Saltwater Acute value (AWQC-SA; U.S. EPA, 1986) at nearshore Stations NSB-1 through NSB-7, while zinc exceeded the AWQC-SA at Stations NSB-4 and NSB-5 (Figure 4.2-31). In general, mercury and nickel approached or exceeded EPA Ambient Water Quality Criterion - Saltwater Chronic (AWQC-SC) throughout the study area (Figure 4.2-32). Nickel exceeded the AWQC-SC at intertidal Stations NSB-1 through NSB-5 and NSB-7.

#### 4.2.4. Fecal Pollution Indicators in Sediments

Sediment and bivalve tissue samples collected in the McAllister Point study area were analyzed for fecal pollution indicator bacteria (raw data presented in Appendix B-2). Fecal pollution indicator bacteria are commonly used to assess the sanitary quality of marine environments. Studies have consistently shown a direct association between inputs of sewage and other fecal materials and the concentrations of fecal pollution indicator bacteria in water, sediments, and marine organisms (APHA, 1970; U.S. FDA, 1992b; Institute of Medicine, 1991; Mitchell, 1978). Sediment results are

reported here as an indication of CoC exposure pathways; tissue-based fecal concentrations are reported in Section 5.3 as an indicator of possible effects.

The relative concentrations of selected indicators provide inference as to the source and history of sewage-related contamination. Untreated contamination is suspected when elevated levels of vegetative bacteria, such as total and fecal coliforms and fecal streptococci, as well as spore-forming bacteria, such as *Clostridium perfringens*, are observed. Untreated contamination sources include animal waste (shore birds), run-off (fertilizers and animals wastes), human wastes (boat discharges) and untreated or improperly treated sewage effluent. *C. perfringens* bacteria may produce an endospore which is very resistant to adverse environmental conditions, and thus allows for extended survival (Cabelli, 1978). Therefore, elevated levels of *Clostridium perfringens* serve as an indicator of historic fecal exposure.

Sediment samples collected from Stations NSB-1 and NSB-3 exhibited elevated densities of total coliforms, fecal streptococci, and *Clostridium perfringens* relative to Stations NSB-5 and NSB-7 (Table 4.2-4). The apparent gradient in densities indicates a source of untreated or improperly treated sewage or other fecal material to the north of the site, perhaps via Gomes Brook (Figure 3.5-2). Indicator densities for the sediment samples collected from the remaining four stations, MCL-11, MCL-12, MCL-13, and MCL-15 also generally exhibited this trend. *C. perfringens* exhibited markedly elevated levels (9200 - >16,000 CFU/100 g) relative to other fecal indicators, suggesting that the area has historically experienced fecal pollution.

The spatial trend in the relative densities of indicators suggest a decrease with proximity to Station NSB-5 which would indicate this area of the landfill is not a source of the indicators. Elevated sediment-associated *Clostridium* at offshore (MCL) sites relative to NSB sites is possibly due to these sites being more depositional in nature, allowing spores to accumulate, as demonstrated by finer grained sediments

(Figure 4.2-1). Although non-point sources of fecal indicators might occur due to landfill seeps, the fecal pollution indicator concentrations are inversely related to the xenobiotic (human-made contaminant) distributions. Thus, the data suggest that there does exist alternative transport pathways for contaminants to the study site, but this mechanism cannot explain the measured CoC distribution pattern.

### 4.3. UNCERTAINTY

Contaminant sources from the landfill areas have been well characterized based on the previous studies. However, the exposure pathways as reflected by the first through Fourth Tier models (Section 3.3) are necessarily conceptual and cannot account for all the complexities, including proximal and distal sources and receptors, of a natural ecosystem. These uncertainties also are driven by incomplete knowledge of the chemical behavior of the CoCs, even though considerable information is available on solubility, partitioning, and toxicity for several analytes. Nonetheless, existing information on the chemical contaminants and a reasonably thorough understanding of the bay ecosystem have allowed sufficient and relevant data to be targeted, collected, and interpreted for the risk assessment.

Fate and transport evaluations for the exposure assessment focused on spatial (horizontal) and vertical (sediment layering) patterns as well as data comparability among the study phases (i.e., temporal consistency). The placement of sampling stations was largely based on providing “visually complete” (essentially “gridded”) coverage of the various habitats (e.g., intertidal and offshore). Station placement was guided additionally by results from prior phases; however, visual coverage was still the principal method applied. The uncertainty associated with any sampling station is whether it is truly representative of the habitat and impact/reference zone being evaluated. Collection of station replicates is one method that allows assessment of

within-station variability (i.e., the representativeness of a sample), although only single samples were collected per station for this study. Further, various methods of statistical power analysis can be used to determine, for example, the number of randomly placed samples that are necessary to characterize, with a desired level of confidence, a particular sampling regime (habitat and zone).

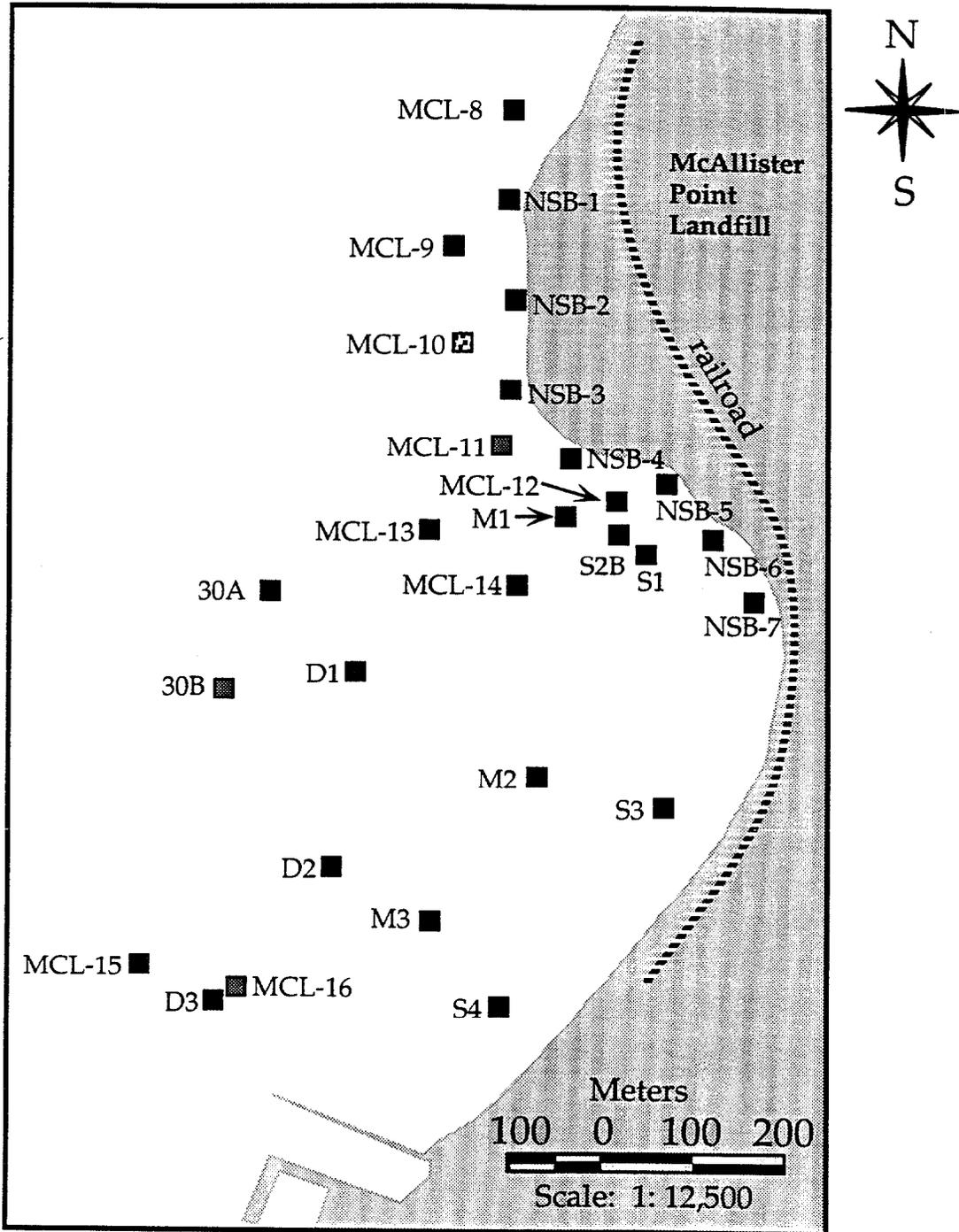
*Station selection.* For the present study, comparison of the data variability among stations is the primary method used to assess adequacy and representativeness of the sampling grid. Conceptually, this is fundamental to ongoing debates about the uncertainties of extrapolations (and assumptions) from point measurements to broader spatial areas. Quantitative approaches using Geographic Information Systems (GIS) technology have been recently reported (Clifford *et al.*, 1995) which appear to provide an effective, unbiased method for estimating spatial extent of risk, with minimum uncertainty and maximum data usage. These techniques will undoubtedly be more prevalent in future studies where large databases are available for quantitation.

*Temporal/spatial variability.* An area of uncertainty for the assessment is the temporal comparability of data among the phases. The general study design assumes that there have not been substantial changes in environmental conditions and chemical contaminant concentrations at individual sampling sites, as representative of particular habitat and sampling zones, over the various phases. However, in practice interannual and seasonal variations occur in every environment, thereby changing to some degree the conditions that influence contaminant sources, exposure pathways, and receptors. This was exemplified by changes in CoC concentrations at some locations as a result of a sediment erosion event. Further, the navigational methods used to locate stations varied somewhat among the phases, with the present study using more accurate methods ( $\pm 3$  m) than earlier phases (Loran Navigation:  $\pm 100$  m). Nonetheless, the validity of the assumptions concerning insignificant temporal changes was tested by

comparing the differences between concentrations of representative CoCs and toxicity for sets of stations that were located very near or coincident with each other, but which were sampled during different phases (see Section 4.2).

Finally, the exposure point estimates are based on representative chemical analytes that, due to practicality, are a subset of the total possible compounds that could be analyzed. However, the analytes have been carefully selected as a result of extensive screening and analyses during the present and previous phases and are considered to be appropriately conservative and representative of source contaminants. Calculations of SEM for use in comparisons with AVS values utilize sediment data on copper, zinc, lead, nickel, cadmium, and mercury. Each of these metals, except mercury, is commonly accepted as reacting appropriately in the presence of sulfides to fulfill the assumptions of the AVS paradigm. However, there is ongoing debate about the need to include mercury in the calculations. This is because mercury can react in a manner that is more similar to an organic compound than a divalent metal. For this assessment, mercury has been included in the SEM calculation; and in some cases its incorporation does affect the final SEM/AVS ratios. Further, the inclusion of an additional metal primarily serves to increase the ratio value, thereby representing a more conservative effects measure.

Figure 4.2-1. Grain size characteristics of surface sediments from McAllister Point Landfill and Jamestown Cranston Cove.



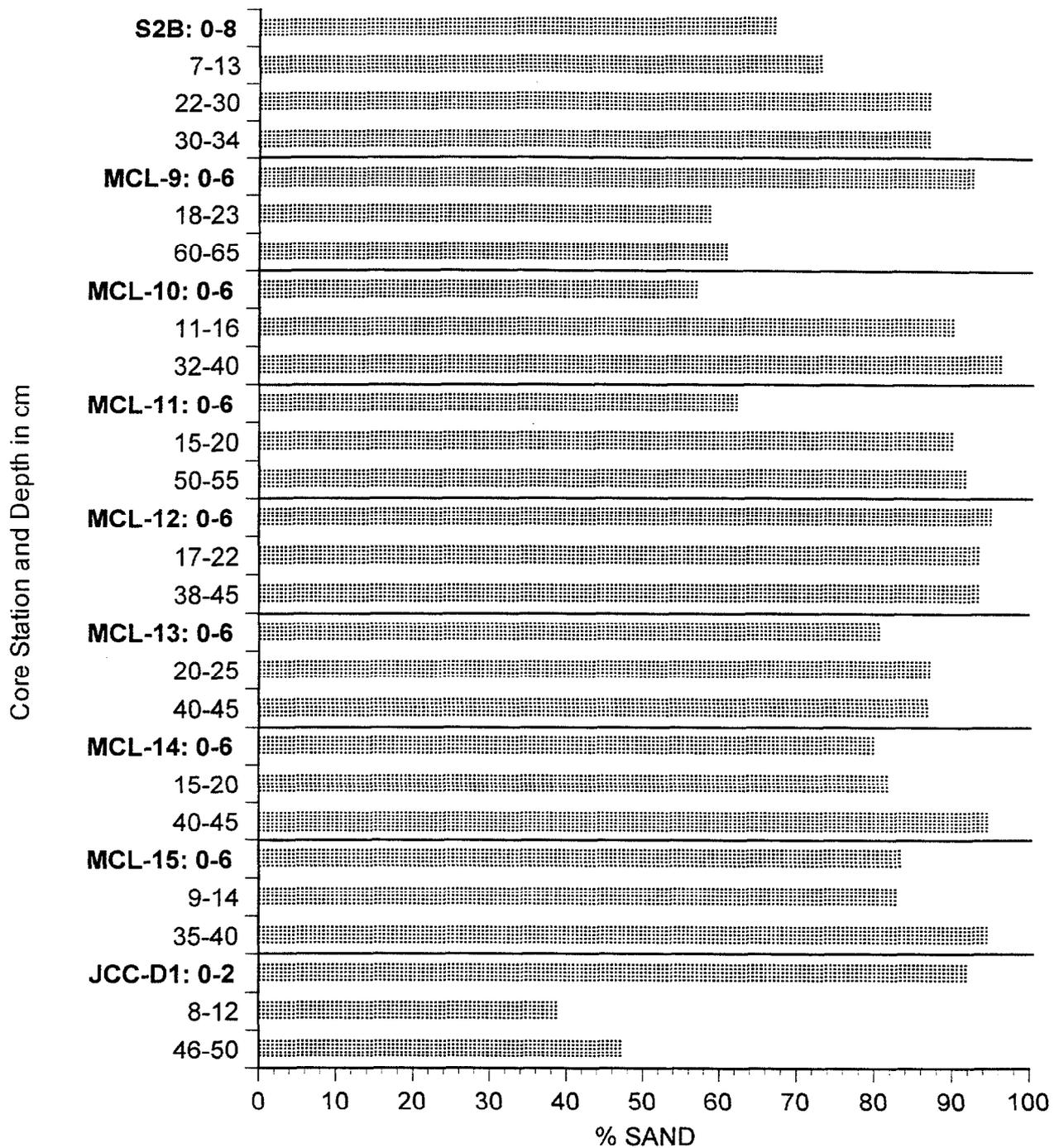


Figure 4.2-2. Percentage of sand versus depth in cm for piston cores from McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

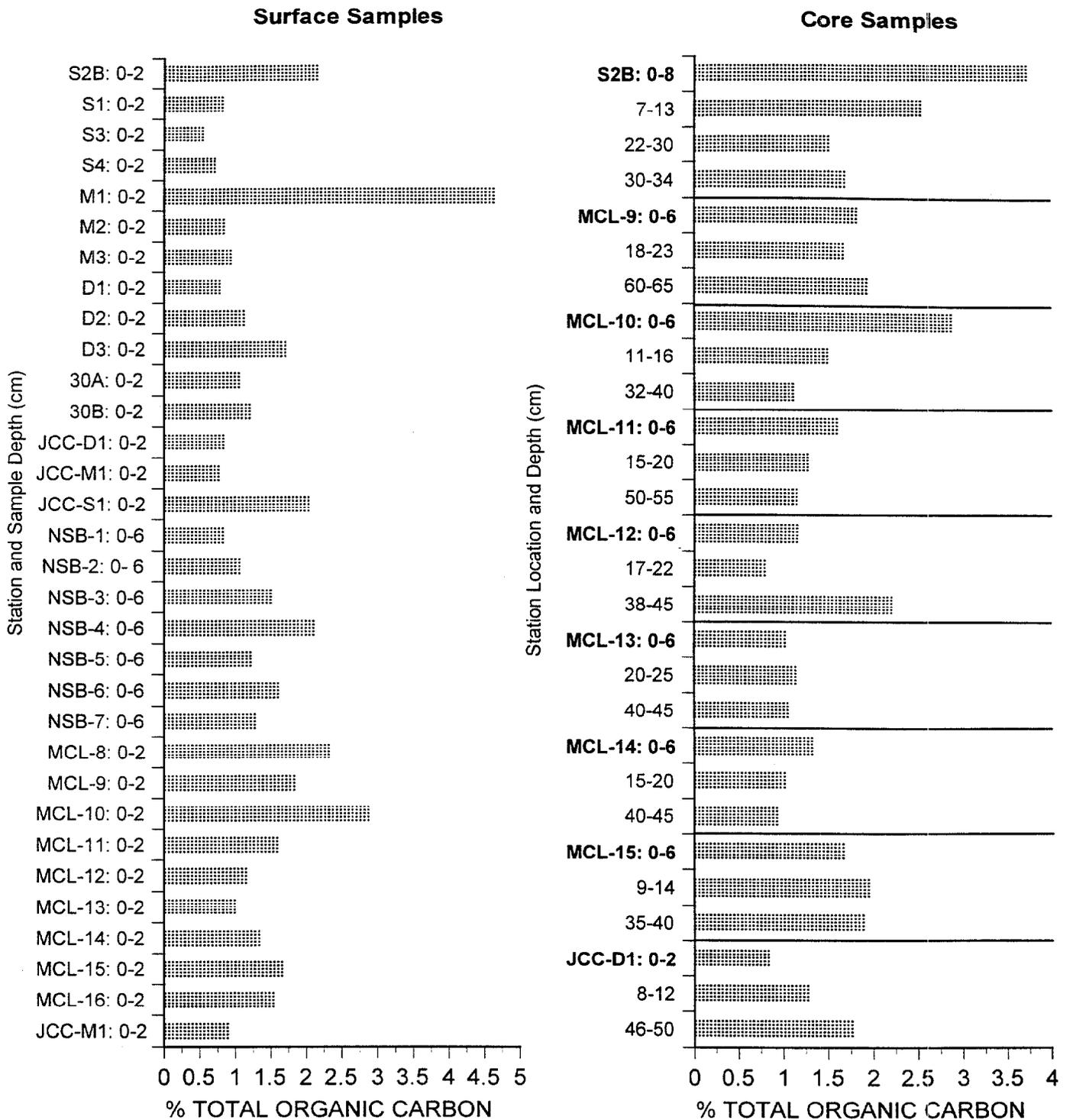


Figure 4.2-3. Organic carbon content of surface sediment samples and piston core samples from McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

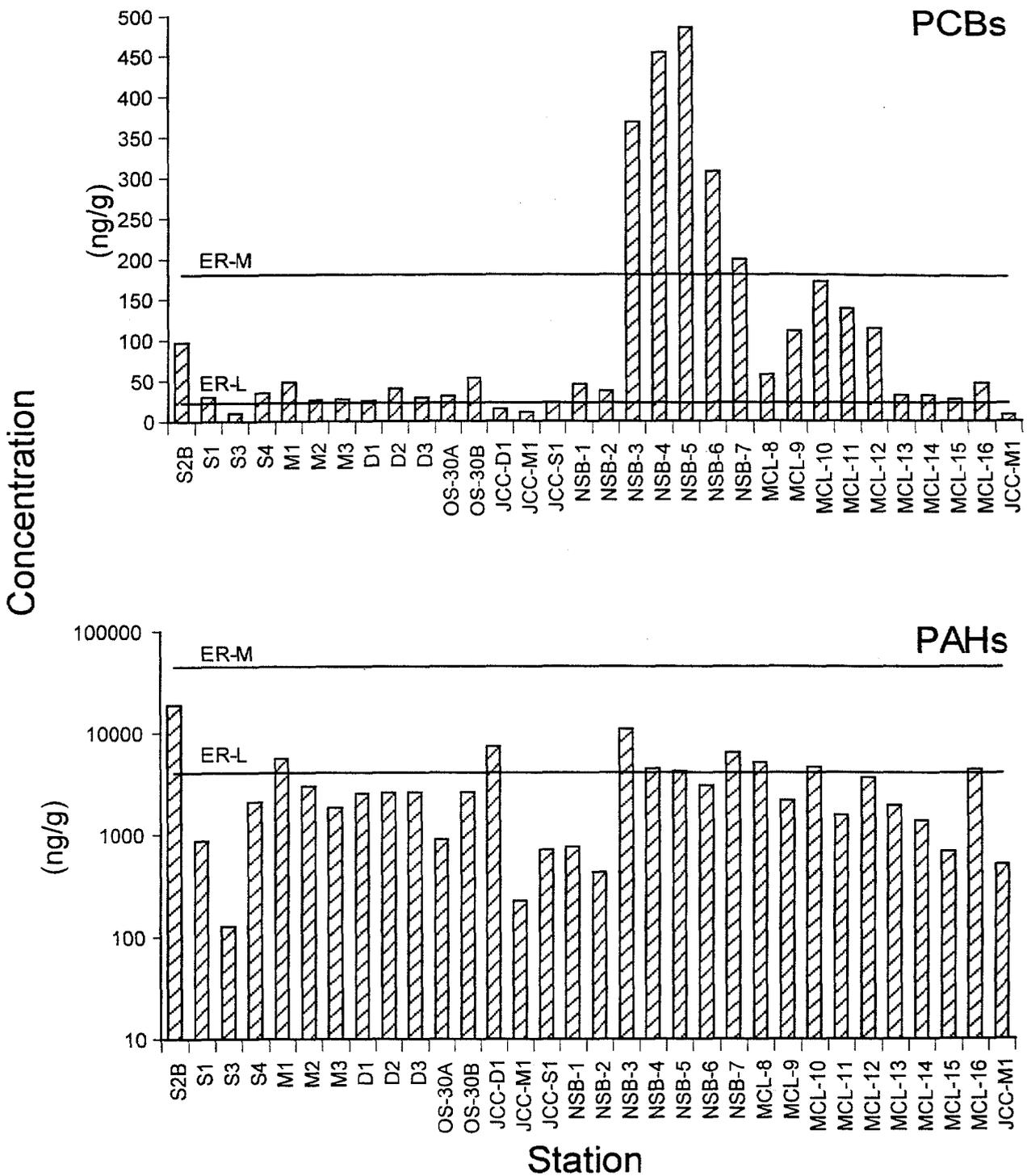


Figure 4.2-4. Concentration (ng/g dry weight sediment) of organic contaminants in surface sediments from the McAllister Point study area and Jamestown Cranston Cove (JCC). The sample depth at sites NSB-1 through NSB-7 is 0-6cm. The depth at all other sites is 0-2cm. The horizontal lines are the ER-L and ER-M guidelines (Long *et al.*, 1995).

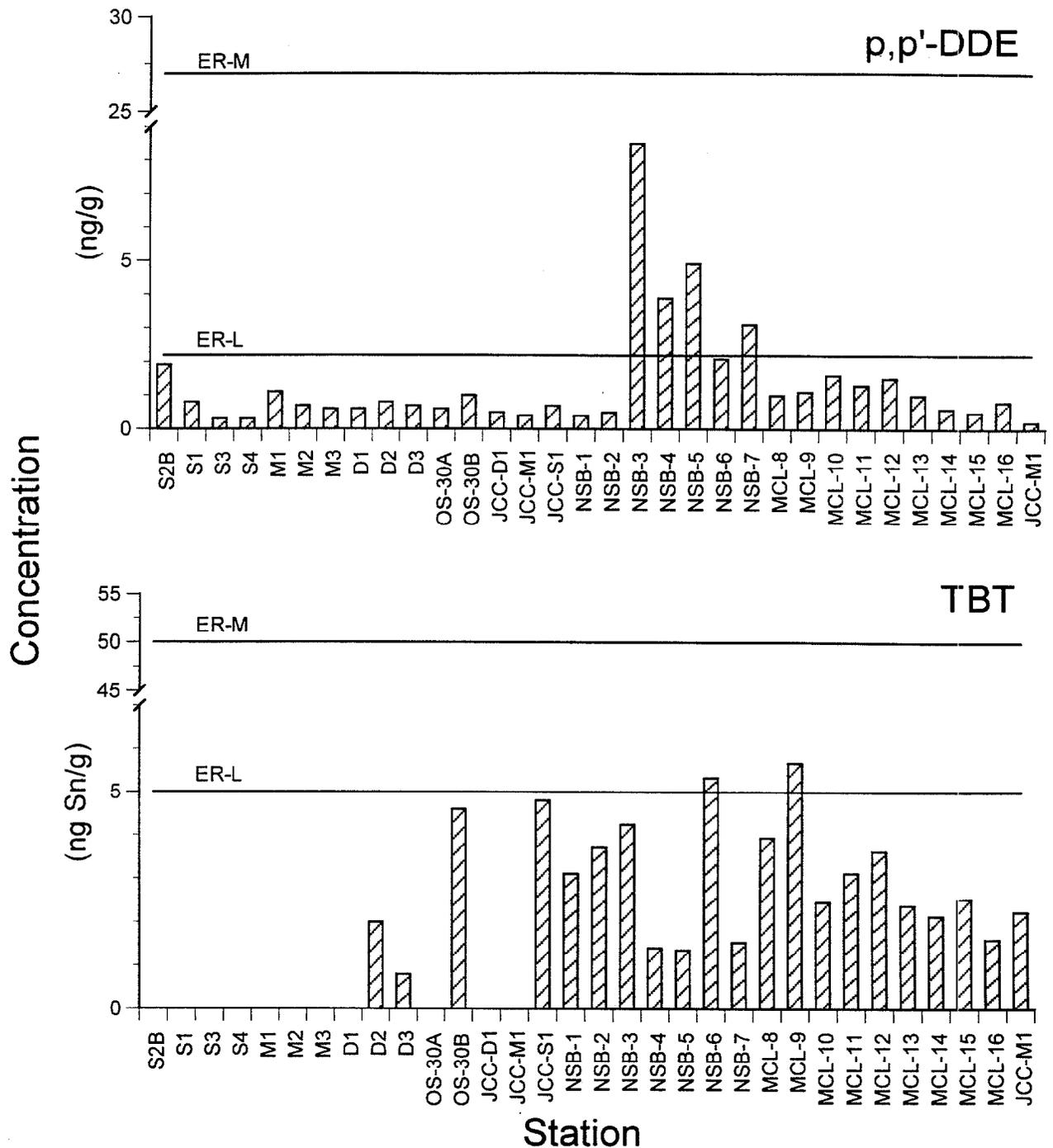


Figure 4.2-4 (continued). Concentration (ng/g dry weight sediment) of organic contaminants in surface sediments from the McAllister Point study area and Jamestown Cranston Cove (JCC). The sample depth at sites NSB-1 through NSB-7 is 0-6cm. The depth at all other sites is 0-2cm. The horizontal lines are the ER-L and ER-M guidelines (Long *et al.*, 1995).

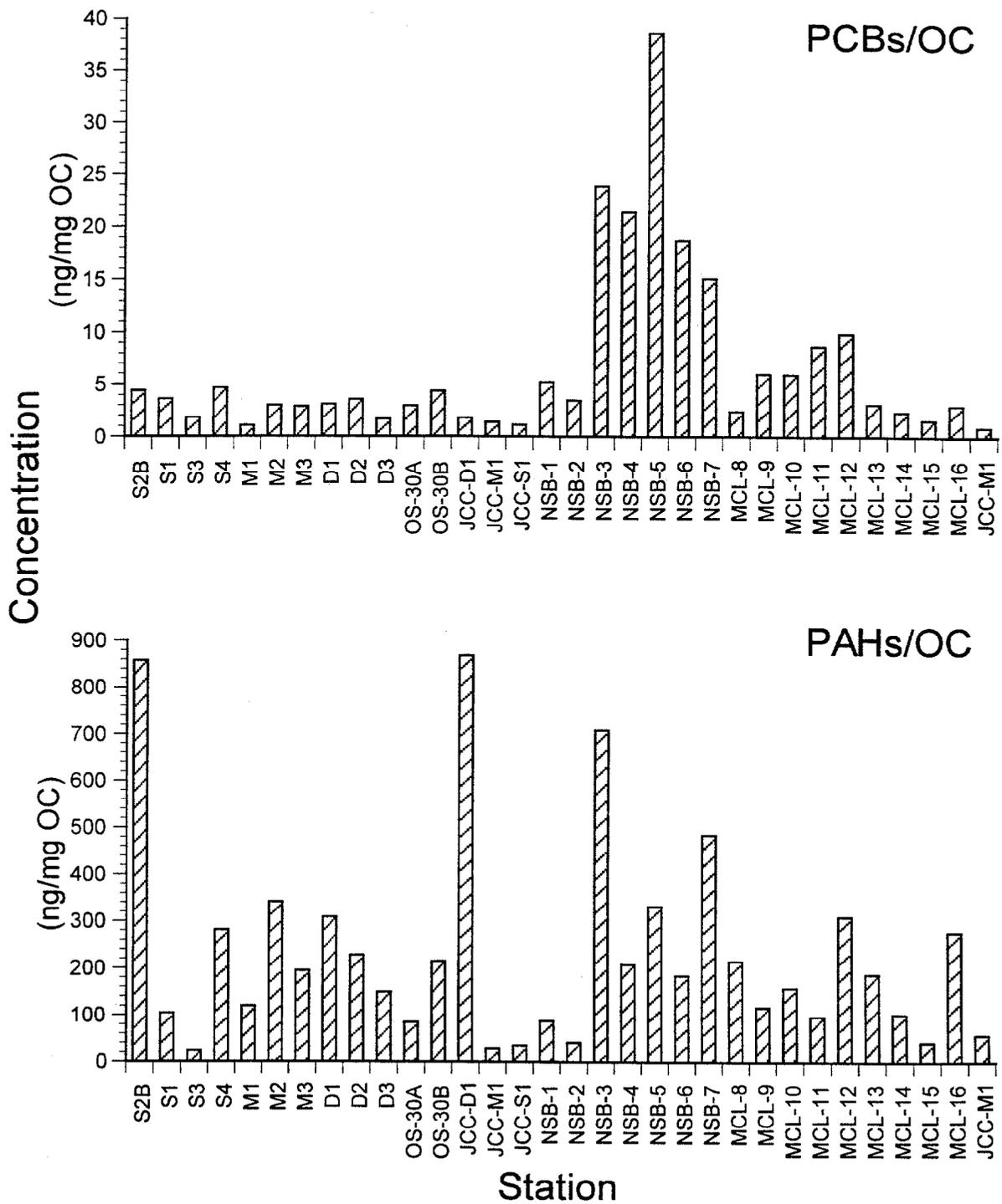


Figure 4.2-5. Concentration (ng/mg OC) of organic contaminants normalized to organic carbon (OC) in surface sediments from the McAllister Point study area and Jamestown Cranston Cove (JCC). The sample depth at sites NSB-1 through NSB-7 is 0-6cm. The depth at all other sites is 0-2cm.

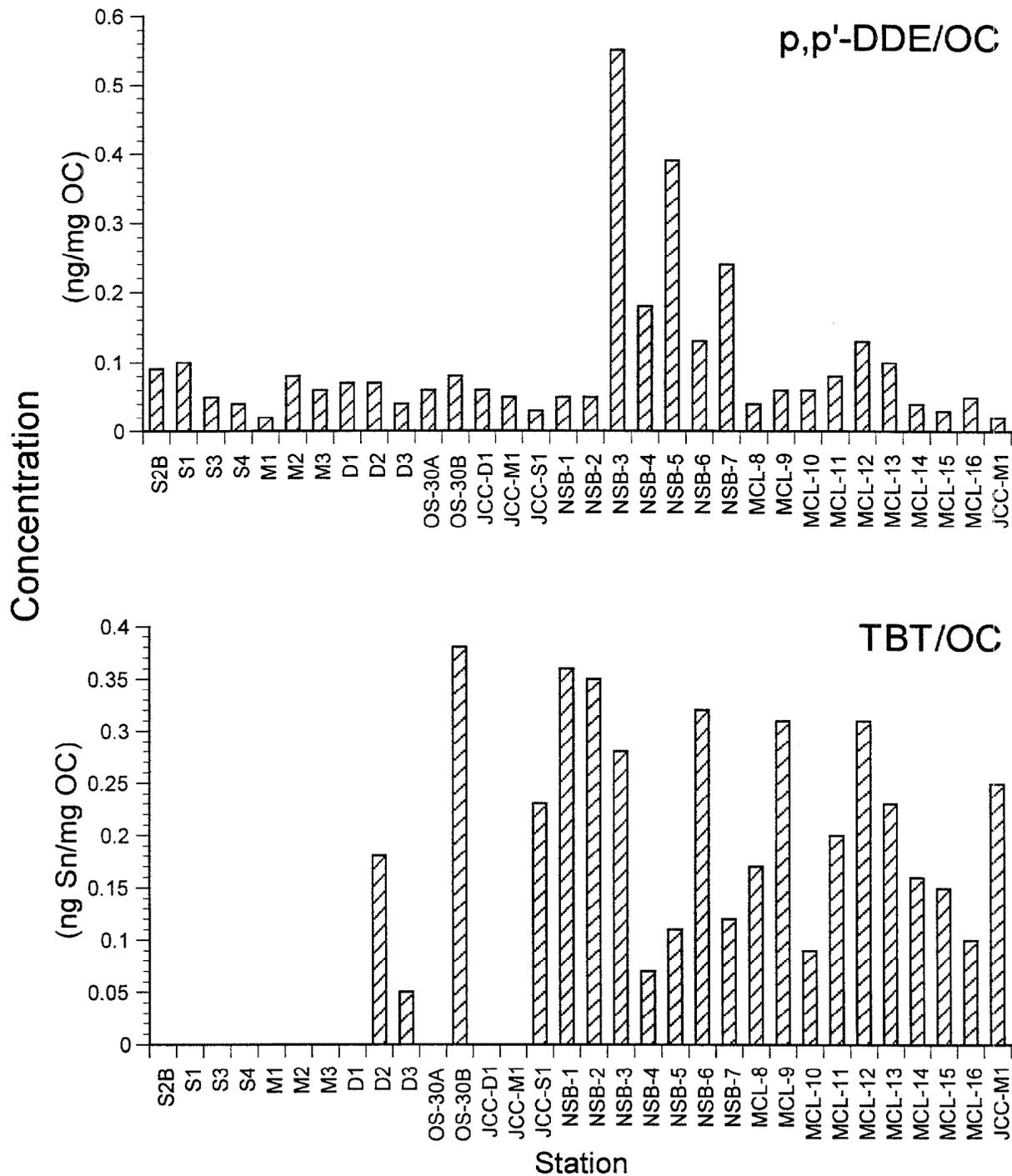


Figure 4.2-5 (continued). Concentration (ng/mg OC) of organic contaminants normalized to organic carbon (OC) in surface sediments from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The sample depth at sites NSB-1 through NSB-7 is 0-6 cm. The depth at all other sites is 0-2 cm.

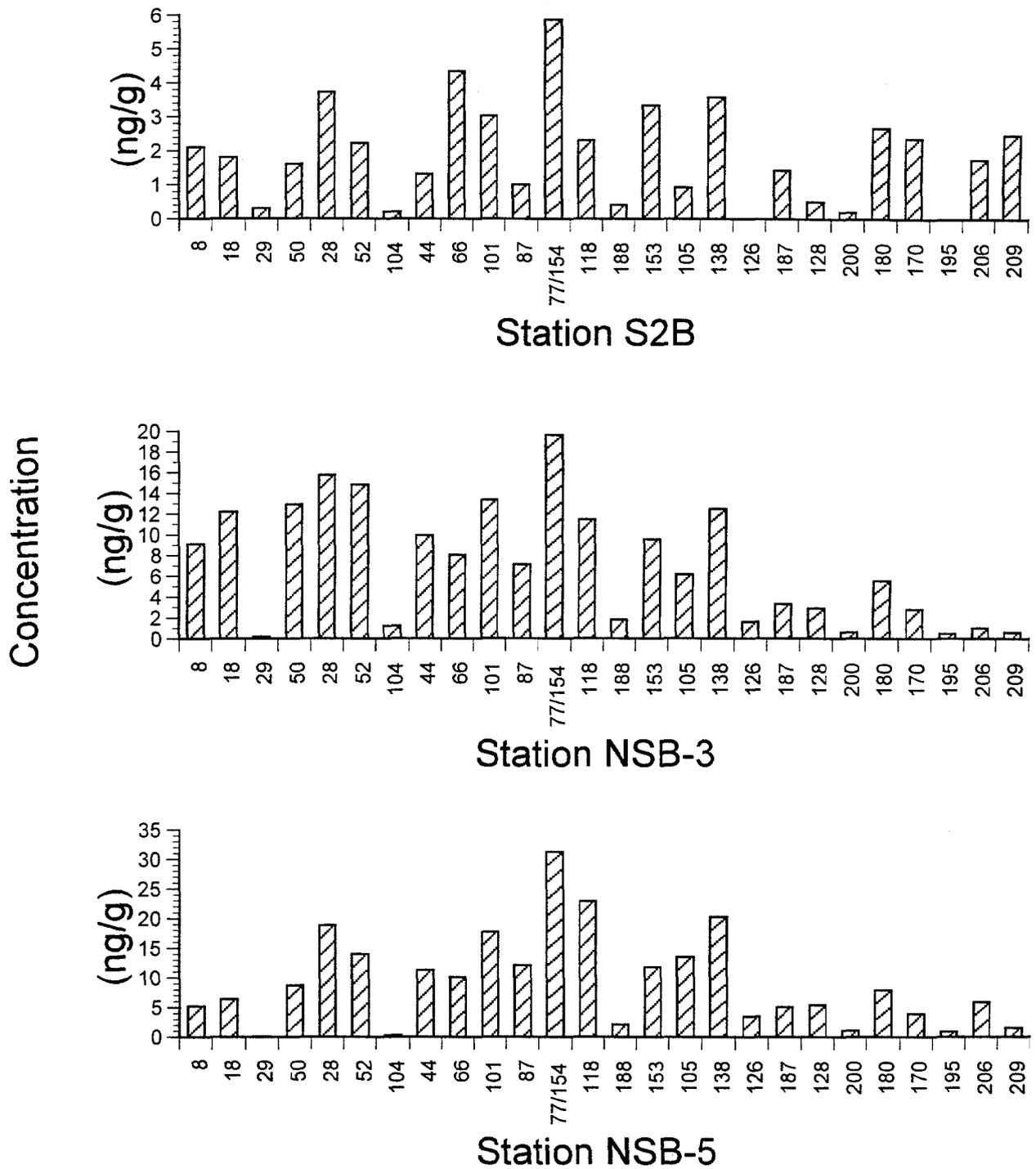


Figure 4.2-6. Concentration (ng/g dry weight sediment) of PCB congeners in surface sediments from selected stations in the McAllister Point Landfill study area. The depth of Stations NSB-3 and NSB-5 is 0-6 cm, while that of Station S2B is 0-2 cm.

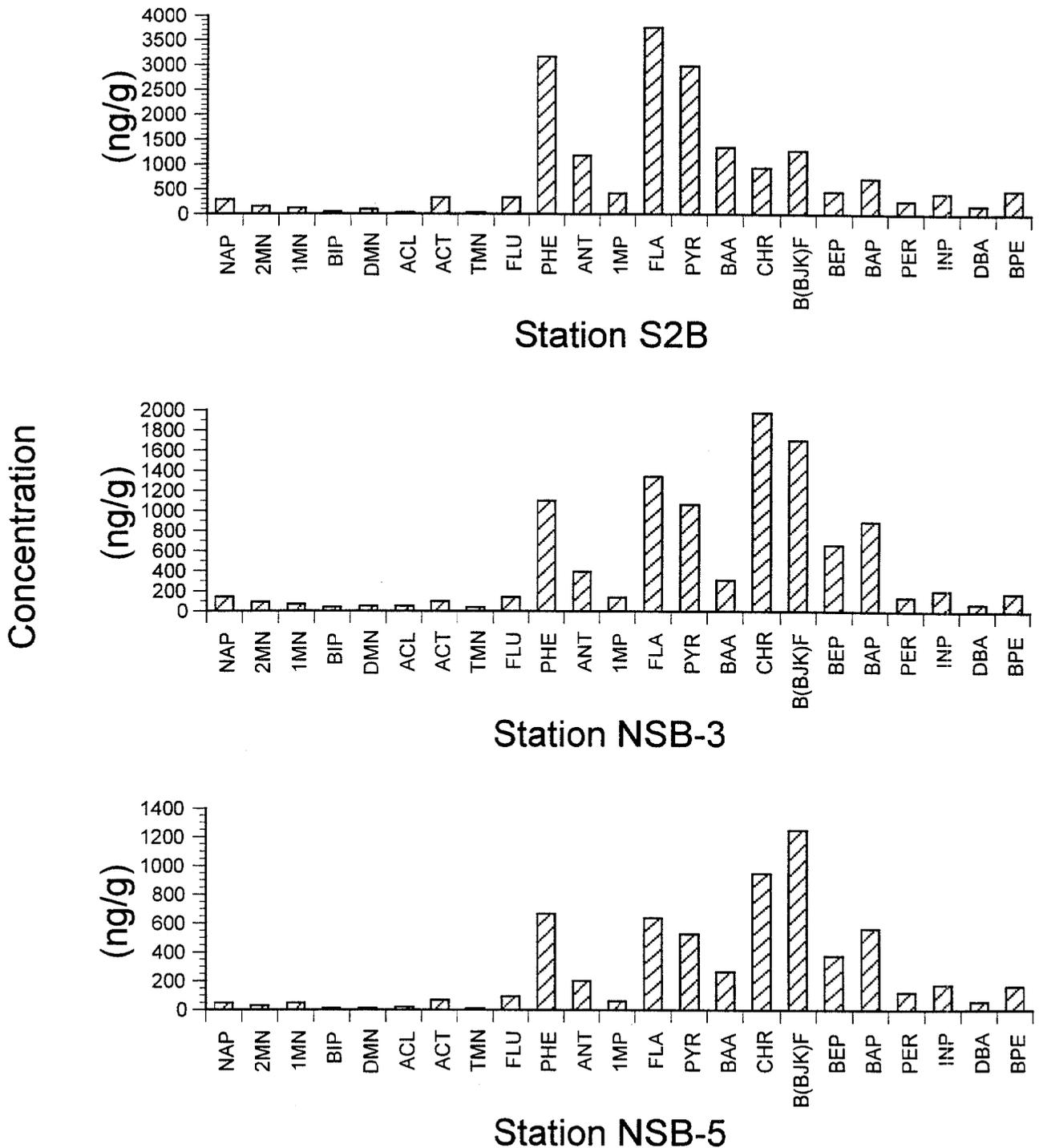


Figure 4.2-7. Concentration (ng/g dry weight sediment) of PAH compounds in surface sediments from selected stations in the McAllister Point Landfill study area. The depth of Stations NSB-3 and NSB-7 is 0-6 cm, while that of Station S2B is 0-2 cm.

Figure 4.2-7B. Explanation of PAH codes for Figure 4.2-7.

COMPGRP	Code	Analyte Name	CAS - NO.
PAH	1MN	1-Methylnaphthalene	90-12-0
PAH	1MP	1-Methylphenanthrene	832-69-9
PAH	2MN	2-Methylnaphthalene	91-57-6
PAH	ACL	Acenaphthylene	208-96-8
PAH	ACT	Acenaphthene	83-32-9
PAH	ANT	Anthracene	120-12-7
PAH	B(BJK)F	Benzo(k)fluoranthene	207-08-9
PAH	BAA	Benzo(a)anthracene	56-55-3
PAH	BAP	Benzo(a)pyrene	50-32-8
PAH	BEP	Benzo(e)pyrene	192-97-2
PAH	BIP	Biphenyl	92-52-4
PAH	BPE	Benzo(g,h,i)perylene	191-24-2
PAH	CHR	Chrysene	218-01-9
PAH	DBA	Dibenz(a,h)anthracene	53-70-3
PAH	DMN	2,6-Dimethylnaphthalene	581-42-0
PAH	FLA	Fluoranthene	206-44-0
PAH	FLU	Fluorene	86-73-7
PAH	INP	Indeno(1,2,3-cd)pyrene	193-39-5
PAH	NAP	Naphthalene	91-20-3
PAH	PER	Perylene	198-55-0
PAH	PHE	Phenanthrene	85-01-8
PAH	PYR	Pyrene	129-00-0
PAH	TMN	1,6,7-Trimethylnaphthalene	2245-38-7

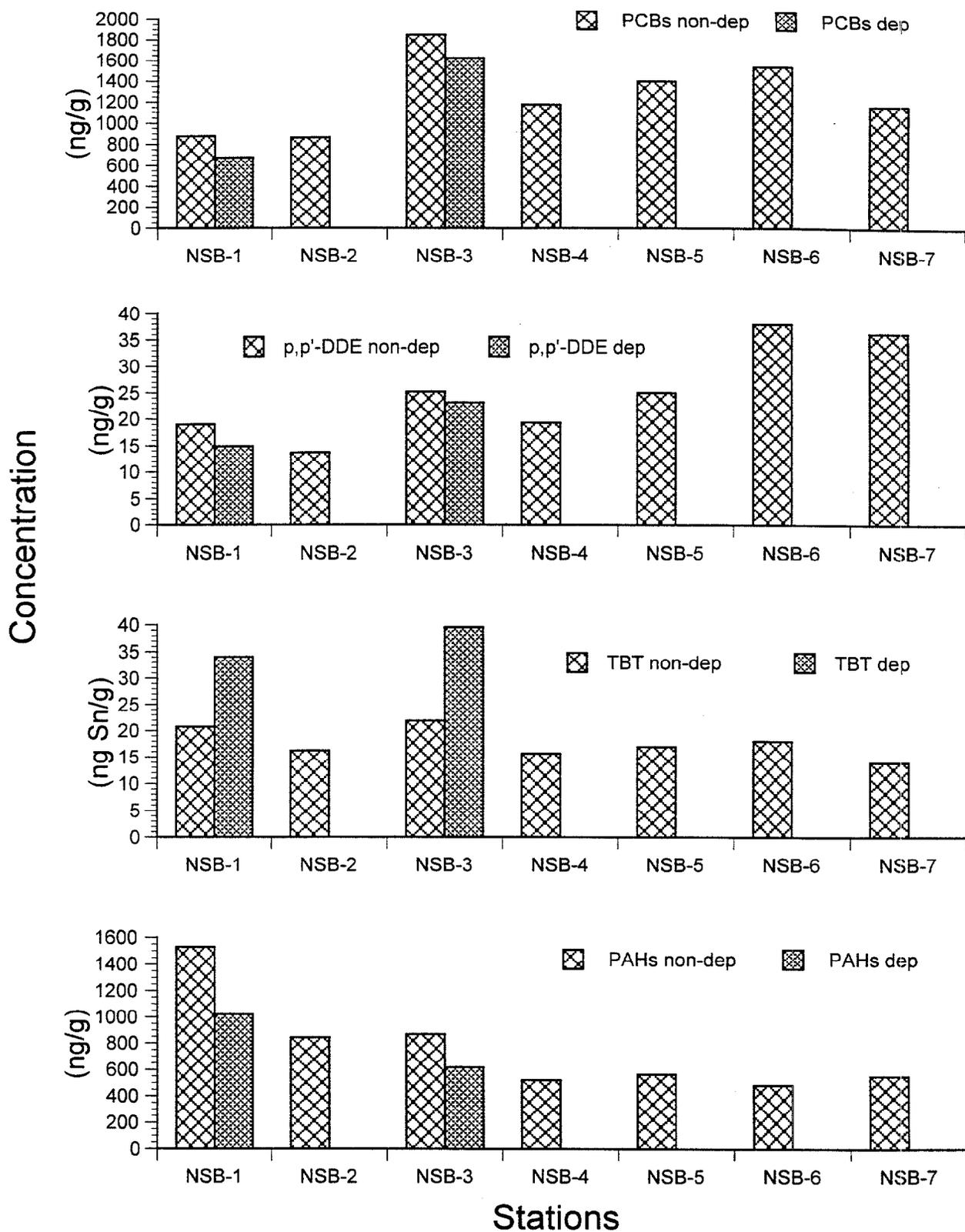


Figure 4.2-8. Concentration (ng/g dry weight tissue) of organic contaminants in blue mussels from the McAllister Point study area (non-dep = not depurated samples; dep = depurated samples).

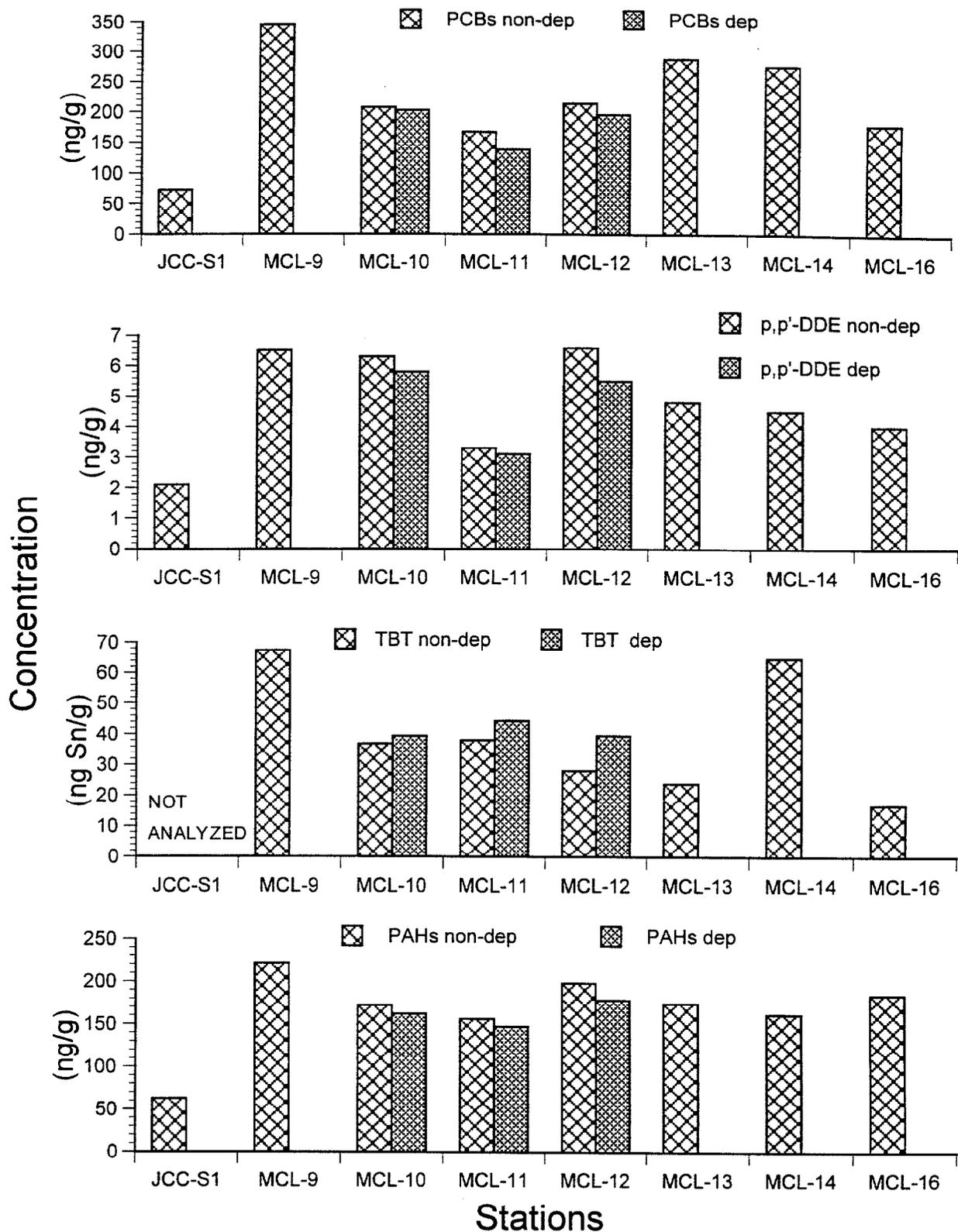


Figure 4.2-9. Concentration (ng/g dry weight tissue) of organic contaminants in hard clams from the McAllister Point Landfill study area and the Jamestown Cranston Cove (JCC) reference station JCC-S1 (non-dep = non-depurated samples; dep = depurated).

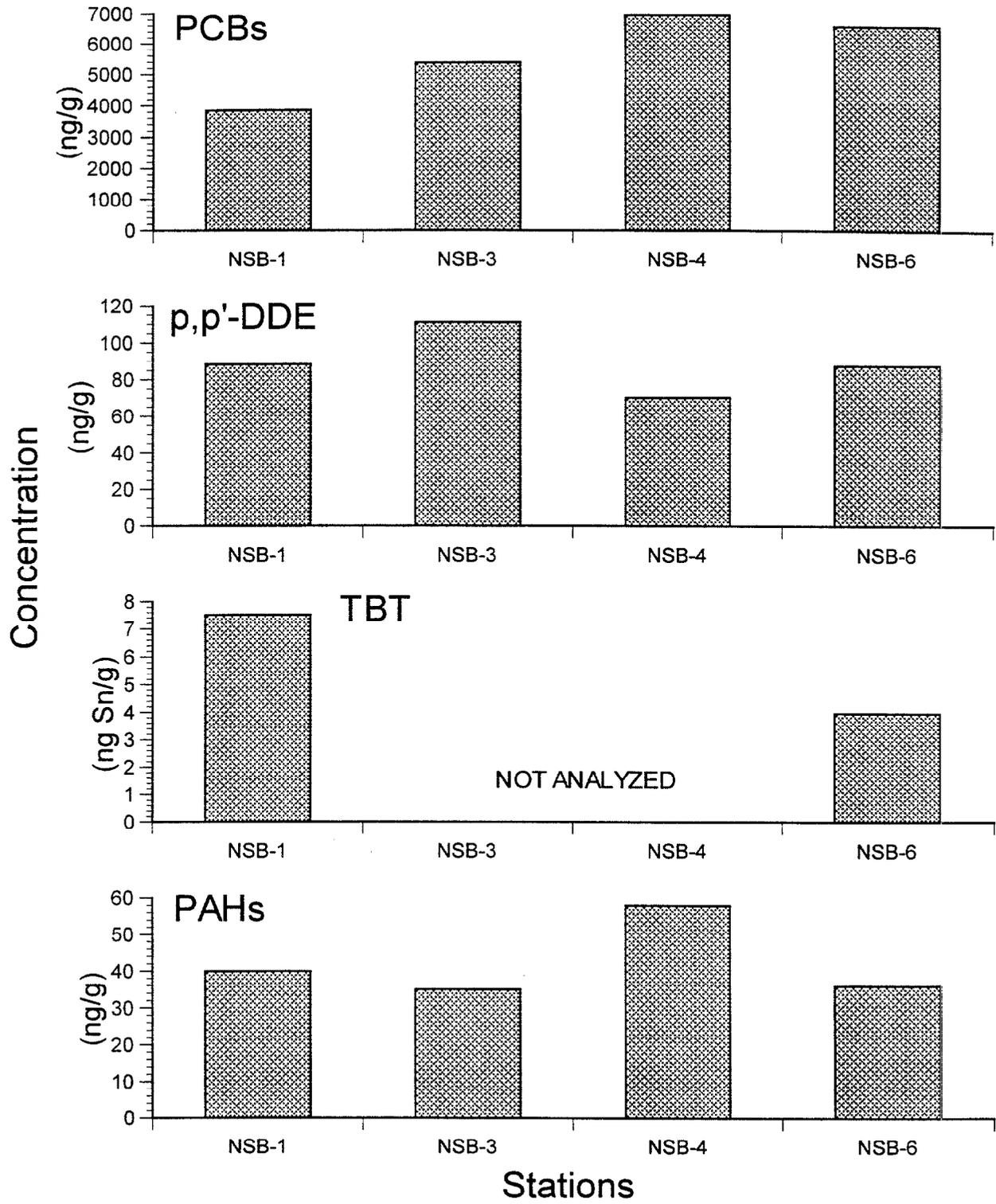


Figure 4.2-10. Concentration (ng/g dry weight tissue) of organic contaminants in fish (cunner) from the McAllister Point Landfill study area.

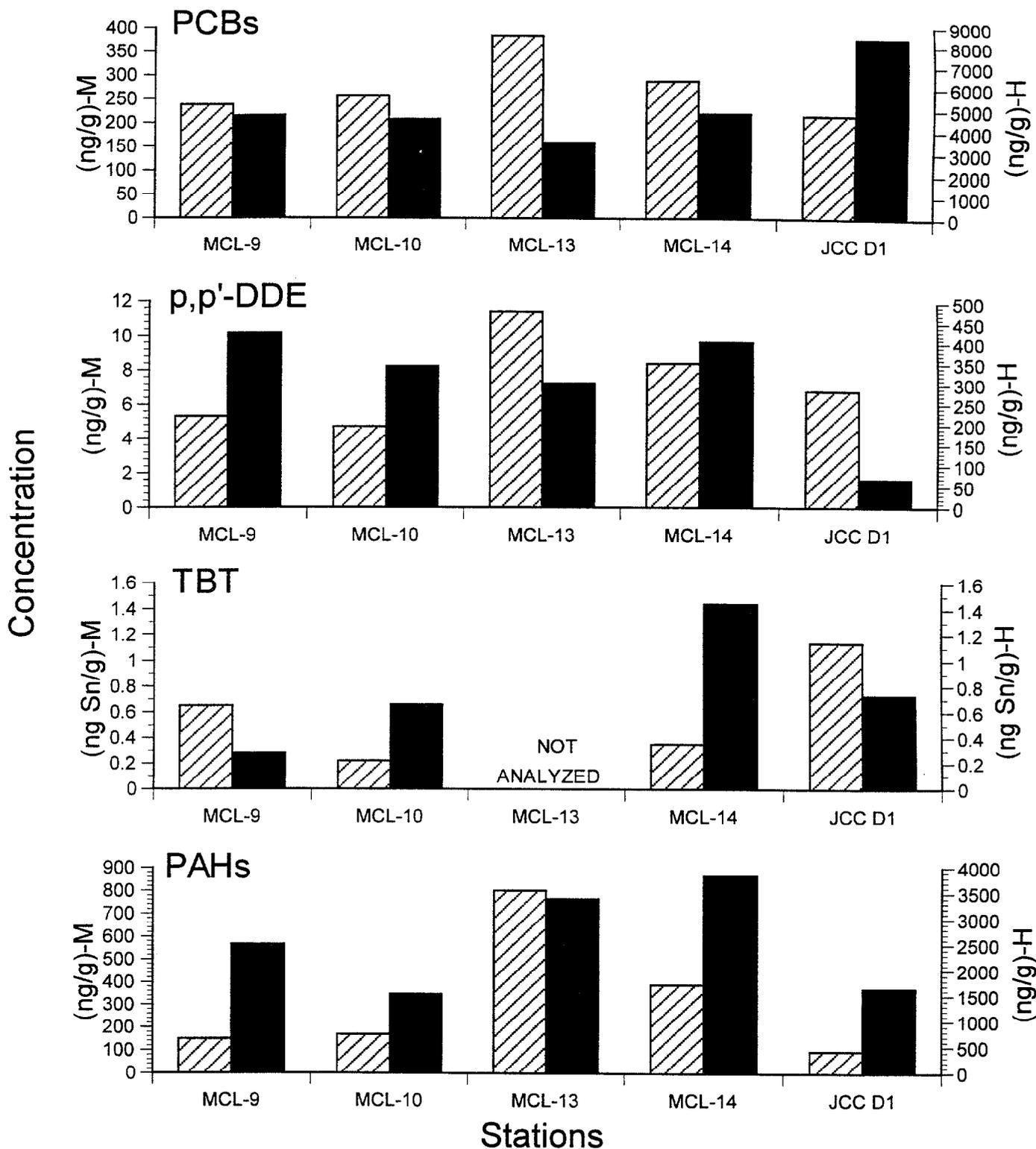


Figure 4.2-11. Concentration (ng/g dry weight tissue) of organic contaminants in lobster muscle (M) and hepatopancreas (H) from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference Station JCC-D1. Patterned bars indicate muscle concentrations; solid bars indicate hepatopancreas concentrations. Note scale difference for muscle and hepatopancreas, except for TBT.

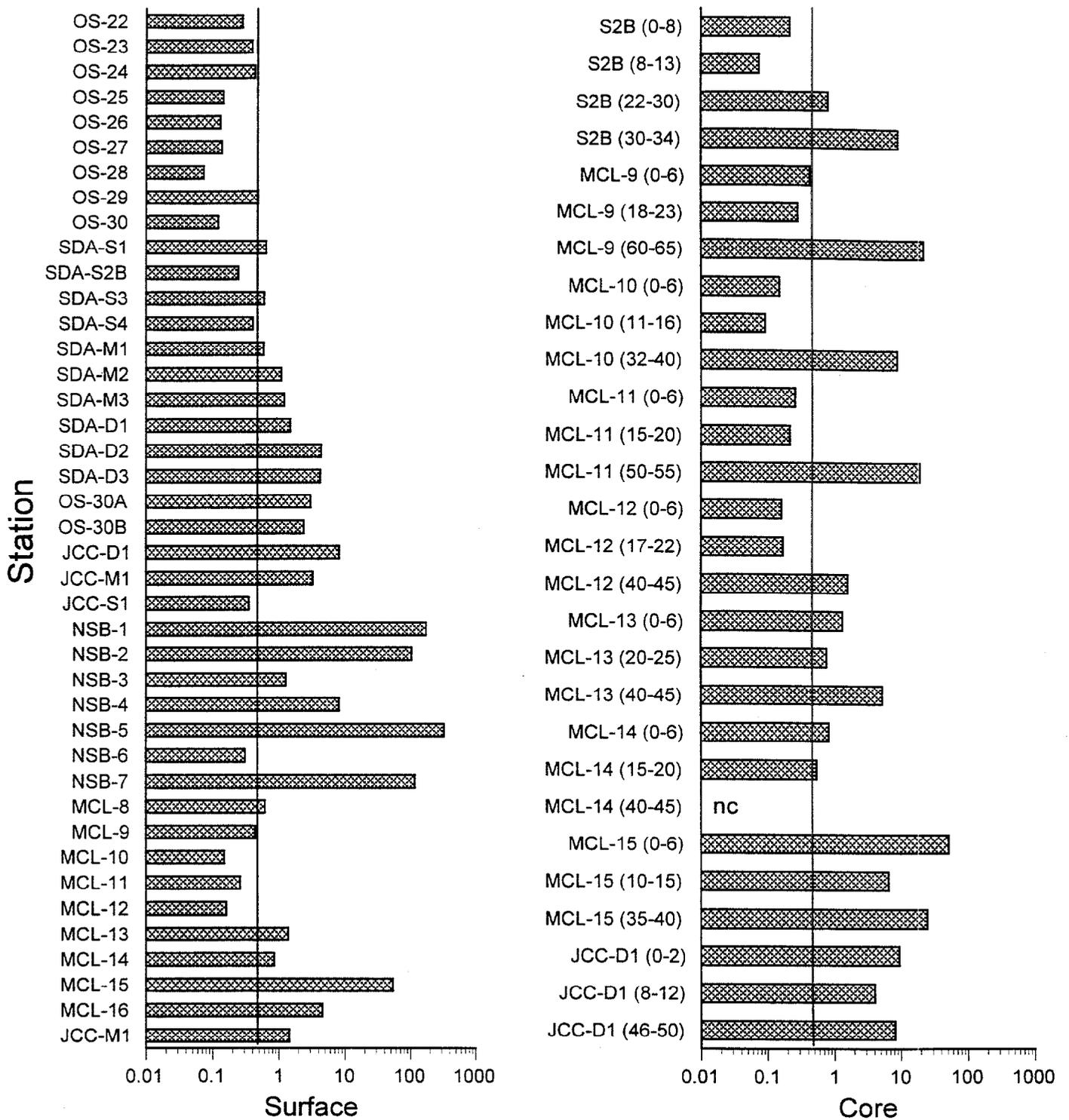


Figure 4.2-12. SEM/AVS ratios for surface and core samples from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Numbers in parenthesis indicates core depths. Lines at 0.5 indicate potential bioavailability of metals. Values greater than 0.5 indicate bioavailability and potential toxicity. "nc" indicates ratio not calculated due to non-detected levels of AVS.

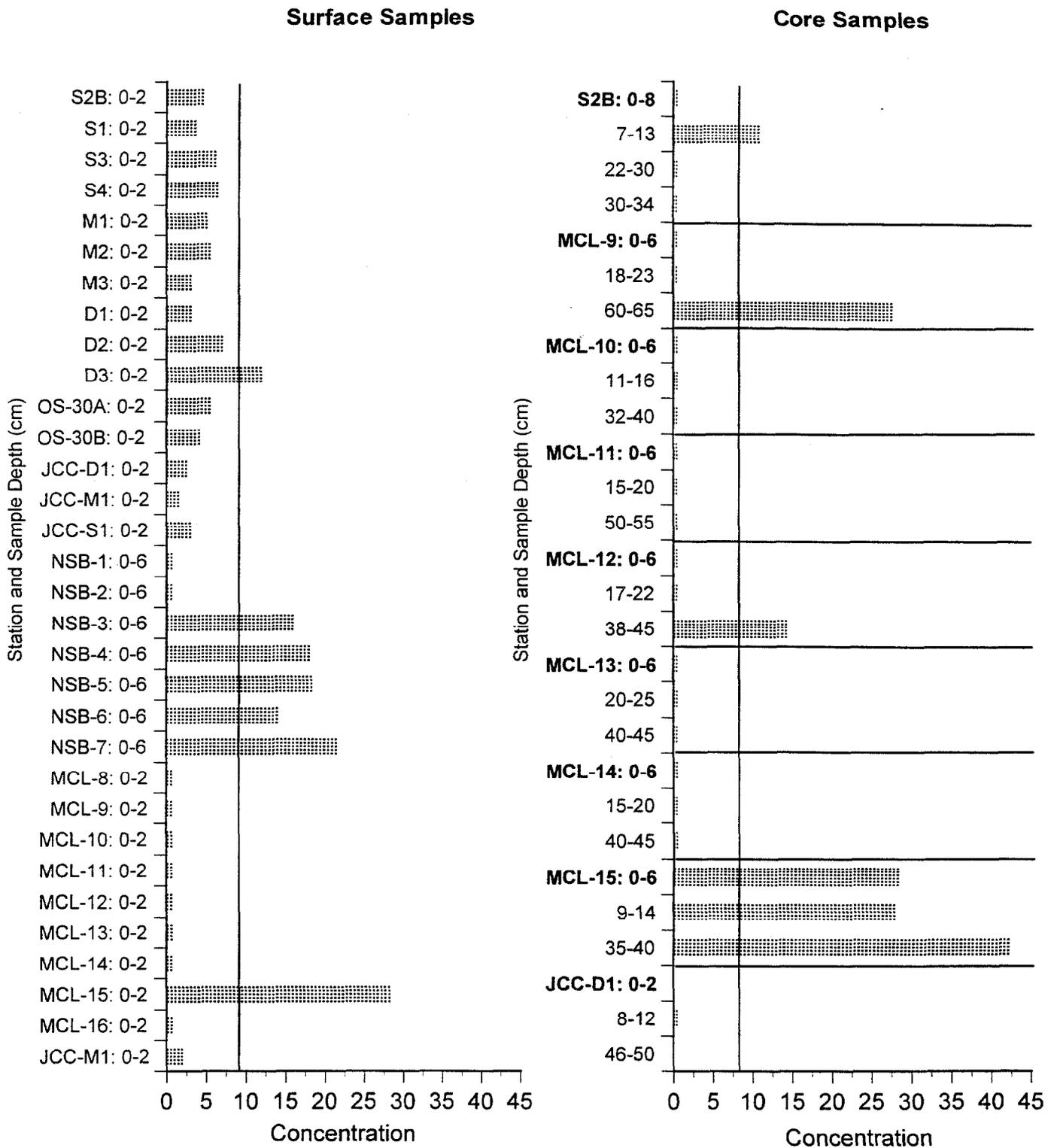


Figure 4.2-13. Arsenic concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical line indicates NOAA ER-L Guideline (Long *et al.*, 1995).

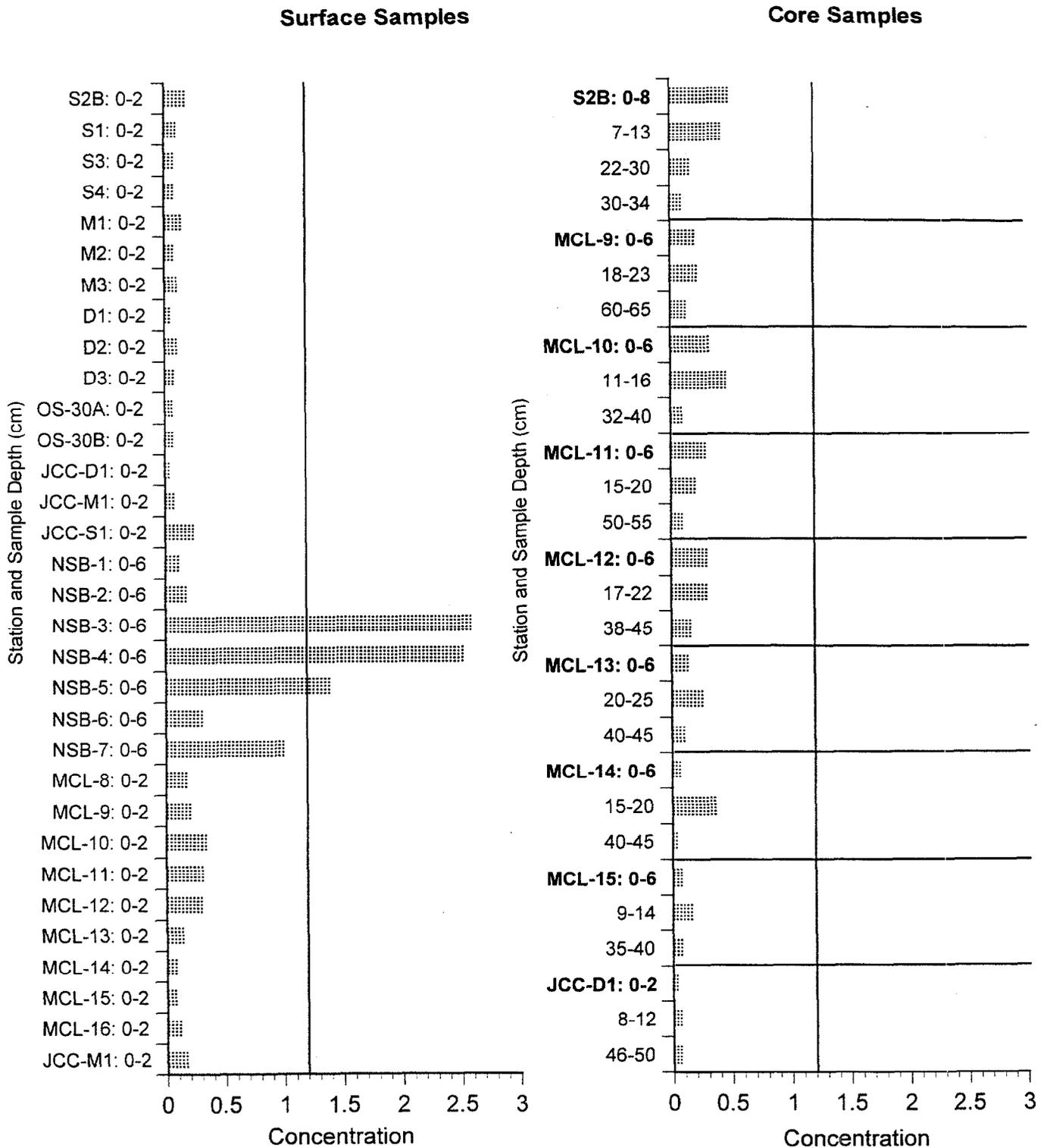


Figure 4.2-14. Cadmium concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical line indicates NOAA ER-L Guideline (Long *et al.*, 1995).

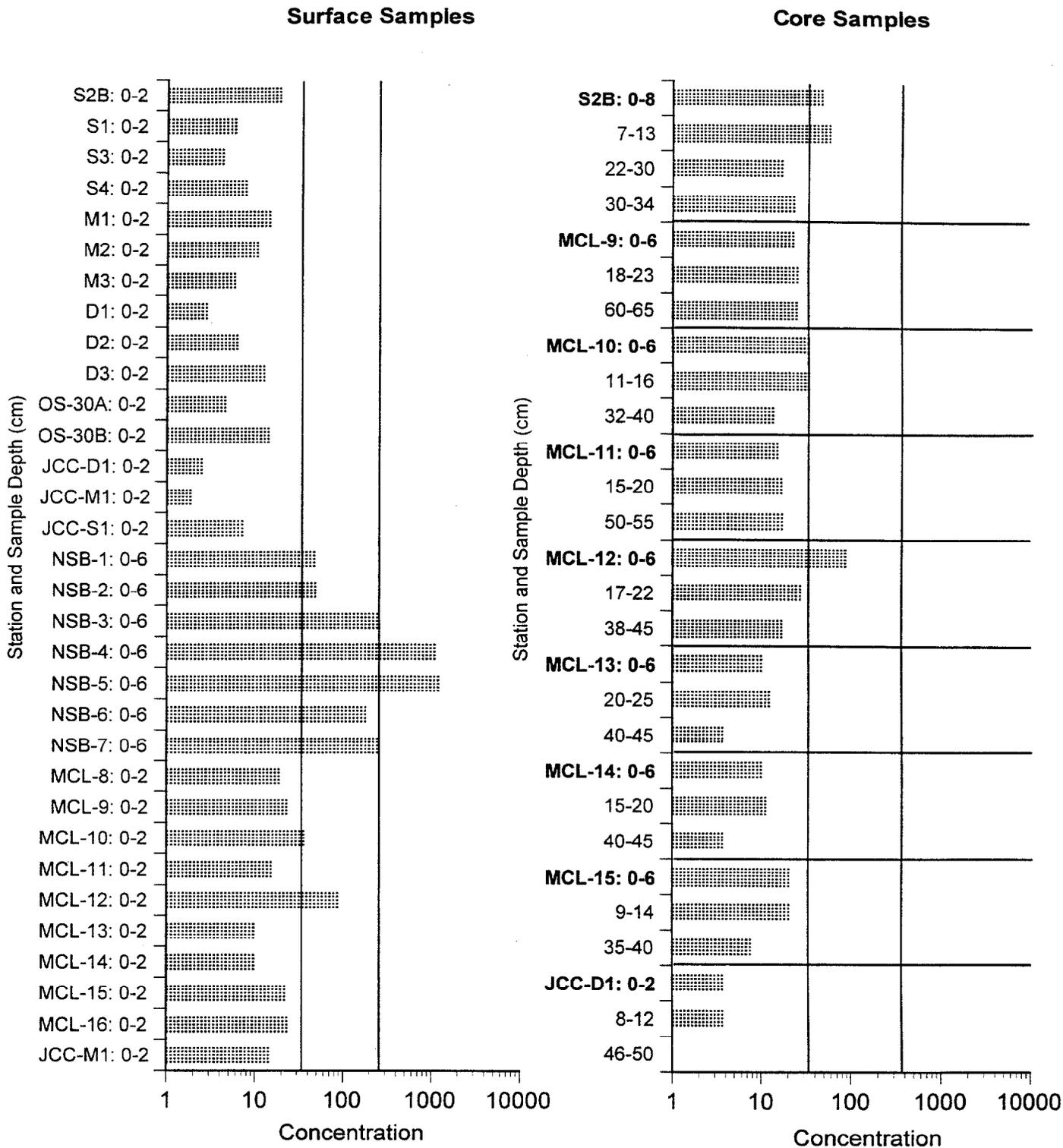


Figure 4.2-15. Copper concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical lines indicate NOAA ER-L and ER-M Guidelines (Long *et al.*, 1995).

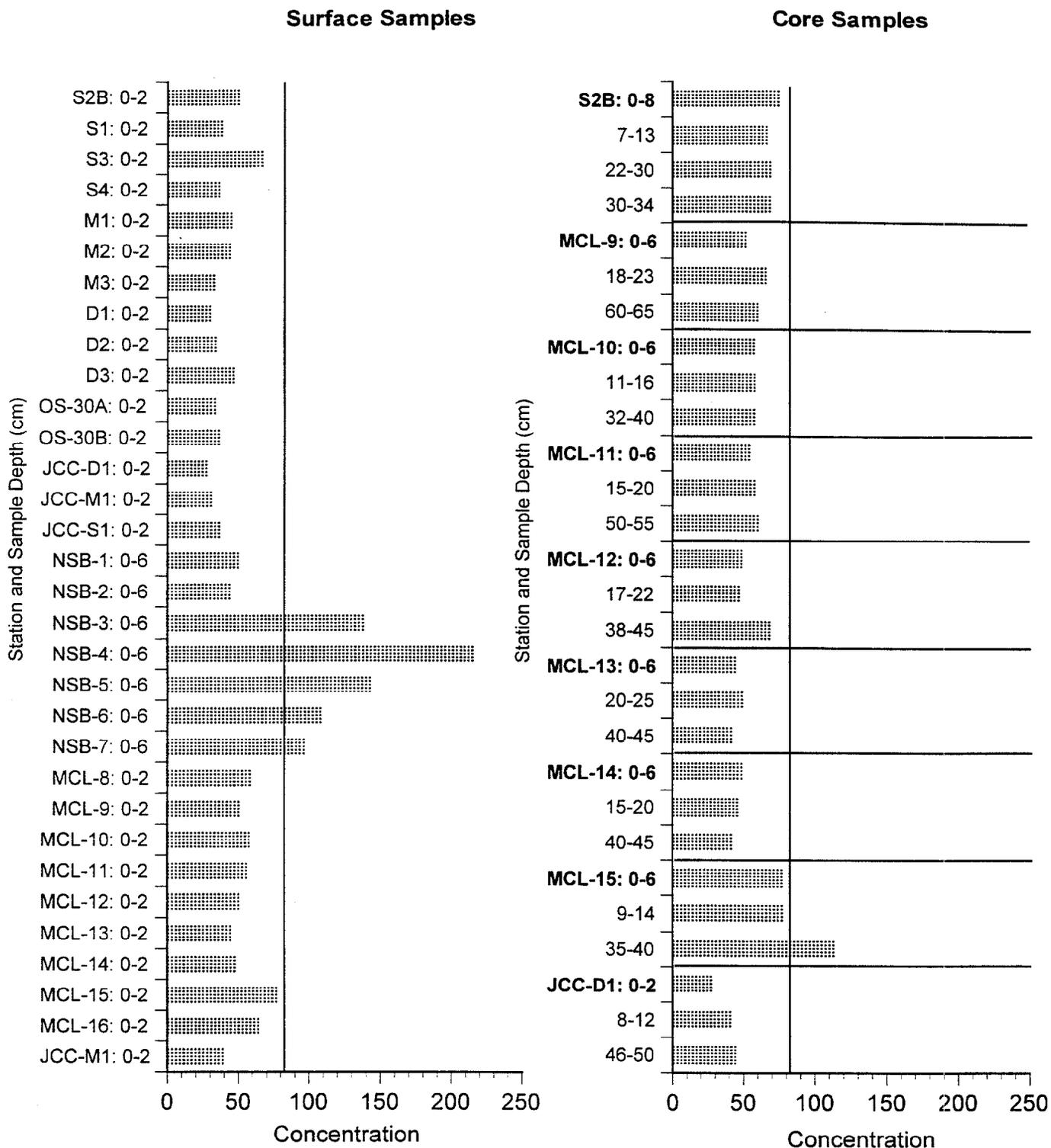


Figure 4.2-16. Chromium concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical line indicates NOAA ER-L Guideline (Long *et al.*, 1995).

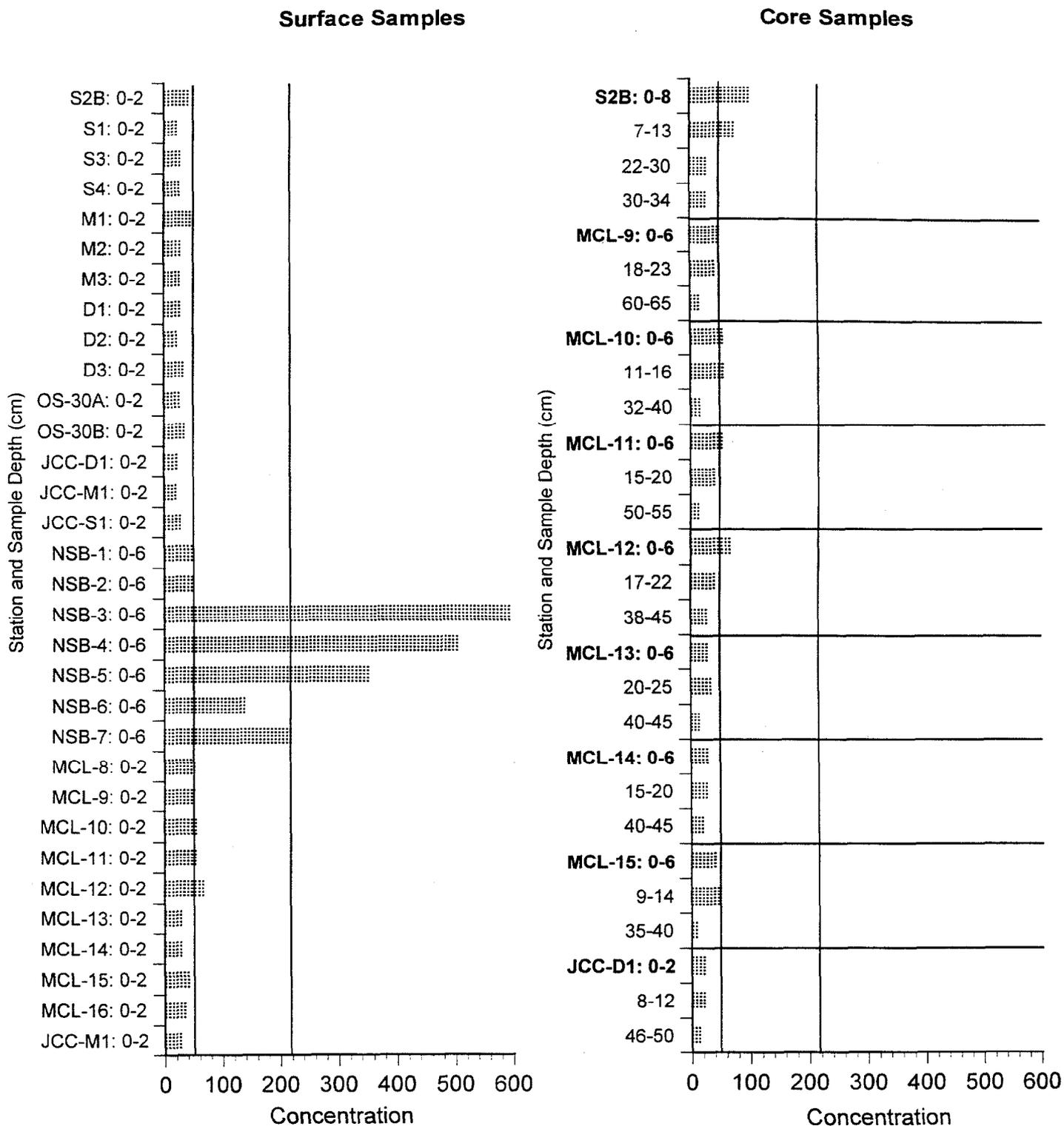


Figure 4.2-17. Lead concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical lines indicate NOAA ER-L and ER-M Guidelines (Long *et al.*, 1995).

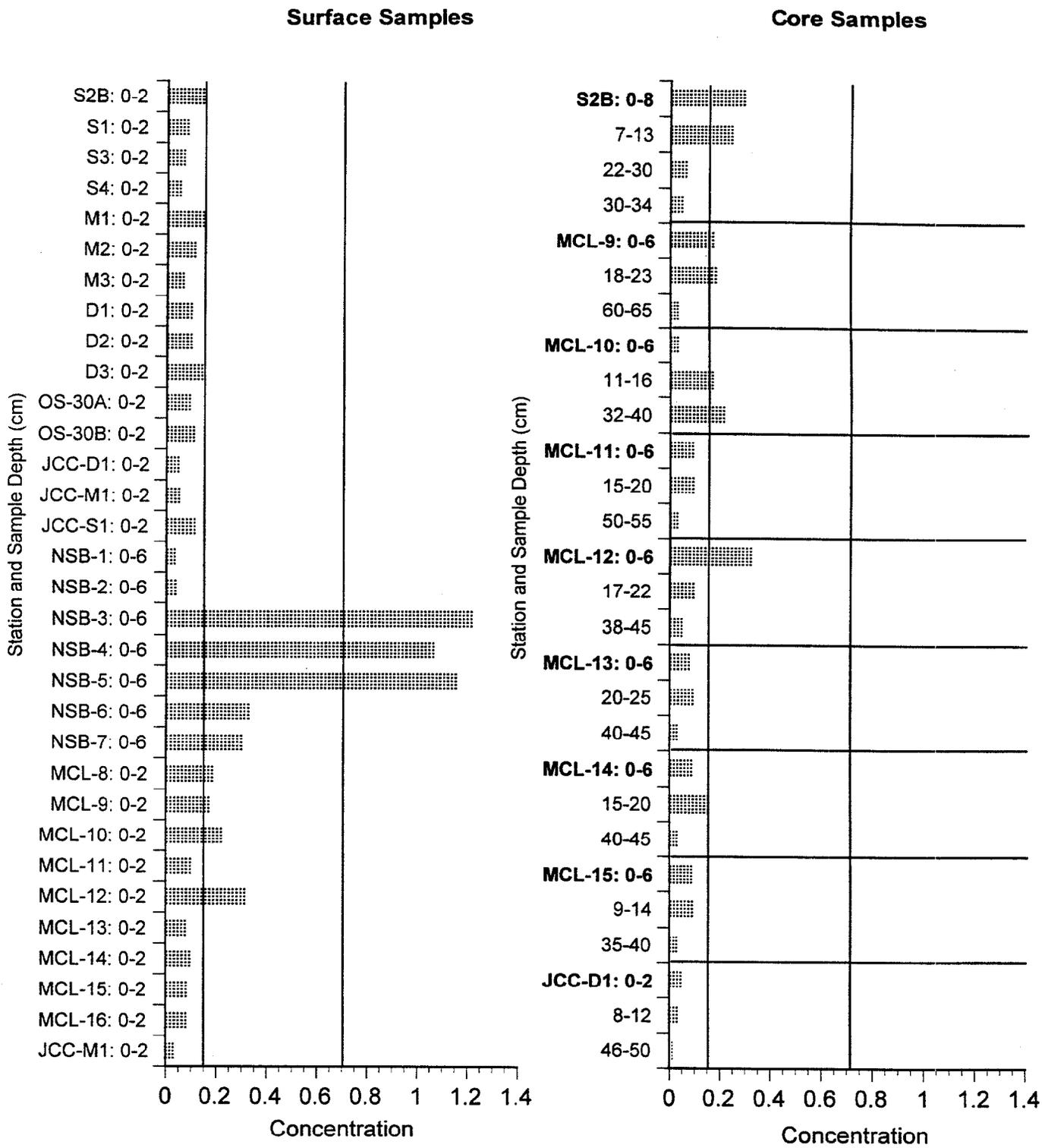


Figure 4.2-18. Mercury concentrations (µg/g dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical lines indicate NOAA ER-L and ER-M Guidelines (Long *et al.*, 1995).

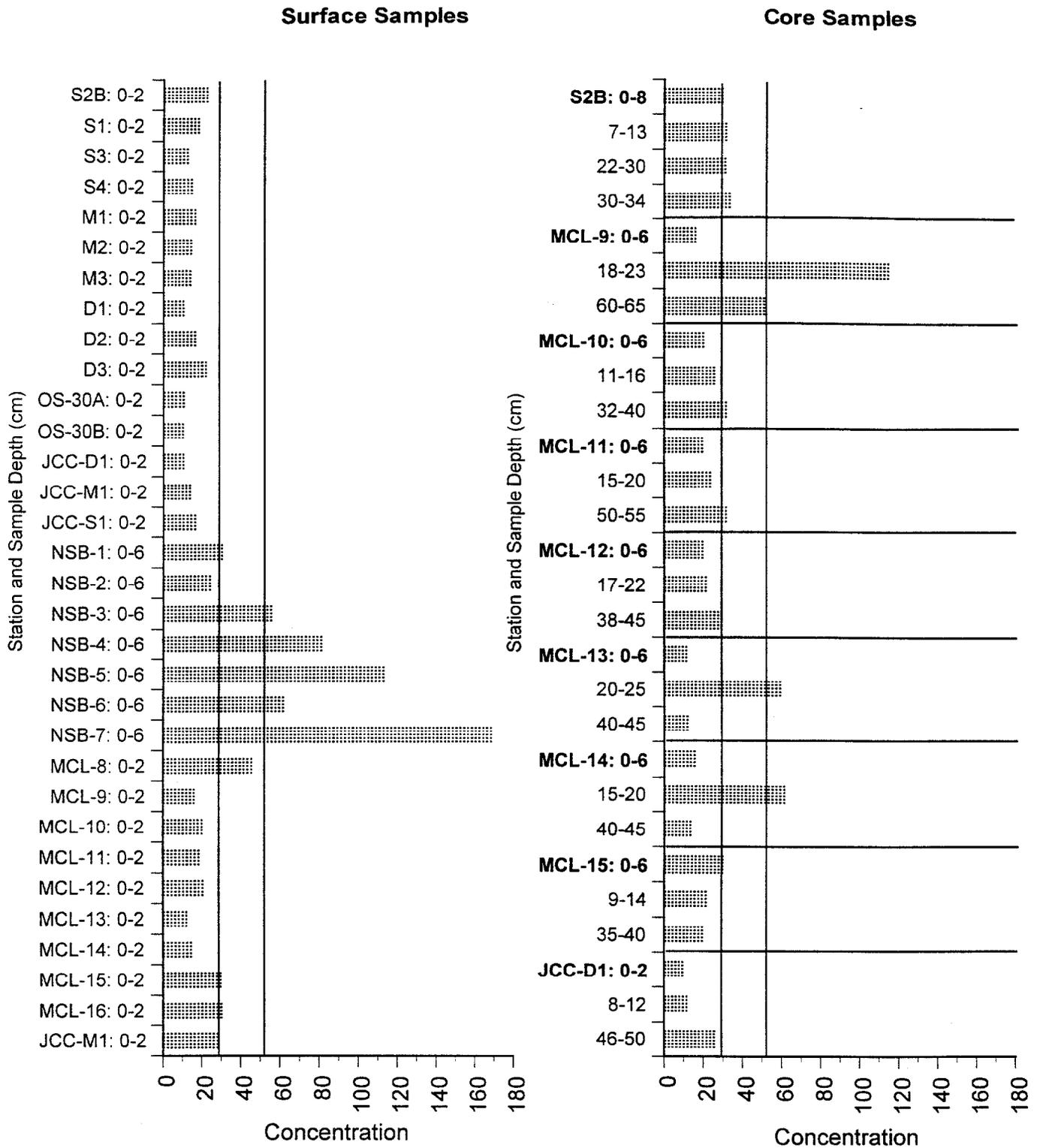
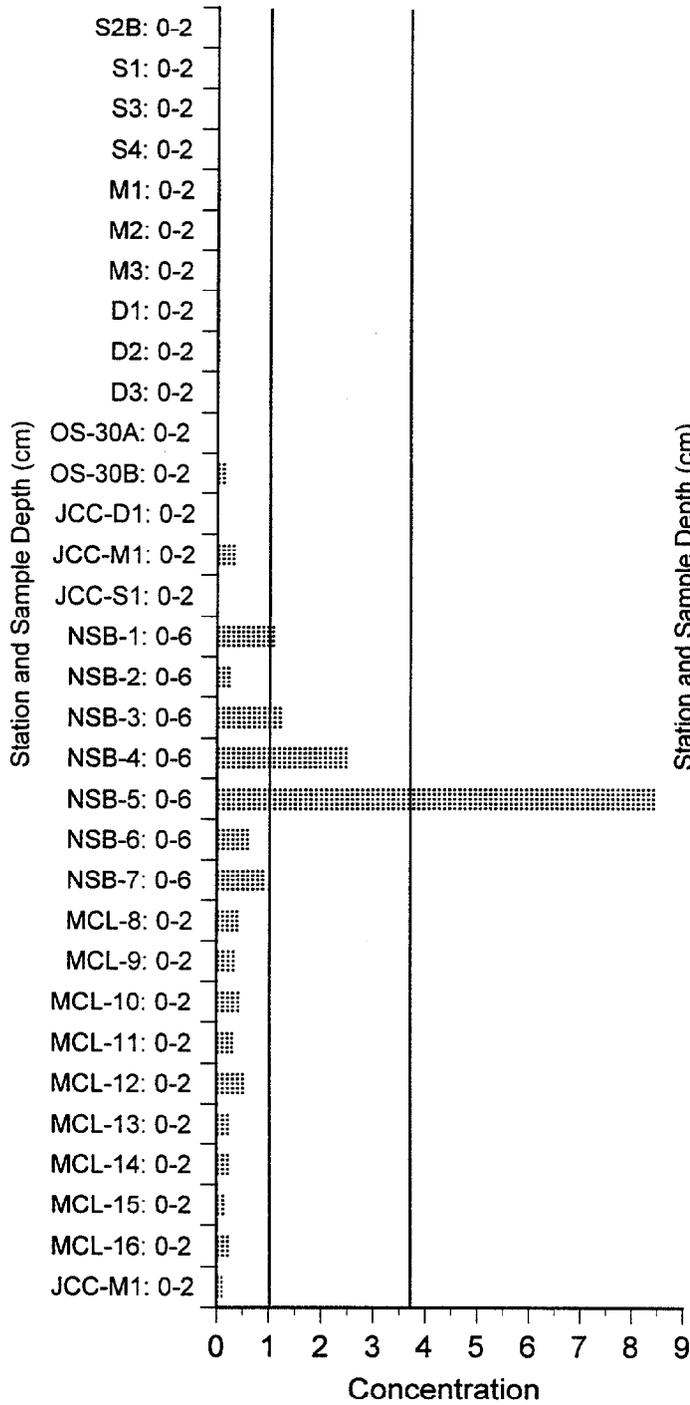


Figure 4.2-19. Nickel concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical lines indicate NOAA ER-L and ER-M Guidelines (Long *et al.*, 1995).

### Surface Samples



### Core Samples

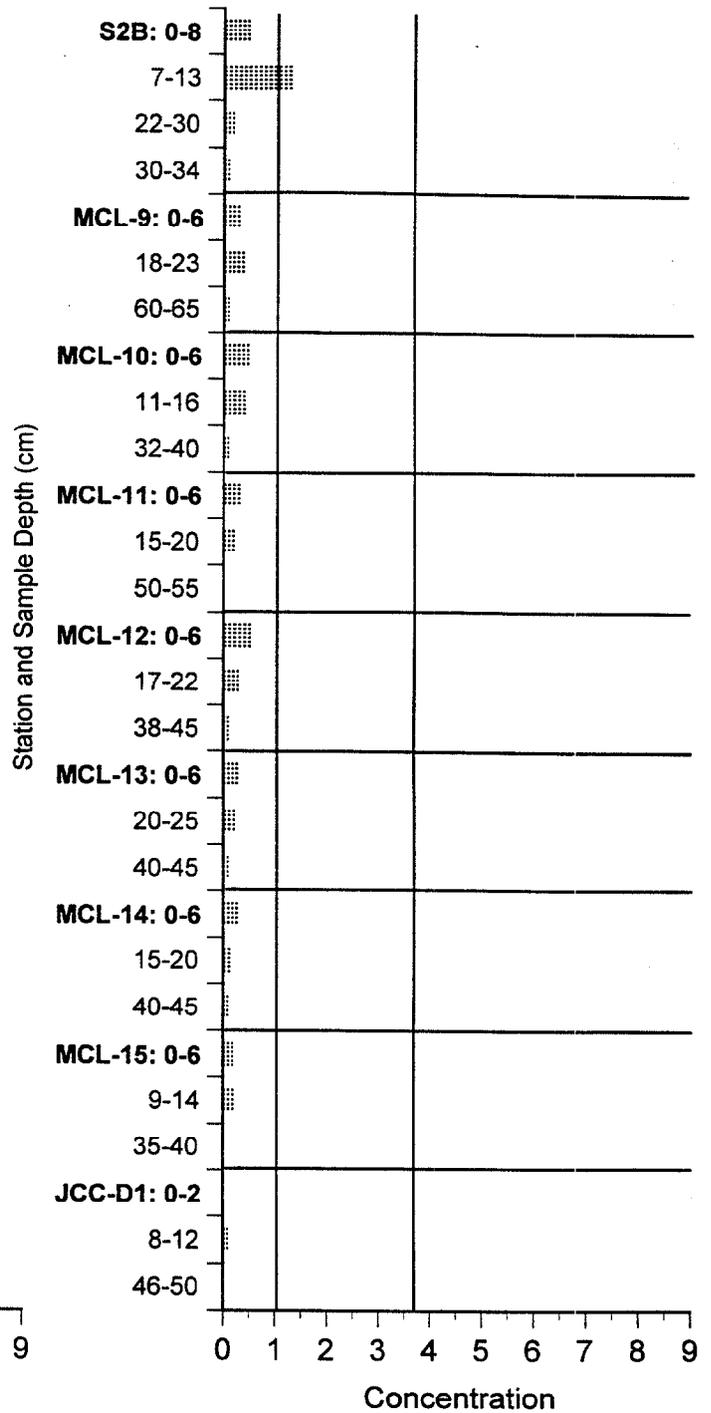


Figure 4.2-20. Silver concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical lines indicate NOAA ER-L and ER-M Guidelines (Long *et al.*, 1995).

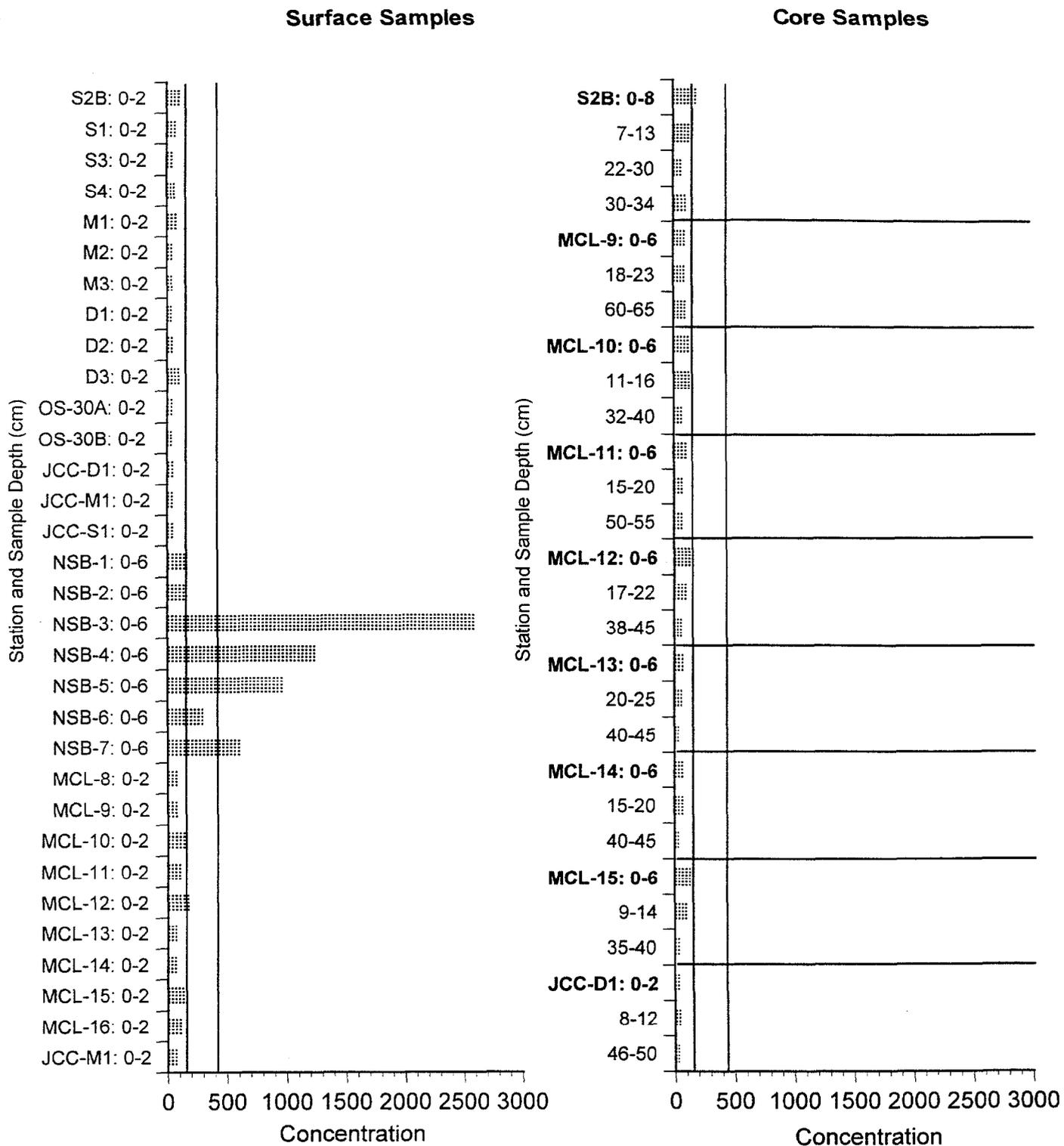
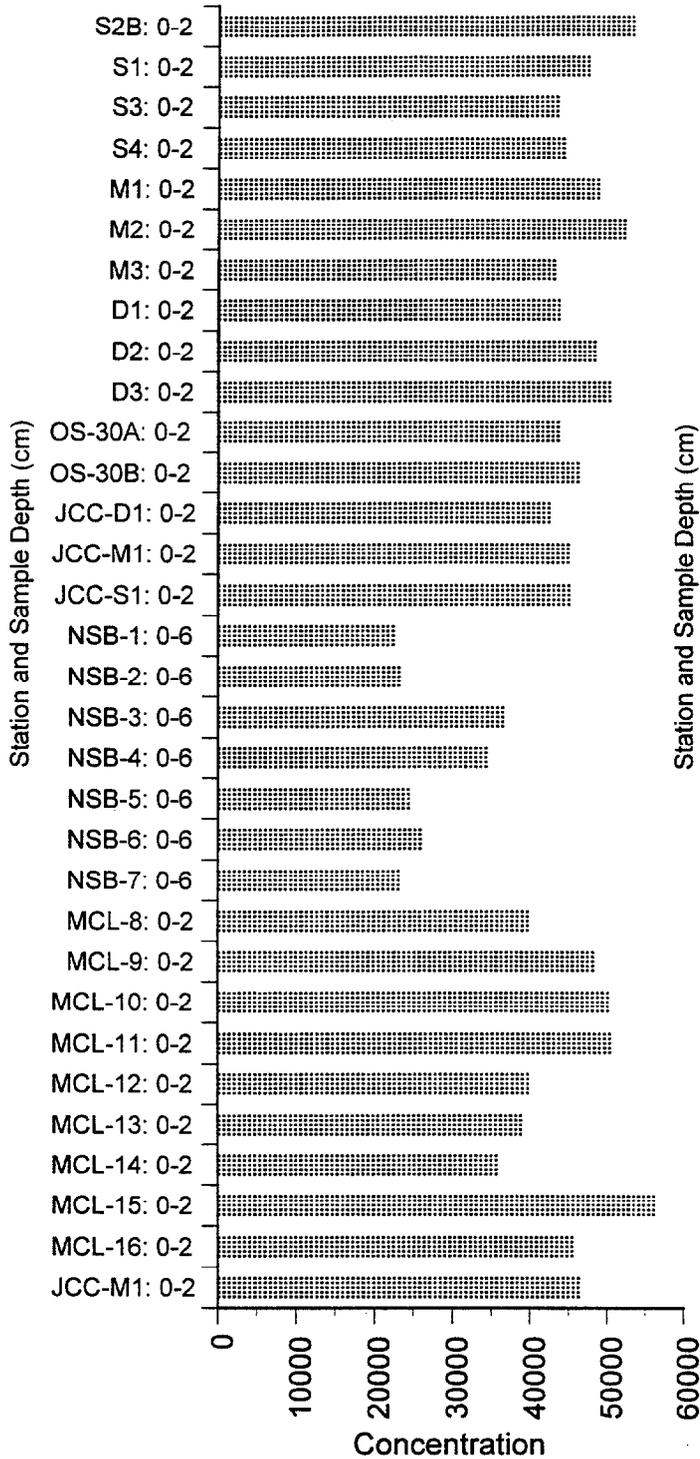


Figure 4.2-21. Zinc concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical lines indicate NOAA ER-L and ER-M Guidelines (Long *et al.*, 1995).

### Surface Samples



### Core Samples

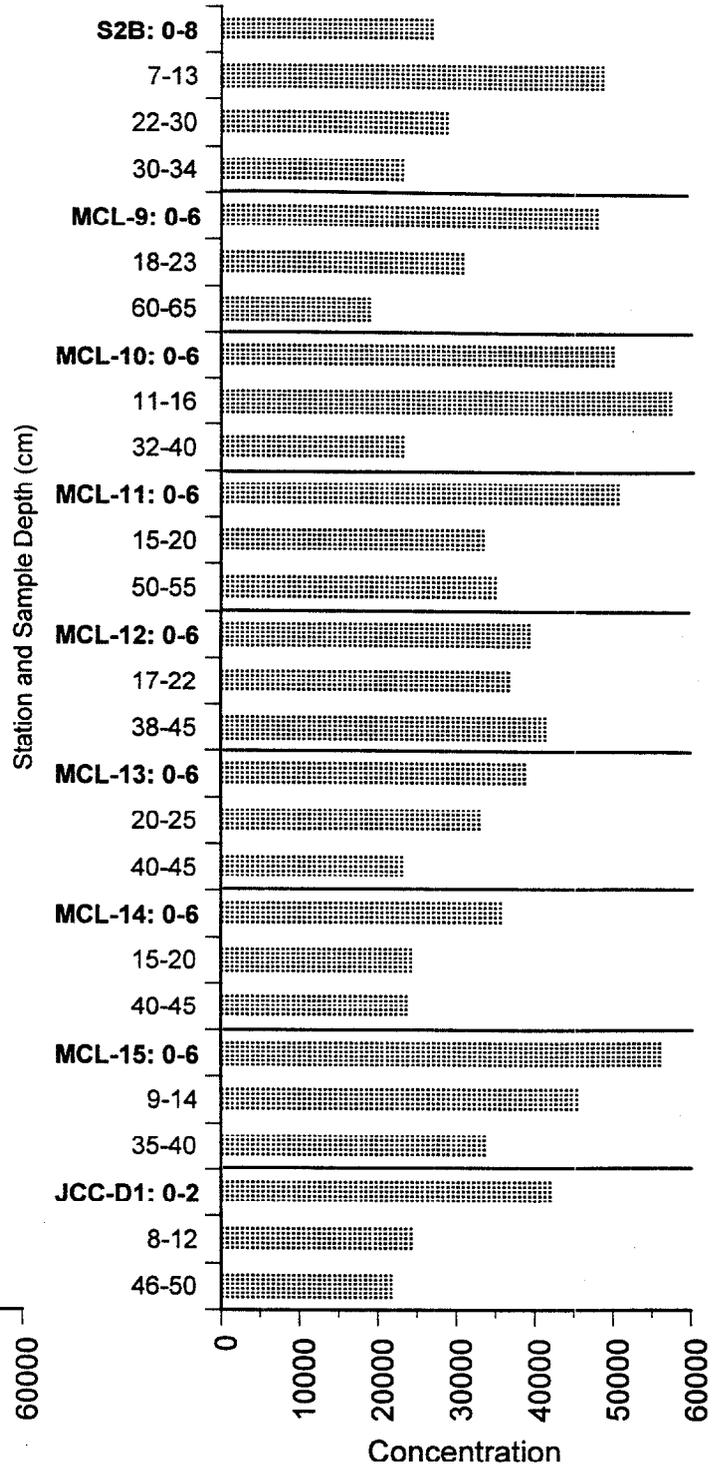


Figure 4.2-22. Aluminum concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

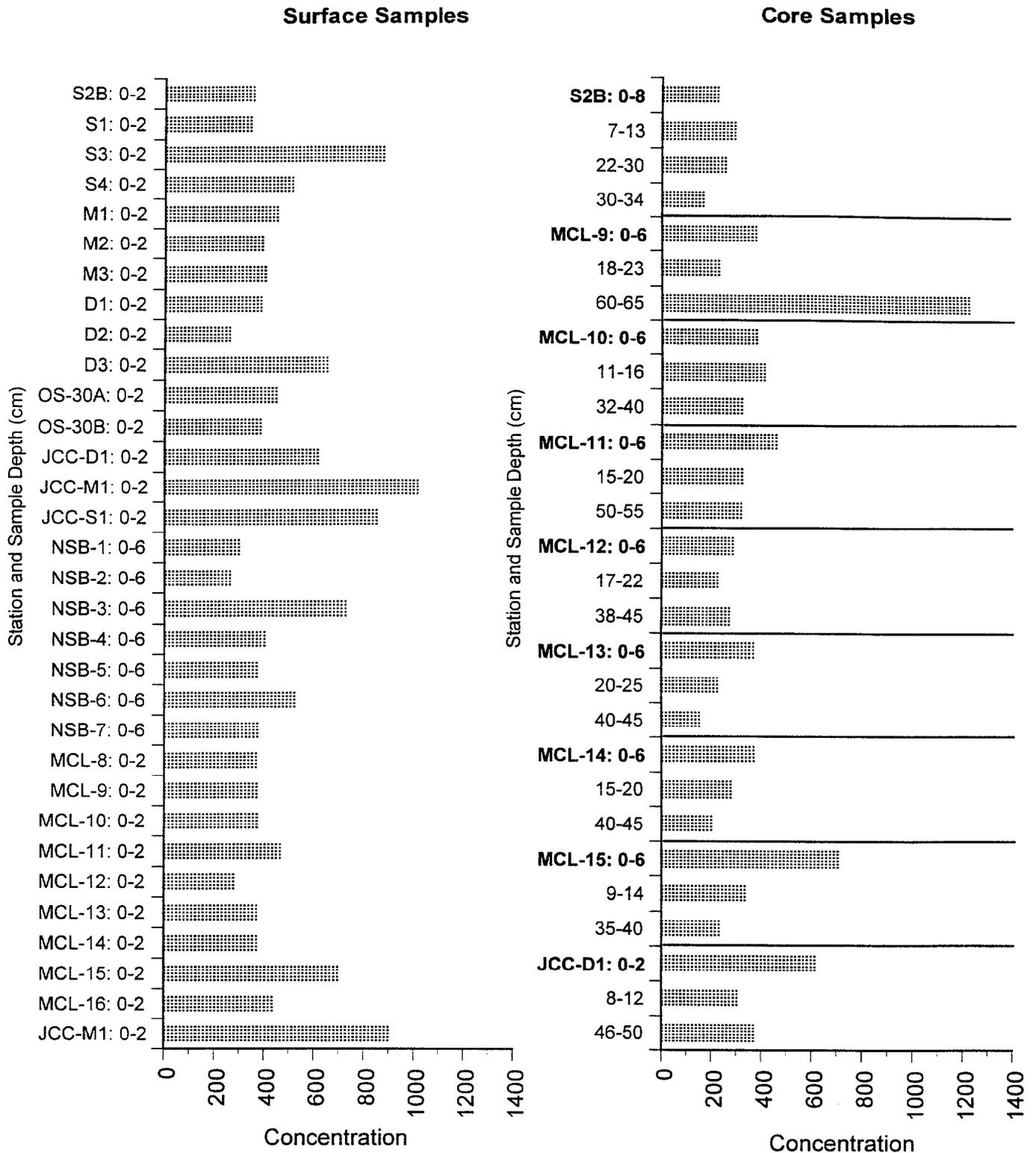


Figure 4.2-23. Manganese concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

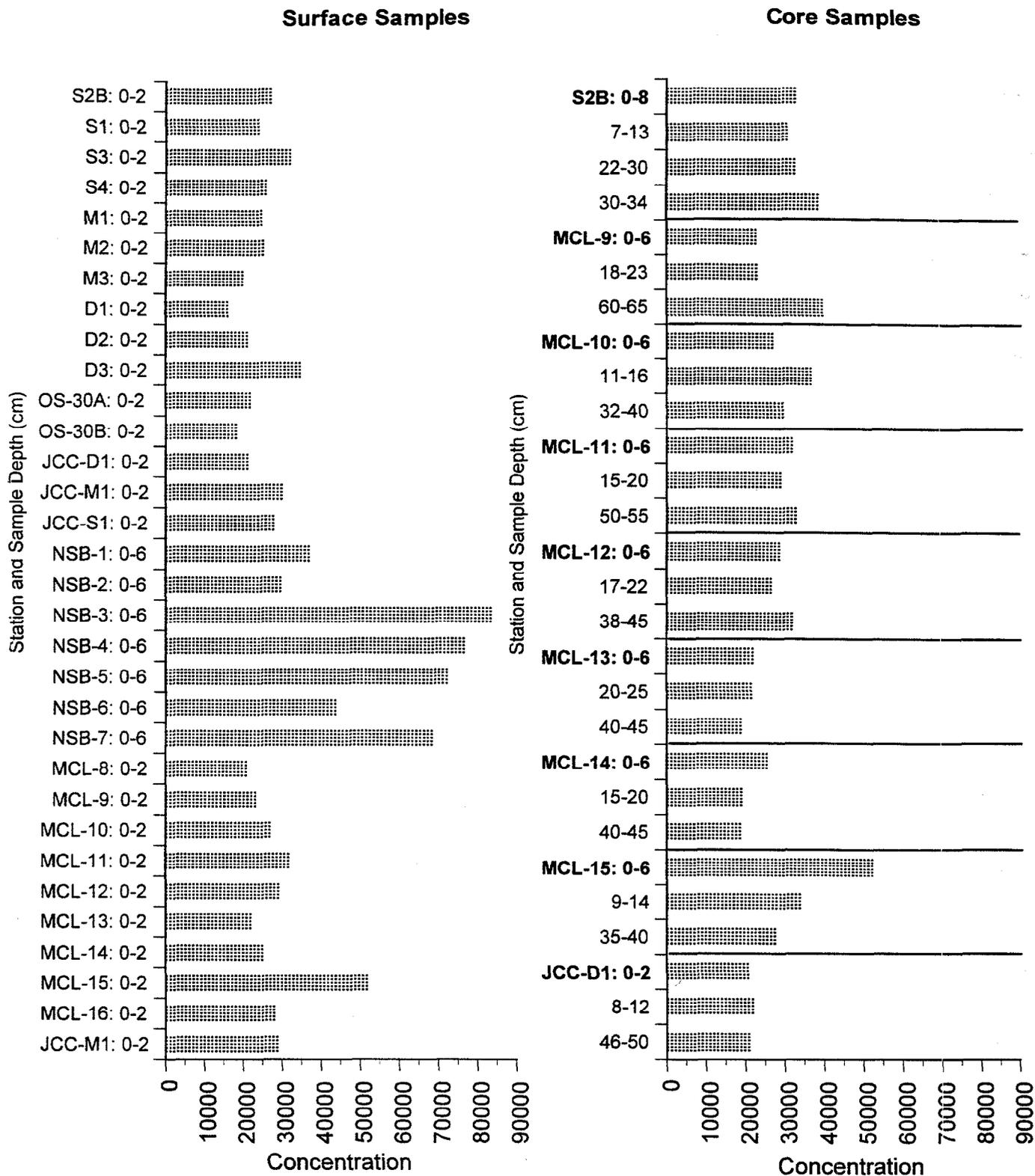


Figure 4.2-24. Iron concentrations ( $\mu\text{g/g}$  dry weight) in surface and core sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

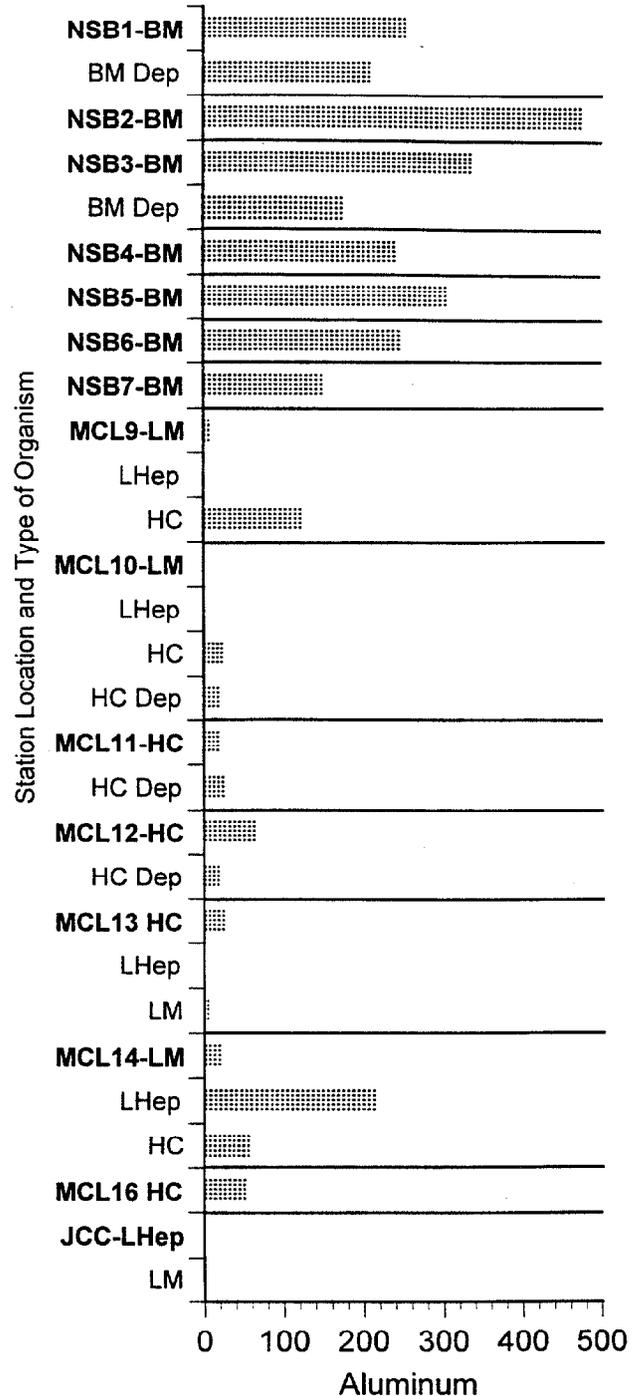
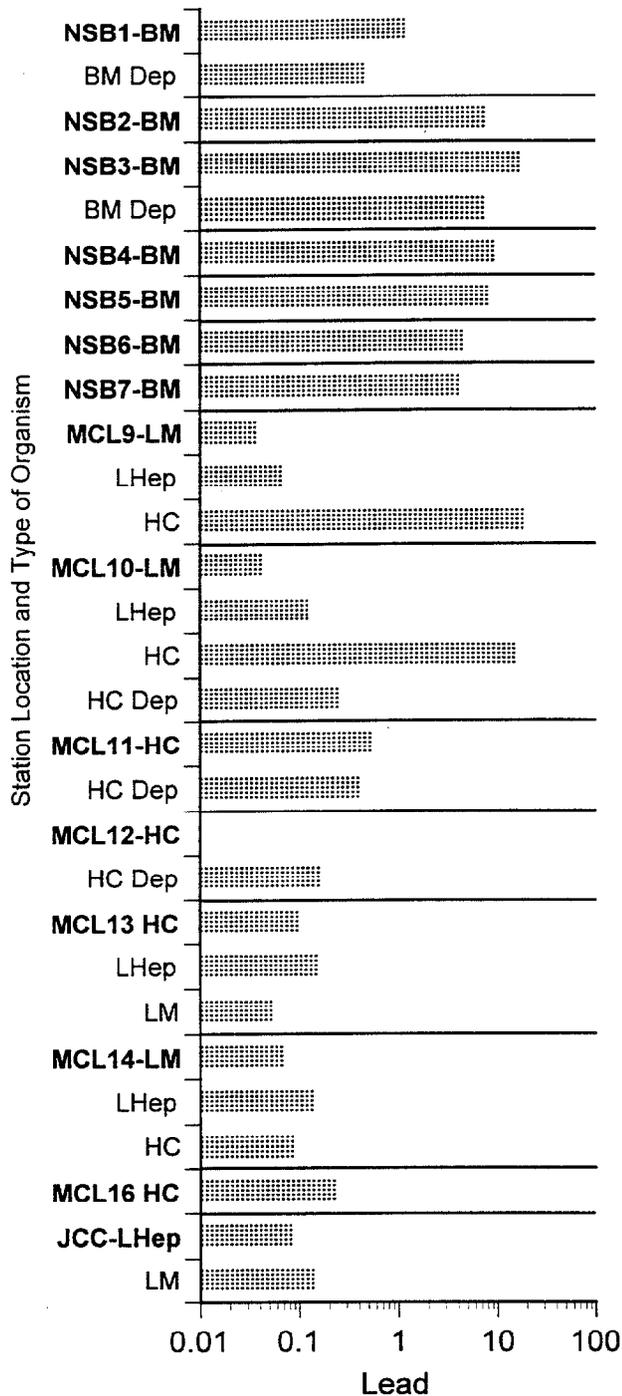


Figure 4.2-25. Tissue concentrations of Lead and Aluminum ( $\mu\text{g/g}$  dry weight) for blue mussels (BM), depurated blue mussels (BM Dep), hard clams (HC), depurated hard clams (HC Dep), lobster muscle (LM), and lobster hepatopancreas (LHep) from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

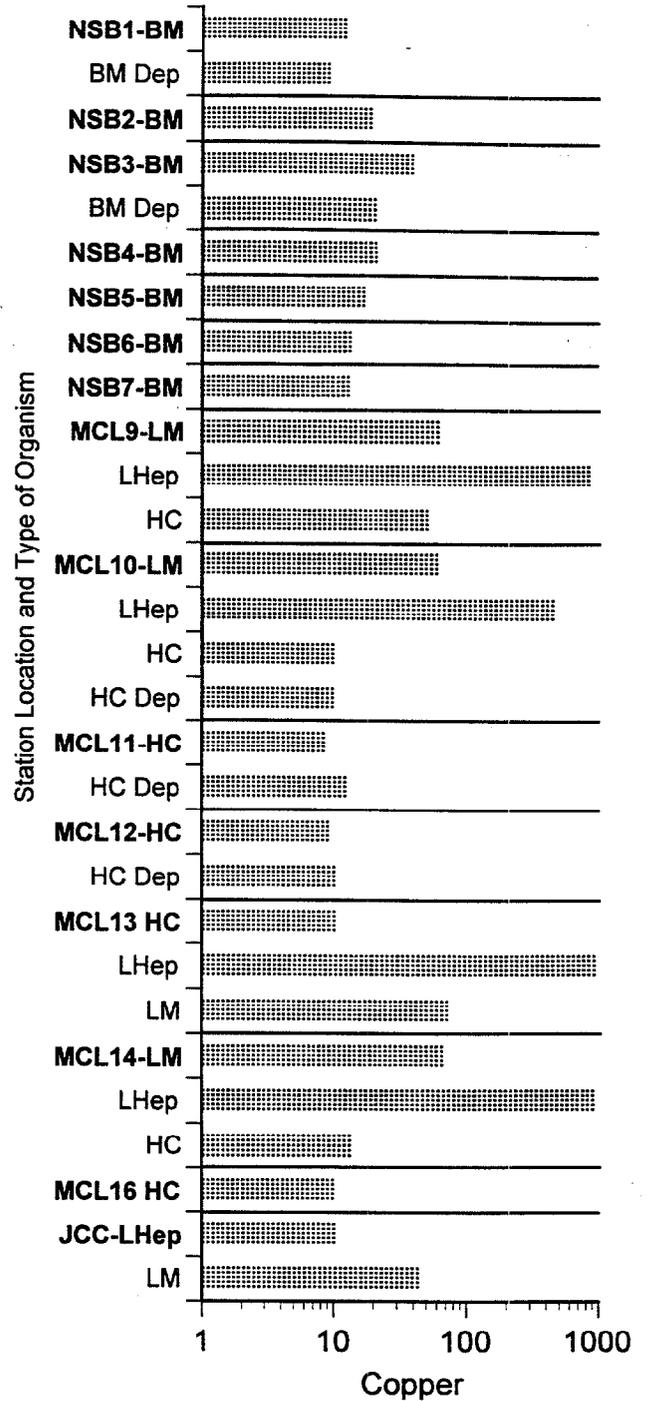
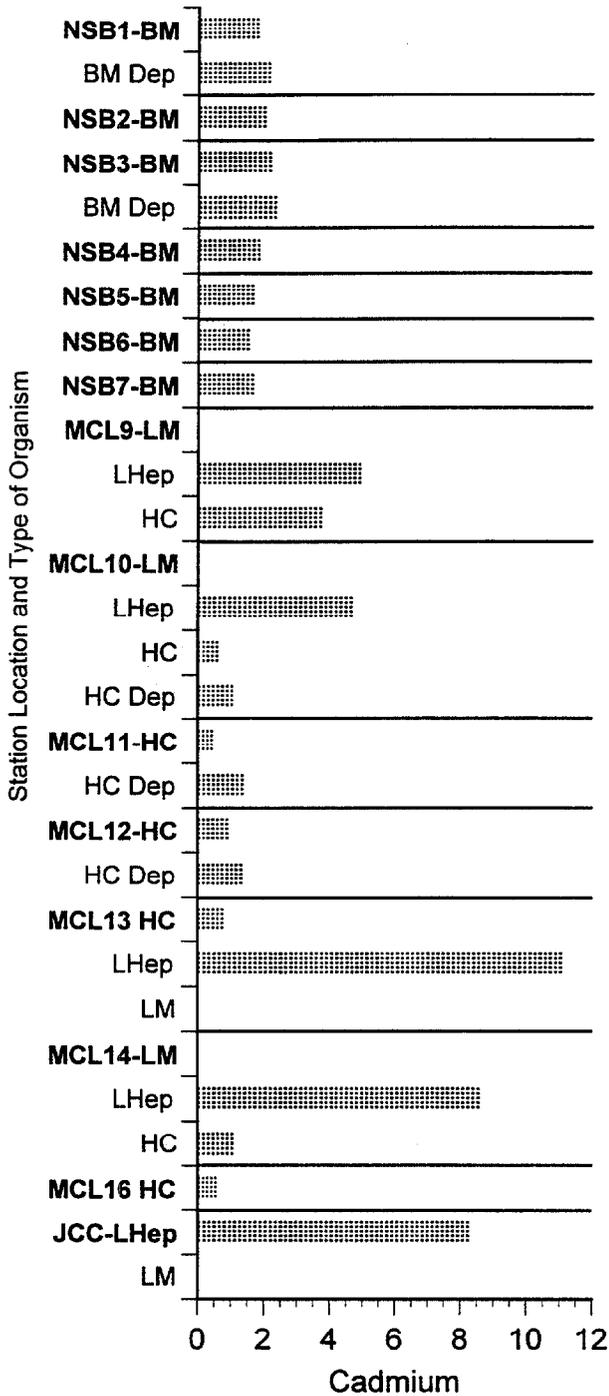


Figure 4.2-26. Tissue concentrations of Cadmium and Copper ( $\mu\text{g/g}$  dry weight) for blue mussels (BM), depurated blue mussels (BM Dep), hard clams (HC), depurated hard clams (HC Dep), lobster muscle (LM), and lobster hepatopancreas (LHep) from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

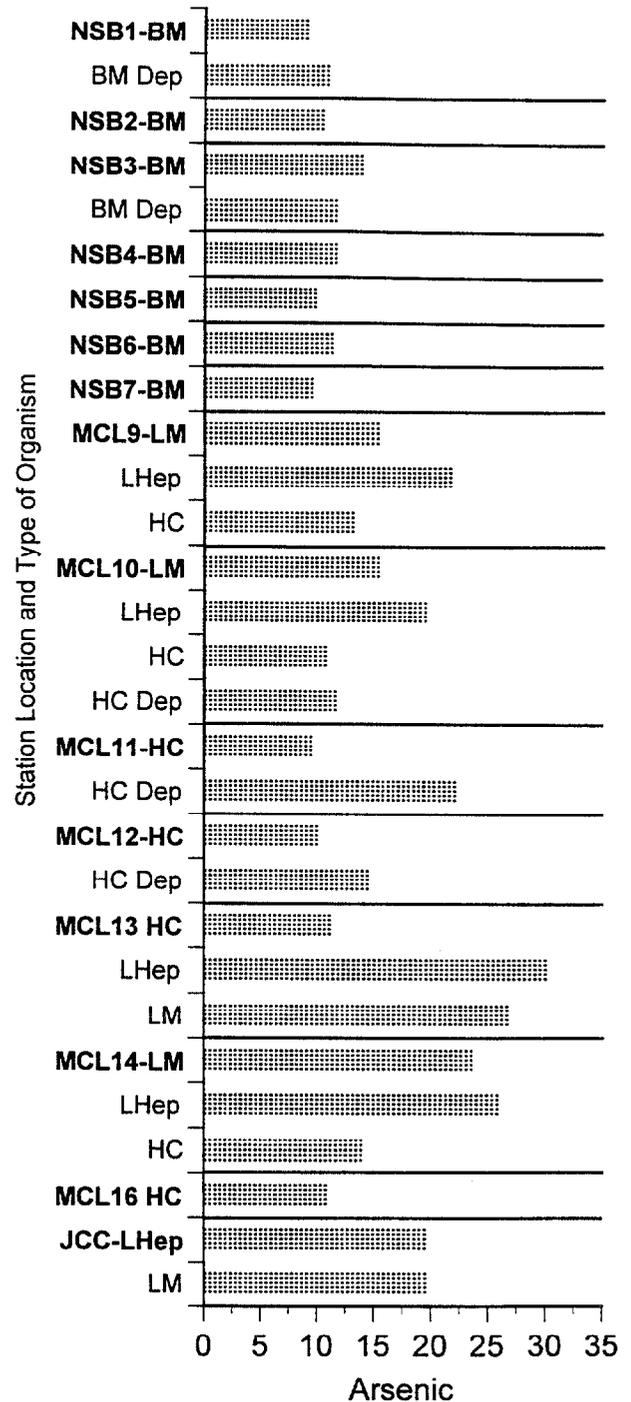
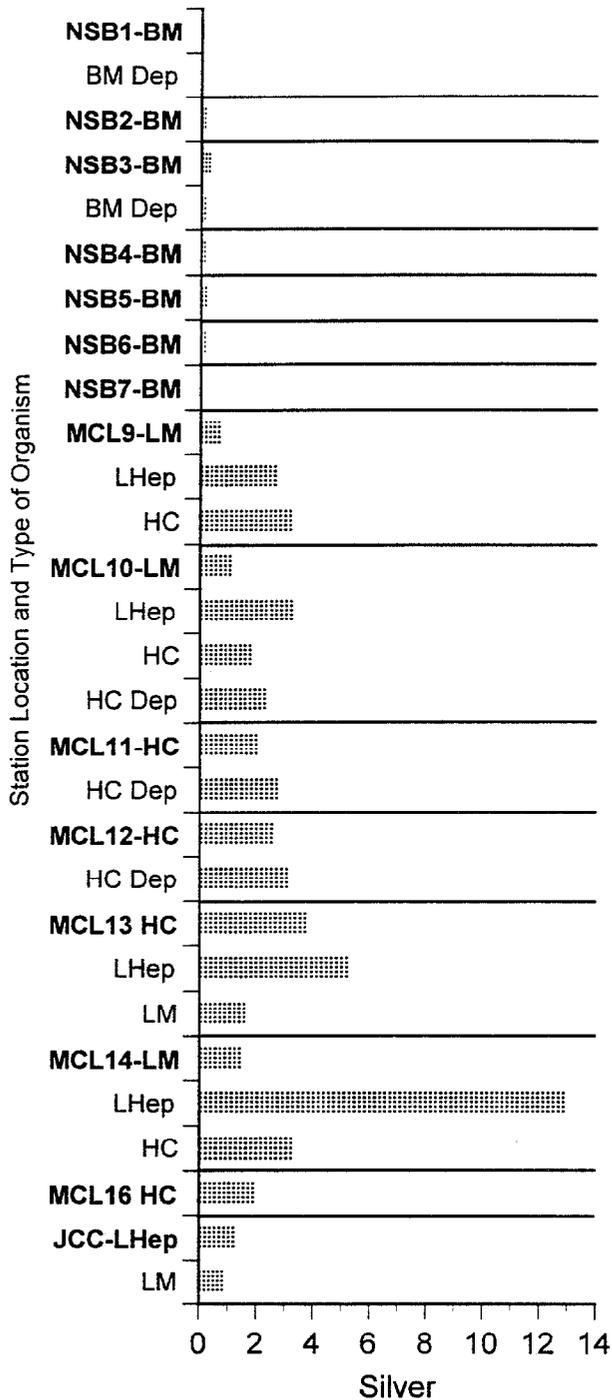


Figure 4.2-27. Tissue concentrations of Silver and Arsenic ( $\mu\text{g/g}$  dry weight) for blue mussels (BM), depurated blue mussels (BM Dep), hard clams (HC), depurated hard clams (HC Dep), lobster muscle (LM), and lobster hepatopancreas (LHep) from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

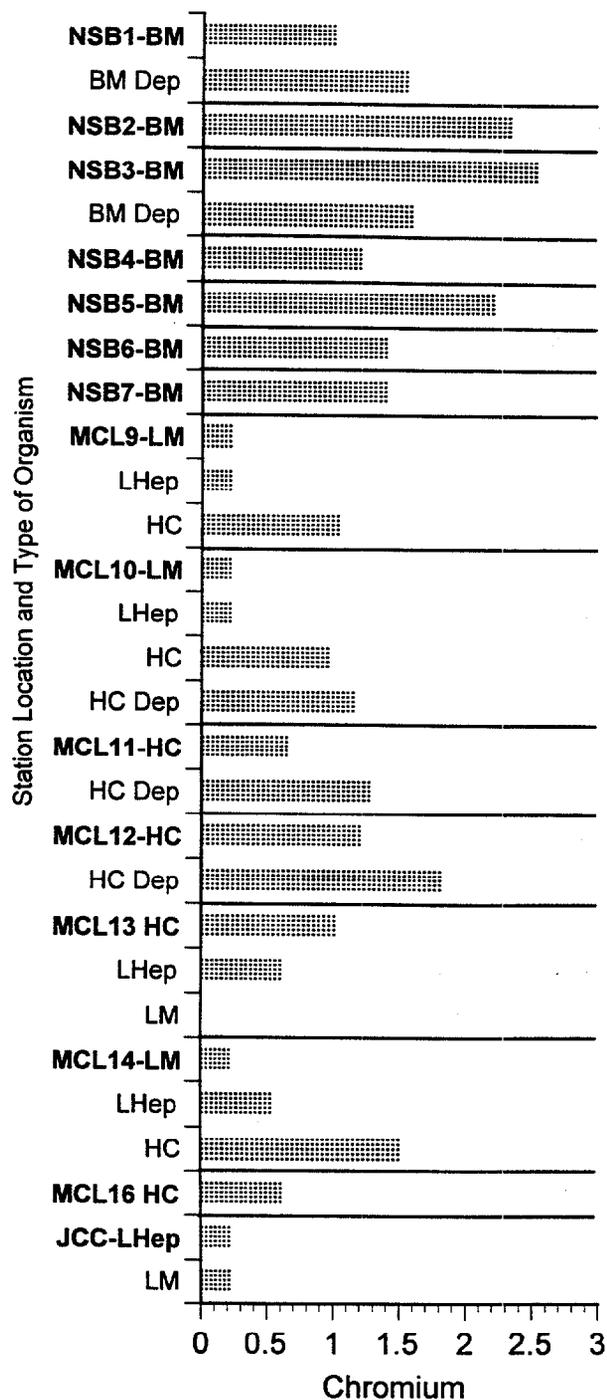
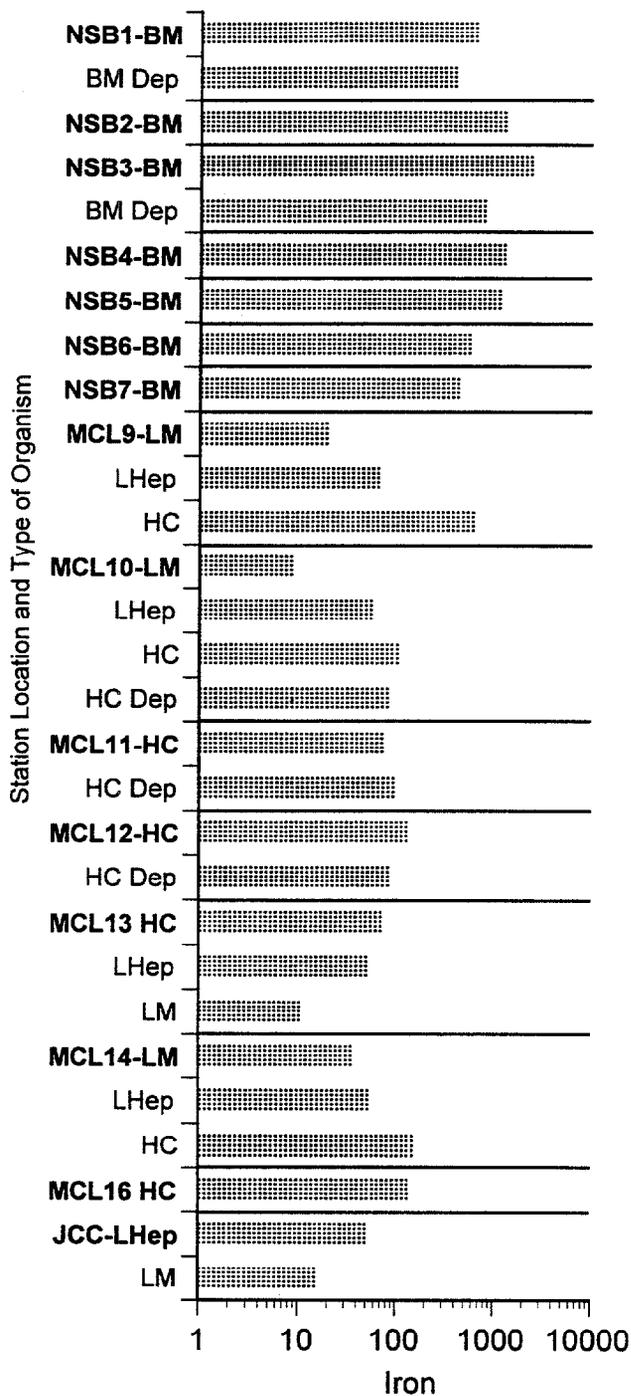


Figure 4.2-28. Tissue concentrations of Iron and Chromium ( $\mu\text{g/g}$  dry weight) for blue mussels (BM), depurated blue mussels (BM Dep), hard clams (HC), depurated hard clams (HC Dep), lobster muscle (LM), and lobster hepatopancreas (LHep) from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

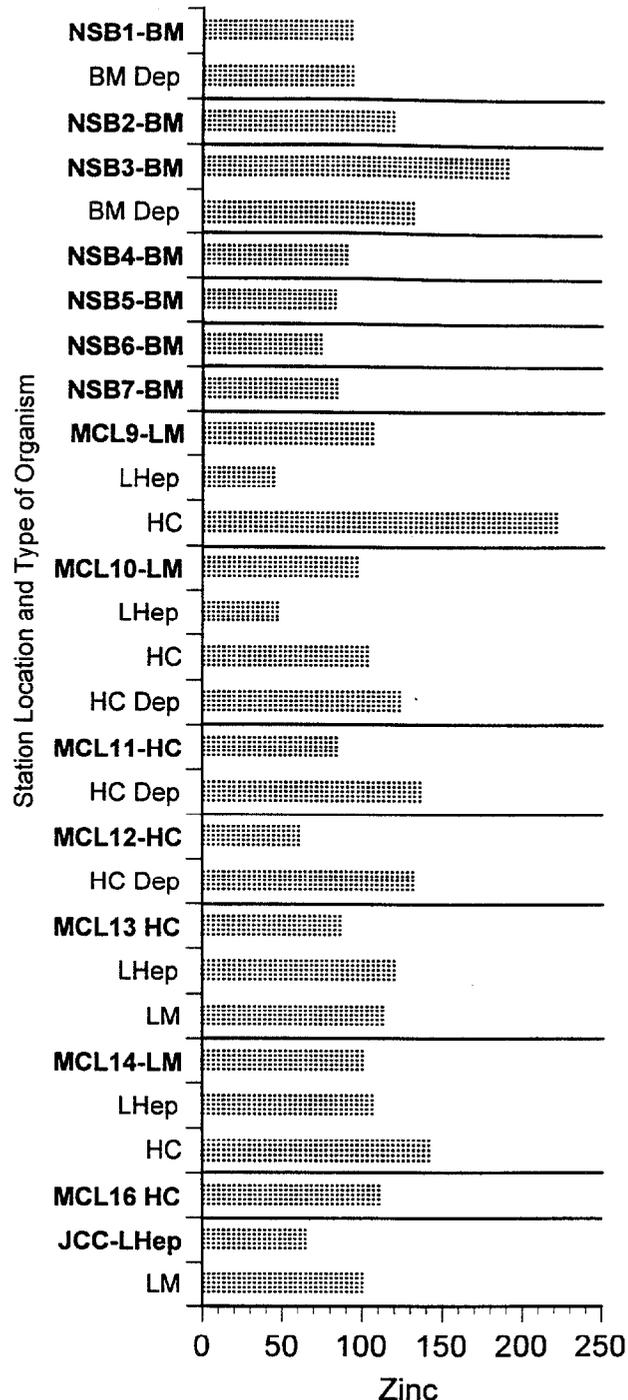
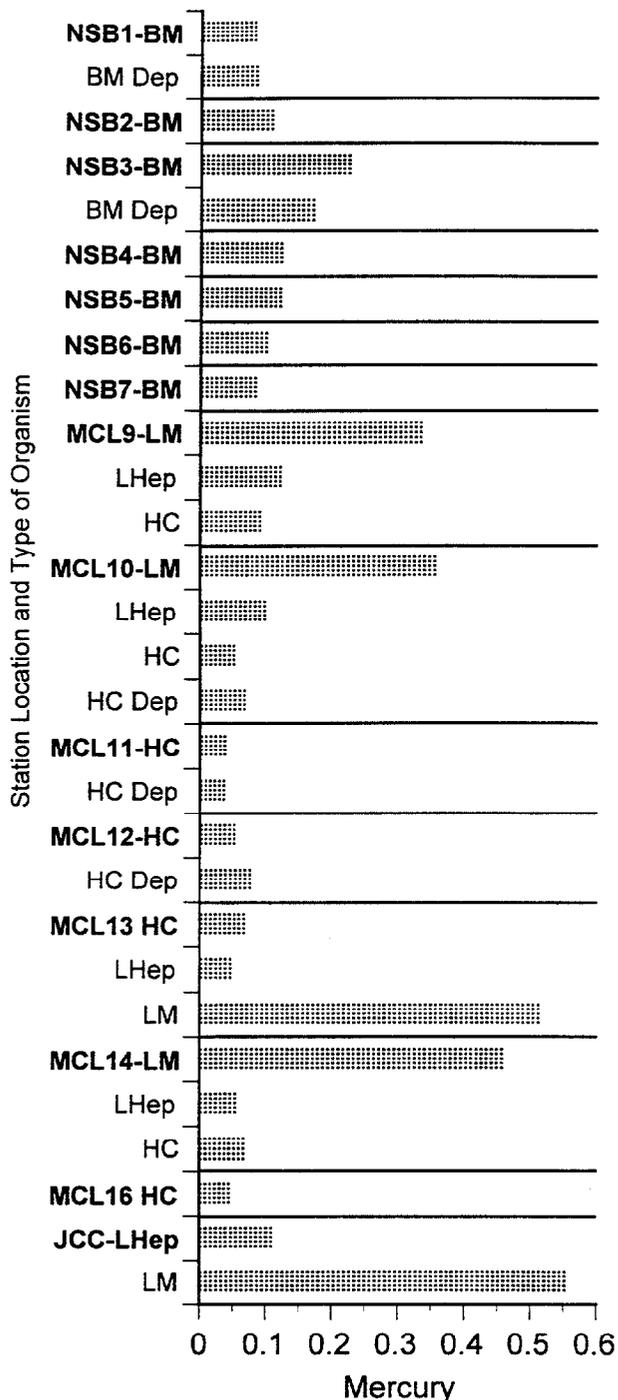


Figure 4.2-29. Tissue concentrations of Mercury and Zinc ( $\mu\text{g/g}$  dry weight) for blue mussels (BM), depurated blue mussels (BM Dep), hard clams (HC), depurated hard clams (HC Dep), lobster muscle (LM), and lobster hepatopancreas (LHep) from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

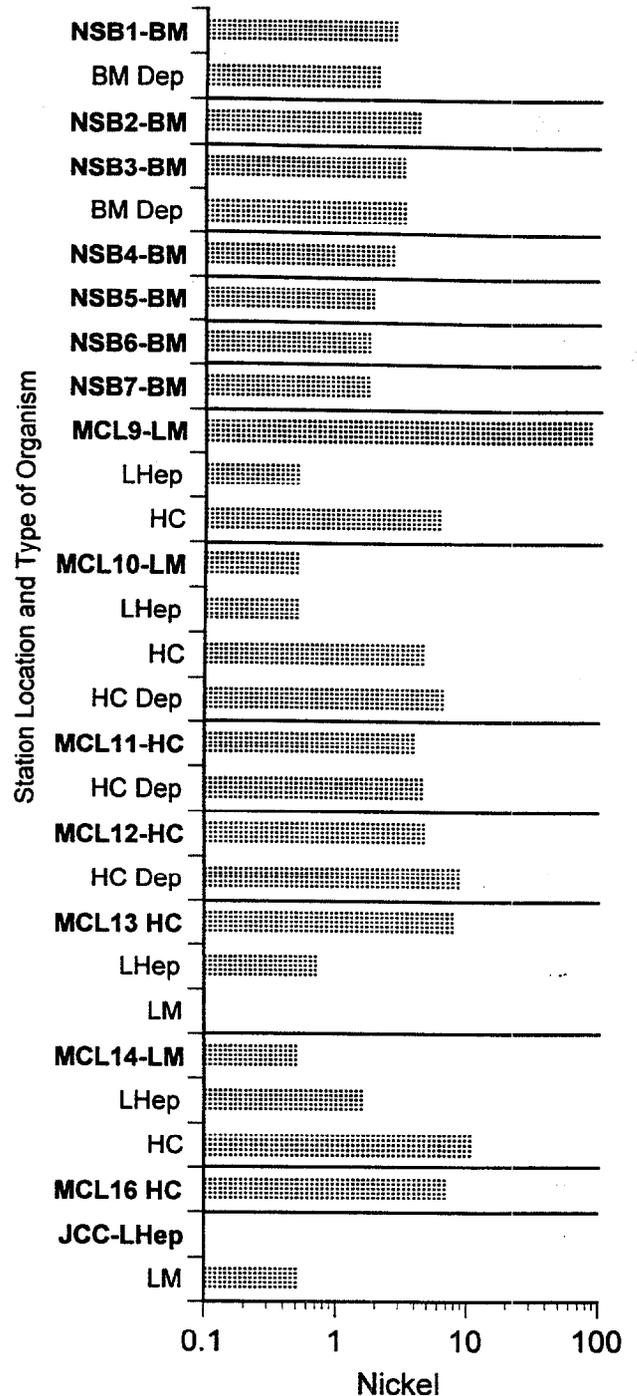
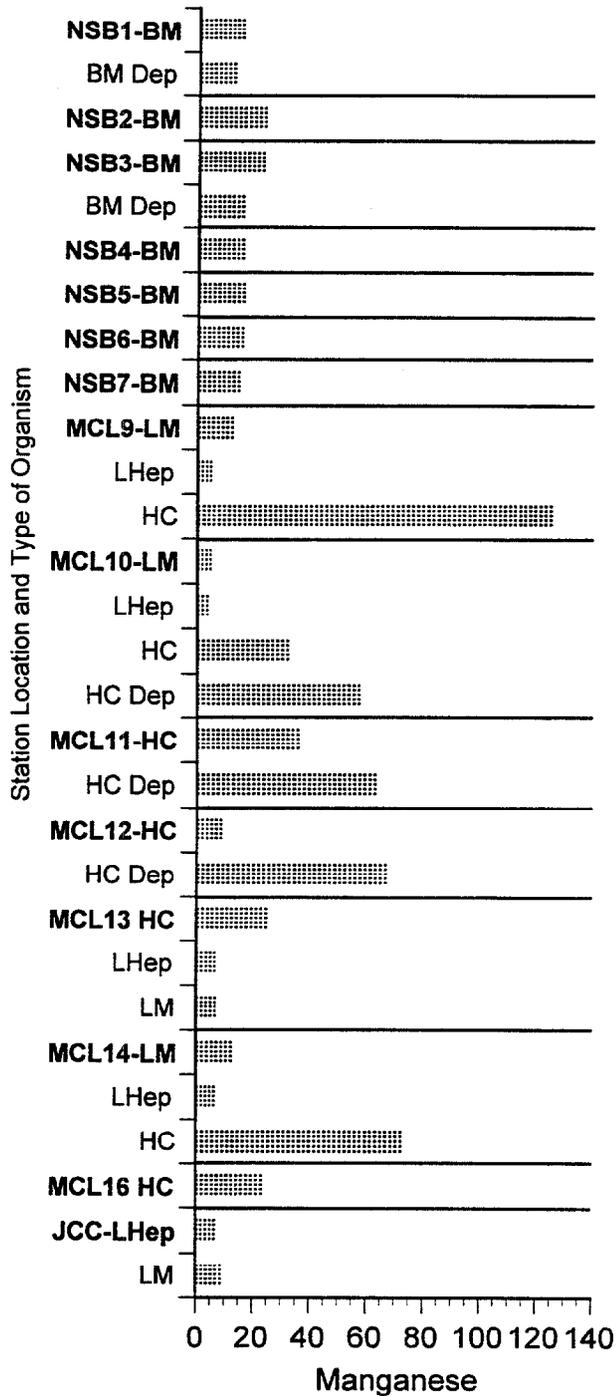


Figure 4.2-30. Tissue concentrations of Manganese and Nickel ( $\mu\text{g/g}$  dry weight) for blue mussels (BM), depurated blue mussels (BM Dep), hard clams (HC), depurated hard clams (HC Dep), lobster muscle (LM), and lobster hepatopancreas (LHep) from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC).

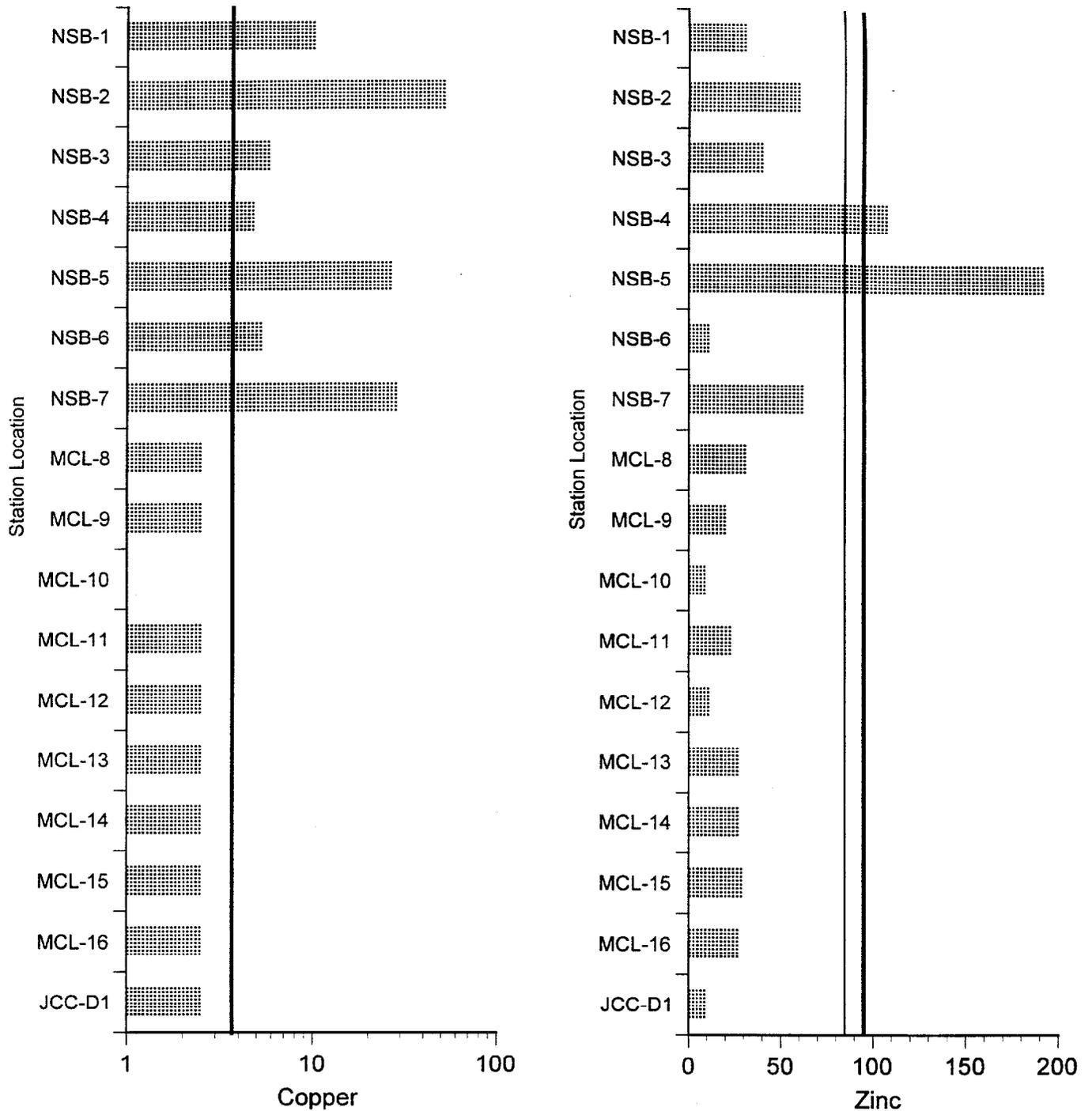


Figure 4.2-31. Concentrations of Copper and Zinc ( $\mu\text{g/L}$ ) in porewaters from surface sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical lines represent the EPA saltwater water quality screening criteria for chronic (light) and acute (bold) exposures.

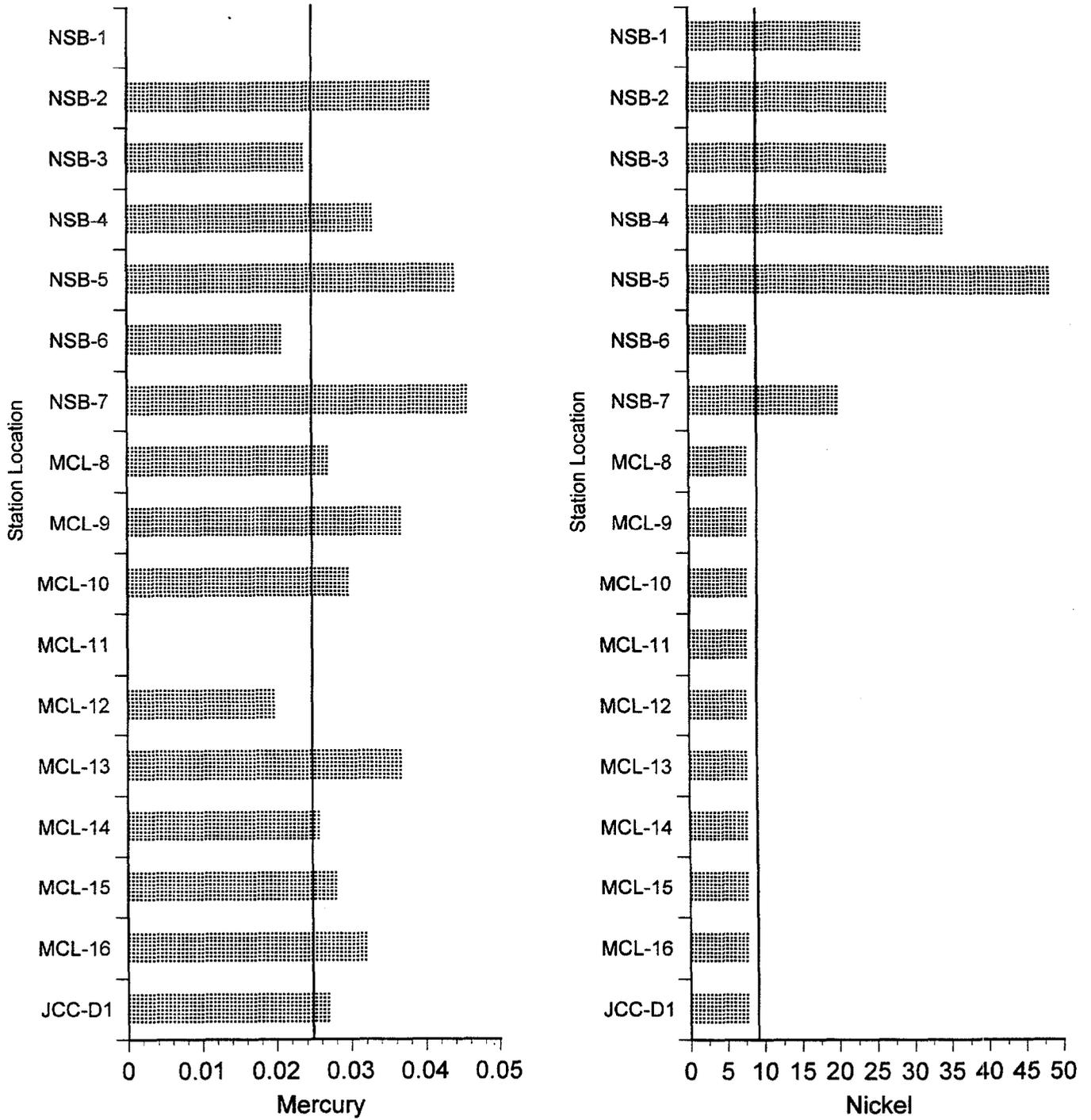


Figure 4.2-32. Concentrations of Mercury and Nickel ( $\mu\text{g/L}$ ) in porewaters from surface sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC). The vertical lines represent EPA saltwater water quality screening criteria for chronic exposure.

Table 4.2-1. Concentrations (ng/g sediment) of organic contaminants in sediment cores collected during Phase I/II investigations of the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location.

Station--> depths cm.	Sum of PCBs (ng/g)								
	S2B	JCC-D1	MCL-9	MCL-10	MCL-11	MCL-12	MCL-13	MCL-14	MCL-15
0-2 (ss)	96	14	--	--	--	--	--	--	--
0-6 (ss)	--	--	111	172	139	115	32	32	27
0-8	237	--	--	--	--	--	--	--	--
8-15	312	20	--	140	--	--	--	--	18
15-30	39	--	55	--	41	42	33	31	--
30-45	17	--	--	8	--	9	3	4	3
46-55	--	3	3	--	2	--	--	--	--

Station--> depths cm.	p,p'-DDE (ng/g)								
	S2B	JCC-D1	MCL-9	MCL-10	MCL-11	MCL-12	MCL-13	MCL-14	MCL-15
0-2 (ss)	1.9	0.5	--	--	--	--	--	--	--
0-6 (ss)	--	--	1.1	1.6	1.3	1.5	1.0	0.6	0.5
0-8	5.8	--	--	--	--	--	--	--	--
8-15	4.3	0.3	--	1.5	--	--	--	--	0.5
15-30	0.7	--	1.0	--	0.8	0.9	1.0	0.8	--
30-45	0.4	--	--	0.1	--	0.1	0.1	0.1	<0.05
46-65	--	<0.05	0.1	--	<0.05	--	--	--	--

Station--> depths cm.	Tributyltin (ng/g)								
	S2B	JCC-D1	MCL-9	MCL-10	MCL-11	MCL-12	MCL-13	MCL-14	MCL-15
0-2 (ss)	<1.0	<1.0	--	--	--	--	--	--	--
0-6 (ss)	--	--	5.67	2.47	3.13	3.63	2.39	2.14	2.54
0-8	7.76	--	--	--	--	--	--	--	--
8-15	8.56	6.82	--	3.22	--	--	--	--	lost
15-30	NA	--	1.34	--	1.55	1.39	7.93	1.86	--
30-45	NA	--	--	NA	--	NA	NA	NA	NA
46-65	--	NA	NA	--	NA	--	--	--	--

Station--> depths cm.	Sum of PAHs (ng/g)								
	S2B	JCC-D1	MCL-9	MCL-10	MCL-11	MCL-12	MCL-13	MCL-14	MCL-15
0-2 (ss)	18,800	7300	--	--	--	--	--	--	--
0-6 (ss)	--	--	2160	4550	1550	3600	1930	1370	690
0-8	25,600	--	--	--	--	--	--	--	--
8-15	18,500	44,600	--	2480	--	--	--	--	561
15-30	3290	--	4540	--	2790	1130	1110	1260	--
30-45	531	--	--	153	--	272	141	66	33
46-65	--	35	49	--	30	--	--	--	--

NA = Not Analyzed; ss = surface sediment.

Table 4.2-2. Phase III concentrations (ng/g dry weight) of organic contaminants in surface and core sediments collected in resampling of the McAllister Point Landfill study area.<sup>1</sup>

Station	Chemical Concentrations			
	Surface Sediments <sup>3</sup>		Core Sediments <sup>3,4</sup>	
	Total PCBs	Total PAHs	Total PCBs	Total PAHs
NSB-1	117	113		
NSB-2	134	264	1040	4090
NSB-2-FD <sup>2</sup>	129	587		
NSB-3	250	480	196	3130
NSB-4	5599	3490	3270	24000
NSB-5	1204	2500	566	3710
NSB-6	249	18100	398	10500
NSB-7	339	1910		
S2B	62.7	1260		
S2C <sup>3</sup>	184	19900		
MCL-8	88.9	2210		
MCL-9	90.4	2400		
MCL-10	112	2180	177	2160
MCL-11	57.3	1020		
MCL-12	192	12000	390	11800
M1	40.9	2130		
MCL-13	34.8	1270		
MCL-14	46.8	1350		

Total PCBs **ER-L**=22.7 ng/g; **ER-M**=180 ng/g; Total PAHs **ER-L**=4022 ng/g; **ER-M**=44,792 ng/g.  
 Bordered cells indicate >30% Relative Percent Difference (RPD) increase in 1996 versus 1995.  
 Lightly shaded cells indicate Phase III concentration >ER-L Guidelines.  
 Darkly shaded cells indicate Phase III concentration >ER-M Guidelines.  
 See Appendix A-1-1 and Table 4.2-1 for prior phase data.

- 1 - Prior studies: TRC, 1994; URI/SAIC, 1995; current study: SAIC/URI, 1996.
- 2 - NSB-2-FD compared to prior study values for NSB-2.
- 3 - S2C surface and NSB core sediment not sampled in prior phases, thus RPD not calculated.
- 4 - Phase III cores depth 0-18 cm; Phase II core sediments sampled only at MCL-10 and MCL-12; Phase II MCL-10 core concentrations average of 0-6 cm and 8-15 cm depths (Table 4.2-1); Phase II MCL-12 core concentrations average of 0-6 cm and 15-30 cm depths (Table 4.2-1).

Table 4.2-3. Phase III concentrations ( $\mu\text{g/g}$  dry weight) of selected metals contaminants in surface and core sediments collected in resampling of the McAllister Point Landfill study area.<sup>1</sup>

Station	Chemical Concentrations					
	Surface Sediments <sup>3</sup>			Core Sediments <sup>3,4</sup>		
	Copper	Lead	Zinc	Copper	Lead	Zinc
NSB-1	29.5	17.8	160			
NSB-2	7629	5405	2135	615	365	4660
NSB-2-FD <sup>2</sup>	821	1269	1195			
NSB-3	1006	718	2878	211	1051	860
NSB-4	8466	1478	6913	1384	1181	24468
NSB-5	591	526	2132	1227	739	1289
NSB-6	165	135	252	68.5	126	265
NSB-7	177	215	1576			
S2B	25.1	33.1	2.30			
S2C <sup>3</sup>	51.5	70.2	103			
MCL-8	26.2	44.3	83.7			
MCL-9	24.5	44.1	65.1			
MCL-10	250	61.0	650	102	46.6	197
MCL-11	12.9	28.0	2.30			
MCL-12	49.4	58.6	287	40.5	75.3	289
M1	14.5	25.7	2.30			
MCL-13	13.2	25.1	2.30			
MCL-14	4.50	28.3	862			

Copper ER-L=34.0  $\mu\text{g/g}$ ; ER-M=270  $\mu\text{g/g}$ ; Lead ER-L=46.7  $\mu\text{g/g}$ ; ER-M=218  $\mu\text{g/g}$ ; Zinc ER-L=150  $\mu\text{g/g}$ ; ER-M=410  $\mu\text{g/g}$ . Bordered cells indicate >30% Relative Percent Difference (RPD) increase in 1996 versus 1995.

Lightly shaded cells indicate Phase III concentration >ER-L Guidelines.

Darkly shaded cells indicate Phase III concentration >ER-M Guidelines.

See Appendix A-1-2 for prior phase data.

1 - Prior studies: TRC, 1994; URI/SAIC, 1995; current study: SAIC/URI, 1996.

2 - NSB-2-FD compared to prior study values for NSB-2.

3 - S2C surface and NSB core sediment not sampled in prior phases, thus RPD not calculated.

4 - Phase III cores depth 0-18 cm; Phase II core sediments sampled only at MCL-10 and MCL-12; Phase II MCL-10 core concentrations average of 11-16 cm and 32-40 cm depths (Appendix A-1-2); Phase II MCL-12 core concentrations average of 17-22 cm and 38-45 cm depths (Appendix A-1-2).

Table 4.2-4. Fecal pollution indicator concentrations in sediments collected from the McAllister Point Landfill study area.<sup>1</sup>

Station	Total Coliforms		Fecal Coliforms		Fecal Streptococci		<i>Clostridium perfringens</i>		Overall Ranking <sup>3</sup>
	CFU <sup>2</sup> /100 g		CFU <sup>2</sup> /100 g		CFU <sup>2</sup> /100 g		CFU <sup>2</sup> /100 g		
NSB-1	2200	+++	45	-	480	++	1700	+++	+++
NSB-3	9200	+++	45	-	340	+	3500	+++	+++
NSB-5	45	-	<18	-	20	-	45	-	-
NSB-7	20	-	<18	-	<18	-	490	++	+
MCL-11	150	+	37	-	45	-	9200	+++	++
MCL-12	130	+	78	-	170	+	9200	+++	++
MCL-13	20	-	20	-	20	-	>16,000	+++	++
MCL-15	68	-	20	-	45	-	16000	+++	++

1 - Indicator-specific rankings: "-" = <100 CFU/100 g; "+" = 100-350 CFU/100 g (low); "++" = >350 CFU/100 g (intermediate); "+++" = >1000 CFU/100 g (high).

2 - CFU = Colony forming units

3 - Overall Ranking: "+++" = intermediate (++) or higher exposure observed for two or more indicators, one of which indicates high (+++) exposure; "++" = intermediate (++) exposure observed for two or more indicators or high (+++) exposure for one indicator; "+" = low (+) exposure observed for two or more indicators or intermediate (++) exposure for one indicator; "-" = low (+) exposure observed for only one indicator or no exposure for all indicators. See text in Section 6.0-2.

## 5.0. ECOLOGICAL EFFECTS ASSESSMENT

Ecological effects are quantified from the relationships between exposure patterns and resulting responses of ecological systems, as determined from measurement endpoints identified during Problem Formulation (Section 3). Ecological effects assessments include literature-reported evaluations of the known effects of CoCs to receptors of concern (Section 5.1); direct measurement of the toxicity of exposure media (Section 5.2) to appropriately sensitive marine species (the amphipod *Ampelisca* and the sea urchin *Arbacia*, respectively); site-specific investigations of the abundance and condition of receptors of concern (Section 5.3); and collation of toxicity-based criteria and standards for exposure media identified in exposure pathways (Section 5.4). Uncertainty associated with these assessments is discussed in Section 5.5.

### 5.1. KNOWN EFFECTS OF CoCs

Contaminants of concern as identified in Section 3 consist primarily of PAHs, the chlorinated pesticide p,p'-DDE, PCBs, the metals Ag, As, Cd, Cr, Cu, Hg, Ni, Pb and Zn, and tributyltin (TBT).

Potential effects of the CoCs on biological receptors are influenced strongly by their chemical behavior, solubility, and toxicity. For example, Ni, Cu, Cd, and Cr<sup>+6</sup> have relatively high solubility and thus higher dissolved phase concentrations than many organic contaminants, such as PAHs and relatively insoluble metals, (e.g., Ag, Pb, Zn, and Cr<sup>+3</sup>). Subsequently, dissolved contaminants may be transported throughout the water column by current and tidal flows, while contaminants associated with particles tend to be transported horizontally, commonly settling to the bottom in sediment

depositional areas. Once on the bottom, the sediment particles can be transported as bedload or resuspended, resulting in redistribution of the contaminants. Dissolved or particle/sediment-bound contaminants may be available to biological receptors in the water column, pore waters and sediments, potentially resulting in biological uptake and/or direct toxicological effects. Impacts to organisms can then be strongly influenced, for example, by the affinity of various contaminants for tissue lipids and the type of cellular or subcellular effects associated with particular compounds and elements.

The following describes the chemical behavior and known effects of key contaminants of concern.

*Arsenic.* Arsenic in surface water can undergo complex patterns of transformation, including oxidation-reduction reactions, biotransformation, precipitation, and adsorption, resulting in extremely mobile behavior in aquatic systems. Sorption of arsenic onto clays, iron oxides, manganese compounds, and organic material is a typical fate. Sediment can serve as a reservoir for arsenic, and sediment-bound arsenic (arsenate/arsenite) that has been methylated by aerobic and anaerobic bacteria may be released back into the water column (ATSDR, 1987a). Bioconcentration of arsenic occurs in aquatic organisms, primarily in algae and lower invertebrates. Biomagnification in aquatic food webs does not appear to be significant, although some fish and invertebrates contain high levels of arsenic compounds that are relatively inert toxicologically (ATSDR, 1987a). Arsenic in seafood occurs primarily as complex methylated or organic chemical species which are less toxic and more readily excreted than inorganic arsenic. The Effects Range-Low (ER-L) and Effects Range-Medium (ER-M) benchmarks, defined by Long *et al.* (1995) as the lower 10<sup>th</sup> and 50<sup>th</sup> percentiles of all concentrations of a contaminant observed to cause a biological effect, over a range of studies and species, are 8.2 and 70 mg/kg, respectively (also see Section 5.4). Acute responses to inorganic arsenic in water-only exposures were observed in marine

organisms at 2,319 ppm (Long and Morgan, 1990). During tests of sediments from Commencement Bay, where arsenic concentrations ranged between 2,257 to 28.3 mg/kg, mortality ranged between 15.7 and 2.5% to the amphipod *Rhepoxynius abronius*, respectively (Long and Morgan, 1990). Arsenic concentrations as high as 1,005 mg/kg were detected in Puget Sound sediments where highly toxic (i.e., 95% mortality) responses were indicated, and concentrations as low as 22.6 mg/kg where survival was >87% (Long and Morgan, 1990). Severe mortality (i.e., 100%) to the polychaete *Nereis virens* was observed during exposures to Black Rock Harbor sediment where the arsenic concentration was 1.88 mg/kg (Long and Morgan, 1990).

There is good evidence that arsenic is carcinogenic in humans, although evidence of arsenic-induced carcinogenicity in animals is mostly negative. In addition, very high oral doses of sodium arsenite may be teratogenic and ferotoxic. Arsenic is a weak inducer of chromosomal aberrations, and is a known teratogen in vertebrates (Eisler, 1988). Arsenic exposure may produce behavioral impairment, and leads to death at high concentrations. In aquatic invertebrates, arsenic exposure may lead to decreased growth, reproductive impairment, and death. Pre-exposure to sublethal levels of arsenic may result in increased tolerance to this element upon re-exposure (Eisler, 1988). It is generally agreed that inorganic arsenic is more toxic than organic arsenic, and that trivalent forms are more toxic than are pentavalent forms.

*Cadmium.* Cadmium in the water column may partition to dissolved and particulate organic carbon. Cadmium speciation yields primarily the divalent form of the metal,  $Cd^{+2}$ , between pH 4.0 and pH 7.0 (ASTDR, 1987b, Stephenson *et al.*, 1989). Studies indicate that the divalent cadmium ion is responsible for observed biological effects. Acid volatile sulfides can influence the toxicity and bioaccumulation of cadmium in sediments. Cadmium is not a highly mobile element in the aquatic food web, nor does it biomagnify (Kay, 1985). Studies with zebrafish indicate no maternal transfer of cadmium to young, and cadmium measured in bird eggs was not a reliable indicator of

environmental exposure (Kay, 1985). Tissue residue toxicity relationships for cadmium may be variable because detoxification processes allow organisms to sequester this metal in various unavailable forms while analytical measurements continue to detect its presence (Klerks and Bartholomew, 1991). Whole body residues may fail to predict effects concentrations at the organ level because concentrations in target organs may be larger than whole body residues (McKinney, 1993). In freshwater studies, cadmium has been associated with high mortality, reduced growth, and inhibited reproduction (Eisler, 1985). Generally, resistance to cadmium was higher in marine organisms when compared to freshwater species (Eisler, 1985). Marine organism LC<sub>50</sub>s ranged from 320 to 430 µg/L, whereas effects in freshwater organisms have been observed at 1-2 µg/L (Eisler, 1985).

In tests of Puget Sound sediment, statistically significant effects were noted in the amphipod, oyster larvae, and Microtox™ bioassays, at cadmium concentrations ranging between 6.7 and 9.6 mg/kg (Long and Morgan, 1990). Cadmium concentrations of 1.2 and 1.7 mg/kg were measured in tests of San Francisco Bay sediments and caused significant toxicity in the amphipod and bivalve larvae bioassays, respectively (Long and Morgan, 1990). Highly toxic effects (i.e. 75% mortality) were noted in amphipod tests of Commencement Bay sediments with 41.6 mg/kg cadmium (Long and Morgan, 1990). Low abundances of echinoderms and arthropods were observed in Southern California where cadmium concentrations were 6.2 and 4.3 mg/kg, respectively (Long and Morgan, 1990). Complete mortality was observed in tests using the polychaete *Nereis verens* exposed to Black Rock Harbor sediments at 1.6 mg/kg cadmium (Long and Morgan, 1990). Baltimore Harbor sediments were toxic to mummichogs and spot, where the cadmium concentration in these sediments was 22.8 mg/kg (Long and Morgan, 1990). The ER-L and ER-M benchmarks for cadmium over a range of studies and species, are 1.2 and 9.6 mg/kg, respectively (Long *et al.*, 1995).

*Chromium.* Chromium(+6) occurs only rarely in nature, except from anthropogenic contamination, because it is readily reduced to chromium(+3) in the presence of oxidizable organic matter. However, chromate and dichromate (chromium(+6)) compounds are stable in many natural waters because of the low concentration of reducing material, and thus may undergo intermedia transport. In contrast, chromium(+3) compounds, the form most commonly observed in biological systems, are generally insoluble in water. The effects of chromium on wildlife, fish, and invertebrates have been summarized by Eisler (1986a):

"...chromium is mutagenic, carcinogenic, and teratogenic to a wide variety of organisms, and Cr<sup>+6</sup> [hexavalent Cr] has the greatest biological activity. However, information is lacking on the biological activities of water soluble Cr<sup>+3</sup> [trivalent Cr] compounds, organochromium compounds, and their ionic states. Aquatic plants and marine polychaete worms appear to be the most sensitive groups tested. In exposures to Cr<sup>+6</sup>, growth of algae was inhibited at 10.0 ppb, and reproduction of marine polychaete worms was inhibited at 12.5 ppb. At higher concentrations, Cr<sup>+6</sup> is associated with abnormal enzyme activities, altered blood chemistry, lowered resistance to pathogenic organisms, behavioral modifications, disrupted feeding, histopathology, osmoregulatory upset, alterations in population structure and species diversity indices, and inhibition of photosynthesis. Not all sublethal effects observed were permanent, but the potential for acclimatization of organisms to Cr is not well documented."

Cr exposure at high concentrations can produce death. Sensitivity to Cr varies widely among species, even among those which are closely related (Eisler, 1986a). Chromium(+6) is classified as a human carcinogen, but chromium(+3) still is being evaluated for its carcinogenic potential. Most of the chromium in aquatic environments eventually is expected to precipitate in sediments. The ER-L and ER-M values for

chromium in sediments are 81 and 370 mg/kg, respectively (Long *et al.*, 1995). Acute toxicity to marine organisms in water-only exposures is evident at concentrations ranging from 2000 to 105,000 ppm (Long and Morgan, 1990). Tests with the amphipod *Rhepoxynius abronius* indicate toxic effects in sediments from Commencement Bay where chromium levels ranged between 16.2 to 19.7 mg/kg (Long and Morgan, 1990). Survival greater than 97% was observed in the polychaete *Neanthes arenaceodentata* exposed to San Diego Bay sediment with 299.5 mg/kg chromium (Long and Morgan, 1990).

*Copper.* The two processes that primarily influence the fate of copper in the aquatic environment are sorption and chemical speciation. Speciation is determined by the oxidation-reduction potential of the copper compound and the media pH. In contaminated settings, copper may form complexes with organic material in the water; however, copper ultimately settles out of the water column and is deposited in sediments. Various processes including sorption onto clay minerals, hydrous iron, manganese oxides, and organic material reduce the level of copper compounds in aquatic media. In organically rich sediments, the sorbed and precipitated copper may become redissolved through complexation and can persist in the water for long periods.

Copper is an essential element for most organisms, although the distinction between deficiency and toxicity in some organisms, including algae and some invertebrates, is small if there is limited ability to control absorption. Fish are sensitive to copper, and it is thought that their gills do not provide an effective barrier to absorption (Hammond and Beliles, 1980). Copper is toxic to aquatic plants and animals at relatively low levels. In addition to affecting survival, Cu exposure has been associated with development of histopathological lesions in mollusks and fish (Martin, 1977; Gardner and LaRoche, 1973), inhibition of egg hatching in fish (Gardner and LaRoche, 1973), impairment of fertilization and larval development in polychaetes and echinoderms (Reisch, 1964; Young and Nelson, 1974; Bougis, 1965), and

retardation of growth in hydroids (Karbe, 1972). Cu is particularly active in disruption of enzymatic systems (Albergoni and Piccinni, 1983). Copper is not strongly bioaccumulated and does not appear to transfer significantly through aquatic (or terrestrial) food webs. Bioconcentration factors are in the range of 10 to 100, although in some mollusks it can reach 30,000 (U.S. EPA, 1984). This may be because copper proteins in the blood of many bivalves act as oxygen carriers. For example, American oysters have been documented to have tissue concentrations of 1,500 mg/kg (Hammond and Beliles, 1980). Acute toxic effects on lower marine biota have been demonstrated at water concentrations ranging from 5.8 to 600 µg/L (U.S. EPA, 1986; Anderson *et al.*, 1991). The ER-L and ER-M values for copper in sediments are 34 and 270 mg/kg, respectively (Long *et al.*, 1995). In water-only exposures, acute responses of marine organisms were observed at concentrations ranging from 5.8 to 600 ppm (Long and Morgan, 1990). Mortality responses in the amphipod *Rhepoxynius abronius* ranged between 79% to 13% in sediments from Commencement Bay where corresponding copper concentrations ranged between 2820 to 85.1 mg/kg (Long and Morgan, 1990). In oyster bioassays, a highly toxic developmental response (i.e., >44% abnormal larvae) was observed in tests of sediments with 918 mg/kg copper from Commencement Bay (Long and Morgan, 1990). Eighteen to 67% mortality to *Rhepoxynius abronius* was observed in tests of sediments from San Francisco Bay, where copper concentrations were between 72 and 85 mg/kg, respectively (Long and Morgan, 1990). No survival was observed in exposures of the polychaete *Nereis virens* to Black Rock Harbor sediment with 612 mg/kg copper (Long and Morgan, 1990).

*Lead.* The chemistry of lead in aqueous solutions is highly complex because of its occurrence in many forms, although it has a tendency to form compounds of low solubility. The divalent form (Pb<sup>+2</sup>) is the stable ionic species of lead. Hydroxide, carbonate, sulfide and, more rarely, sulfate may act as solubility controls. Lead may occur either as adsorbed ions or surface coatings on sediment mineral particles, or it

may be carried as a part of suspended living or non-living organic matter in the water (ATSDR, 1988a). The ER-L and ER-M values for lead in sediments are 46.7 and 218 mg/kg, respectively (Long *et al.*, 1995). In freshwater tests, adverse effects to test organisms occur between 1.3 and 7.7 ppm (Long and Morgan, 1990). Studies indicate that marine organisms in water-only exposures are more sensitive (Long and Morgan, 1990). The proposed marine water quality standard for California is 8 ppm (Long and Morgan, 1990). Statistically significant responses to amphipods, oyster larvae, and Microtox™ were observed in Puget Sound sediment tests at concentrations ranging between 530 and 660 mg/kg (Long and Morgan, 1990).

Lead can bioaccumulate in some bivalves, such as mussels, but does not appear to bioaccumulate in fish. In vertebrates, Pb is known to modify the structure and function of the kidney, bone, central nervous system, and the hematopoietic system, and produces adverse biochemical, histopathological, neuropsychological, ferotoxic, teratogenic, and reproductive effects. Inhibition of blood delta aminolevulinic acid dehydratase (ALAD), an enzyme critical in heme formation, has been observed as a result of exposure to Pb in invertebrates, birds, and a variety of marine fish. At sufficiently high concentrations, Pb effects manifest in estuarine organisms as reduced growth, fecundity, and survivorship. Lead is classified as a probable human carcinogen, based on animal (primarily rat) studies (Eisler, 1988a).

*Mercury.* Mercury forms a wide variety of complexes with organic ligands, the compounds of which (e.g., methylmercury) are toxicologically and environmentally significant (Nriagu, 1979). Mercury is very persistent when released into the environment, with the major removal mechanism occurring by adsorption onto particles and subsequent settlement to sediments. Mercury can become methylated to a highly toxic form, methylmercury, by biological and chemical processes (Nriagu, 1979). Methylation occurs most readily under anaerobic conditions. Mercury has no known essential status or function in organisms, and is a mutagen and teratogen (U.S. EPA,

1985a). Bioaccumulation and toxic effects of mercury in aquatic systems are highly complex and are influenced by water temperature, salinity, hardness, pH, age of an organism, prior exposure, reproductive state (related to lipid content), trophic level, and metabolism. Mercury is considered to be one of the most toxic of the heavy metals (Nriagu, 1979). At higher concentrations, mercury is toxic to a wide range of marine invertebrates and fish, and its acute toxicity varies among species. For instance, Hg is acutely toxic to the mysid shrimp *Mysidopsis bahia* at concentrations as low as 3.5 µg/L, whereas the acute value for winter flounder is 1,678 µg/L (U.S. EPA, 1985a).

In addition to mortality, Hg exposure can result in impairment of reproduction, development, and growth in estuarine plants and animals. For example, productivity and time to first reproduction in *Mysidopsis* was affected in 28-d life cycle tests at mercury concentrations of 1.6 µg/L. Methylmercury can be bioconcentrated and biomagnified through aquatic food webs, with higher concentrations generally observed at the higher trophic levels (e.g., carnivorous fish and piscivorous birds; Nriagu, 1979). Concentrations of mercury in ocean sediments have been shown to be reflected in the tissues of epifauna (Klein and Goldberg, 1970). Bioconcentration factors range upwards to almost 200,000 for marine zooplankton (Hirota *et al.*, 1983), and transfer rates in piscivorous fish and birds have been documented up to 36,000 (Eisler, 1981). Adverse effects on reproduction in birds have been demonstrated at concentrations as low as 5 mg/kg. The ER-L and ER-M values for mercury in sediments are 0.15 and 0.71 mg/kg, respectively (Long *et al.*, 1995). Acute toxicity in water-only exposures of mercury to marine organisms is observed between 3.5 and 1,678 ppm (Long and Morgan, 1990). Statistically significant responses in Puget Sound sediment tests were observed at concentrations ranging between 0.4 and 2.1 mg/kg to amphipods, oyster larvae, and Microtox™ (Long and Morgan, 1990). Highly toxic (i.e., 67 and >78% mortality) effects to the amphipod *Rhepoxynius abronius* were observed in tests of Commencement Bay and San Francisco Bay sediments with 11.2 mg/kg and 1.0 mg/kg mercury, respectively (Long and Morgan, 1990).

*Nickel.* Very little information on the fate of nickel in the environment could be found in the literature. It is generally characterized as moderately soluble in water. U.S. EPA (1986) provides the following insights into the adverse effects of nickel:

"Mechanisms of nickel toxicity are varied and complex, and, as with other heavy metals, significant effects occur at cell membranes and membranous tissues, such as gills. In fish, hematological effects such as hyperglycemia, lymphopenia, and erythrocytosis have been reported in association with nickel intoxication..."

Nickel exposure has resulted in reduced photosynthesis in aquatic plants (plankton and macrophytes), inhibition of enzyme systems in a variety of organisms, stunted growth and development, reproductive impairment, and at sufficiently high levels, death. Exposure levels associated with these effects are summarized in U.S. EPA (1986). Nickel is classified as a human carcinogen (U.S. EPA, 1992a). The ER-L and ER-M values for nickel in sediments are 20.9 and 51.6 mg/kg, respectively (Long *et al.*, 1995). Acute toxicity to marine organisms has been observed in water-only exposures to nickel at 151.7 ppm (Long and Morgan, 1990). In tests with sediments from Puget Sound, statistically significant toxicity to amphipods, oyster larvae, and Microtox™ was observed at concentrations ranging between 28 and >120 mg/kg (Long and Morgan, 1990). Exposures of the amphipod *Rhepoxynius abronius* indicated highly toxic responses (i.e., 67 and >78% mortality) to Commencement Bay and San Francisco Bay sediments with 41 mg/kg and 113 mg/kg nickel, respectively (Long and Morgan, 1990). No survival was observed in the polychaete *Nereis virens* when exposed to 52.0 mg/kg nickel in Black Rock Harbor sediment (Long and Morgan, 1990).

*Silver.* The toxicity of silver to aquatic life is apparently dependent on water hardness: the harder the water, the higher the silver concentration that is needed to be toxic. Silver and its compounds have high chronic toxicity to aquatic life. As with all of

the CoCs discussed in this section, the adverse effects of silver include impairments to survival, growth, development, and reproduction in estuarine organisms. Quoting from the U.S. EPA (1987):

"Symptoms of silver intoxication in aquatic organisms appear to be similar to those caused by other heavy metals. Separation and disruption of the gill epithelium is frequently observed, resulting in esphisia. Damage may be the result of silver ions reacting directly at the gill membrane, or as an indirect result of hematological osmotic imbalances."

Such effects on gill structure often manifest as impairments to respiration, an effect particularly noted on mollusks (U.S. EPA, 1987). Other effects noted in laboratory exposures (summarized in U.S. EPA, 1987) include reductions in chlorophyll *a* in phytoplankton populations, ionic imbalance in polychaete coelomic fluid, histopathological changes, impairment of fertilization success and abnormal larval development, and disruption of enzymatic systems. There is no conclusive evidence that silver is carcinogenic to humans. The ER-L and ER-M values for silver in sediments are 1.0 and 3.7 mg/kg, respectively (Long *et al.*, 1995). Studies indicate that marine water-only concentrations of silver should not exceed 2.3 ppm (Long and Morgan, 1990). In Puget Sound sediment tests, statistically significant toxicity to amphipods, oyster larvae, and Microtox™ were observed at concentrations ranging between >0.6 and >3.7 mg/kg. In San Francisco Bay sediments, toxicity to amphipods and oyster larvae occurred at concentrations ranging between 1.1 and >8.6 mg/kg (Long and Morgan, 1990). Tests of sediments from Commencement Bay and San Francisco Bay indicate highly toxic effects (i.e., >78 and 67%, respectively) to the amphipod *Rhepoxynius abronius* at silver concentrations of 0.2 and 1.7 mg/kg, respectively (Long and Morgan, 1990). Survival in the sanddab *Citarichtys stigmaeus* was >82% when exposed to San Diego sediments with 0.8 mg/kg silver (Long and Morgan, 1990).

*Zinc.* Sorption onto sediments is probably the most common fate of zinc in the aquatic environment (Eisler, 1993). Small amounts may be partitioned into the dissolved phase through speciation into soluble zinc compounds. Formation of complexes with organic and inorganic ligands may increase the mobility of zinc in aquatic media, but these complexes also have a tendency to be adsorbed more strongly onto sediments. The ER-L and ER-M values for zinc in sediments are 150 and 410 mg/kg, respectively (Long *et al.*, 1995). Acute LC<sub>50</sub>s for marine fish in water-only exposures to zinc range from 192 to 320,400 ppm (Long and Morgan, 1990). Chronic responses of marine mysids in water-only exposures were noted at 120 ppm (Long and Morgan, 1990). Statistically significant responses in Puget Sound sediment tests were observed at concentrations ranging between 870 and 1600 mg/kg to amphipods, oyster larvae, and Microtox™ (Long and Morgan, 1990). A highly toxic response in the amphipod *Rhepoxynius abronius* was observed during testing of sediment with 707 mg/kg zinc from Puget Sound (Long and Morgan, 1990). A significant toxic response (i.e., 43% mortality) was observed during exposures of the amphipod *Rhepoxynius abronius* to San Francisco Bay sediment with 158 mg/kg zinc (Long and Morgan, 1990). No survival was observed in the polychaete *Nereis virens* exposed to Black Rock Harbor sediment with 334 mg/kg zinc (Long and Morgan, 1990).

Zinc is an essential element in maintaining many physiological processes, and zinc deficiency can result in severe adverse effects on growth, reproduction, and survival in plants and animals. However, exposure to excess concentrations of zinc can result in a range of adverse physiological and ecological effects. According to Eisler (1993):

"The most sensitive aquatic species were adversely affected at nominal water concentrations between 10 and 25 µg Zn/L, including representative species of plants, protozoans, sponges, mollusks, crustaceans, echinoderms, fish, and amphibians. Acute LC<sub>50</sub> (96-h) values were

between 32 and 40,930 µg/L for freshwater invertebrates, 66 and 40,900 µg/L for freshwater teleosts, 195 and >320,000 µg/L for marine invertebrates, and 191 and 38,000 µg/L for marine teleosts. Acute toxicity...was markedly affected by the age and nutrient status of the organism... Pancreatic degeneration occurred in ducks fed diets containing 2,500 mg Zn/kg ration. Ducks died when fed diets containing 3,000 mg Zn/kg feed."

Thus, according to Eisler (1993), adverse effects include decreased growth, survival, and reproduction. Some noncarcinogenic effects of zinc to humans and animals are evident, but information on carcinogenic effects could not be located in the literature.

*Butyltins.* Butyltin compounds include dibutyltin (DBT), monobutyltin (MBT), and tributyltin (TBT). Generally, DBT and MBT represent biodegradable and therefore less toxic degradation products of TBT. Of the three, TBT is the more prominent compound found (Clarke *et al.*, 1988; Bryan and Gibbs, 1991; Fent and Hunn, 1995). Tributyltin leached from anti-fouling paints inhibits the attachment of fouling organisms (sessile invertebrates) and has been shown to be toxic even at very low concentrations (Wade *et al.*, 1990). In fact, studies have demonstrated that TBT is toxic at concentrations far below those indicated for other marine pollutants (Clarke *et al.*, 1988). TBT concentrations in aquatic sediments likely reflect partitioning between butyltins and suspended particles in the water column, although up to 99% of the TBT may reside in the sediments. Nonetheless, TBT contaminated sediments can represent a substantial source of organotin to aquatic waters (Huggett *et al.*, 1986, as reported in Wade *et al.*, 1990). Studies by Wade *et al.* (1990) determined that the average ratio of TBT concentrations in bivalves compared to sediments collected nearby is 18 (range 6.8 - 57 in coastal waters of the U.S.), suggesting a moderate bioaccumulation potential. Fent and Hunn (1995) indicate that because TBT has significant lipid solubility with log

*n*-octanol/water partition constant of ca. 3.5, a high affinity for bioaccumulation may exist. Although regulations have been introduced and water column concentrations have declined, TBT concentrations in sediments have persisted at levels high enough to induce chronic effects in susceptible aquatic organisms such as marine mollusks (Fent and Hunn, 1995). Fent and Hunn (1995) suggest that degradation of TBT is slow in sediments, with a half-life in the range of two to three years. Others have indicated a half-life of less than one year for aerobic sediments and two years for anaerobic sediments (Bryan and Gibbs, 1991). In addition, other studies have shown the ability of some species of fish, crustaceans, bivalves, and microorganisms to bioconcentrate TBT to levels which are orders of magnitude higher than the exposure concentration (Clarke *et al.*, 1988).

Acute effects of TBT have been observed in the water column where TBT concentrations of 1 ng/L have been associated with reduced reproduction (i.e. egg laying) in the freshwater snail (Fent and Hunn, 1995). Histologic alterations were observed in young European minnows exposed to 0.8 µg/L TBT (Fent and Hunn, 1995). Reduced growth was noted in long-term exposures of rainbow trout yolk sac fry to 0.2 µg/L TBT, and a NOEC of 0.04 µg/L TBT was estimated for this organism (Fent and Hunn, 1995). Immunotoxic effects were observed in the guppy at 0.32 µg/L TBT. In studies of the zooplankton, *Acartia tonsa*, reductions in survival in acute tests were observed at the lowest measurable concentration, 0.029 µg/L, and NOECs and LOECs for survival during chronic tests were 0.024 and 0.017 µg/L, respectively (Bushong *et al.*, 1990). In studies of sediments, data indicate that concentrations of TBT are one to several thousand times higher than concentrations found in the overlying water (Bryan and Gibbs, 1991). In an assessment of sediments, bivalves were virtually eliminated when TBT concentrations exceeded 0.8 µg/g (Fent and Hunn, 1995). Although ER-L and ER-M ranges are unavailable for TBT, studies have shown that mollusks respond to TBT concentrations in sediments as low as 10 ng/g, while some copepod

crustaceans, echinoderms, polychaetes, tunicates, phytoplankton, and fish respond to TBT concentrations which range between 10 and 100 ng/g (Bryan and Gibbs, 1991).

*Polycyclic Aromatic Hydrocarbons (PAHs)*. High molecular weight (HMW) PAHs (e.g., chrysene, benzo(a)anthracene, benzo(b)fluoranthene, benzo(k)-fluoranthene, benzo(a)pyrene, and dibenzo(a,h)anthracene) typically have low solubility in water, high partition coefficients (i.e., higher affinity for organic matter, such as in soil and sediments, than water), and slow degradation. Based on the low water solubility and high affinity to organic matter, significant leaching of HMW PAHs into groundwater is not expected. Solubility of PAHs generally decreases with increasing molecular weight; the less soluble the PAH compound, the more likely it will adsorb to soil or sediment particles. The primary removal mechanisms for PAHs in aquatic environments are by volatilization, photochemical reactions, and microbial degradation (ATSDR, 1989b). ER-L and ER-M sediment values in  $\mu\text{g}/\text{kg}$  for various PAH CoCs are 16 and 500 for acenaphthene, 44 and 640 for acenaphthylene, 85.3 and 640 for anthracene, 261 and 1,600 for benzo(a)anthracene, 430 and 1,600 for benzo(a)pyrene, 384 and 2800 for chrysene, 63.4 and 260 for dibenz(a,h)anthracene, 19 and 540 for fluorene, 600 and 5,100 for fluoranthene, 240 and 1,500 for phenanthrene, and 665 and 2,600 for pyrene, respectively (Long *et al.*, 1995).

Amphipods, oyster larvae, and Microtox™ exhibited statistically significant responses to acenaphthene in Puget Sound sediment tests at concentrations ranging between 500 and 630  $\mu\text{g}/\text{kg}$  (Long and Morgan, 1990). A significant toxic response (i.e., 43% mortality) was observed during exposures of the amphipod *Rhepoxynius abronius* to San Francisco Bay sediment with 7.6  $\mu\text{g}/\text{kg}$  acenaphthene (Long and Morgan, 1990). A highly toxic response was observed during tests using *Rhepoxynius abronius* (i.e., 80% mortality) in sediment from Commencement Bay with 654  $\mu\text{g}/\text{kg}$  acenaphthene (Long and Morgan, 1990). Significant toxicity was observed in the

amphipod *Ampelisca abdita* exposed to Black Rock Harbor sediment with 30 µg/kg acenaphthene (Long and Morgan, 1990).

In Puget Sound sediment tests, amphipods, oyster larvae, and Microtox™ exhibited statistically significant responses to anthracene at concentrations ranging between 960 and 1,900 µg/kg (Long and Morgan, 1990). Bioassays of San Francisco Bay sediments using bivalve larvae and amphipods indicated significant effects at 24 µg/kg and 1,100 µg/kg anthracene, respectively (Long and Morgan, 1990). Tests of sediments from Commencement Bay and Eagle Harbor, Washington, were highly toxic to the amphipod *Rhepoxynius abronius* at 363 and 7,597 µg/kg anthracene, respectively (Long and Morgan, 1990). In tests with the fish *Leiostomus xanthurus*, the 24-hr and 28-day LC<sub>50</sub>s for anthracene were 147,840 and 6,600 µg/kg, respectively (Long and Morgan, 1990).

Effects of benzo(a)anthracene were observed in bivalve larvae and the fish *Leiostomus xanthurus* when concentrations ranged from 60 µg/kg (in tests of sediments from San Francisco Bay) to 350,000 µg/kg (in bioassays of sediments from the Elizabeth River), respectively (Long and Morgan, 1990). In tests of Puget Sound sediment, statistically significant effects to amphipods, oyster larvae, and Microtox™ were observed between 1,300 and 1,600 µg/kg benzo(a)anthracene (Long and Morgan, 1990). Statistically significant toxicity in the bivalve and amphipod bioassays was observed in exposures to sediments from San Francisco Bay with 60 and 1,100 µg/kg of benzo(a)-anthracene (Long and Morgan, 1990). Amphipod mortality exceeded 80% in tests of sediments from Commencement Bay and Eagle Harbor where benzo(a)anthracene concentrations were 931 and 11,088 µg/kg, respectively (Long and Morgan, 1990). In tests with the fish *Leiostomus xanthurus*, the 24-hr and 28-day LC<sub>50</sub>s for benzo(a)anthracene were 196,000 and 8,750 µg/kg, respectively (Long and Morgan, 1990).

Effects of benzo(a)pyrene were observed in bioassays of sediments from San Francisco Bay and Lake Union, Washington, where concentrations ranged from 400 to 220,000 µg/kg (Long and Morgan, 1990). In tests of Puget Sound sediment, statistically significant effects to amphipods, oyster larvae, and Microtox™ were observed between 1,600 and 2,400 µg/kg benzo(a)pyrene (Long and Morgan, 1990). Statistically significant toxicity in the bivalve and amphipod bioassays was observed in exposures to sediments from San Francisco Bay with >1,800 and 1,300 µg/kg of benzo(a)-pyrene, respectively (Long and Morgan, 1990). Amphipod mortality exceeded 80% in tests of sediments from Commencement Bay and Eagle Harbor where benzo(a)pyrene concentrations were 1,192 and 3,485 µg/kg, respectively (Long and Morgan, 1990). In tests with the fish *Leiostomus xanthurus*, the 24-hr and 28-day LC<sub>50</sub>s for benzo(a)pyrene were 55,160 and 2,462 µg/kg, respectively (Long and Morgan, 1990).

Responses were observed in amphipod and fish tests of sediments from San Francisco Bay and the Elizabeth River, with chrysene concentrations ranging from 80 µg/kg to 317,000 µg/kg, respectively (Long and Morgan, 1990). In tests of Puget Sound sediment, statistically significant effects to amphipods, oyster larvae, and Microtox™ were observed between 1,400 and 2,800 µg/kg chrysene (Long and Morgan, 1990). Statistically significant differences in the bivalve larval and amphipod bioassays were indicated when San Francisco Bay sediment concentrations of chrysene were 1,700 and 2,100 µg/kg, respectively (Long and Morgan, 1990). Amphipod mortality exceeded 80% in tests of sediments from Commencement Bay and Eagle Harbor, where chrysene concentrations were 1,363 and 10,574 µg/kg, respectively (Long and Morgan, 1990).

Effects were observed when dibenz(a,h)anthracene concentrations were as low as 42 µg/kg in bivalve larval bioassays of San Francisco Bay sediments (Long and Morgan, 1990). In tests of Puget Sound sediment, statistically significant effects to

amphipods, oyster larvae, and Microtox™ were observed between 230 and 260 µg/kg dibenz(a,h)-anthracene (Long and Morgan, 1990). Statistical differences in the bivalve larval and amphipod bioassays with San Francisco Bay sediments were indicated when concentrations of dibenz(a,h)anthracene were 260 and 300 µg/kg, respectively (Long and Morgan, 1990). Amphipod mortality exceeded 80% in tests of sediments from Commencement Bay and Eagle Harbor, where dibenz(a,h)anthracene concentrations were 72 and 263 µg/kg, respectively (Long and Morgan, 1990). Significant toxicity to bivalve larvae was observed in sediments from Eagle Harbor with 63 µg/kg dibenz(a,h)-anthracene (Long and Morgan, 1990).

The amphipod *Grandidierella japonica* exhibited a significant response to sediment from southern California with 11 µg/kg fluorene (Long and Morgan, 1990). The 24-hour and 28-day LC<sub>50</sub>s for the fish *Leiostomus xanthurus* exposed to Elizabeth River sediments, were 700,000 and 17,500 µg/kg fluorene, respectively (Long and Morgan, 1990). Liver somatic condition indices were elevated in winter flounder exposed to 220,550 µg/kg fluorene in spiked sediment bioassays (Long and Morgan, 1990). Mixed function oxygenase (i.e., P450) induction in winter flounder liver and kidney was elevated in spiked sediment tests with 176,510 and 285,290 µg/kg fluorene (Long and Morgan, 1990). In tests of Puget Sound sediment, statistically significant effects to amphipods, oyster larvae, and Microtox™ were observed at 540 µg/kg fluorene (Long and Morgan, 1990). Statistical differences in the bivalve larval and amphipod bioassays with San Francisco Bay sediments were indicated when concentrations of fluorene were 11 and 210 µg/kg, respectively (Long and Morgan, 1990).

Bioassays of sediments from southern California and the Elizabeth River indicated significant responses to amphipods at 382 µg/kg fluoranthene (Long and Morgan, 1990). In tests of Puget Sound sediment, statistically significant effects to amphipods, oyster larvae, and Microtox™ were observed between 1,700 and 3,900

µg/kg fluoranthene (Long and Morgan, 1990). Statistical differences in the bivalve larval and amphipod bioassays with San Francisco Bay sediments were indicated when concentrations of fluoranthene were 2,000 and >3,700 µg/kg, respectively (Long and Morgan, 1990). The 24-hour and 28-day LC<sub>50</sub>s for the fish *Leiostomus xanthurus* exposed to Elizabeth River sediments, were 327,200 and 59,250 µg/kg fluoranthene, respectively (Long and Morgan, 1990).

Responses in bivalve larval bioassays were observed using sediments from San Francisco Bay with 88 µg/kg phenanthrene (Long and Morgan, 1990). In tests of Puget Sound sediment, statistically significant effects to amphipods, oyster larvae, and Microtox™ were observed between 1,500 and 5,400 µg/kg phenanthrene (Long and Morgan, 1990). Amphipod mortality exceeded 80% in tests of sediments from Commencement Bay and Eagle Harbor, where phenanthrene concentrations were 2,838 and 33,603 µg/kg, respectively (Long and Morgan, 1990). Significant amphipod mortality (i.e., 67%) was observed in tests of San Francisco Bay sediments with 242 µg/kg phenanthrene (Long and Morgan, 1990). The 24-hour and 28-day LC<sub>50</sub>s for the fish *Leiostomus xanthurus* exposed to Elizabeth River sediments, were 2,363,200 and 105,500 µg/kg phenanthrene, respectively (Long and Morgan, 1990). Elevated liver somatic condition indices were observed in winter flounder exposed to 340 µg/kg phenanthrene in spiked sediment tests (Long and Morgan, 1990). Mixed function oxygenase (i.e., P450) induction in winter flounder liver and kidney was elevated in spiked sediment tests with 270 and 429 µg/kg phenanthrene, respectively (Long and Morgan, 1990).

Elevated liver somatic condition indices were observed in winter flounder exposed to 360 µg/kg pyrene in spiked sediment tests (Long and Morgan, 1990). Mixed function oxygenase (i.e., P450) induction in winter flounder liver and kidney was elevated in spiked sediment tests with 300 and 182 µg/kg pyrene, respectively

(Long and Morgan, 1990). In tests of Puget Sound sediment, statistically significant effects to amphipods, oyster larvae, and Microtox™ were observed between 2,600 and 4,300 µg/kg pyrene (Long and Morgan, 1990). Amphipod mortality exceeded 80% and 65% in tests of sediments from Commencement Bay and San Francisco Bay, where pyrene concentrations were 1,820 and 777 µg/kg, respectively (Long and Morgan, 1990). The 24-hour and 28-day LC<sub>50</sub>s for the fish *Leiostomus xanthurus* exposed to Elizabeth River sediments, were 1,350,000 and 33,750 µg/kg pyrene, respectively (Long and Morgan, 1990).

PAHs as a group contain a number of individual organic compounds, as discussed above, and thus may vary in toxicity and ecological effects. According to Eisler (1987):

"A wide variety of PAH-caused adverse biological effects have been reported in numerous species of organisms under laboratory conditions, including effects on survival, growth, metabolism, and especially tumor formation. Inter- and intraspecies responses to carcinogenic PAHs were quite variable, and were significantly modified by many chemicals including other PAHs that are weakly carcinogenic or noncarcinogenic. Until these interaction effects are clarified, the results of single substance laboratory tests may be extremely difficult to apply to field situations of suspected PAH contaminants."

Responses to Total PAHs were observed in sediment tests where concentrations ranged between 870 and 21,200,000 µg/kg (Long and Morgan, 1990). In tests of Puget Sound sediment, statistically significant effects to amphipods, oyster larvae, and Microtox™ were observed with 5,200 µg/kg low molecular weight PAHs and between 12,000 and 18,000 µg/kg high molecular weight PAHs (Long and Morgan, 1990). Statistically significant effects in the bivalve larval development and amphipod

bioassays were observed in tests of sediments from San Francisco Bay with 870 and >15,000 µg/kg Total PAHs, respectively (Long and Morgan, 1990). Toxic responses (i.e., >80% amphipod mortality and >44% abnormal larval development) were noted in tests of Commencement Bay sediments with 6,977 and 3,835 µg/kg low molecular weight PAHs and with 9,794 and 9,042 µg/kg high molecular weight PAHs, respectively (Long and Morgan, 1990). Negative growth was noted in nematode bioassays using Hudson-Raritan Estuary sediments with 42,769 µg/kg Total PAHs (Long and Morgan, 1990). Elevated liver somatic condition indices were observed in winter flounder exposed to 228,722 µg/kg Total PAHs in spiked sediment tests (Long and Morgan, 1990). Mixed function oxygenase (i.e., P450) induction in winter flounder liver and kidney was elevated in spiked sediment tests with 183,060 and 295,860 µg/kg Total PAHs, respectively (Long and Morgan, 1990). The 24-hour and 28-day LC<sub>50</sub>s for the fish *Leiostomus xanthurus* exposed to Elizabeth River sediments, were 530,000 and 21,200,000 µg/kg Total PAHs, respectively (Long and Morgan, 1990).

In addition to the interactions alluded to by Eisler (1987), an understanding of the potential bioaccumulation (and hence potential effects) of PAHs is confounded by the fact that many aquatic vertebrate (primarily fish) and, to a lesser degree, some invertebrate (polychaetes, crustaceans, and mollusks) species possess enzymatic systems which support metabolism of PAHs (National Research Council of Canada, 1983), such that the level of exposure of these organisms to PAHs cannot be directly inferred from the PAH concentrations present (or absent) in their tissues. Such enzymatic systems have also been observed in some bacteria, fungi, and algae. With respect to PAH activation and carcinogenesis, the National Research Council of Canada (1983, p. 13) states:

"Structure-activity relationships for mutagenic and carcinogenic activity seem to favor 4-, 5- and 6-ring PAHs rather than smaller or larger compounds. It is believed that PAHs require metabolic activation to exert

their carcinogenic effects...with carcinogenesis being initiated by the binding of electrophilic metabolites to critical cellular constituents.

Enzymes other than mixed function oxidase (MFO), which may influence the rate of production or destruction of reactive metabolites, are found in aquatic animals and may play an important role in toxicity."

Hence, although the metabolism of PAHs is more common for aquatic vertebrates, significant food chain transfer may occur between invertebrates (such as bivalves, e.g. mussels, clams) that do not metabolize PAHs and vertebrates (e.g. seabirds) whose diet may consist of substantial quantities of these prey types. This exposure pathway is addressed in the current investigation as the Fourth Tier model for avian aquatic receptors (Figure 3.4-7).

*Mirex.* Mirex, a chlorinated insecticide, is the active ingredient used in bait to control the fire ant, harvester ant, and the Texas leaf-cutting ant. Mirex, marketed under the trade name Dechlorane, is also used in flame-retardant coatings. Mirex is a white, odorless, crystalline solid, partially soluble in some solvents, and only slightly soluble in water (i.e. maximum solubility in water is 0.20 mg/L at 24°C). In aquatic tests with phytoplankton, photosynthesis was inhibited 16%, 10%, 33%, and 19% after exposure to 1 ppb mirex for 5, 10, 15, and 20 days, respectively (Verschueren, 1983). In tests of mirex using the freshwater cnidarian *Hydra spp.*, 1-day, 2-day, 3-day, 4-day, 5-day, and 6-day LC<sub>50</sub>s were measured at 100,000, 682, 23, 4, 1, and 0.5 ppm, respectively (Verschueren, 1983). In the 96-hour test with the algae, *Tetrahymena pyriformis*, growth was inhibited by 96% at 0.9 ppb mirex (Verschueren, 1983). Decreased feeding activity was observed in the adult polychaete *Arenicola cristata* during 30 day exposures to <0.003 - 0.062 µg/L, and decreased survival of prey (i.e., *Palaemonetes vulgaris*) was observed in tests with the fish *Lagodon rhomboides* during 13 day exposures to 0.025 - 0.046 µg/L (Verschueren, 1983). EC<sub>50</sub>s and LC<sub>50</sub>s for the juvenile pink shrimp (*Penaeus duorarum*), blue crab (*Callinectes sapidus*), eastern

oyster (*Crassostrea virginica*), and the fish (*Leiostomus xanthurus*) were 720, 2000, 2000, and 2000 µg/L, respectively (U.S. EPA, 1987b). Increased mortality was observed in acute and chronic tests with shrimp, blue crabs, fiddler crabs, and finfish exposed to particles of fire ant bait (0.3% mirex) in food and/or water (Lowe *et al.*, 1971).

*DDE.* DDE, a metabolite of DDT, is very persistent in the environment. Few specific data are available regarding the environmental fate of DDE; however, both DDT and DDD in water are subject to sedimentation, volatilization, photodegradation, and food web uptake. DDT is absorbed by humans in direct proportion to dietary exposure. Human epidemiological data are not available for DDE, although based on its structural similarity to DDT, it is classified as a probable human carcinogen. Bioconcentration factors for DDE are from  $10^3$  to  $10^5$ . The ER-L and ER-M values in sediments for DDE are 2.2 and 27 µg/kg, respectively (Long *et al.*, 1995). Statistically significant responses to DDE were noted in tests of sediments from Puget Sound in the amphipod bioassay and in the evaluation of benthic community composition where DDE concentrations were 15 and 9 µg/kg, respectively (Long and Morgan, 1990). Statistically significant effects in the bivalve larval development and amphipod bioassays were observed in tests of sediments from San Francisco Bay with 2.2 µg/kg DDE (Long and Morgan, 1990). Significant toxicity to amphipods and bivalve larvae were noted in tests of sediments from San Francisco Bay with 1 and 3 µg/kg DDE (Long and Morgan, 1990).

*PCBs.* PCBs, also known by the commercial name Aroclors, vary substantially in their chemical, physical, and biological properties based on their degree of chlorination (Eisler, 1987). The less chlorinated Aroclors will sorb less strongly onto sediments than the highly chlorinated components. Sediment and suspended particulate transport is the dominant mode of PCBs in aqueous solutions. The ER-L and ER-M values for total PCBs in sediments are 22.7 and 180 µg/kg, respectively (Long *et al.*, 1995).

Responses to PCBs were observed in tests of marine sediments where concentrations ranged between 36.6 and 10,800 µg/kg (Long and Morgan, 1990). In tests of Puget Sound sediment, statistically significant effects to amphipods, oyster larvae, and Microtox™ were observed between 130 and 2,500 µg/kg PCBs (Long and Morgan, 1990). Statistically significant effects in the bivalve larval development and amphipod bioassays were observed in tests of sediments from San Francisco Bay with 54 and 260 µg/kg PCBs, respectively (Long and Morgan, 1990). Toxic responses (>80% amphipod mortality and >44% abnormal bivalve larval development) were noted in tests of Commencement Bay sediments with 38 and 368 µg/kg PCBs, respectively (Long and Morgan, 1990). Negative growth was noted in nematode bioassays using Hudson-Raritan Estuary sediments with 638 µg/kg PCBs (Long and Morgan, 1990).

PCBs as a group contain a number of individual congeners which vary with respect to toxicity. Exposure to PCBs in various combinations has resulted in effects on growth of phytoplankton through impairment of photosynthesis and cell division, and has been shown to influence competitive interactions between phytoplankton species (Mosser *et al.*, 1972; Fisher *et al.*, 1974). PCBs also affect reproduction in fish (Hansen *et al.*, 1974), growth in bivalves (Parrish *et al.*, 1972), molting physiology of crustaceans (Fingerman and Fingerman, 1977), and may adversely affect population dynamics in fish (Munns *et al.*, 1995). Hansen *et al.* (1974) demonstrated the adverse influence of PCB exposure (as Aroclor 1254) on recruitment and development of benthic and epibenthic estuarine communities in laboratory exposure systems. At high enough concentrations, PCBs cause death in a number of estuarine organisms (Hansen, 1974).

In summary, the CoCs identified in Section 3.3 can be characterized by their tendency to be associated with dissolved or particulate/sediment fractions, assuming that other, non-contaminant related factors (e.g. TOC, AVS) are similar in concentration:

- Dissolved fraction components - salts of nickel, copper, cadmium, and chromium(+6) have a tendency to be more prevalent in the dissolved phase than those of other metals when conditions permit;
- Particulate/sedimentary fraction components - PAHs, PCBs, DDE, silver, lead, zinc, arsenic, mercury, and chromium(+3) have a greater tendency to be particle-associated than the above group, again assuming comparable geochemical conditions.

This information will be used to aid in the interpretation of contaminant distribution, bioaccumulation and toxicity as discussed in the following sections.

## 5.2. TOXICITY EVALUATIONS

Site-specific evaluations of bulk surface sediments and porewaters were conducted using the 10-day amphipod (*Ampelisca abdita*) mortality test and the sea urchin (*Arbacia punctulata*) sperm cell toxicity test, respectively. Both tests are directed tools to evaluate the bioavailability of contaminants in the respective media. Comparison of these results to Phase I toxicity data provides a more detailed spatial evaluation of potential impacts to aquatic biota.

### 5.2.1. Phase I and II Sediment Toxicity: Amphipod Test Results

*Background.* The 10-day amphipod test has been used extensively to assess the toxicity of laboratory-spiked and field-collected sediments to benthic organisms (DiToro *et al.*, 1992, Scott and Redmond, 1989; Long and Morgan, 1990). In addition, *Ampelisca abdita* has been used routinely for sediment toxicity tests conducted by SAIC in support of numerous EPA programs (SAIC, 1990a; SAIC, 1991; SAIC, 1992a; and SAIC, 1993a). It was the most sensitive species tested in the U.S. EPA/USACE Field Verification Program, and has formed the toxicological basis for EPA research on

the availability of metals in relation to acid volatile sulfides in marine sediments (Gentile *et al.*, 1987 and DiToro *et al.*, 1992). It has been used to characterize the toxicity of sediments from the Calcasieu River, LA, covering a broad range of salinity and grain size (SAIC, 1990b). *Ampelisca abdita* was the first species used to demonstrate the toxicity of sediments from New Bedford Harbor, MA, and subsequently was used to assess the effectiveness of capping procedures as part of a Pilot Dredging Project on site remediation techniques (USACE, 1989). Tests of sediments from New York Harbor have been conducted for EPA Region II and the New York District (SAIC, 1992b; SAIC, 1994a and SAIC, 1995). SAIC also completed a series of 10-day amphipod tests for NOAA to characterize toxicity of sediments from the Hudson-Raritan Estuary, Long Island Sound, Boston Harbor, and Tampa Bay (SAIC, 1992c; SAIC, 1992d; SAIC, 1993b and SAIC, 1994b).

*Test procedures.* Amphipod tests (5 replicates each) were conducted on surface sediments from 32 Phase I and Phase II sampling stations comprising seven intertidal stations at the base of McAllister Point Landfill ("NSB" sites), stations to the south and west of the landfill ("S", "M" and "D" stations), stations offshore of the McAllister Point Landfill (MCL stations), and reference site Jamestown Cranston Cove (JCC) intertidal and offshore stations (JCC-S1 and JCC-M1, respectively). Amphipods were exposed to test sediments for 10 days under static conditions, following ETC SOPs developed according to ASTM and EPA guidelines (ASTM 1990 and U.S. EPA 1994; Appendix B-1-1). Water quality parameters were monitored throughout the test; temperature, salinity, dissolved oxygen (DO), and pH were measured in two replicates selected through a computerized random and blind sampling process, twice during each test. In addition, samples were analyzed for ammonia (following methods of Bower and Holm-Hansen, 1908) to address the continuing concern and debate over the potential toxic effects of ammonia in static sediment toxicity tests. Sub-samples of sediments were collected for porewater analyses after sediments were press-sieved and homogenized before placement into test chamber.

Performance control sediments were collected during May 1994 from the U.S. Army Corps of Engineers New England Division central Long Island Sound (LIS) reference station. Sediments from this reference station have been used for the COE Disposal Areas Monitoring System, the Field Verification Program, and EPA's EMAP Virginian Province in 1990-1993. The sediments from this site are fine-grained (>90% silt-clay) and have an organic carbon content of about 2%. An extensive database has demonstrated its non-toxic nature in solid-phase tests with *A. abdita*. The survival of *A. abdita* exposed to this collection of LIS sediment was consistent with all previous LIS collections used at the ETC (November 1989, May 1991, and August 1993). Performance control survival for the 34 of the most recent tests performed at the ETC are presented in Appendix B-1-1; the control survival range for these tests is 84% to 98%.

*Data analyses.* Stations with a mean survival less than that of the LIS performance control were compared statistically to the control using a one-way, unpaired *t*-test ( $\alpha=0.05$ ) assuming unequal variance. Data were not transformed since an examination of a large historical data set from the ETC has shown that *A. abdita* percentage survival data meet the requirement of normality. Significant toxicity for *A. abdita* has been defined as survival statistically less than the performance control and  $\leq 80\%$  of the mean control survival (U.S. EPA, 1994). Statistical power curves created from SAIC's extensive testing database with *A. abdita* show that the power to detect a 20% difference from the control is approximately 90%. Sampling stations with toxicity results statistically different than the performance control,  $\leq 80\%$  of the control, and  $\leq 60\%$  of the control were flagged.

*Phase I and II Results.* Toxicity testing results from Phases I and II were merged in order to provide more complete representation of conditions in the study area. To facilitate the intercomparison of data sets, the station-specific toxicity was normalized to the mean performance control. Mean sample survival for McAllister Point Stations,

normalized to performance controls, ranged from 0 to 103% (Table 5.2-1). High toxicity (e.g., “\*\*\*\*”) was observed at Station NSB-5, where survival was both statistically lower than the performance control and <10% of the mean control survival. Intermediate toxicity (e.g., “\*\*\*”) was observed at Stations NSB-1 and NSB-4, where survival was both statistically lower than the performance control and <60% of the mean control survival. Reduced toxicity (e.g., “\*\*”) was observed at Stations NSB-3, NSB-6, NSB-7, and S2B was both statistically lower than the performance control and 60-80% of the mean control survival. Two stations, MCL-13 and JCC-S1, were statistically lower than the performance control, but not <80% of the mean control survival (e.g., “\*”). All other stations tested exhibited no effect.

Unionized ammonia in the overlying water exceeded the NOEC of 0.40 mg/L at pH 7.7 (U.S. EPA, 1994) on three occasions, however none occurred in samples for which significant survival effects <80% were observed (Appendix B-1-1). Total ammonia in porewater was elevated above the NOEC (30 mg/L) at six stations (MCL-9 through MCL-12, MCI-16, and JCC-S1); unionized ammonia in porewater was elevated above the NOEC at six stations (S3, MCL-10 through MCL-12, MCL-16, and JCC-S1). Survival at Station JCC-S1 exhibited statistically significant difference from control; otherwise, no survival effects were noted at these stations (Table 5.2-1).

The repeated analyses of sediment toxicity at the reference site JCC-M1 between Phase I and Phase II gave very similar results (97% as compared to 102% of the control), indicating spatial/temporal variability is relatively unimportant with regard to the observed trends. Survival in JCC-S1 sediments was 83%, but high unionized ammonia concentrations (0.54 mg/L) were above the NOEC (0.4 mg/L) and thus may have contributed to the toxicity of the sample at this station.

### 5.2.2. Phase I and II Porewater Toxicity: Sea Urchin Test Results

The chronic toxicity of porewaters obtained from sediments collected from McAllister Point Landfill study area and Jamestown Cranston Cove was assessed to evaluate the bioavailability and biological effects of interstitial water sediment contaminants to benthic organisms. Sediment porewater toxicity was determined using the sea urchin (*Arbacia punctulata*) fertilization test according to SOP No. SCT-01 in Appendix B-1-2. This assay is used routinely by the U.S. EPA and by National Pollutant Discharge Elimination System (NPDES) permittees to determine ambient and effluent water quality, and to evaluate the effects of pollutants on aquatic life (U.S. EPA, 1988).

The purple sea urchin, *Arbacia punctulata*, occurs along the North American east coast from Cape Cod to Florida. They live in widely separated aggregations on rocky and shelly bottoms or adhere to rocks. Their life cycle includes a period of planktonic embryo-larval development, followed by settlement and metamorphosis in the adult life stage. Sea urchin gametes have become widely used and popular subjects for toxicological studies (Bay *et al.*, 1993).

*Methodology.* Sea urchin tests were conducted on surface sediment porewaters (Table 5.2-2) from the same suite of stations as performed for bulk sediment tests with amphipods. Porewater was extracted for testing according to methods described by Winger and Lasier (1991). An SOP is attached in Appendix B-1-2. Briefly, porewater was extracted by inserting a fused-glass air stone attached with plastic tubing to a 50 cc syringe. A vacuum was created by retracting and bracing the syringe plunger. Extractions were performed overnight in the dark at 4°C before testing. Samples for which porewater was unobtainable were saturated for six hours with filtered natural seawater collected from lower Narragansett Bay on an incoming tide. Porewater samples were filtered through a 0.45 µm filter for testing. The sea urchin fertilization

test was conducted following ETC SOPs according to U.S. EPA procedures (U.S. EPA, 1988).

One mL suspensions of eggs and sperm from each of two replicates was transferred to a Sedgwick-Rafter counting chamber and eggs examined using a compound microscope (100X). One hundred eggs were examined for fertilization as indicated by the presence of a membrane surrounding the egg. A third replicate was examined when data varied by more than 10%. The performance control was natural seawater (NSW) collected on the test date from lower Narragansett Bay, RI during an incoming tide after passage through a 0.45  $\mu\text{m}$  filter. Fertilization results for *Arbacia* eggs and sperm exposed to NSW in this test were consistent with all previous NSW collections at the ETC; performance control data for the last 35 sea urchin tests was 90% to 100% (Appendix B-1-2).

Extracted porewater samples were analyzed for ammonia to address the potential toxic effects of this compound. Unionized ammonia was calculated using measured total ammonia values and concurrent measurements of pH and salinity, and mean test temperature. The calculations were based on information provided in Whitfield (1978).

*Data Analysis.* The number of fertilized eggs per 100 were recorded on laboratory data sheets, and were then entered into a computer spreadsheet for statistical analyses. Stations with mean fertilization less than that of the natural seawater (NSW) performance control were compared statistically to the control using a two-sample, one-way, unpaired Student's *t*-test assuming unequal variance ( $\alpha=0.05$ ), which tests the hypothesis that the means of the NSW control and the sample are equal. Station samples with an alpha or p value less than or equal to 0.05 indicate statistical significance and were flagged; and samples with fertilization  $\leq 70$  and  $\leq 50\%$  of the control were also flagged.

*Phase I and II Results.* As with amphipod data, toxicity testing results from the Phase I and Phase II sea urchin tests were merged in order to provide more complete representation of conditions in the study area. To facilitate the intercomparison of data sets, the station-specific toxicity was normalized to the mean performance control. Results indicated high toxicity (e.g., mean fertilization, expressed as a percentage of the performance control, and  $\leq 10\%$  of the mean control) for three stations, D3, NSB-4, and JCC-S1 (Table 5.2-2). Intermediate toxicity (e.g., mean fertilization, expressed as a percentage of the performance control, and  $\leq 50\%$  of the mean control) was observed at three stations, S2B, M1, and NSB-5 (Table 5.2-2). Stations NSB-3, S3, and JCC-D1 exhibited low toxicity (e.g., mean fertilization percentages that were both statistically lower and 50-70% of the control mean), while Stations OS-30B, M2, M3, S1, S4, NSB-7, MCL-12, MCL-13, MCL-14, MCL-16, and JCC-M1 exhibited fertilization percentages which were statistically lower but not  $\leq 70\%$  of the control mean. All other stations exhibited no effect. The repeated analyses of porewater toxicity at the reference site JCC-M1 between Phase I and Phase II gave somewhat dissimilar results (89% as compared to 79% of the control, respectively), perhaps indicating the degree to which seasonality or sampling variability at this site may alter toxicity findings. The toxicity observed at Station D3 in the Phase I sample was not observed in the Phase II samples from similarly-located stations (i.e., MCL-15 and MCL-16), suggesting that the apparent toxicity at D3 is not widespread, or that seasonal changes have occurred between Phase I and Phase II investigations.

Porewater total and unionized ammonia measurements for the sea urchin test are also presented in Table 5.2-2. Total ammonia values ranged from 0.35 to 32.85 mg/L, while unionized ammonia values ranged from 0.01 to 0.435 mg/L. Although a NOEC for total and unionized ammonia are not available for the sea urchin embryo test, the  $EC_{50}$  for total and unionized ammonia for sea urchin fertilization is 20 mg/L and 0.6 mg/L, respectively (NOAA, 1994). Ammonia concentrations at S2B, M1, S4, and JCC-S1 exceeded the total ammonia criteria but are well below the unionized ammonia

criteria. This suggests that the reduced fertilization observed at these stations is not likely to be a result of ammonia present in the sample.

Overall, conclusions from the combined amphipod and sea urchin tests were in agreement, indicating greatest toxicity for intertidal Stations NSB-3 through NSB-5 and subtidal Station S2B (Figure 5.2-1). However, unlike amphipod toxicity results, sea urchin toxicity results did not indicate impacts at intertidal sites along the northern and southern intertidal areas of the landfill (NSB-1 and NSB-6 to NSB-7, respectively). Furthermore, based on sea urchin fertilization, toxicity was observed for several "Southern Depositional Area" stations sampled in Phase I and at the reference stations, yet no similar impacts were observed at these stations based amphipod survival (Table 5.2-1). The cause for these disparate results is unknown, but is likely related to differential sensitivity to CoCs in the test species.

### 5.2.3. Post-erosion Toxicity Assessment

In October and November 1996, samples of sediment were obtained from splits of chemistry stations identified in Section 4.2, and analyzed for toxicity to invertebrate animals. Sample locations are presented in Figure 3.6-2. As discussed in Sections 3 and 4, these tests were conducted to assess the potential for increased CoC exposure as a result of sediment erosion in the intertidal zone of McAllister Point Landfill after the construction of the landfill revetment. Laboratory reports, raw data, and complete details of sample handling, storage and testing are contained in the *Technical Memorandum for Phase III Investigations* (Brown and Root Environmental, 1996).

*Sediment toxicity to amphipod survival.* The acute toxicity of sediments from selected stations in the vicinity of McAllister Point Landfill was determined to assess the biological effects of sediment contaminants and to evaluate the change in the bioavailability of contaminants in bulk sediments due to the sediment erosion event.

Sixteen sediment samples were evaluated for toxicity using the 10-day *Ampelisca abdita* amphipod test. Sample testing at NSB-3 was not possible due to insufficient sample volume.

The test endpoint was adult survival. Stations with a mean survival less than that of the LIS performance control were compared statistically to the control using a two-sample student's t-test (assuming unequal variances). Data were analyzed and toxicity was determined in the manner described in Section 5.2.1. The data were flagged where survival was statistically significantly less than control (\*), and less than 80% and 60% of the performance control.

Summary survival data are presented in Table 5.2-3, with comparisons to pre-revetment toxicity results. Mean sample survival, normalized to performance controls, ranged from 15 to 98%. Post-revetment mean survival at Stations NSB-2, NSB-4 and NSB-5 (15, 24, and 37%, respectively) was both statistically different from the performance control and <60% of the mean control survival, while survival for Station NSB-7 (63%) was both statistically different than the performance control and <80% of the mean control survival. Water quality parameters for temperature, salinity, and dissolved oxygen measured in the overlying water of chambers during Phase III tests were within acceptable limits (Brown and Root, 1996). The overlying water unionized ammonia NOEC of 0.40 mg/L at pH 7.7 (U.S. EPA, 1994) was exceeded on one occasion (Station S2C), but effects on survival (e.g. <80%) were not observed at this location.

*Elutriate toxicity to sea urchin fertilization and larval development.* The chronic toxicity of elutriates prepared from core sediments collected in the McAllister Point Landfill study area, was assessed with the purple sea urchin *Arbacia punctulata* to evaluate the biological effects of resuspended sediment contaminants to water column organisms.

Fertilization and larval development success were used as test endpoints. Responses were measured in each of three concentrations per station/sample, from which a point estimate of the concentration that would cause a given percent inhibition in fertilization/development is calculated (called the inhibition concentration (IC)).

Sediments from seven sites were collected between 8 October and 5 November 1996. Elutriates were prepared by adding homogenized sediment to filtered (0.45  $\mu\text{m}$ ) natural seawater collected from Narragansett Bay, RI on an incoming tide in a 1:4 volumetric ratio. The mixture was stirred for 30 minutes by hand and then settled for one hour. The supernatant was siphoned off and was used to prepare dilutions. Dilutions were prepared by mixing the supernatant with filtered (0.45  $\mu\text{m}$ ) natural seawater (NSW) collected from lower Narragansett Bay on an incoming tide. Elutriate dilutions (10%, 50%, and 100%) as well as a NSW performance control (0%) were tested.

Stations with mean fertilization less than that of the NSW performance control were compared statistically to the control. The linear interpolation method available on ToxCalc (version 4.0.8) from TidePool Scientific Software was used to calculate the IC values of samples where statistically significant responses were noted in one or more of the elutriate dilutions. For this analysis, the Inhibitory Concentration (IC), a point estimate of the elutriate concentration that would cause a 10% reduction in sea urchin fertilization/development ( $\text{IC}_{10}$ ), was calculated for each station;  $\text{IC}_{10}$ s are presented in Tables 5.2-4 and 5.2-5. Samples with an alpha or p value less than or equal to 0.05, indicating statistical significance were flagged (e.g, “\*”). The data were further flagged where statistically significant difference from the control occurred and the  $\text{IC}_{10}$  was less than 70% (“\*+”), less than 50% (“\*++”), and less than 10% (“\*+++”).

The results for sea urchin fertilization success are presented in Table 5.2-4. The indicator of the overall exposure-response relationship,  $\text{IC}_{10}$ s varied over a relatively

narrow range from most toxic (13.3%) at Station MCL-12 to least toxic (36.2%) at Station NSB-6 (Table 5.2-4). It was notable, however, that a greater separation in apparent toxicity was observed for the 50% elutriate exposure, where Stations NSB-2, NSB-5 and NSB-12 exhibited fertilization success less than 10% while at other stations the response improved to greater than 40%. Total ammonia and unionized ammonia were measured in elutriates of sediments used for the Phase III larval development tests and assumed to be comparable for the fertilization tests. Ammonia did not exceed the IC<sub>50</sub> thresholds of 20.0 mg/L (NOAA, 1994) and >0.60 mg/L, respectively (Carr *et al.*, *in press*).

Results for the sea urchin larval development test are presented in Table 5.2-4. IC<sub>10</sub>s for larval development reflected a broader range, but a comparable rank order sensitivity from 6.3% at Station NSB-2 to greater than 100% at Station NSB-6 (Table 5.2-5). As observed for the fertilization test, a greater separation in apparent toxicity was observed for the 50% elutriate exposure, where in this case, Stations NSB-2 and NSB-5 exhibited fertilization success less than 5%, while at other stations the response improved to greater than 70%. As discussed above, total ammonia and unionized ammonia were not present at toxic concentrations.

*Comparison of pre- and post-erosion results.* Results of amphipod survival in bulk sediments and sea urchin fertilization in sediment elutriates collected during the Phase III McAllister Point Landfill study area resampling event were compared to toxicity results from Phase II testing. Rankings from Phase III testing are presented in Table 5.2-6. Stations for which the relative percent difference (RPD) was greater than 30% between in 1995 and 1996 are indicated by bordered cells. Stations which exhibited RPD > 30% and for which the toxicity ranking increased from 1995 to 1996 (for example, Phase II “\*++” and Phase III “\*+++”) are indicated by shaded cells. Findings of the larval development tests, conducted only in Phase III, are also presented as an indicator of concurrence among test endpoints.

A comparison of amphipod sediment toxicity results between pre- and post-erosion conditions demonstrates that no toxicity was observed in subtidal sediment Stations MCL-8 to MCL-12 for either sampling event (Table 5.2-6). Post-erosion toxicity was significantly higher (RPD > 30%) than pre-erosion toxicity at Stations NSB-2 and NSB-4; however, Station NSB-4 exhibited 10-50% survival in both Phase II and Phase III, while survival at Station NSB-2 dropped to less than 10% of control (Table 5.2-6). In contrast, post-erosion toxicity was lower than that for pre-erosion conditions at Stations S2B and NSB-6.

Adverse effects rankings for sea urchin successful fertilization in 100% elutriate of sediments collected during Phase III are also presented in Table 5.2-6, with comparisons to the Phase II porewater results for the same species and endpoint. Toxicity of elutriates from all Phase III stations, with the exception of Station NSB-4, was greater than 30% higher than that observed in porewater testing during Phase II, and exhibited an increased toxicity ranking (Table 5.2-6).

The uniformly low success of sea urchin fertilization in Phase III sediment elutriates, in contrast to amphipod survival in the parent bulk sediment samples and generally high sea urchin larval development success in the corresponding elutriate samples, suggested that an alternate mechanism unrelated to sediment-associated CoCs may have contributed to observed response. In reviewing the methodology, a hypothesis was developed that suspended particulates present in sediment elutriates of were affecting fertilization results through interference with either the mobility of the sperm or penetrability of the egg.

To test this hypothesis, elutriates were prepared using sediment from central Long Island Sound (LIS) using standard methods. Chemical analysis has indicated the non-toxic nature of the LIS sediment and has been successfully employed as a performance control in toxicological determinations for a variety of laboratory marine

organisms (i.e. *Mysidopsis bahia*, *Ampelisca abdita*, *Leptocheirus plumulosus*, *Eohaustorius estuarius*, *Rhepoxynius abronius*) confirming the non-toxic nature of the LIS sediment. The elutriate sample was split and a portion was centrifuged until all visible material was sedimented in the bottom of the tube. The supernatant was aspirated and used as dilution water for testing.

A standard dilution series (i.e., 0, 6.25, 12.5, 25, 50, and 100%) was conducted where 0% represented the centrifuged sample and 100% represented the original elutriate preparation. In addition, suspended solids determinations were performed on centrifuged and non-centrifuged sub-samples according to the methods provided in *Standard Methods for the Examination of Water and Wastewater* (APHA, 1992).

Test results are presented graphically in Figure 5.2-2. Regression analysis indicated that the sea urchin fertilization success declined significantly as suspended particulate concentration increased ( $y = -6.1x + 99.5$ ;  $p < 0.05$ ). The correlation coefficient ( $r = -0.93$ ) indicated a strong negative linear relationship. Thus, the presence of suspended particulates in elutriates of sediments at concentrations above ~ 4 mg/L can cause apparent toxicity in the sea urchin fertilization test even when contaminants are absent, and may have additive effects when toxicity is contaminant-induced. These findings imply that, while the sea urchin fertilization test has proven to be a valuable tool in assessing sediment porewater toxicity, it may be more problematic for testing elutriates by standard methods where unsettled particles may remain in the water column, since it would be difficult to separate the CoC-induced toxicity from the suspended solids effect.

In summary, the combined results of the Phase III toxicity tests indicate greatest concurrence among all three endpoints at Stations NSB-2, NSB-4 and NSB-5, suggesting continued and/or elevated toxicity at these stations. Station MCL-12 and, to a lesser extent Stations MCL-10 and NSB-3, suggest possible increased toxicity based

on two of the three endpoints. These three stations are within the expected range of seasonal scour due to winter storms (discussed in Section 3). The remaining stations suggest little evidence of increased toxicity as a result of the erosion event.

### 5.3. BIOLOGICAL FIELD INVESTIGATIONS

#### 5.3.1. Infaunal Distribution and Abundance

Benthic organisms at sites adjacent to the McAllister Point Landfill were sampled and identified in order to detect existing environmental stresses and to provide information on the biology of the area. Several programs have sought to develop a system of biotic condition indicators sensitive to the biological integrity of sites (Messer, 1990). For example, a benthic index, developed by the EMAP-Estuaries Virginian Province Demonstration Program (Schimmel *et al.*, 1994), makes use of the mean number of infaunal species per grab, biomass/abundance ratio for all species, and mean abundance of opportunistic pollution-tolerant species to discriminate between degraded and reference sites. Although the identification of opportunistic species has been the subject of debate, there is agreement that some capitellid and spionid polychaetes belong in this category.

*Methods.* Sample locations for benthic community structure determinations are shown in Figure 3.1-2. Different procedures were used to study intertidal and subtidal areas. At intertidal stations (NSB-1 to NSB-7), single samples were taken in sediment with and without embedded clusters of blue mussels (*Mytilus edulis*) between March 28 and April 3, 1995. These were termed "mussel bed" and "sediment", respectively. Sediment was removed from circular areas of 240 cm<sup>2</sup> with a trowel since it was not possible to penetrate the shingle beach with a core tube. Embedded mussels were not found at Station NSB-7; a cluster of mussels was instead taken from a rock surface at

this location on July 20, 1995. In addition, intertidal reference samples were not taken because of the difficulty of matching the unique sub-environments found within the disposal area. At subtidal stations, duplicate samples were obtained on April 26, 1995, with a Smith-McIntyre grab, modified to take a 500 cm<sup>2</sup> sample. Subtidal reference samples were obtained at three depths in Cranston Cove, Jamestown, Rhode Island.

Samples were sieved to 0.5 mm, preserved, and invertebrates removed and identified to species where possible. Notes were made of the size distribution of key species. Organisms were archived for possible additional taxonomic determination and population analysis. Counts of organisms recovered are given in Appendix B-4. Damaged or immature specimens which were difficult to identify were entered as known species or combined in more inclusive taxa to simplify interpretation of changes in species number.

#### 5.3.1.1 Intertidal Habitat Results

The small number of samples and variability of the habitat did not allow statistical comparisons among stations. The following observations are based on inspection of the data.

*Mussel beds versus sediment.* Counts of the 20 most abundant species in mussel bed and sediment samples are shown in Table 5.3-1. Complete data are provided in Appendix B-4. In both mussel and sediment sub-habitats, oligochaetes (*Oligochaeta spp.* and *Pelosclex benedeni*) were the overwhelming numerical dominants. Juvenile *Mytilus edulis* (blue mussels) were much more abundant among mussel clump samples than in the co-located sediment samples. Other species found to be relatively more abundant in mussel samples included sessile epifauna (e.g., anemone); tube-dwellers (*Corophium: amphipoda*); and more motile epifauna (*Littorina: periwinkles*, *Hyale plumulosa: beach hopper amphipod*). An abundance of small

*Nemertinea* were observed and presumed to be mussel predators. The herbivorous gastropod, *Lacuna vincta*, was found associated with macroalgae.

The fauna found within the NSB-7 mussel cluster (sampled July 1995) differed from embedded mussel bed samples taken from Stations NSB-1 through NSB-6 (sampled in March/April, 1995). Station NSB-7 had lower numbers of oligochaetes and nemertines, and higher numbers of the polychaete *Polydora cornuta* and the amphipod *Melita nitida*. It cannot be determined whether these differences are related to CoCs from McAllister Point Landfill, or instead related to habitat and seasonal differences.

In "sediment" samples, epifauna were less important, and the majority of species were infaunal polychaetes. Other infauna, such as the oligochaete *Peloscolex benedeni*, also had lower mean densities in sediment vs. mussel habitat (127 versus 614 per sample, respectively). A few numerically important taxa which were equally abundant in both habitat types included *Oligochaeta* spp., and the polychaetes *Fabricia sabella*, *Harmothoe* spp., *Neanthes succinea*, *Streblospio benedicti*. Species numbers in mussel bed samples (24/station) were higher than in sediment samples (17/station) (Table 5.3.1).

*Effect of slope.* The shore between Stations NSB-1 and NSB-4 is steep with a narrow intertidal zone and no obvious fresh water discharge. In contrast, the shoreline between Stations NSB-5 and NSB-7 has a low slope, areas of standing water and fresh water discharge. For mussel samples, there was no change in epifaunal species number or abundance between high and low areas. For both mussel and sediment samples, however, the change from high to low slope (Stations NSB-4 to NSB-5) was correlated with increased density of several infaunal species (*Marenzelleria viridis*, *N. succinea*, *Pygospio elegans*, *S. benedicti*, *Mya arenaria*, and *Mercenaria mercenaria*).

### 5.3.1.2 Subtidal Habitat Results

Table 5.3-2 presents counts for the 20 most abundant species found in subtidal samples at the McAllister Point Landfill study area (MCL stations) and the Jamestown Cranston Cove reference area (JCC stations). Counts for all species are given in Appendix B-4. An average of 54.4 species per sample were found at the MCL stations versus 42 species per sample at the reference stations. Polychaetes were the most diverse major taxa, but both mollusks and crustacea were well represented.

*Dominants.* A variety of life forms were found among numerical dominants at the MCL subtidal stations. These include a small deposit-feeding polychaetes (*Mediomastus ambiseta*, *Montocellina baptisteeae*, *Tharyx acutus*, and *Aricidea catherinae*), as well as small deposit-feeding *Oligochaeta spp.* Suspension-feeding bivalve mollusks, including juvenile blue mussels (*M. edulis*), were found at all stations. High densities of the little black mussel (*Musculus niger*) were found at Station MCL-9. Other abundant species were *Polydora caulleryi*, a surface-feeding, tube-dwelling polychaete; *Microdeutopus anomalus*, an epifaunal, suspension-feeding amphipod crustacean; *Tellina agilis*, a surface deposit-feeding bivalve; *Harmothoe spp.*, a motile epifaunal predatory polychaete; and *Leptocheirus pinguis*, a suspension-feeding amphipod which occupies "U"-shaped burrows.

Divers and bullrakers were used to successfully collect *Mercenaria mercenaria* from Stations MCL-10, MCL-12, MCL-13, and MCL-16. *P. morrhua* was sampled at Stations MCL-12, MCL-13, and MCL-16. Other large species which were observed in grab samples included the polychaetes *Ninoe nigripes*, *Arabella ornata*, *Cirriformia grandis*, *Heteromastus filiformis* and *Glycera americana*. Adult hard clams (*Mercenaria mercenaria*) and another bivalve (*Pitar morrhua*) are probably the macroinvertebrate biomass dominants in this area, but were not quantified as they were not sufficiently dense to be collected during microinvertebrate sampling.

*Species Diversity.* The number of species per sample (39–67) found at subtidal stations near McAllister Point landfill, was high relative to previous subtidal surveys in lower Narragansett Bay (French *et al.*, 1992), in that the MCL stations included many more motile and sessile epifaunal species (Table 5.3-2). However, the habitat samples described by French *et al.* (1992) included soft-bottom stations which lacked the pebble and shell material found to dominate at MCL stations. It would appear that the variability of sediment textures, as well as the availability of hard substrate at MCL stations provided a more diverse habitat, thus promoting coexistence of a large variety of species, although many were represented by few individuals (in the ten MCL samples taken at five subtidal stations, twenty-six species were represented by only one individual).

Species diversity in lower Narragansett Bay can be related to the presence of estuarine and shelf fauna (e.g., offshore). In the present study, primarily shelf species were found at MCL stations, an observation attributed to the more stable environmental conditions of open coastline and deeper waters, (e.g., reduced seasonal variation in temperature, low salinity variation, little wave effect, and reduced suspended sediment loads) which are not conducive to the more opportunistic estuarine species. In contrast, benthic assemblages in relatively deep water off Coasters Harbor were found to have high species number (City of Newport, 1985; French *et al.*, 1992), but were composed of primarily estuarine species.

*Numerical abundance.* Inspection of numerical abundance data (Table 5.3-2 and Appendix B-4) indicate that while some species (e.g., *M. ambiseta*, *M. baptisteeae*, *M. edulis*, *T. agilis*) were found in comparable abundance along the coast adjacent to McAllister Point Landfill (i.e., a transect from Station MCL-9 through MCL-14), other species were more abundant at northern stations (MCL-9 through MCL-12; *M. niger*, *P. morrhuanus*, and the amphipods *Ampelisca vadorum*, *Corophium acutum* and *Paracaprella tenuis*) versus the southern station (MCL-14). A third group of

species (e.g., *Oligochaeta* spp., *H. filiformis*, and *C. capitella*) was more abundant at Stations MCL-12 and MCL-14.

The greater number of shells and pebbles found at Stations MCL-9 through MCL-12 would be favorable to shelf epifauna, such as *M. niger*, *C. acutum*, and *P. tenuis*, partly explaining reduced abundances at Stations MCL-13 and MCL-14, where this substrate was lacking and finer-grained sediments prevailed. The increased number of estuarine species observed at Stations MCL-12 and MCL-14, including deposit-feeding polychaetes (e.g., *H. filiformis* and *C. capitella*) and oligochaetes, is probably due to the fact that these stations having a higher proportion of fine sediments.

*Reference stations.* Counts for reference stations (JCC-S1, JCC-M1, and JCC-D1) are reported in Appendix B-4. An average of 42 species per sample were found at JCC stations, the same as at Station MCL-14, but less than the average of all MCL stations (54.4). The shallower reference stations (JCC-S1 and JCC-M1) have shells and pebbles on the sediment surface similar to Stations MCL-9 through MCL-12, and the type of species found at these stations was generally similar to those found at MCL stations. In general, however, both species numbers and densities were reduced at shallow reference stations compared to MCL stations.

The deep reference station (JCC-D1) differed considerably from all MCL stations in both substrate and fauna. Here, the sediment consisted of fine sand without shells and pebbles, and the dominant organisms were juvenile amphipods (*Leptocheirus pinguis*). Sub-dominants included the polychaetes *Nephtys incisa* and *Ninoe nigripes*; the small amphipod *Phloe pollex*; and the deposit-feeding bivalves; *Nucula annulata* and *N. delphinodonta*. These species are commonly found on silt and sand substrates within the middle and lower portions of Narragansett Bay (French *et al.*, 1992). This substrate is not represented at McAllister Point. The initial study of Cranston Cove

reference stations (TRC, 1994) also found changes in substrate and faunal composition between shallow and deep locations which were very similar to the trends reported here.

### 5.3.2. Bivalve Condition Indices

The health of biota including bivalves and fish under varying environmental conditions has frequently been assessed through measures of growth rate, condition index and survival rate (Brown and Hartwick, 1988). Condition indices based on allometric relationships were developed primarily for detection of the ecophysiological status of animals in an aquaculture setting (e.g., Lucas and Benninger, 1985), but have received expanded use in water quality monitoring programs (Lawrence and Scott, 1982). Mann (1978) and Lucas and Benninger (1985) recommended the use of the dry tissue weight to dry shell weight ratio index, where low index values reflect energy deficits resulting from environmental stress or loss of gametes. Another condition index, dry tissue weight to shell volume (calculated from length) ratio, has been used, where the proportion of internal shell body occupied by tissue relative to shell size reflects the status (related to fitness) of bivalve metabolic reserves (Brown and Hartwick, 1988). In addition, the ratio of shell weight to shell length is useful as an indicator of shell thickness. Enhanced shell thickness is interpreted as an indication of stunted shell growth due to crowding or other environmental influences.

Three bivalve condition indices, including dry tissue weight to length, dry tissue weight to shell weight, and shell weight to length ratios, as well as fish weight to length ratios, were calculated from samples collected during August 1995 (raw data presented in Appendix B-3). The database includes two bivalve species; blue mussels (*Mytilus edulis*) for intertidal stations and hard shell clams (*Mercenaria mercenaria*) at the offshore, subtidal stations. The sampling strategy for mussels included a randomized design, whereby four 1-m<sup>2</sup> quadrants were sampled quantitatively for the occurrence of

the target species. Due to the natural variability in bivalve distributions, it was common for the same species to be found at only 2-3 of the four replicates from each station. Sample analyses for hard shell clams were based on composites of multiple bull-rake retrievals.

Sample analyses for condition indices involved the selection of five to seven individuals from each replicate such that the full size range was adequately characterized. The mean of CI values derived from each replicate was calculated and used as the CI index for the station. Standard deviations about the mean were calculated where possible.

*Results.* The condition indices for the two bivalve species are presented in Figure 5.3-1. No statistically significant differences ( $P_F < 0.05$ ) in the indices for blue mussels were observed (Figure 5.3-1.A). Because mussels were not available at the Cranston Cove reference site, and given that sites NSB-8 through NSB-10 are remote to potential impacts of the landfill, these stations were evaluated as a point of reference. Tissue weight/length ratios appear reduced at Stations NSB-2 and NSB-7 relative to NSB-8 through NSB-10. Additionally, the shell weight to length index were reduced slightly at NSB-2 and NSB-5 to NSB-7, when compared to NSB-8 through NSB-10. The tissue weight to shell weight index was more variable within station, and not statistically different (Stations NSB-5 and NSB-3 were different at  $P_F = 0.10$ ).

For hard clams, no significant differences in condition indices were observed, although the data suggest somewhat elevated shell thickness at sites MCL-12 and MCL-13 relative to stations north (MCL-10) and south (MCL-16) of this area.

Limited fish samples were available for calculation of the indices, such that statistical evaluation of the data were not possible. No apparent trends in fish weight to length ratios were observed among three stations at which fish were obtained.

Interpretation of adverse impact based on CI data must be considered carefully in context with contaminant exposure concentrations, as well as other potentially confounding factors, such as temperature, food supply, and substrate type. For example, it is difficult to discern differences based on reference data, because of potential differences in water temperature and/or food supply. However, these factors are perhaps less important given that such water quality gradients (exclusive of anthropogenic factors) on small spatial scales (20-50 m) are unlikely to exist. Substrate type can be an important factor although the rocky intertidal habitat was fairly comparable over the distributional range of stations. The results must therefore be interpreted in conjunction with other exposure and effects indicators to determine if these data support or contradict the prevailing weight of evidence; such interpretation is presented in the risk analysis (Section 6).

### 5.3.3. Fecal Pollution Indicators in Mussels

Bivalve tissue collected in the McAllister Point study area were analyzed for fecal pollution indicator bacteria (raw data presented in Appendix B-2). Fecal pollution indicator data were used to assess the sanitary quality of the marine environment, which might adversely impact growth, as well as potential CoC sources in the waste stream, as described in Section 4.2.4.

Fecal pollution indicator densities in bivalve tissues are presented in Table 5.3-3. Mussel specimens were collected at NSB stations, while hard clams were collected at MCL stations. Indicator densities for bivalve tissues collected from Station NSB-5 were low for total coliforms, fecal coliforms, and fecal streptococci. Although the levels of the aforementioned indicators were low, *Clostridium perfringens* densities showed an elevated density of 2400 CFU/100 g. Occurrence of elevated *C. perfringens* in absence of other indicators is suggestive of historic fecal pollution to this site.

In contrast, indicator densities for mussel samples collected from Stations NSB-1, NSB-3 and NSB-7 were somewhat elevated for total coliforms, fecal coliforms, and fecal streptococci relative to NSB-5, indicating a fresh source of fecal pollution to these areas. In addition, markedly elevated levels (3500 - 5400 CFU/100 g) of *Clostridium perfringens* were observed at these stations, suggesting that these areas presently and historically have been exposed to fecal pollution. Data from MCL-13 suggests recent contamination that has not been persistent historically. Finally, moderate indicator densities for the bivalve samples collected from Stations MCL-12, MCL-16 and JCC-D1 are characteristic of an area receiving low level of untreated or improperly treated fecal material.

All bivalve sample analyses demonstrated the presence of one or more of the fecal pollution indicators. The relative densities of indicators suggest new sources of fecal contamination to the study area, but decrease with proximity to Station NSB-5. Elevated sediment-associated *Clostridium* at offshore (MCL) sites relative to NSB sites is possibly due to these sites being more depositional in nature, allowing spores to accumulate, as demonstrated by finer grained sediments (Figure 4.2-1). This pattern also discounts the contribution of shore birds as a source of fecal pollution. Thus, higher bivalve *Clostridium* may be due to a species effect, since mussels were collected from NSB stations, while hard clams were collected from MCL stations.

Similarity among stations within each of the inshore and offshore zones also suggests that the historic sources of fecal pollution are regional, and possibly involve non-point sources as might occur due to landfill seeps. In addition, fecal pollution indicators near NSB-4 through NSB-6 are inversely related to the xenobiotic (human-made contaminant) distributions. Thus, the data suggest that there does exist a transport pathway for contaminants to the study site, but this mechanism cannot explain the apparent CoC distribution pattern.

#### 5.4. EXISTING TOXICITY-BASED CRITERIA AND STANDARDS

Toxicity-based criteria and standards provide the basis for comparing expected or actual environmental concentrations of contaminants to toxicological benchmark concentrations, thereby allowing an estimation or quantification of risk. For the present risk assessment, the primary benchmarks utilized were 1) ER-L/ER-M values presented in Long *et al.* (1995); 2) EPA Chronic Water Quality Criteria (WQC) for marine waters (used in this study to assess chemical concentrations in groundwater from the landfill and sediment porewater), and 3) EPA Sediment Quality Criteria (SQC), as presently available or as predicted from WQC and partitioning parameters (see Section 3.3).

*ER-L/ER-M values.* The ER-L and ER-M concentrations correspond to the lower 10<sup>th</sup> and 50<sup>th</sup> percentiles, respectively, of all concentrations of a contaminant observed to cause a biological effect, over a range of studies and species (Long and Morgan, 1990; Long *et al.*, 1995). Conceptually, ER-Ls are similar to LOELs (lowest observed effect levels), which represent the lowest toxicant concentration observed in bioassays to cause biological effects. Another type of benchmark, AET (Apparent Effects Threshold; PTI, 1988, U.S. EPA, 1989a) developed to address individual contaminants in field sediments, represents the level of individual chemicals above which statistically significant biological effects are always expected to occur. As demonstrated in Section 3, the ER-L values are typically more conservative (i.e., correspond to lower benchmark levels) than AET or SQC values, usually representing concentrations that are an order of magnitude lower. Consequently, ER-Ls are utilized primarily for this assessment, with ER-Ms used as an upper range benchmark.

*Water Quality Criteria.* The EPA Water Quality Criteria (WQC) were used to calculate screening benchmarks for landfill groundwater samples collected during the Phase II RI (TRC, 1994), and for porewater metals samples collected in Phase II of the present investigation. Water-based aquatic life criteria are based on the total

recoverable concentration of the metal as sampled from test chambers during aquatic toxicity tests. However, it is recognized that the dissolved metal concentration more closely approximates the bioavailable fraction of the metal in the water column than does the total extractable concentration. Lussier *et al.* (1995) addressed this issue through conduct of paired (unfed and fed) toxicity tests, and derived conversion factors between the dissolved phase effect concentration and the total recoverable concentration. For most metals (As, Cd, Cr<sup>6+</sup>, Pb, Ni, Se, and Zn), the conversion factor was 0.95 or greater, indicating the presence of food (which could alter metal bioavailability) had minimal effect on the derived Water Quality Criteria. One exception was Cu, where the conversion factor (0.83) indicates that the expected dissolved phase concentration is 83% of the promulgated criteria. The implication of these findings is that WQC provide protective prediction of dissolved (measured) metal concentrations in organically rich environments (e.g., porewater). For this evaluation, the Hazard Quotients were calculated as the ratio of the porewater concentration to the criterion level for individual analytes.

*Equilibrium Partitioning.* For non-ionic organic chemicals, the criteria concentration for sediment (SQC), was derived directly from the product of the Water Quality Criteria (WQC), Final Chronic Value (FCV), and the organic carbon partition coefficient ( $K_{oc}$ ) of the chemical. The primary assumption inherent in this approach is that the vector of chemical exposure (e.g., interstitial water versus sediment organic carbon) under equilibrium conditions is not an important determinant in chemical bioavailability. This is because the compound has reached a state of equal fugacity (chemical activity) among porewater, sediment, and biota matrices. Furthermore, the SQC approach assumes that the WQC FCV is the appropriate effects concentration for protection of benthic organisms (i.e., that the WQC FCV and the FCV derived for benthic species do not differ).

The Fourth Tier conceptual models presented in Figure 3.5-4 through Figure 3.5-7 present pathways for exposure by CoCs to receptors of concern. Inherent in these models are potential differences in the habitat or feeding mode of epibenthic and infaunal organisms, which could result in more limited exposure to non-ionic organic contaminants (e.g., epibenthic or filter-feeding organisms receive significantly less exposure and are more sensitive than infaunal deposit feeding species), as discussed above. However, recent research has found that non-ionic organic chemical exposures for various species both within and among differing habitat groups are similar (Tracey and Hansen, 1996). The present assessment also provided suitable data for demonstration of this relationship (Section 6.3).

## 5.5. UNCERTAINTY

Numerous assumptions concerning the applicability of 1) toxicity evaluations, 2) biological field investigations, and 3) particular benchmarks as criteria and standards to evaluate impacts to biota are made that bear upon the certainty of risk derived from these effects based measures.

*Toxicity evaluations.* The evaluation of ecological effects of contaminated sediments using toxicity tests is essential because chemical concentrations alone are not accurate predictors of biological effects. The principal advantage of the sediment toxicity testing approach is that they are performed in a manner comparable to WQC derivation exercises, (e.g., mortality or sublethal effects are observed), hence the data are directly comparable to these criteria. Uncertainties associated with toxicity testing conducted in the present study are that the responses may not be chemical-specific and the responses observed may not represent chronic effects. In addition, there is uncertainty in the comparability between the sediment test species and the water test species upon which the WQC are derived (EPA, 1989b). With regard to porewater and

elutriate tests, there is uncertainty as to the comparability of CoCs measured by the porewater extraction method versus the elutriate extraction method, and its relation to Phase I/II porewater toxicity results and Phase III elutriate toxicity results for *Arbacia* fertilization.

*Biological investigations.* Field survey approaches, such as the benthic community structure and condition endpoints measured in this study, have the advantage of providing assessments of *in situ* effects without significant sampling artifact. However, methods for analyses of the data, particularly for community structure, are not standardized and thus difficult to compare between studies. Large field sampling programs have attempted to develop reliable benthic community indices of impacts with limited success (Schimmel, 1994). Often, a large amount of field data is required, including both seasonal and spatial coverage, such that benthic impacts can be discerned. Additional uncertainty exists in the taxonomic identification of species, as well as their enumeration and relative sensitivity to various pollutants.

There exists a lack of historical data from quantitative benthic community studies of the Newport area before 1985. In the present study, similar sampling and sample preparation techniques were used as in three recent surveys in the area (City of Newport, 1985; French *et al.*, 1992; TRC, 1994), yet there is variation between each study in the level of identification of some taxa and differences in the identifications based on recent taxonomic research. Techniques are available to improve comparability between studies, such as combining species into higher taxa, or focusing on long-lived species and on seasonally stable parameters.

The number of samples which could be examined was limited by a large volume of coarse material and high densities of organisms; this habitat patchiness increased the within-station variability, and hence lowered the certainty with which between station differences could be discerned. Sources of natural variation (grain size, water content

of intertidal sediments, presence of mussels, depth, and tidal circulation) were also large, so as to again hinder discrimination between stations with regard to potential effects of contaminants. Possible indicators of stress were seen in samples close to contaminant sources in both intertidal and subtidal samples. Elevated sediment and residue concentrations of fecal pollution indicators were observed, and related to possible adverse exposure and effects; however, the true relationship (direct or indirect) is not well understood.

*Benchmarks.* As summarized in Section 5.4, the derivation of ER-L/ER-Ms is based on very conservative assumptions concerning use of the lower 10<sup>th</sup> and 50<sup>th</sup> percentiles, respectively, of all concentrations of a contaminant that have been observed to cause biological effects. In the derivation of screening criteria (Section 3.3), ER/L values in particular are lower by one order of magnitude for most parameters (including AET values, representing the only other effects-based benchmark that is commonly applied), and hence were most often the benchmark of lowest value for each CoC. The uncertainty is the level of conservatism that is appropriate to assess ecological risk. ER-Ls are used in this assessment to provide a protective evaluation; however, these benchmarks may be overly conservative as they do not account for site-specific factors that can mitigate (buffer) the responses of ecological systems to particular contaminants (e.g., TOC). However, the incorporation of toxicity data at various effects levels and for species from different phyla and trophic levels is an attempt to add another measure of realism to the final assessment. A further issue with the use of the ER-L/ER-Ms (or AETs) is the relatively limited list of chemicals for which values are available compared to the overall list of contaminants from the study. This uncertainty was partly addressed by comparisons against reference site concentrations. In other areas, "surrogate" benchmark values were used for similar compounds to allow evaluation of potential environmental impact. However this approach also suffers from being potentially overly conservative (as discussed in Section 3.3).

*Use of Surrogate/Indicator Species.* The species evaluated in this study, including hard shell clams, mussels, cunner, and two bioassay organisms (*Ampelisca* and *Arbacia*), as well as benthic community structure measurements are used as indicators of the assumed general response of the various communities within the study region. These species represent a variety of biological endpoints which have been shown to be sensitive to contaminant inputs and whose relationship to a particular habitat and community is well established. It also was important to maintain consistency in the use of the same species, for purposes of data comparability, among the various phases of the risk assessment. Nonetheless, the use of surrogate or indicator species is associated with some undefined level of uncertainty since one or a few species cannot duplicate precisely the response of the numerous species that comprise the various communities of the region.

**Toxicity**

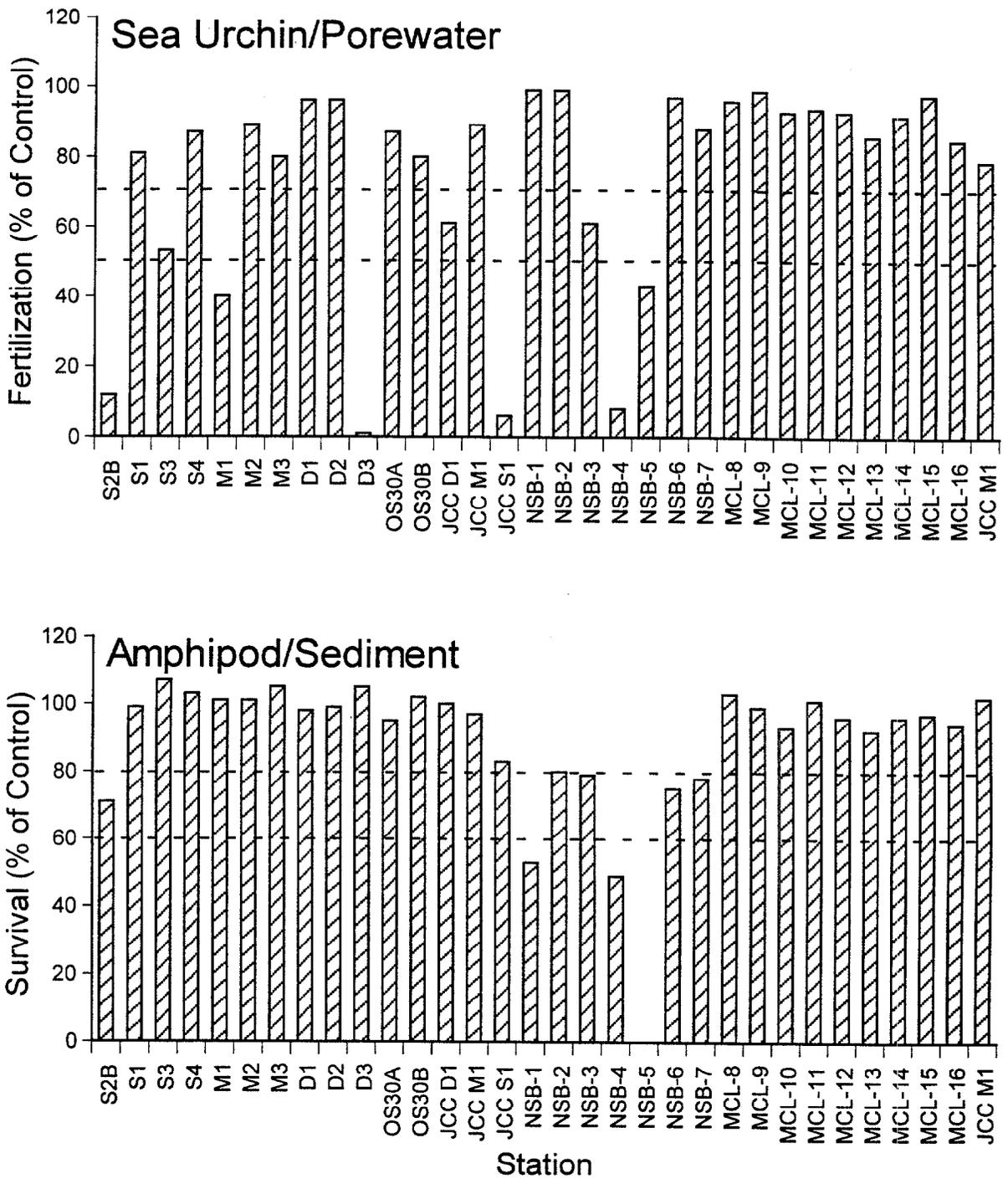


Figure 5.2-1. Sea urchin (*Arbacia*) fertilization and amphipod (*Ampelisca*) survival results for the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Dashed lines indicate the effects threshold values.

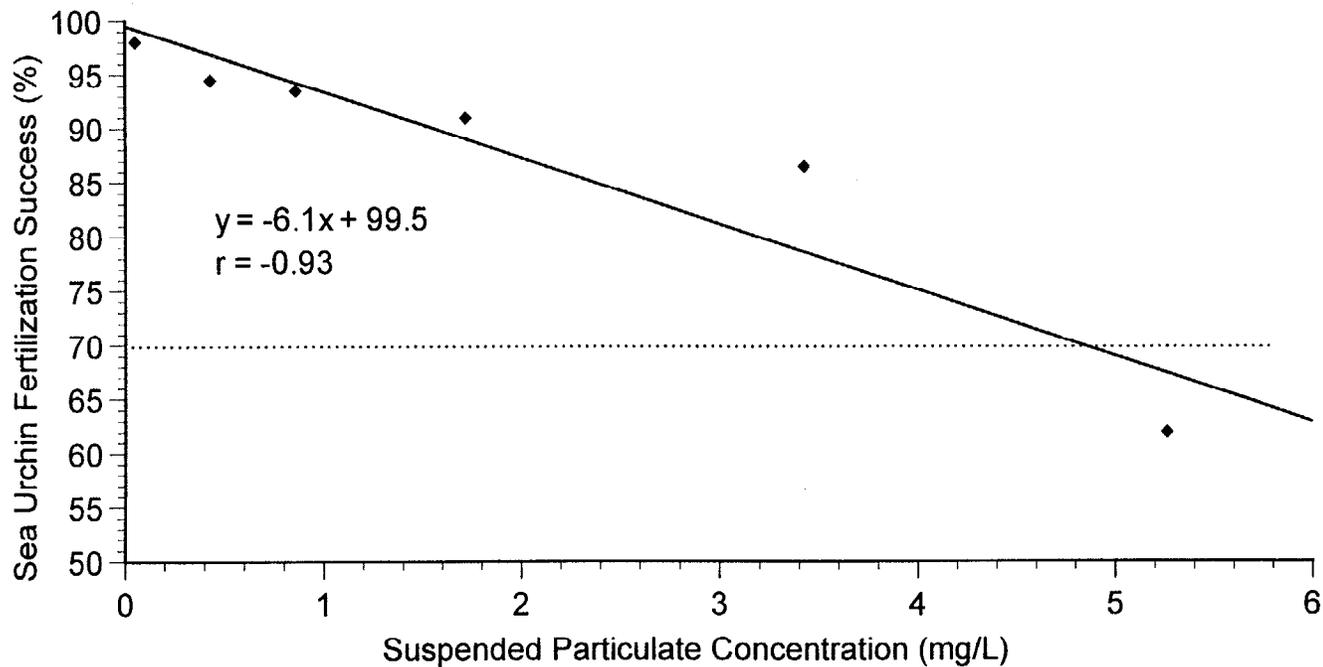


Figure 5.2-2. Sea urchin fertilization success (%) versus suspended particulate concentration (mg/L) in elutriates prepared from performance control sediments collected from the Central Long Island Sound control location. Dotted line indicates effects threshold value (i.e., 70% fertilization success).

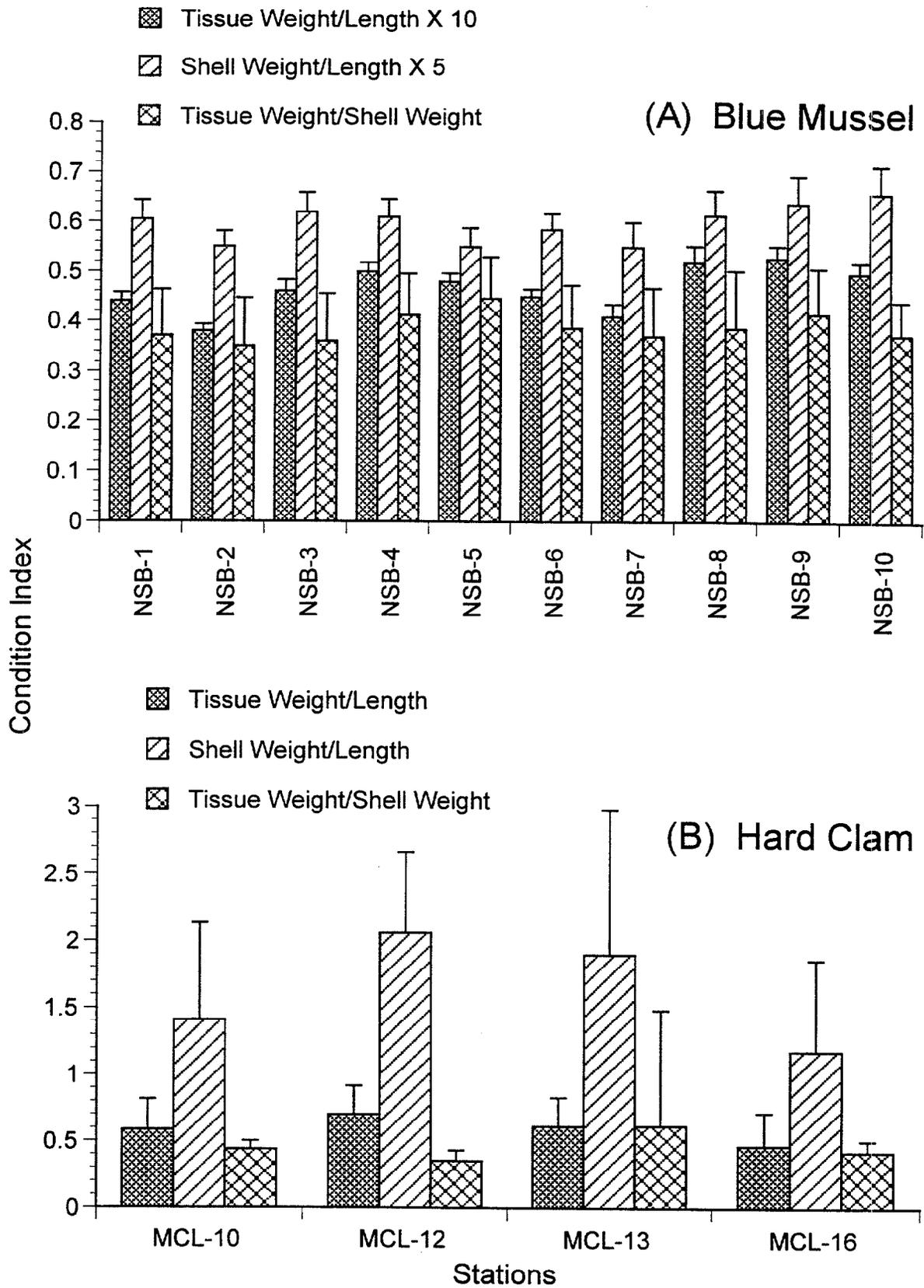


Figure 5.3-1. Condition Indices for (A) blue mussels and (B) hard clams for the McAllister Point Landfill Marine Ecological Risk Assessment. In (A), Y-axis scale is adjusted for clarity.

Table 5.2-1. Sediment toxicity results using the amphipod (*Ampelisca abdita*) 10-day bioassay survival test on surface sediments from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location.

Study	Station	Total Ammonia (mg/L)	Unionized Ammonia (mg/L)	% of Control	Comment
Phase I	S2B	25.9	0.15	71.3	*+
	M1	21.8	0.23	101	-
	OS-30A	14.5	0.24	95.4	-
	OS-30B	7.27	0.12	102	-
	D1	4.84	0.06	97.7	-
	D2	6.66	0.07	98.9	-
	D3	18.4	0.16	105	-
	M2	17.4	0.13	101	-
	M3	7.53	0.08	105	-
	S1	10.0	0.13	98.9	-
	S3	14.1	0.43	107	-
	S4	20.9	0.24	103	-
	JCC-D1	13.1	0.16	100	-
	JCC-M1	18.0	0.15	96.6	-
JCC-S1	32.8	0.54	82.8	*	
Phase II	NSB-1	7.56	0.16	52.6	***
	NSB-2	9.70	0.27	80.4	-
	NSB-3	12.7	0.34	79.4	*+
	NSB-4	7.05	0.20	49.0	***
	NSB-5	2.29	0.11	0.00	****
	NSB-6	6.48	0.14	75.3	*+
	NSB-7	2.48	0.10	78.4	*+
	MCL-8	0.35	0.00	103	-
	MCL-9	34.7	0.27	99.2	-
	MCL-10	38.1	0.77	92.6	-
	MCL-11	30.2	0.67	101	-
	MCL-12	36.4	0.86	96.1	-
	MCL-13	20.6	0.38	91.6	*
	MCL-14	16.9	0.25	95.8	-
	MCL-15	14.4	0.37	96.8	-
	MCL-16	31.4	0.48	93.7	-
JCC-M1	26.0	0.38	102	-	

"-" = no statistically significant reduction from control;

"\*" = sample survival was statistically lower than the performance control;

"\*+" = sample survival was both statistically lower than the performance control and between 60-80% of the performance control;

"\*\*\*" = sample survival was both statistically lower than the performance control and less than 60% of the performance control;

"\*\*\*\*" = sample survival was both statistically lower than the performance control and less than 10% of the performance control.

Table 5.2-2. Sediment porewater toxicity results using the sea urchin (*Arbacia punctulata*) fertilization index on surface sediments from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location.

Study	Station	Total Ammonia (mg/L)	Unionized Ammonia (mg/L)	% of Control	Comment
Phase I	S2B	25.9	0.15	12.1	*++
	M1	21.8	0.23	40.3	*++
	OS-30A	14.6	0.25	87.2	-
	OS-30B	7.27	0.12	80.2	*
	D1	4.84	0.06	95.7	-
	D2	6.66	0.07	95.7	-
	D3	18.4	0.16	1.07	*+++
	M2	17.4	0.14	89.3	*
	M3	7.53	0.08	80.2	*
	S1	10.0	0.13	80.7	*
	S3	14.1	0.44	53.1	*+
	S4	20.9	0.24	87.2	*
	JCC-D1	13.1	0.16	61.0	*+
	JCC-M1	18.0	0.15	88.8	-
JCC-S1	32.9	0.54	6.06	*+++	
Phase II	NSB-1	5.75	0.14	99.2	-
	NSB-2	2.32	0.05	98.7	-
	NSB-3	19.9	0.43	61.2	*+
	NSB-4	2.22	0.06	8.55	*+++
	NSB-5	1.16	0.04	43.1	*++
	NSB-6	4.43	0.09	96.9	-
	NSB-7	1.05	0.02	88.2	*
	MCL-8	4.16	0.05	96.3	-
	MCL-9	7.45	0.29	99.5	-
	MCL-10	5.15	0.06	93.2	-
	MCL-11	5.66	0.05	93.7	-
	MCL-12	6.89	0.07	92.7	*
	MCL-13	0.57	0.01	86.9	*
	MCL-14	0.49	0.01	91.6	*
	MCL-15	0.35	0.01	97.9	-
	MCL-16	0.46	0.01	85.9	*
	JCC-M1	6.84	0.08	79.3	*

"-" = no statistically significant reduction from control;

"\*" = sample fertilization was statistically lower than the performance control;

"\*+" = sample fertilization was both statistically lower than the performance control and between 50-70% of the performance control;

"\*++" = sample fertilization was both statistically lower than the performance control and less than 50% of the performance control;

"\*+++)" = sample fertilization was both statistically lower than the performance control and less than 10% of the performance control.

Table 5.2-3. Results of Phase III amphipod (*Ampelisca abdita*) survival tests with sediments collected from the McAllister Point Landfill study area.

Station <sup>1</sup>	Ammonia (mg/L) <sup>2</sup>		Amphipod Survival	
	Total	Unionized	(% Control) <sup>3</sup>	Flag <sup>4</sup>
NSB-1-R	0.01	0.00	90.5	-
NSB-2-R	0.66	0.02	14.7	***
NSB-3-R			NA	
NSB-4-R	0.00	0.00	24.2	***
NSB-5-R	0.54	0.02	36.8	***
NSB-6-R	1.06	0.06	90.5	-
NSB-7-R	0.89	0.03	63.2	***
S2B-R	4.66	0.28	97.8	-
S2C	6.88	0.54	92.3	-
M1-R	3.39	0.22	93.4	-
MCL-8-R	4.87	0.32	97.8	-
MCL-9-R	4.75	0.35	93.4	-
MCL-10-R	4.32	0.35	92.3	-
MCL-11-R	1.83	0.09	97.8	-
MCL-12-R	4.24	0.32	94.8	-
MCL-13-R	2.29	0.15	93.4	-
MCL-14-R	0.74	0.04	90.1	-

1 - "R" designation denotes location was resampled in Phase III.

2 - Ammonia measurements from overlying water column.

3 - Survival in Long Island Sound sediment used as control response for all treatments.

4 - Rankings for *Ampelisca* survival:

"-" = no statistically significant reduction in survival from control;

\*\*\* = survival statistically significantly lower than control;

"+" = \*\*\* and survival between 60-80% of control;

"++" = \*\*\* and survival less than 10-60% of control;

"+++" = \*\*\* and survival less than 10% of control.

Table 5.2-4. Results of Phase III sea urchin (*Arbacia punctulata*) fertilization tests in elutriates of sediments collected from the McAllister Point Landfill study area.

Station <sup>1</sup>	Ammonia (mg/L)		Fertilization Success (% Fertilized Eggs)				IC <sub>10</sub> <sup>3</sup> (%)	Flag <sup>4</sup>
	Total	Unionized	Elutriate Concentration (% Full Strength Elutriate)					
			0% <sup>2</sup>	10%	50%	100%		
NSB-2-R	0.51	0.02	98.7	95.7	8.0	6.7	13.6	***
NSB-3-R	0.21	0.01	98.7	95.3	43.0	5.0	16.1	***
NSB-4-R	0.30	0.00	98.7	97.3	57.7	10.3	21.4	***
NSB-5-R	0.02	0.00	98.7	98.0	8.0	8.3	16.1	***
NSB-6-R	0.36	0.01	98.7	96.3	85.3	84.7	36.2	***
MCL-10-R	3.30	0.04	98.7	94.3	65.3	11.0	17.5	***
MCL-12-R	4.70	0.05	98.7	95.3	7.3	8.0	13.3	***

1 -"R" designation denotes location was resampled.

2 - Control value for experiment, assumed for all treatments.

3 - Inhibition Concentration - 10% (concentration of elutriate causing 10% reduction in test response)

4 - Rankings for *Arbacia* successful fertilization:

"-" = no significant reduction in successful fertilization from control;

"\*" = one or more dilutions statistically < control;

\*+ = <70% elutriate concentration is toxic;

\*++ = <50% elutriate concentration is toxic;

\*+++ = <10% elutriate concentration is toxic.

Table 5.2-5. Results of Phase III sea urchin (*Arbacia punctulata*) larval development toxicity tests in elutriates of sediments collected from the McAllister Point Landfill study area.

Station <sup>1</sup>	Ammonia (mg/L)		Larval Development Success (% Normal Development)				IC <sub>10</sub> <sup>3</sup> (%)	Flag <sup>4</sup>
	Total	Un-ionized	Elutriate Concentration (% Full Strength Elutriate)					
			0% <sup>2</sup>	10%	50%	100%		
NSB-2-R	0.51	0.02	92.3	80.4	0.8	0.7	6.3	*+++
NSB-3-R	0.21	0.01	92.3	84.2	88.9	82.8	94.5	*
NSB-4-R	0.30	0.00	92.3	90.1	70.5	31.9	21.3	*++
NSB-5-R	0.02	0.00	92.3	86.5	4.9	0.9	11.0	*+++
NSB-6-R	0.36	0.01	92.3	91.7	89.8	86.8	>100	-
MCL-10-R	3.30	0.04	92.3	88.2	83.4	72.9	51.3	*+
MCL-12-R	4.70	0.05	92.3	84.4	70.6	58.5	12.2	*+++

1 -"R" designation denotes location was resampled.

2 - Control value for experiment, assumed for all treatments.

3 - Inhibition Concentration - 10% (concentration of elutriate causing 10% reduction in test response)

4 - Rankings for *Arbacia* successful larval development:

"-" = no significant reduction in normal development from control;

"\*" = one or more dilutions statistically < control;

\*+ = <70% elutriate concentration is toxic;

\*++ = <50% elutriate concentration is toxic;

\*+++ = <10% elutriate concentration is toxic.

Table 5.2-6. Comparison between Phase I/II (pre-erosion) and Phase III (post-erosion) amphipod (*Ampelisca abdita*) survival in bulk sediment and sea urchin (*Arbacia punctulata*) fertilization in 100% elutriates prepared from sediments collected in the McAllister Point Landfill study area, and Phase III sea urchin larval development results in elutriates prepared from sediments collected in the McAllister Point study area.

Station	Laboratory Toxicity <sup>A</sup>		
	Amphipod Survival <sup>1</sup>	Sea Urchin Fertilization <sup>2</sup>	Sea Urchin Development <sup>3</sup>
NSB-1-R	-		
NSB-2-R	***	****	****
NSB-3-R		****	-
NSB-4-R	***	***	***
NSB-5-R	***	****	***
NSB-6-R	-	-	-
NSB-7-R	*+		
S2B-R	-		
S2C	-		
M1-R	-		
MCL-8-R	-		
MCL-9-R	-		
MCL-10-R	-	***	*+
MCL-11-R	-		
MCL-12-R	-	****	***
MCL-13-R	-		
MCL-14-R	-		

A - Blank indicates no data available or not calculated.

Bordered cells indicate Relative Percent Difference (RPD) >30%.

Shaded cells indicate toxicity ranking increase in post-erosion sediments or elutriates.

1 - Amphipod survival rankings (% of control, see Table 5.2-3):

"-" = no significant reduction in survival from control;

"\*" = statistically significantly lower survival than control;

"\*+" = "\*" and survival between 60% and 80%; "\*\*\*\*" = "\*" and survival between 10% and 60%;

"\*\*\*\*" = "\*" and survival less than 10%.

2 - Sea urchin fertilization rankings (% fertilization success, see Table 5.2-4):

Phase I/II porewater fertilization test compared with Phase III elutriate fertilization test;

"-" = no significant reduction in successful fertilization from control;

"\*" = successful fertilization in 100% elutriate statistically significantly lower than control;

"\*+" = "\*" and successful fertilization between 50% and 70% in 100% elutriate;

"\*\*\*\*" = "\*" and successful fertilization between 10% and 50% in 100% elutriate;

"\*\*\*\*" = "\*" and successful fertilization less than 10% in 100% elutriate.

3 - Sea urchin larval development rankings (IC10, see Table 5.2-5):

no prior phase data for sea urchin development; RPDs not calculated;

"-" = no significant reduction in normal development from control;

"\*" = one or more dilutions statistically < control; "+" = <70% elutriate concentration is toxic;

"\*+" = <50% elutriate concentration is toxic; "\*\*\*\*" = <10% elutriate concentration is toxic.

Table 5.3-1. Intertidal benthic invertebrate community structure summary for the McAllister Point Landfill Ecological Risk Assessment. Density of the 20 most abundant species, number of species, and number of individuals are listed.

NSB STATION NUMBER	NUMBER OF INDIVIDUALS RECOVERED FROM 240CM <sup>2</sup> SAMPLES																	
	SEDIMENT								MUSSEL BED									
	1	2	3	4	5	6	7	sed	1	2	3	4	5	6	7	mussel		
SHORE PROFILE	steep slope.....				shallow slope.....				avg	steep slope.....				shallow slope.....				avg
ANTHOZOA																		
anemone sp.			1						2	1	10	33	5					
NEMERTINEA																		
Nemertinea 1	3	1	8	5					4	30	40	41	12	30	1			
Nemertinea 2		1							16	29	12	13	4	14	1			
PLATYHELMINTHES																		
Platyhelminthes sp.	55	9		14			20			7		3						
ANNELIDA																		
POLYCHAETA																		
Capitella capitata	1		14	15	101	11	1				11	1	7	2	1			
Fabricia sabella		3	13	2		1			7	4		3	77	5				
Harmothoe extenuata	1	3		1	1	11			6	4		3	11		5			
Marenzelleria viridis					9	25	12							2	1			
Neanthes succinea					6	26	12						13	1	51			
Polydora cornuta															93			
Pygospio elegans					81							1	4		1			
Spio sp.						45							5	3				
Streblospio benedicti					209	83							67	28	5			
Spionidae combined	1	1	1	7	299	164	12		0	0	0	1	76	100				
OLIGOCHAETA																		
Oligochaeta spp.	1948	3251	530	17	53	301	16		862	3914	153	1707	1102	264	126			
Pelosclex benedeni	52	242	73	217	182	122	3		9	577	526	1111	1007	1020	49			
MOLUSCA																		
GASTROPODA																		
Littorina littoria		7		5	4	1	13		19	25	10	58	21	33				
BIVALVIA																		
Mytilus edulis spat	48	43	25	73	176	73	2		857	861	441	534	233	194				
CRUSTACEA																		
AMPHIPODA																		
Corophium acutum		2	5	2		2			12	2	3	19	17					
Hyale plumulosa			1			1			4			107	198	6	27			
Melita nitida						6							1	1	74			
Total species	8	11	9	11	11	15	9	10.6	23	22	17	30	32	27	25	25.1		
Total individuals	2109	3563	670	358	1121	872	91	1255	1848	5484	1222	3668	2821	1655	510	2458		

Table 5.3-2. Subtidal benthic invertebrate community structure summary for the McAllister Point Landfill Ecological Risk Assessment. Density of the 20 most abundant species, number of species, and number of individuals are listed.

STATION NUMBER SAMPLE	NUMBER OF INDIVIDUALS RECOVERED FROM 0.05M GRAB SAMPLES										MCL avg	JCC Stations Jamestown Cranston Cove						JCC avg
	MCL Stations McAllister Point											S1.1	S1.2	M1.1	M1.2	D1.1	D1.2	
	9.1	9.2	10.1	10.2	11.1	11.2	12.1	12.2	14.1	14.2								
<b>POLYCHAETA</b>																		
Aricidea catherinae	5		2	1	2		2	5	161	116				72	134	1	2	
Capitella capitata	0	0	1	0	7	6	3	3	6	0		2	10	1	0	0	1	
Exogone sp.	8	4	10		9	9	6	8	7	5		1		1				
Harmothoe extenuata	26	55	40	42	70	26	65	32	8	4		74	9	14	6		1	
Heteromastus filiformis					1	3	15	13	5	5				2				
Lumbrineris tenuis	49	40		18	27	30			2	1				2				
Macroclymene zonalis	31	4	7	39	6	4		1	6	11			1	18		2		
Mediomastus ambiseta	138	279	71	290	423	515	194	273	369	110		39	107	215	214	77	204	
Montocellina baptistae	105	112	107	21	107	122	37	2						11	4	3	1	
Ninoe nigripes	20	10	2	8		1			4	7				1	1	37	27	
Polycirrus medusa	10	37	13	18	13	9	32	15				7	3		2	1	2	
Polydora caulleryi	36	18	26	145	85	105	59	103	91	39		36	15	43	30	1	3	
Prionospio heterobranchia	51	24	1	22	7	12	2	11	48	11		5	6	135	80		4	
Spionidae (combined)	94	65	39	190	126	158	73	139	147	74		42	23	203	177	3	8	
Spiophanes bombyx		5	8	7	19	20	2	7	6	21				18	41			
Streblospio benedicti					5	2	1	8							6			
Tharyx acutus	4	26	1	15	89	36	255	6		5		33	56	7	3	1		
<b>OLIGOCHAETA</b>																		
Oligochaeta spp.	95	45	30	66	204	253	629	492	67	31		120	139	63	27	17	17	
<b>MOLLUSCA</b>																		
<b>GASTROPODA</b>																		
Turbonilla interrupta	1		1		4	26	10	11	3	39		3	2	2	7	4		
<b>BIVALVIA</b>																		
Musculus niger	303	26		3		1												
Mytilus edulis	52	98	690	60	97	28	65	26		29		2	1	2		2		
Pitar morrhuanus	9	2	3	3	5					1					1	3	3	
Tellina agilis	16	19	37	23	42	36	28	51	21	28		13	9	64	80	3	1	
<b>CRUSTACEA</b>																		
Ampelica vadorum/abdita	15	5	2	1	3	1	1											
Corophium acutum	5	2	38	8	16		3			1		1	1	3	17			
Leptocheirus pinguis	67	32		13	2	30		6	26	14		1		12	27	720	808	
Microdeutopus anomalus	14	39	59	14	65	18	59	37	3	3		41	3	3				
Paracaprella tenuis	2	3	7	1	1	2				2								
Total species	65	63	57	59	66	54	46	50	39	45	54.4	40	39	49	45	39	40	
Total individuals	1183	997	1300	933	1438	1378	1580	1220	869	528	1143	498	491	775	756	1121	1287	

Table 5.3-3. Fecal pollution indicator concentrations in bivalve tissue collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference locations.<sup>1</sup>

Station	Total Coliforms CFU <sup>2</sup> /100 g		Fecal Coliforms CFU <sup>2</sup> /100 g		Fecal Streptococci CFU <sup>2</sup> /100 g		<i>Clostridium perfringens</i> CFU <sup>2</sup> /100 g		Overall Ranking <sup>3</sup>
NSB-1	490	++	45	-	110	+	3500	+++	+++
NSB-3	170	+	20	-	230	+	3500	+++	++
NSB-5	68	-	<18	-	78	-	2400	+++	++
NSB-7	330	+	170	+	110	+	5400	+++	++
MCL-11	330	+	330	+	<18	-	230	+	+
MCL-12	2400	+++	2400	+++	45	-	230	+	+++
MCL-16	330	+	330	+	<18	-	230	+	+
JCC-D1	330	+	330	+	<18	-	230	+	+

1 - Indicator-specific rankings: "-" = <100 CFU/100 g; "+" = 100-350 CFU/100 g (low); "++" = >350 CFU/100 g (intermediate); "+++ = >1000 CFU/100 g (high).

2 - CFU = Colony forming units

3 - Overall Ranking: "+++ = intermediate (++) or higher effect observed for two or more indicators, one of which indicates high (+++) effect; "++ = intermediate (++) effect observed for two or more indicators or high (+++) effect for one indicator; "+" = low (+) effect observed for two or more indicators or intermediate (++) effect for one indicator; "-" = low (+) effect observed for only one indicator or no effect for all indicators. See text in Section 6.0-2.

## 6.0. RISK CHARACTERIZATION

Risk Characterization phase for the McAllister Point ERA includes the evaluation of Exposure Assessment and Effects Assessment Weights of Evidence (WoE). The five principal WoE of Exposure Assessment (presented in Sections 6.1 and 6.2) include:

- Comparisons of sediment concentrations with ER-L and ER-M criteria;
- Comparisons of porewater concentrations with Water Quality Criteria;
- Assessment of divalent metal (SEM) bioavailability;
- Sediment fecal pollution indicator concentrations; and
- Evaluation of site tissue CoC concentrations relative to reference locations (Tissue Concentration Ratios).

The corresponding WoE for Effects Assessment (presented in Sections 6.3 to Section 6.5) are:

- Analysis of CoC bioaccumulation in fish, bivalves and lobster, and related potential impacts due to ingestion of these aquatic biota by avian predators;
- Evaluation of toxicity and comparison of these results with CoC sediment and porewater concentrations; and
- Analysis of CoC concentration versus effects measurements.

Each WoE also has multiple supporting indicators, such as analyte-specific Hazard Quotients for sediments and porewater, TCR values for each of the aquatic receptors (mussels, clams, lobster, fish), amphipod and sea urchin toxicity, etc.. These indicators are intended to increase the certainty of the assessment with regard to the

presumption of adverse exposure or effects conditions. The individual indicators within each Weight of Evidence (WoE) were interpreted and summarized using semi-quantitative ranking schemes so as to allow the synthesis of the overall probability of adverse Exposure/Effects (E/E) indicated for each of the primary weight of evidence (discussed in Section 6.01). As an additional step in the synthesis of exposure and effects WoE for characterization of risk in the study area (Section 6.6), station data from the four sampling events (TRC/BOS, Phase I, Phase II and Phase III) were grouped into Ecological Exposure Zones (EEZs) as discussed in Section 6.02. A final, but critical element of the risk characterization is an analysis of uncertainties associated with the above interpretations (Section 6.7).

#### 6.0.1. Characterization of Adverse Exposure/Effects

The individual indicators within each Weight of Evidence (WoE) were interpreted and summarized using semi-quantitative ranking schemes so as to allow the synthesis of the overall probability of adverse Exposure/Effects (E/E) indicated for each of the primary weight of evidence. Comparability of ranking strategies for the synthesis of indicators within the various WoE was deemed necessary in order to provide a consistent evaluation of exposure/effects (E/E) data. Thus, for the majority of WoE, the quantity and nature of indicator data permitted the development and application of the following E/E ranking strategy, as follows:

<i>Baseline Adverse E/E Probability (-):</i>	Baseline (-) ranking for all indicators, or low (+) ranking observed for only one indicator;
<i>Low Adverse E/E Probability (+):</i>	Low (+) ranking observed for two or more indicators, or intermediate (++) ranking for only one indicator;

<i>Intermediate Adverse E/E Probability (++):</i>	Intermediate (++) ranking observed for two or more indicators, or high (+++) ranking for only one indicator;
<i>High Adverse E/E Probability (+++):</i>	High (+++) ranking observed for two or more indicators.

For the two WoE where this approach was not deemed appropriate (SEM bioavailability and benthic community data), the overall rank was taken as the maximum the indicator specific values.

The above ranking strategy is intended to characterize the extent and pervasiveness of CoC-related exposure or effects. For the exposure WoE, for example, the extent to which CoC concentrations in various matrices (sediment, porewater, tissue) exceed benchmarks and how often this exposure/effect was observed among the individual WoEs. The above rankings for exposure-based WoE do not consider exposure-response relationships; this information is incorporated into the effects-based WoE evaluation. In addition, this type of ranking scheme is intended only as a qualitative tool. The ranking approach is based on best professional judgement, since the "true" ecological risk of, for example, benchmark exceedence or observed toxicity, is not presently known. Hence, the risk manager is encouraged to keep in mind the nature of the risk ranking approach when evaluating the general outcome of the risk assessment.

#### 6.0.2. Ecological Exposure Zones

As an additional step in the summarization of exposure and effects WoE, Ecological Exposure Zones (EEZs) were delineated based on an understanding of the general hydrographic, bathymetric and habitat characteristics of the area, as well as trends in spatial distribution and composition of contaminants found in sediments and

tissues, the distribution of effects, and the proximity among sampling stations in the study area. Figure 6.0-1 shows the eight EEZs that have been identified for the McAllister Point ERA, containing stations from four sampling events (TRC/BOS, Phase I, Phase II and Phase III investigations). The CoC-related characteristics of each zone are discussed in the appropriate WoE section. A brief description of the natural characteristics of these zones is included below:

*Zone 1: Landfill Intertidal North.* A steep-sloping intertidal, this zone includes Station NSB-1. This station was selected to characterize the northern extent of intertidal environment assessed for potential landfill-related impacts. This zone contains habitat for primarily epifaunal macroinvertebrate communities and blue mussels growing on and between large boulders. Small fish, including cunner, are likely to occupy the habitat. Shore birds including the herring gull and great blue heron may feed upon the epibenthic communities and fish located in this area. This zone faces west-northwest and is the most highly exposed zone in the study area to both winter storm conditions and swell resulting from summer sea breezes. Gomes Brook drains into Narragansett Bay to the north of this zone. The substrate is rocky/sandy sediment as typical of a high energy intertidal environment. Some sediment erosion was observed toward the southern end of this zone as assessed during the Phase III investigation.

*Zone 2: Landfill Intertidal Middle.* This west/southwest-facing zone includes the intertidal habitat Stations NSB-2, NSB-3, NSB-4, and NSB-5. This area has a habitat generally comparable to that of Zone 1. This zone also has the greatest degree of visible refuse and sediment staining. This area represents the furthest point of extension of the landfill into Narragansett Bay. The substrate is rocky/sandy sediment as typical of a high energy intertidal environment. As in Zone 1, this region provides habitat for mussels and small fish which may be consumed by shore birds such as the

gull or heron. This zone was also the region where, prior to the Phase III investigation, the greatest degree of sediment erosion was observed to have occurred.

*Zone 3: Landfill Intertidal South.* This zone includes Stations NSB-6 and NSB-7, and comprises a shallow-sloping rock/pebble beach environment with relatively low surface relief, lacking larger rocks found in zones to the north. This zone is southwest-facing and is moderately exposed to wave action during summer sea breeze conditions but is shielded by the landfill from northeasterly storms, a condition which has allowed the development of a sand/pebble beach which may vary in extent depending upon seasonal cycles of sand migration. The area appears influenced by creek drainage from a culvert located southeast of NSB-7. Sparse eel grass has been observed to the southwest of NSB-7. These stations were selected to characterize the southern extent of the intertidal environment and associated potential for landfill-related impacts. As with the northern and middle intertidal zones, this region provides habitat for macrobenthos, mussels and small fish which may be consumed by shore birds.

*Zone 3A.* This isolated area is located approximately 50 meters offshore of Zone 3, and includes Stations S2B, S2C, M1, MCL-12, and TRC Station OS-28. This area has been given separate designation due to unique chemical and toxicological conditions observed during the present and previous studies (discussed below). A sand bar was observed to have formed in this zone after the sediment erosion event. This transitional habitat from shallow to deep water would be expected to contain macrobenthos, as well as mussels and small fish where hard substrate is available, but may also be frequented by more mobile fish species such as winter flounder. Water depths exceed 3 m, limiting availability of prey to avian predators.

*Zone 4: Landfill Subtidal - Nearfield.* This area includes Stations MCL-8 to MCL-11, which define an area of approximately 50 m wide which runs the length of the landfill immediately off shore of the intertidal Stations NSB-1 through NSB-4. This

habitat is characterized as silty sand and supports relatively sparse populations of hard clams, but abundant lobsters; sidescan imagery of the area reveals that larger boulders are also present in this area. Winter flounder may also range into this area, feeding on macrobenthos, although the primary habitat for this species is expected to be the deeper offshore flats which better support its primary food resource (nereid worms).

*Zone 5: Landfill Subtidal - Farfield.* This area defines the subtidal environment offshore of McAllister Point Landfill seaward of Zone 4, and includes TRC/BOS Stations OS-22 through OS-27. Hard clams were collected by TRC (1994) from this area. Numerous floats for lobster traps are visible in the area, suggesting suitable habitat for this species. Winter flounder would be expected to occupy this region. Maps of regional geology and a side scan sonar survey of the area suggest sand and silty sand bottom with boulders.

*Zone 6: "Southern Depositional Area".* This region extends from the Coddington Cove breakwall south of the site to the north to Zones 3, 3A, and 5, as defined above, and extends seaward from the intertidal zone adjacent to NETC properties to offshore areas of approximately 12 m depth. This area was sampled primarily in Phase I to determine whether sediment potentially originating from McAllister Point may have been deposited there. Stations sampled in this zone include Stations S1 through S4, M2 and M3, D1 through D3, MCL-13 through MCL-16, OS-30A, OS-30B, and TRC/BOS Stations OS-29 and OS-30. Sidescan of the area shows relatively featureless relief of sediment characterized as silty sand, except for a deltoid-shaped region of disturbed sediment extending away from the Coddington Cove breakwall. This habitat is expected to contain macrobenthic communities, hard clams and lobsters.

*Zone 7: Reference.* Includes reference stations at Jamestown Cranston Cove (JCC), including shallow (<3 m; JCC-S1), mid-depth (3-5 m; JCC-M1) and deeper water (>10 m; JCC-D1) stations. The area receives freshwater input from Carr Creek on

Conanicut Island and has viable eelgrass beds nearshore. The nearshore macrobenthos resembles that offshore of McAllister Point (Zone 4), while the deep reference station contain macroinvertebrate species typical of shelf communities. Sparse numbers of hard clams are apparent; lobsters also appear to occupy the area based on commercial trap deployments.

## **6.1. COMPARISON OF CoC CONCENTRATIONS WITH CRITERIA AND STANDARDS**

In this section, concentrations of contaminants of concern (CoC) are compared with effects-based screening benchmarks. For sediments, comparisons were made against the NOAA ER-L and ER-M values (Long *et al.*, 1995). Porewater concentrations were compared against EPA Chronic and Acute Water Quality Criteria. For each matrix, Hazard Quotients were developed, calculated as the measured concentration at the station divided by the benchmark concentration. An additional contaminant class-level index, called the Hazard Index, was included, being calculated as the sum of analyte-specific Hazard Quotients within the PAH and metal CoC classes. This latter analysis was intended to provide a means of evaluating potential risks posed by analytes acting in an additive manner; however the Hazard Index does not address potential synergistic or antagonistic interactions among contaminants. Sediment Hazard Quotients (HQs) and Hazard Indices (HIs) are presented numerically in Appendix Tables A-2-1.1 (relative to ER-L) and A-2-1.2 (relative to ER-M).

### **6.1.1. Bulk Sediment Contaminants**

*Sediment Organics Hazard Quotients.* Comparisons of site concentrations of organics relative to ER-L and ER-M guidelines are illustrated in Figures 6.1-1 and Figure 6.1-2, respectively. Total PAHs, Total PCBs, and the pesticide p,p'-DDE generally exhibited similar spatial and temporal trends (Figure 6.1-1). Total PAH

concentrations exceeded the ER-L (4,022 ng/g) at TRC/BOS Stations OS-22, OS-25 through OS-30, at Phase I/II Stations NSB-3, NSB-4, NSB-5, NSB-7, MCL-8, MCL-10, MCL-16, S2B, and M1, at Phase III Stations NSB-6-R, MCL-12-R, and S2C, and at reference Station JCC-D1 (Figure 6.1-1). Similarly, p,p'-DDE concentrations exceeded the ER-L (2.2 ng/g) at Phase I/II Stations NSB-3 through NSB-7 (Figure 6.1-1). Neither Total PAHs or p,p'-DDE exceeded the ER-M guidelines (40,000 and 22 ng/g, respectively).

Total PCBs exceeded the ER-L (22.7 ng/g) throughout the study area and at the reference location, with the exception of Station S3, and reference Stations JCC-D1 and JCC-M1 (Figure 6.1-1). Total PCB concentrations also exceeded the ER-M (180 ng/g) by more than two-fold at Phase II Stations NSB-3 through NSB-7, Phase III Stations NSB-3-R through NSB-7-R, and between one- and two-fold at Phase II Stations MCL-10 and MCL-11, and Phase III Stations S2C and MCL-12-R (Figure 6.1-2). Furthermore, ER-M HQs at Phase III Stations NSB-4-R and NSB-5-R were approximately 31 and 6.7, respectively, as compared to HQs  $\approx$  3 at Phase II Stations NSB-4 and NSB-5 (Figure 6.1-2). Highest HQs were generally observed for stations in Zones 2, 3, and 3A (see Figure 6.0-1). Tributyltin concentrations did not exceed the U.S. EPA suggested lower effects benchmark (20 ng Sn/g at 2% TOC) for any stations measured (U.S. EPA, 1997).

*Sediment Metals Hazard Quotients.* Concentrations of metals relative to ER-L and ER-M guidelines are illustrated in Figures 6.1-3 and 6.1-4, respectively. ER-L-based Hazard Quotients for metals (Figure 6.1-3) generally reflect the trends observed for the organic CoCs. With the exception of arsenic at Phase II Station MCL-15 (HQ = 3.5) and Ni at Phase II Station MCL-8 (HQ = 8), the most impacted areas are in Zones 2, 3, and 3A, particularly for Cu, Pb, Hg, Ni, Ag, and Zn (Figure 6.1-3). Comparisons against ER-M values suggest particularly high adverse exposure due to copper at

Phase III Stations NSB-2 -R and NSB-4-R (HQ>28), as well as Zn at Phase III Station NSB-4-R (HQ>16; Figure 6.1-4).

*Hazard Indices.* ER-L Hazard Quotients discussed above for PAH and metal contaminants were summed by CoC class to derive Hazard Indices (Figure 6.1-5). HIs for other CoC classes were not calculated because only one benchmark was available for each. Metal HIs showed that the highest values were at stations in Zone 2, particularly Phase III Stations NSB-2-R and NSB-4-R (HI=387 and HI=372, respectively). Generally, greatest exposure due to PAHs was observed for stations in Zone 3A, with highest HIs observed for TRC/BOS Station OS-28 (HI=90), Phase II Station S2B (HI=92) and Phase III Stations MCL-12-R and S2C (HI=63 and HI=112, respectively). However, Phase III Station NSB-6-R (HI=85) also exhibited PAH exposure equivalent to that observed for Zone 3A stations.

*Sediment Hazard Quotient Rankings.* Table 6.1-1 presents Hazard Quotient rankings for selected analytes by zone and station. Rankings are based on NOAA ER-L and ER-M guidelines (Long *et al.*, 1995), as follows:

- “-” CoC concentration does not exceed the ER-L value (ER-L HQ<1);
- “+” CoC concentration equals or exceeds the ER-L value (ER-L HQ≥1);
- “++” CoC concentration equals or exceeds the ER-M value (ER-M HQ≥1); and
- “+++” CoC concentration exceeds the ER-M value by two-fold or greater (ER-M HQ≥2).

*Ecological Exposure Zone-based* exposure rankings were performed as described at the beginning of Section 6.0; these rankings are carried forward into the Exposure-Based Weights of Evidence Summary (Table 6.6-1).

Although adverse exposure due to Total PCBs are indicated throughout the study area, including some reference locations, in general, HQs are greatest in Zone 2, with lower HQs in Zones 3, 3A, and 4 (Table 6.1-1). However, Zone 2 also appears to be adversely impacted by high concentrations of a number of metals; a similar trend is noted in Zone 3 sediments for lead, nickel and zinc. Zone 3A appears to be most highly impacted by PAHs, and to a lesser extent Total PCBs. With some exceptions for individual sampling stations, remaining Zones 1, 4, 5, and 6, as well as reference Zone 7, generally exhibit comparatively lower CoC exposure conditions. Impacts due to tributyltin and p,p'-DDE do not appear to generate adverse exposure conditions in the study area.

*Comparison of pre- and post-erosion sediment Hazard Quotients.* A comparison of sediment organic contaminant concentrations between pre-erosion (Phase I/II) and post-erosion (Phase III) sediment Hazard Quotients was performed to assess whether nearshore sediment erosion related to landfill revetment construction (as described in Section 3.1) had increased possible CoC exposure to aquatic biota.

Sediment Hazard Quotients for Total PAHs, p,p'-DDE and Total PCBs measured during previous studies and Phase III are presented in Figures 6.1-1 (ER-L HQ) and 6.1-2 (ER-M HQ). For Total PAHs, notable increases of previously measured ER-L HQs were observed for Phase III Stations NSB-6-R and MCL-12-R (compared with Phase II Stations NSB-6 and MCL-12, respectively). Reductions in Total PAH ER-L HQs were noted for Phase III Stations NSB-3-R and S2B-R (compared with Phase II Stations NSB-3 and S2B, respectively). Total PCBs exhibited increased ER-M HQs at Phase III Stations NSB-4-R (HQ>31) and NSB-5-R (HQ>6) (compared with Phase II Stations NSB-4 (HQ=3) and NSB-5 (HQ=3), respectively). The pesticide p,p'-DDE or tributyltin was not measured in Phase III sediment samples due to relatively low concentrations observed in Phase I/II samples.

Sediment Hazard Quotients for metals measured during previous studies and Phase III are presented in Figures 6.1-3 (ER-L HQ) and 6.1-4 (ER-M HQ). Notable increases of previously measured ER-L HQs occurred for mercury at Phase III Station NSB-4-R (compared with Phase II Station NSB-4) and cadmium at Phase III Stations NSB-3-R and NSB-4-R (compared with Phase II Stations NSB-3 and NSB-4, respectively). Notably increased ER-M HQs were observed for nickel, copper, and zinc at Phase III Station NSB-4-R (compared with Phase II Station NSB-4), and copper, silver and zinc at Phase III Station NSB-2-R (compared with Phase II Station NSB-2).

#### 6.1.2. Metals Contaminants in Porewater

Concentration of metals in sediment porewater (measured only during Phase II investigations) were evaluated against EPA Water Quality Criteria - Saltwater Chronic (WQC-SC) and Water Quality Criteria - Saltwater Acute (WQC-SA) concentrations. Raw data are presented in Appendix A-2-3. Data are presented only for CoCs found to exceed the criteria (Cu, Ni and Zn; Figures 6.1-6, 6.1-7, and 6.1-8, respectively). Results for Cu show that porewater concentrations exceeded the WQC-SA at all intertidal stations, but not at subtidal stations (Figure 6.1-6). Nickel exceeded saltwater chronic criteria at six of seven intertidal stations, but not at subtidal stations (Figure 6.1-7). Finally, zinc concentrations exceeded WQC-SA at Stations NSB-4 and NSB-5 (Figure 6.1-8), while all other stations were below AWQC-SC. Hazard Indices for the above metals (sum of station-specific chronic HQs) are shown in Figure 6.1-9. The data suggest highest risk ( $HI > 15$ ) at Stations NSB-2, NSB-5 and NSB-7, and somewhat lower risk ( $HI > 5$ ) at Stations NSB-1, NSB-3 and NSB-4. The remaining stations had relatively low HI values ( $HI < 3$ ).

Hazard Quotient rankings for metal CoCs measured in porewater were developed based on EPA WQC-SC and WQC-SA (EPA, 1989), as follows:

- “-” = CoC concentration does not exceed the WQC-SC value (WQC-SC  $HQ < 1$ );
- “+” = CoC concentration equals or exceeds the WQC-SC value (WQC-SC  $HQ \geq 1$ );
- “++” = CoC concentration equals or exceeds the WQC-SA value (WQC-SA  $HQ \geq 1$ ); and
- “+++” = CoC concentration exceeds the WQC-SA value by two-fold or greater (WQC-SC  $HQ > 2$ ).

*Ecological Exposure Zone-based* exposure rankings were performed as described at the beginning of Section 6.0; results are carried forward into the Exposure-Based Weights of Evidence Summary (Table 6.6-1). In general, adverse porewater exposure conditions were high for Zone 2, whereas low adverse exposure conditions were observed for Zones 1, 4, and 6 (Table 6.6-1). No apparent adverse exposure conditions were observed for Zone 3A and Reference Zone 7.

### 6.1.3. Simultaneously Extracted Metals

The bioavailability and thus potential toxicity of divalent metals in sediments are believed to be predictable from measures of Simultaneously Extracted Metals (SEM) concentrations relative to Acid Volatile Sulfides (AVS) in the sediment matrix (DiToro *et al.*, 1994). The concentration of SEM is operationally defined by the chemical extraction procedure, which is less robust in comparison to conventional (e.g. strong acid) sediment digestion methods. The concentration of AVS is also operationally defined by the extraction procedure (i.e. sulfides released during sample acidification, hence "acid volatile"). Because sulfides in sediments form stable bonds with metals under anoxic conditions, toxicity of metals is limited when the molar concentration of AVS exceeds that of SEM (DiToro *et al.*, 1996).

Because sulfides are easily oxidized to sulfates which do not bind metals, the interpretation of metal bioavailability must consider possible scenarios which may control AVS concentrations, including seasonality, but also sample handling and processing artifacts. Three measures of SEM bioavailability in sediments collected during the TRC/BOS, Phase I and Phase II studies are presented in Table 6.1-2, including SEM concentration, SEM/AVS, and SEM-AVS.

An SEM/AVS ratio of 1.0 has been recommended as a threshold value of potential metal bioavailability (DiToro *et al.*, 1991), a value of 0.5 has been conservatively adopted for this ERA to allow for seasonal variation in AVS concentration. An SEM-AVS concentration of 5  $\mu\text{mol/g}$  dry weight was previously shown to be an approximate threshold for toxicity to amphipods by the National Sediment Quality Inventory (U.S. EPA, 1996). Total SEM concentration was also adopted for this investigation as a conservative measure, applicable if all AVS were lost from the sediment.

Raw data for SEM and AVS are presented in Appendix A-1-4. SEM bioavailability rankings for each of the metrics, as well as an overall SEM ranking, have been applied, as follows:

Indicator-specific ranking:

- SEM concentration:  $\leq 5 \mu\text{mol/g} = \text{"-"}, >5 \mu\text{mol/g} = \text{"+"}$ ;
- SEM/AVS:  $\leq 0.5 = \text{"-"}, >0.5 = \text{"+"}$ ;
- SEM-AVS:  $\leq 5 \mu\text{mol/g} = \text{"-"}, >5 \mu\text{mol/g} = \text{"+"}$

Overall SEM Exposure Ranking:

- "-" = no observed exposure for any indicator;
- "+" = exposure observed in one indicator only;
- "++" = exposure observed in two indicators; and
- "+++" = exposure observed in all indicators.

As discussed in Section 4.0, SEM concentration divided by AVS concentration (SEM/AVS) reveals several stations with ratios greater than 0.5, including all stations in Zones 1 and 2, most stations in Zone 6 (with the exception of Stations OS-29, OS-30, and S4), Stations NSB-7, M1, and MCL-8, and reference Stations JCC-D1 and JCC-M1 (Table 6.1-2). An SEM-AVS concentration of 5  $\mu\text{mol/g}$  dry weight was exceeded for Stations NSB-1, NSB-2, NSB-4, NSB-5 and NSB-7. Finally, SEM concentrations would exceed the SEM-AVS threshold value of 5  $\mu\text{mol/g}$  at Stations NSB-1 through NSB-5, NSB-7 and M3, assuming a total absence of AVS in the sediment.

Applying the overall SEM ranking criteria, the results show that greatest bioavailability and potential toxicity due to SEM metals were observed for Zones 1, 2, and, to a somewhat lesser extent, in Zone 3 (Table 6.1-2). Additionally, low but possible SEM-related toxicity was observed for Zone 6 and reference Zone 7. The Overall SEM Exposure Ranking is carried forward into the Exposure-Based Weights of Evidence Summary (Table 6.6-1).

## **6.2. ASSESSMENT OF TISSUE RESIDUE EXPOSURE IN TARGET RECEPTORS**

This section evaluates tissue residues in target species as indicators of CoC-related exposure. CoC exposure was assessed by comparison of site tissue residue concentrations with reference tissue residue concentrations (Tissue Concentration Ratios).

Site vs. Reference Tissue Concentration Ratios (TCRs) were employed to evaluate the potential significance of CoC tissue residues in target species. The analysis involves the comparison of receptor- and analyte-specific tissue body burdens from the McAllister Point Landfill study area stations against the corresponding data for the Jamestown Cranston Cove reference location. Comparisons of site tissue

concentrations against reference stations were made only for the same species and analytes. For this analysis, species- and analyte-specific data collected from the reference stations were numerically averaged to yield a single best estimate for the reference-based value. For organics data, tissue concentrations were normalized to the lipid content of the organism. Site and reference values below the Method Detection Limit (MDL) were not used to calculate TCRs in this analysis. In the present study, the availability of some species limited the biomass of tissue available for chemical analysis. To augment the data, reference station metals data for hard clams, as well as organic and metal data for mussels, were employed as reported in TRC (1994). The TRC reference station was located on the Conanicut Island shore just north of the reference station used in this study (Jamestown Cranston Cove). In addition, reference data for mummichog fish were obtained for Jamestown Potter Cove (Figure 3.1-1) as reported by SAIC and URI (1996).

The TCR rankings for organic contaminants and metals in target receptors from the McAllister Point Landfill study area, presented in Table 6.2-1, were based on PAHs, Total PCBs, p,p'-DDE, tributyltin, and all nine anthropogenic metals. Complete results are presented in Appendix Tables A-2-2.1 through A-2-2.5. Results were ranked according to the following method:

- “-” indicates  $TCR \leq 1$ ;
- “+” indicates  $TCR > 1$ ;
- “++” indicates  $TCR > 10$ ; and
- “+++” indicates  $TCR > 40$ .

Table 6.2-1a summarizes TCR rankings by zone and station. For PAHs, the results generally suggest the highest enrichment in lobsters collected from Zone 6, and in blue mussels collected from Zone 3. Metals were elevated in lobsters collected from stations in Zones 4 and 6 (particularly copper), and hard clams collected from Zone 5.

From Table 6.2-1b which presents TCR rankings by species, it is readily apparent that blue mussels consistently exhibited high CoC enrichments, particularly for PAHs, p,p'-DDE, and TBT; at Station NSB-7, TCRs exceeding forty were observed for all three of these CoCs. TCRs greater than forty were observed for copper in lobster hepatopancreas (HPP) at all stations, and for nickel in lobster muscle (MUS) at Station MCL-9, while PAH TCRs greater than ten were observed for lobster muscle at Stations MCL-13 and MCL-14. In particular, the lobster muscle TCR for fluorene at Station MCL-13 was greater than 40. Hard clams from many stations exhibited PAH and lead TCRs greater than ten, while cunner generally exhibited only slight enrichment of PAHs (metals were not measured in cunner).

In summary, TCRs indicated that the greatest CoC exposure to target species occurred in Zones 3, 4, and 6. PAHs and metals were the primary CoC classes of concern, though possible impacts were observed for stations in Zones 2 and 3 due to p,p'-DDE and TBT. These species-specific results are carried forward to the exposure-based weight of evidence summary presented in Section 6.6 (Table 6.6-1).

### **6.3. ANALYSIS OF BIOACCUMULATION AND TROPHIC TRANSFER**

In sections below, the relationships between contaminant exposure and tissue residue concentration for organics (Section 6.3.1) and metals (Section 6.3.2), and the trophic transfer of metals and organics to avian receptors feeding on aquatic receptors (Section 6.3.3), are discussed. These relationships are presented in a framework intended to elucidate the essential operative transport and fate mechanisms that control chemical bioavailability and trophic transfer in the exposure pathway models for target receptors (outlined in Sections 3.4 and 3.5). Using these exposure pathway models, the relative degree of CoC bioavailability in fish (organics only), bivalves and lobster in

the McAllister Point Landfill study area versus reference stations, is discussed with respect to differences between species and habitat.

For organics (PCBs, PAHs, pesticides, and TBT), exposure pathway differences were evaluated through the use of Biota-Sediment Accumulation Factors (BSAFs), while for metals, Bioaccumulation Factors (BAFs) were developed. BSAF factors are based on Equilibrium Partitioning (EqP) theory, whereby non-ionic organic contaminants are assumed to be at steady state between the carbon-normalized sediment concentration and the lipid-normalized tissue concentration (DiToro, *et al.*, 1991). A similar partitioning model for inorganic contaminants does not presently exist, hence the ratio of tissue to sediment concentration (BAF) is used as a tool to assess bioavailability. Factors were calculated for each CoC-receptor pairing for the above groups, including depurated and non-depurated mussels and clams, lobster muscle and hepatopancreas, and fish tissues.

### 6.3.1. Analysis of Organic Contaminant Bioaccumulation

The purpose of this analysis is to identify potential differences in organic contaminant exposure for target species representing different habitat or feeding types. For each organic contaminant class (PCBs, PAHs, pesticides, and butyltins), exposure pathway differences were evaluated through Tissue Residue - Exposure Relationships, as well as Biota-Sediment Accumulation Factors (BSAFs).

*Tissue Residue - Exposure Relationships.* Figure 6.3-1 gives a comparison of contaminants in hard clams and surface sediments from seven stations in the McAllister Point study area. The strongest relationship was for TBT ( $R^2 = 0.308$ ,  $n = 7$ ), but even this trend was not significant at the 90% confidence level. Corresponding values for mussels and surface sediments are given in Figure 6.3-2. In these regressions, Total PCBs showed the strongest relationship ( $R^2 = 0.466$ ,  $n = 7$ ), which was significant at the

90% confidence level. Comparisons of mussel tissue contaminants/% lipid and sediment contaminants/OC were also performed; results for these regressions are shown in Figure 6.3-3. Again, there were no significant correlations for any of the organic contaminants presented in Figure 6.3-3, nor for other CoCs measured in the study.

*Biota-Sediment Accumulation Factors (BSAFs).* The above analyses indicate that regression techniques did not effectively explain potential differences in contaminant exposure for different target species. An alternate approach to the above regression analyses was taken, involving the calculation of Biota-Sediment Accumulation Factors (BSAFs), which is the lipid normalized concentration of the CoC in an organism ( $\mu\text{g/g}$  lipid) divided by the organic carbon-normalized concentration of the same chemical in sediment ( $\mu\text{g/g}$  OC).

BSAFs from the present assessment were compared for similarity of central tendency as grouped by chemical class (PCBs, PAHs, pesticides, and butyltins). Plots of BSAFs for each species by compound class are presented in Figure 6.3-4. The box plots present the median value, as well as the range in values  $\pm 25\%$  about the median (box top and bottom), and the vertical lines represent the outside range or "whiskers" (approximately = 95% confidence limits). Asterisks and open circles are values exceeding the 95% and 99% confidence limits of the data set, respectively. The dashed line represents the mean of species-specific median values for each compound class. Data were included only for those analyte pairings where both the tissue and sediment concentrations were detected (e.g., substituted  $\frac{1}{2}$  MDL values were not used), so as to obtain a more accurate assessment of the data.

For the present study, the median BSAF values for PAHs, PCBs, pesticides and butyltins were 0.11, 3.8, 4.5, and 1.7, respectively (Figure 6.3-4). There was considerable overlap in central tendency about the median BSAF value for all species,

and little difference was observed between depurated and non-depurated bivalves. Butyltins were perhaps the exception, where lobster and fish BSAFs were lower than mussels and clams. However, the applicability of the EqP model to the TBT contaminant class may not be entirely appropriate, as these analytes are not entirely non-polar.

These results are similar to BSAF values calculated from literature values for infaunal deposit feeders, scavengers, filter feeders and benthically-coupled fish (Tracey and Hansen, 1996), where BSAFs for PAHs were uniformly lower (mean 0.34) than PCB (1.03) or pesticide (1.36) classes. In the Allen Harbor ERA (SAIC, 1995b), these values were 0.27, 1.57 and 1.62, respectively. Hence, the BSAF demonstrate that for species of varying habitat, including intertidal, subtidal (mussels and clams), scavenger (lobsters) and epibenthic predators (fish), the bioavailability of organic chemicals, and thus the functional contaminant exposure pathways, are similar. For this reason, a single exposure pathway model appears appropriate to predict the ultimate fate (i.e., tissue accumulation) of organic contaminants for target receptors of concern.

### 6.3.2. Analysis of Metals Bioaccumulation

The ratio of CoC tissue residues in bivalves and lobster at McAllister Point stations relative to co-located sediment concentrations, called Bioaccumulation Factors (BAFs), were analyzed as done above for organics to elucidate potential differences between species, tissue type and gut contents with regard to metal bioavailability for target receptor species. Factors governing differential metals bioaccumulation among species are poorly understood relative to that for organic contaminants as discussed in Section 6.3.1, such that analyses are conducted on a metal-by-metal basis.

BAFs were calculated for each CoC-receptor pairing and compared between the McAllister Point and reference stations. BAFs for each of the three species analyzed

for metals (mussels, clams and lobsters) were calculated for all CoC-receptor pairings. In addition, three non-CoC metals (Fe, Mn and Al) were evaluated to assess patterns that might reflect differential bioavailability due to crustal components. As with BSAF plots, the BAF box plots represent several measures of central tendency (refer to Section 6.3.1 for explanation of box plots). The median metal pairing of each species was used to calculate the overall BAF for the species.

Results show both species-specific and metal-specific differences in BAF values (Figure 6.3-5). For Hg, elevated tissue residues in lobster muscle were notable, whereas little differences were observed among depurated and non-depurated bivalves. BAFs for Fe, Al, and Pb were highest for mussels in the intertidal, intermediate for subtidal hard clams and lobster HPP and lowest for lobster muscle. In contrast, As, Cu, Ag and Zn exhibited a somewhat inverse pattern of bioavailability to that of Fe, Al and Pb; BAFs were lower for mussels in the intertidal than for hard clams and lobster from subtidal environments. Among the latter group, BAFs for Cu in lobster were notably higher than BAFs for hard clams. A third group of metals, including Ni and Mn, displayed higher BAFs for hard clams, intermediate for mussels, and lowest for lobster. Finally, a fourth group, including Cr, Hg, and Cd were similar for bivalves, but higher or lower for lobster.

The overall pattern of BAFs for metals was found to fall into four groups relative to the propensity for accumulation into tissues:

- |               |    |                                  |
|---------------|----|----------------------------------|
| High:         | 1) | As (17.2), Ag (5.2), Cd (4.9)    |
| Intermediate: | 2) | Zn (0.8), Hg (0.6), Cu (0.6)     |
| Low:          | 3) | Ni (0.08), Mn (0.04), Cr (0.02)  |
| Very Low:     | 4) | Pb (.006), Fe (.003), Al (.001). |

The bioavailability of sediment-associated heavy metals is related to the concentration of iron oxides in sediment as well as insoluble sulfides (Bryan and Langston, 1992). Hence, differences in the geological characteristics of the sediments between intertidal and subtidal zones can have a profound effect on metal bioavailability.

The most bioavailable metals include arsenic, silver and cadmium. Arsenic exhibits extremely mobile behavior in aquatic systems; surface water can undergo complex patterns of transformation including oxidation-reduction reactions, biotransformation, precipitation, and adsorption. These metals are relatively mobile in the aquatic environment based on relatively high solubility compared to most heavy metals. Because As and Ag exhibited highest BAFs for subtidal species, this would imply that direct dissolved phase exposure from landfill seeps (which would dilute quickly) is not the probable transport pathway. Instead, the data suggest that remobilization of these metals via resuspension or ingestion are the more important exposure route to target receptors.

Metals in the fourth group (Fe, Pb and Al) are the least bioavailable forms, and all are highly particle associated. Largest differences between depurated and non-depurated bivalve tissues were observed for these metals, which is consistent with the fact that a large fraction of the metal is not bioavailable to the organisms. Their particle-associated nature suggests that these metals are unlikely to be transported far from the source, which is supported by the fact that greater BAFs for these metals were identified in mussels inhabiting the intertidal zone.

Metals in the second and third groups, above, tend to show comparable BAFs among intertidal and subtidal species. Mercury, copper and zinc are very persistent when released into the environment, with the major removal mechanism occurring by adsorption onto particles and subsequent settlement to sediments. Mercury is well

known to bioaccumulate in marine organisms, and has received considerable attention because of its toxicity relative to other metals (Wren *et al.*, 1995). Nickel, manganese and chromium(+6) are generally characterized as moderately soluble in water, whereas  $\text{Cr}^{+3}$ , the form most commonly observed in biological systems and likely the form measured in the present study, is generally insoluble in water. Thus, the bioavailability of these metals with intermediate BAFs are most likely affected by a variety of processes ranging from dissolved-particulate partitioning to internal metabolic regulation.

In summary, the observed bioavailability of metals in this study is consistent with the known behavior of metals with respect to mobility and solubility, as well as the habitat characteristics of the McAllister Point Landfill intertidal and subtidal environments.

### 6.3.3. Trophic Transfer of CoCs to Avian Receptors

The potential for adverse effects to avian aquatic predators from the ingestion of contaminated food within the McAllister Point Landfill study area was assessed by comparison of prey Exposure Point Concentrations (EPC) and prey-associated CoC dosage (Dose) to appropriate Toxicity Reference Values (TRV-EPC and TRV-Dose, respectively) derived using a Hazard Quotient approach following the methods described in Opresko *et al.* (1996):

- 1)  $\text{HQ-EPC} = \text{prey EPC} / \text{TRV-EPC}$ ; and
- 2)  $\text{HQ-Dose} = \text{prey Dose} / \text{TRV-Dose}$ .

In the above equations, TRV-EPC and TRV-Dose benchmarks are defined as the concentration/ingestion rate of CoCs in prey (mg CoC/kg prey; dry weight) which would result in CoC uptake by the avian predator in an amount *equivalent* to the No

Observable Adverse Effects Level (NOAEL; Opresko *et al.*, 1996). When CoC uptake exceeds the benchmark (i.e. HQ > 1), a potential for adverse effects on the receptor is presumed to exist.

Whereas the TRV-EPC method (Equation 1) permits the direct comparison of prey tissue concentrations with the benchmark, and thus requires minimal assumptions, the TRV-Dose method (Equation 2) incorporates site-specific and avian receptor-specific information in the estimation of adverse effects of CoC uptake via ingestion of prey from the study area. Thus, the TRV-EPC method, although conservative, reduces the chance that important CoCs will be overlooked, while the TRV-Dose method increases the potential relevancy of the effects assessment, but may be less conservative because of the potentially greater uncertainty associated with the underlying assumptions required.

For this ERA, a combination of the two approaches (TRV-EPC and TRV-Dose) was implemented to allow better characterization of possible adverse effects due to the trophic transfer of CoCs from prey to avian aquatic predators. Prey-associated EPCs were determined by direct chemical analysis of the prey tissue (See Section 4.2). In the sections below, the methodology for estimation of dose (Section 6.3.3.1), derivation of Toxicity Reference Values (Section 6.3.3.2) and Hazard Quotient results (Section 6.3.3.3) are discussed.

#### 6.3.3.1. Bird Dose Calculations

In the TRV-Dose approach, a target species dosage model was employed to calculate uptake of CoCs as dependent upon exposure factors specific to the Receptor of Concern (RoC); including size-dependent food consumption rate, foraging behavior, migratory behavior, and food preference) and compared to the NOAEL benchmark.

The herring gull (*Larus argentatus*) and great blue heron (*Ardea herodias*) are species representative of aquatic birds which feed on invertebrates and fish in nearshore marine areas such as McAllister Point Landfill (Section 3). The herring gull may be considered omnivorous whereas the great blue heron feeds primarily upon fish (U.S. EPA, 1993). The Oak Ridge National Laboratory (Sample and Suter, 1994) avian predator exposure model was used to estimate the dose, or exposure to CoCs based on ingestion of CoC-contaminated prey in the McAllister Point Landfill study area, as follows:

$$3) \text{ Dose} = \text{EPC} * \text{EF};$$

where:      Dose =      Bird dose of CoC contaminant (mg CoC/kg bird/day, dry weight);

                  EPC =      Exposure Point Concentration of contaminant in prey within on-site foraging area (mg CoC/kg dry wt prey); and

                  EF =      Exposure Factor, or quantity of prey from the study area ingested by the bird per day, (kg dry wt. prey/kg bird/day)

The model assumes that the intake of contaminants via other exposure routes, such as water ingestion, are minimal in comparison to intake via food ingestion. The Exposure Factor is calculated as follows:

$$4) \text{ EF} = \text{FCR} * \text{a/fa} * \text{MF} * \text{FF};$$

where:      FCR =      Food Consumption Rate; (kg total diet/kg bird/day);

                  a/fa =      on-site foraging area/total foraging area of a bird (km<sup>2</sup>/km<sup>2</sup>);

                  MF =      Migration Factor; fraction of the year bird is in the area (yr/yr); and

FF = Feeding Fraction, or contribution of the prey type to the total diet (kg dry wt prey/kg dry wt total diet).

Documentation of the avian aquatic receptor exposure factors for this ERA are provided in Table 6.3-1. Important assumptions underlying the EF estimates include:

- The avian receptors are adults (e.g. body weights for adult specimens are assumed);
- The receptor food consumption rates are accurately predicted from body weight using the appropriate allometric relationships (e.g. heron and gulls have different feeding rates due to body size and behavioral differences);
- The avian receptor usage of the site, and therefore consumption of food from the site, are in proportion to the size of feeding habitat at the site relative to the home range size of the species (Opresko *et al.*, 1996). In this ERA, it is conservatively assumed that the avian predator lives and feeds exclusively at the site (i.e.  $a/fa = 1$ ) as discussed below;
- The birds at the site undergo spring/fall migration as found for other east coast populations (e.g. migration behavior reduces time spent on site);
- The birds may feed exclusively on any of the target receptor prey (serving as suitable surrogates of actual prey species which may be ingested at the site).

The three primary factors which discriminate the two avian predator species are body weight, lifestyle/habitat preference (wading vs. open water), and the total home range/foraging area. While the first two parameters are readily determined from the literature, the third parameter is typically site-specific.

The great blue heron, has a specific habitat preference for shallow (0.5 m) water for wading while fishing. Hence, its home range in the open estuarine environment of

Narragansett Bay is primarily restricted to intertidal areas. For this ERA, it is assumed that the great blue heron feeds exclusively at the site. This assumption is required because the habitat usage patterns for this species are not well known, and hence it is possible that individuals from a colony could heavily utilize habitats in the McAllister Point study area. For similar reasons, the assessment for the herring gull also assumes the receptor feeds exclusively at the site.

Based on input data identified in Table 6.3-1, the CoC-specific exposure factor (EF) for each avian aquatic receptor is calculated according to Equation 4, above, while dose of CoCs to the receptor (mg CoC/kg-bird/day) is calculated according to Equation 3, above.

#### 6.3.3.2. Toxicity Reference Values for Avian Aquatic Receptors.

In this section, the derivation of TRV benchmarks for the contaminants of concern (CoCs) selected for this ERA are presented based on procedures discussed in Opresko *et al.*, 1995).

The TRV-EPC and TRV-Dose benchmarks for the avian receptors selected for this ERA (great blue heron and herring gull) are based on dietary No-Observable-Adverse-Effects-Levels (NOAELs). The NOAELs are derived from literature studies where CoC-contaminated prey were fed to test species and monitored for the highest concentration where adverse effects (e.g. reduced growth and reproduction) were not observed. The benchmarks are converted into values applicable to each avian Receptor of Concern (RoC) as discussed below.

*NOAEL Derivation.* No Observed Adverse Effect Levels (NOAELs) and/or Lowest Observed Adverse Effect Levels (LOAELs) were identified from studies conducted exclusively on avian test species, which include data for domestic and wild

birds. Where possible, aquatic bird test data were selected in preference over data for other bird species.

In cases where only a LOAEL was available, the NOAEL was estimated as being equivalent to 1/10th of the LOAEL. If the only available data consisted of a NOAEL (or a LOAEL) for a subchronic exposure (approximately 10 weeks or less), then the equivalent NOAEL or LOAEL for a chronic exposure was estimated as being 1/10th of the value for the subchronic exposure. If only acute exposure data (LD<sub>50</sub>) were available, an acute-chronic ratio of 8:1 (Shepard, 1995) was applied to first estimate the chronic (e.g. LOAEL) benchmark.

The input data for the derivation of NOAEL-based TRVs, and the resulting TRV-Dose values are provided in Table 6.3-2. NOAEL data are available for seven of eighteen PAHs, two of five pesticides and one of three butyltins measured in the present investigation.

*TRV-Dose Derivation.* The TRV-Dose benchmark was obtained by scaling the test NOAEL on the basis of differences in body size according to (Opresko *et al.*, 1995):

$$5) \quad \text{TRV-Dose} = \text{test NOAEL} \times [\text{test bw/wildlife bw}]^{1/3}$$

where:        RoC bw = body weight of RoC (heron or gull; kg);  
              test bw = body weight of laboratory test species (kg); and  
              test NOAEL = measured adverse effects dosage (mg CoC/kg RoC/day).

As indicated above, the results of the RoC TRV-Dose derivation are provided in Table 6.3-2 under the summary for Toxicity Reference Values.

*TRV-EPC Derivation.* The TRV-EPC benchmark was calculated from the food factor  $f$ , which is the amount of food consumed per unit body weight per day:

$$6) \quad \text{TRV-EPC} = \text{RoC NOAEL}/f \quad (\text{Opresko } et \text{ al.}, 1996)$$

Food factors for aquatic predators were derived from the Food Consumption Rate (FCR, in kg prey dry weight/day) and the receptor body weight (bw in kg):

$$7) \quad f = \text{FCR}/\text{bw} \quad (\text{Opresko } et \text{ al.}, 1995).$$

Food consumption rates (FCR; kg dry wt. prey/kg bird/day) for herring gulls were estimated from the allometric regression model of Nagy (1987), while for great blue heron, the model of Kushlan (1978, as cited in U.S. EPA, 1993) was used (Table 6.3-1). The TRV-EPC values calculated in the above manner are summarized in Table 6.3-2 for gulls and heron.

#### 6.3.3.3. Adverse effects to Avian Aquatic Receptors

The receptor-exposure pathway scenarios evaluated for herring gull and great blue heron include the following species: cunner, blue mussels, lobsters (hepatopancreas and muscle), and hard clams. In reality, herring gulls and great blue herons are not likely to feed on all of the aforementioned species, but consumption of these prey species by avian aquatic predators has been modeled as part of a comprehensive and conservative approach in the assessment of potential risk, assuming that these prey species are surrogates for other organisms which might be part of the diet of gulls or herons. It is also assumed that the predators consume their prey whole. While great blue herons are primarily piscivorous in feeding habit, the literature suggests occasional invertebrate consumption. The herring gull is considered to be omnivorous (U.S. EPA, 1993).

As discussed previously, a sediment erosion event at the toe of the McAllister Point Landfill heightened concern about potential change in CoC exposure concentrations. For the avian predator assessment, the possible impacts of this event were assessed through prediction of CoC concentrations in mussels and fish that may occupy the area of erosion and therefore would be the most likely food source for herons and gulls. CoC concentrations for organics and metals were predicted from BSAF and BAF relationships, respectively. The basis of the relationships between sediment and tissue concentrations is discussed in Section 6.3.2. Using the information derived from the BSAF and BAF models, tissue concentrations can be estimated for organic classes (PCBs, PAHs, p,p'-DDE, TBT) and metals as:

Organics: 7)  $[Tissue] = ([sediment] * BSAF * \% lipid) / \% TOC,$

Metal analytes: 8)  $[Tissue] = [sediment] * BAF.$

The data for these calculations including information on sediment concentration, BSAF and BAF values, %lipid, %TOC is presented in Appendix A-1.

Measured prey concentrations and calculated CoC doses to avian receptors were compared to EPC-based Toxicity Reference Values (TRV-EPC) and Dose-based TRVs (TRV-Dose) to calculate Hazard Quotients which estimate the potential for adverse effects on each avian target species from consumption of CoCs in prey from the study area. The exposure point concentrations of CoCs for the five prey species/tissues (cunner, blue mussels, lobster hepatopancreas and muscle, and hard clams) are provided in Appendices A-1-1 (organics) and A-1-2 (metals). Note that although the avian predators consume live prey, the conversion of tissue CoC concentrations from a dry weight to a wet basis was not required since the TRV benchmarks are expressed on a dry weight basis.

A qualitative summary of potential adverse effects of CoC ingestion on avian aquatic predators, evaluated using prey-, station- and analyte-specific TRV-EPC and TRV-Dose Hazard Quotients, is presented in Table 6.3-3 (data are presented numerically in Appendix A-2-4). Prey-, station- and analyte-specific Hazard Quotients were ranked according to the following method:

- “-” =  $HQ \leq 1$ ;
- “+” =  $HQ > 1$ ;
- “++” =  $HQ > 10$ ; and
- “+++” =  $HQ > 20$ .

For herring gull, Hazard Quotients derived using prey concentration as the benchmark (HQ-EPCs) suggest the greatest adverse effects at Stations NSB-2-R and NSB-4-R (Table 6.3-3A), generally related to metals (Cu, Pb, Ag, and Zn) and PCBs in blue mussels and cunner. Chromium in hard clams at Stations OS-23 through OS-25 was also evident as a CoC of possible concern. A similar comparison using the TRV-Dose benchmark also suggests greatest adverse effects mainly at Station NSB-4-R due to Total PCBs in blue mussels and cunner (Table 6.3-3B). Possible adverse effects due to metals were also identified in blue mussels and cunner at Station NSB-2-R and hard clam Stations OS-23 through OS-25 based on the TRV-Dose HQ (Table 6.3-3B).

Apparent adverse effects to great blue heron generally followed that of the herring gull; HQ-EPCs generally suggest that greatest adverse effects result from ingestion of copper, lead, zinc and PCBs in cunner and blue mussels at Stations NSB-2-R and NSB-4-R, and chromium in hard clams at Stations OS-23 through OS-25 (Table 6.3-3C). As with herring gull, the comparison for great blue herons using the TRV-Dose benchmark suggests that PCBs, Cu and Pb in prey species may also represent a primary source for possible adverse effects in the study area, along with metals such as copper and lead (Table 6.3-3D). The overall station- and benchmark-

specific rankings of potential adverse effects reported in Table 6.3-3 A through D were evaluated as described in Section 6.0 and brought forward into Table 6.3-4, discussed below.

Table 6.3-4 provides an overall summary of adverse effects ranking for each avian predator by station. Results show good agreement between the two avian receptors; contaminants in blue mussels and cunner in Zone 2 appear to provide the most important CoC-avian receptor pathway of concern. These results are brought forward to the effects assessment summary in Table 6.6-2.

#### **6.4. ANALYSIS OF TOXICITY *VERSUS* CoC CONCENTRATIONS**

This section evaluates the relationship between CoC sediment and porewater contamination and toxicity responses for two bioassay species, the amphipod and the sea urchin. As described in Section 6.3, the analysis focuses on elucidation of potential exposure-response relationships. For the Phase I/II assessments, the measurement endpoints evaluated included the following:

- Toxicity of bulk sediments to amphipods (10-day survival); and
- Toxicity of porewater to sea urchins (fertilization success).

Phase III toxicity data on sea urchin fertilization and larval development success in sediment elutriate exposures (discussed in Section 5) is not discussed here due to a lack of matching chemical information on elutriates.

#### 6.4.1. Amphipod Bulk Sediment Toxicity

The amphipod toxicity response to bulk sediment was evaluated by comparison of relationships between survival versus (1) bioavailable metals (SEM- and AVS-related measures), and (2) representative organic contaminants (Total PAH, Total PCB, p,p'-DDE, and tributyltin).

Relationships between amphipod toxicity and three measures of metal bioavailability are presented in Figure 6.4-1. From SEM/AVS relationships, it is apparent that survival declined with increasing SEM/AVS ratios in the range of 150-350 (Figure 6.4-1A), suggesting that observed toxicity at NSB-1 and NSB-5 may be related to metals exposure. Further inspection of the data using the difference of SEM and AVS (SEM-AVS) as the indicator of metal bioavailability reveals that reduced toxicity at NSB-4 may also be related to metal exposure (Figure 6.4-1B). Because of the volatility of AVS in the presence of oxygen, and the possibility that some AVS could be lost during sampling or analysis, it is also instructive to directly examine toxicity vs. SEM metal concentration. The resulting relationship (Figure 6.4-1C) suggests that metals at Station NSB-3 might also exert toxicity if AVS concentrations were sufficiently reduced.

Patterns observed for amphipod toxicity vs. Total PAHs and tributyltins are less suggestive of exposure-response relationships; for example, reduced toxicity was observed at Stations NSB-3 and S2B at high Total PAHs in the sediments. Results of comparisons for total PCBs and p,p'-DDE indicate that although there was no linear relationship with toxicity, Phase II Stations NSB-4 and NSB-5 for p,p'-DDE and both Phase II and Phase III samples for Stations NSB-4 and NSB-5 for PCBs have high contaminant concentrations and are associated with low survival values (Figure 6.4-2).

Organic carbon normalization for the organic CoCs discussed above was performed to further elucidate potential exposure-response relationships (Figure 6.4-3).

Station S2B and reference Station JCC-D1 were observed to have similar Total PAH concentrations (ng/g O.C.). However, TOC concentrations at the reference station (0.9%, Table 4.2-1) were close to 0.5%, at which level carbon normalization of contaminant concentrations has been reported to be less certain (Di Toro *et al.*, 1990, 1992). Patterns observed for PCBs are more suggestive of exposure-response relationships, possibly implicating this chemical group as contributing to the observed toxicity at Stations NSB-3 through NSB-7 for both Phase II and Phase III samples. TOC-normalized pesticide concentrations in sediments are suggestive of an exposure-response relationship (excluding Station NSB-3). TOC-normalized tributyltin exposure-response relationship consistent with the observed toxicity effects at McAllister Point stations.

Thus, the above exposure-response relationships provide support for divalent metal toxicity at three stations, NSB-1, NSB-4 and NSB-5. Additional exposure-response sediment relationships for PAHs may have contributed to toxicity observed at Phase II Stations S2B, NSB-3, NSB-4 and NSB-5, as well as for PCBs at Phase II/III Stations NSB-4 and NSB-5.

#### 6.4.2. Sea Urchin Porewater Toxicity

Sea urchin toxicity responses were used to evaluate porewater toxicity of metals. Responses were compared to CoC concentrations by examining relationships between fertilization success with metals concentrations as discussed for amphipod results in the preceding section.

Results from the SEM/AVS comparison indicated that there was no correlation with fertilization success; only Station NSB-5 was characterized by both high toxicity and high SEM/AVS values (Figure 6.4-4A). Using the difference of SEM and AVS

(SEM-AVS) as the indicator of metal bioavailability, it is apparent that reduced sea urchin fertilization at Station NSB-4 may also be related to metal exposure (Figure 6.4-4B). As described in the amphipod analyses, it is also instructive to examine toxicity vs. SEM metal concentration directly. The resulting relationship suggests that metals at Station NSB-3 might also exert toxicity if AVS concentrations were low (Figure 6.4-4C).

Reduced sea urchin fertilization was also observed at a number of additional stations where exposure-response relationships related to metal toxicity were not observed. Stations in this category included JCC-D1, S3, S2B, M1, JCC-S1 and D3. Unionized ammonia concentrations at Stations S3 (0.43 mg/L) and the reference station JCC-S1 (0.54 mg/L), were high and close to the  $LC_{50}$  (0.6 mg/L) value for this compound. Both stations were located near eelgrass beds with organically rich sediments, that could provide the source of the ammonia. Un-ionized ammonia concentrations for the other stations were substantially below the 0.6 mg/L criterion, therefore observed toxicity at these stations cannot be attributed to ammonia.

Relationships between sea urchin fertilization success and concentration of organic contaminants in McAllister Point sediments are shown in Figure 6.4-5. In general, the comparisons do not suggest exposure-response relationships. For PAHs, the data indicate that despite high organics concentrations at many of the stations, only Station S2B had reduced fertilization related to high total PAH. Although Station NSB-3 had high concentrations of PAHs, PCBs and p,p'-DDE, the sea urchin fertilization result at this stations was not significantly different from the performance control. Similarly, high PCBs and p,p'-DDE at Stations NSB-4 and NSB-5 were correlated with reduced urchin fertilization, but these results are better explained by the metals-related responses. Tributyltin did not appear to contribute to the observed responses. Plots of the above data with sediment concentrations normalized to organic carbon are presented in Figure 6.4-6. Again, effects due to PAHs are suggested at Station S2B

(JCC-D1 was low in TOC concentration as discussed above), while PCBs and p,p'-DDE appear implicated in responses observed at Stations NSB-5 and possibly NSB-4. High concentrations of PAHs, PCBs and p,p'-DDE at Station NSB-3 were not associated with statistically significant reductions in sea urchin fertilization success. The reason for this is unclear, but possibly is related to differences in CoC composition in the porewater. Tributyltin concentrations were also not found to be correlated with sea urchin fertilization success.

Relationships of porewater exposure and toxicity for metals were also investigated to explain observed toxicities among amphipods and sea urchins. Measured porewater concentration data for Hg, Ni, Cu and Zn (the only metals measured above detection) are plotted against toxicity in Figure 6.4-7. Only Zn concentrations were correlated with observed toxicity in both sea urchin and amphipod tests at Stations NSB-4 and NSB-5. As illustrated in Figure 6.4-8, if porewater metals concentrations are expressed in Chronic Water Quality Criteria (CWQC) units, the analysis suggests that Zn, Ni and possibly Cu are above criteria and may contribute to the observed toxicity at Stations NSB-4 and NSB-5.

The results of amphipod and sea urchin results, when considered collectively, support the conclusion that metals are primarily responsible for observed toxicity at the McAllister Point Landfill Stations NSB-1, NSB-3, NSB-4 and NSB-5. Additional stations exhibiting toxicity for both tests included Station S2B which appears related to PAH concentration. CoC concentrations at remaining stations exhibiting toxicity exceed ER-L criteria, including JCC-D1 (PAHs), M1 (PAHs, PCBs and Pb), and D3 (Ni). Finally, toxicity at Station S3 and reference Station JCC-S1 could be partly explained by high unionized ammonia.

It is important to note that the sea urchin test is typically considered a "metals-sensitive" test, whereas the amphipod test appears responsive to both metals and

organic CoC when present in sufficient concentrations. This differential sensitivity may be explained in part by the test method and physiological end point employed; the short-term duration of exposure (60 minutes) for the fertilization test favors toxicity caused by CoCs with higher solubility and membrane permeability, whereas exposure durations of 10 days in the amphipod test permits larger compounds with lower solubility (i.e. organic CoCs) to reach target organs. In the present study, however, organic contaminants are the CoC class which is generally implicated as the cause for observed effects on sea urchin fertilization. It is hypothesized that this response may be more related to interference of chemotaxis by the sperm cells, as opposed to the intracellular effect typically attributed to these compounds (e.g. narcosis, McCarty *et al.*, 1993). However, regardless of the mode of toxic action, the two tests have provided largely similar results.

#### **6.5. ANALYSIS OF EFFECTS MEASUREMENTS VERSUS CoC CONCENTRATION**

This section evaluates the relationships between CoC concentrations and specific effects measurement endpoints, including biotic condition and tissue contaminant levels. The analyses focus on “dose-response” types of relationships (e.g., whether increased levels of contamination are associated with increased effects to biota). This represents some fundamental elements of gradient analysis, with particular benchmarks representing defined threshold levels. Measurement endpoints that are evaluated in this assessment include the following:

- Condition (dry tissue weight to shell length and weight ratio) of mussels and hard clams,
- Infaunal community structure, and

- Correspondence of these measures with those of previous studies at McAllister Point and in the primary literature.

#### 6.5.1. Condition-Exposure Relationships

Variations in biotic condition for bivalves were compared to CoC concentrations by generating scatter plots and performing linear regressions for condition indices versus representative organic and inorganic contaminants. Results of these comparisons for blue mussels and hard clams are shown in Figures 6.5-1 and 6.5-2, respectively. Limited data were available for these statistical analyses. The 95% confidence limits are indicated in the graphs; where the limits are lacking, regressions are not significantly different from zero. The regression analyses indicate generally poor correlations for most condition indices (i.e., no dose-response relationship). Some positive correlations were observed, e.g. mussel tissue weight/shell weight ratios for copper, and tissue weight/shell length ratios vs. total PCBs in clams. However, this relationship would contraindicate adverse exposure effects. The only negative correlations observed for which slopes were different from zero occurred for tissue weight/shell length ratio vs. tributyltin in clams. Even here, correlations are very weak and do not provide significant evidence for adverse impact.

#### 6.5.2. Assessment of Landfill Impacts on Benthic Communities

Visual shoreline observations indicate significant physical habitat disruption due to the presence of solid waste, particularly at Stations NSB-3 and NSB-4. Particles of ash, metal, glass, and iron-stained sediment were very obvious at Stations NSB-3, NSB-4 and NSB-5. In addition, Station NSB-5 was near the open face of the landfill disposal area (opened during capping) and to a seep which may contain waste leachate. CoC concentrations were especially elevated in this area (Section 4).

Changes in intertidal community structure were examined for correspondence with these physical and chemical habitat alterations.

*Intertidal Habitats.* Changes in intertidal community structure included analysis of sediments and mussel clumps as two distinct habitats for benthic fauna. Difference in CoC exposure between samples types is expected because in “sediment” samples, organisms are in direct contact with particulate and porewater contaminants, and therefore have a greater likelihood of impact. There was no consistent relationship between location and number of species per sample (Figure 6.5-3A). In these samples, the pollution-tolerant and opportunistic polychaete, *Capitella capitata*, was in high abundance at Station NSB-3 to NSB-6. Spionid polychaetes (*Pygospio elegans* and *Streblospio benedicti*) were also found to be higher at Stations NSB-5 and NSB-6. In contrast, the pollution-tolerant oligochaete *Pelosclex benedeni* is relatively evenly distributed in sediment samples, and apparently not responsive to existing gradients in contamination, although its abundance relative other non-*P. benedeni* oligochaetes increases from Stations NSB-1 and NSB-2 to a maximum at Stations NSB-4 and NSB-5. The increase in relative abundance of the pollution tolerant species at NSB-3 to NSB-6 suggest possible landfill-related effects on benthic community composition, although the change in slope from northern stations to this area may also explain the observed response.

The impact of the sediment erosion event on the benthic intertidal community is unknown. One can only speculate that the same physical processes occurring prior to the erosion event are likely to be continuing after the erosion event. Thus, it is expected that similar trends in benthic community structure may be reestablished over time without further human intervention, although considerable uncertainty is associated with this assumption.

*Subtidal Habitats.* Measures of benthic faunal quality for MCL stations are presented in Figure 6.5-4. Number of species and individuals do not appear to exhibit spatial trends related to sample location. Numbers of *Capitella capitata* found at Stations MCL-11 through MCL-14 were approximately similar to the reference site shallow (JCC-S1) and mid-depth (JCC-M1) stations. Combined spionid polychaetes have a broad peak between Stations MCL-11 through MCL-14, but were also less than reference site concentrations. One of the spionids, *Streblospio benedicti*, was only found at Stations MCL-11 and MCL-12, and total numbers were very low.

The numbers of *Capitella* and *Streblospio* at the landfill study area stations are consistent with the shallow bottoms of the East Passage of Narragansett Bay, where organic matter may be locally abundant under shell hash (French *et al.*, 1992). Menzie-Cura (TRC, 1994) found high species numbers and faunal densities at all three McAllister Point subtidal stations sampled (OS-22, OS-25, and OS-28), particularly at Station OS-28 (near MCL-14), and concluded that species richness was related to high substrate diversity. These findings are consistent with the present study and suggest that landfill effects on subtidal benthic communities are not readily apparent.

## 6.6. WEIGHT OF EVIDENCE SYNTHESIS

The interpretation of ecological risk in this assessment is based on a weight of evidence approach. The weight of evidence is in turn based on the analysis of exposure and effects data, as well as measures discussed in the previous sections. Uncertainties associated with this interpretation are discussed in Section 6.7. As a framework for discussion of the WoE indicators for various areas of the McAllister Point Landfill intertidal and subtidal environments, the region has been subdivided into discrete "Ecological Exposure Zones" (EEZs) as discussed in Section 6.0.1.

Ecological risks are a function of the severity of potential ecological impacts, the area over which impacts can occur, and the duration of such potential impacts (Suter *et al.*, 1995). Unlike human health risk assessments, there does not exist a standard scale for interpretation of ecological risks. It is therefore necessary to establish tentative boundaries for risk categories, just as benchmark scales have been developed for interpretation of exposure (e.g., NOAA ER-L and ER-M; EPA Acute and Chronic Water Quality Criteria) or effects (e.g., EPA EMAP amphipod survival < 80%).

The following categorization of ecological risks has been developed for the McAllister Point Landfill ERA:

*Baseline* risk is defined as the probability of adverse exposure and/or ecological effects equivalent to that from contamination and other environmental conditions not associated with the site.

A *Low* probability of ecological risks suggests possible, but minimal impacts based on some of the exposure or effects-based weights of evidence, while impacts are undetectable by the majority of exposure and effects-based weights of evidence. Conditions of low risk probability typically lack demonstrable exposure-response relationships.

An *Intermediate* probability of ecological risk occurs for site conditions falling between high and low probabilities of risk. As such, the intermediate risk probability condition is typically characterized by multiple exposure or effects weights of evidence suggesting that measurable exposure or effects, but not both, are occurring at the site. Typically, quantitative exposure-response relationships are lacking. Intermediate risk probability may also be indicated if the spatial extent of apparent impact is highly localized (e.g., a single station), or if the impact occurs for periods of very limited duration.

Conditions indicating *High* probability of ecological risk occurs when numerous weights of evidence suggest pronounced contaminant exposure and effects, the spatial extent of apparent impact is great, the impact is likely to be persistent over long periods of time, and the available data support demonstrable exposure-response relationships.

The intent of the above risk categorizations is not to place rigorous boundaries on actions that risk managers may take with respect to the results of the study, but merely to provide definition and uniformity for the description of risks as discussed in the following section.

In the exposure (Section 4.0) and effects (Section 5.0) sections of this ERA, as well as in the risk characterization (Sections 6.1 through 6.5), the individual weights of evidence were interpreted and summarized using semi-quantitative ranking schemes so as to allow their inclusion into an analysis of the overall risk indicated for each of the primary weight of evidence categories. In Sections 6.6.1 and 6.6.2, below, the process of synthesizing information obtained on individual indicators and translating the result into an exposure/effects Weight of Evidence (WoE) ranking is presented. The primary exposure-based WoEs are Sediment Hazard Quotients, Porewater Hazard Quotients, SEM Bioavailability, Sediment Fecal Indicators and Tissue Concentration Ratios, while Laboratory Toxicity, Field Effects Indicators, and Avian Predators are included as effects-based WoE.

As discussed in Section 6.0 above, a single ranking strategy for the synthesis of WoE indicators was adopted for the ERA in order to provide a consistent synthesis of the data to achieve a coherent evaluation of risk.

Results of the evaluations of the WoE data are presented in exposure and effect WoE summary tables in Sections 6.6.1 and 6.6.2, respectively. In Section 6.6.3, the findings of exposure and effects WoE are evaluated jointly in order to interpret the overall probability of adverse ecological risks by sampling station.

#### 6.6.1. Exposure-Based Weight of Evidence Summary

Exposure-based weights of evidence include Hazard Quotients (HQs) for CoC sediment and porewater contaminants, SEM metal bioavailability, and CoC residues in target species relative to reference as assessed through Tissue Concentration Ratios (TCRs).

*Sediment Hazard Quotients.* Chemical concentrations of CoCs were measured in sediments and compared against ER-L and ER-M benchmarks to elucidate potential adverse effects on target species from exposure to contaminant concentrations in surface sediments (discussed in Sections 6.1.1 and 6.1.2).

The pattern observed in Sediment Hazard Quotient data (Table 6.6-1) reveals that Zone 2 has a high adverse exposure probability (CoCs 2 x ER-M) while an intermediate adverse exposure probability exists for Zones 3 and 3A (CoCs > ER-M). The remaining study area zones have low adverse exposure ranking (CoCs > ER-L; only at the reference location were the sediment CoCs generally below ER-L concentrations (e.g. baseline adverse exposure probability)).

*Porewater Hazard Quotients.* Porewater concentrations were compared to EPA Saltwater Chronic (SC) and Saltwater Acute (SA) criteria to provide additional information as to CoC bioavailability for infaunal organisms.

Zone-based exposure rankings based on Water Quality Criteria were performed on the porewater data as described in Section 6.1.3. In general, an intermediate probability of adverse exposure was observed for Zone 2 ( $SA > CoCs < 2x SA$ ), whereas a low probability of adverse exposure was observed for Zones 1, 4, 5, and 6 ( $CoCs > SC$ ). No apparent risk was observed for Zones 3A, 5, and Reference zone 7 ( $CoCs < SC$ ).

*SEM Bioavailability.* Simultaneously Extracted Metals (SEM) bioavailability is a measure of the simultaneous and cumulative impact of 5 divalent metals (Cu, Cr, Pb, Ni and Zn) on sediment toxicity.

Overall, SEM bioavailability was highest in Zone 2 and somewhat reduced for Zones 1 and Zone 3 (Table 6.6-1). SEM bioavailability was relatively low throughout the remainder of the study area zones and the reference zone.

*Sediment fecal pollution indicators.* Fecal pollution indicators were measured in sediments as an indicator of potential contaminant transport pathways and sewage-related impacts on target species.

Evidence of high sediment fecal pollution was observed for intertidal Zone 1 while the lowest concentrations were observed for Zone 1 (Table 6.6-1). Intermediate fecal indicator concentrations were observed for Zones 2, 3A, 4, and 6. Low fecal indicator concentrations were observed in Zone 3. Data were not available for Zone 5 and Zone 7. Overall, the data suggest that potential sewage-related pathways for contaminant exposure exist in the study area, perhaps coming from the North, but the trend does not explain the occurrence of the highest CoC exposure conditions observed for Zone 2 and Zone 3.

*Tissue Concentration Ratios.* Tissue Concentration Ratios (TCRs) were calculated to elucidate those CoCs and receptors which are chemically enriched at the site relative to regional background conditions.

Overall, Zones 3, 4 and 6 contain biota with residue concentrations that are highly elevated (TCR > 40) with respect to reference locations (Table 6.6-1), while Zone 2 also contains biota with intermediate CoC residue elevations ( $10 > \text{TCR} < 40$ ). In contrast, Zone 3A and Zone 5 generally contain biota with relatively low CoC residue elevations (TCR < 10). As noted previously, TCR values for PAHs in cunner may be underestimates because of PAH metabolism capabilities.

#### 6.6.2. Effects-based Weight of Evidence Summary

*Laboratory Toxicity.* Sediment bioassays with amphipod, *Ampelisca*, and the porewater and elutriate bioassays with the sea urchin, *Arbacia*, are used to assess possible impacts from in-place and resuspended sediments, respectively.

Laboratory toxicity results indicate the greatest likelihood of adverse CoC exposure in Zone 2 (Table 6.6-2), suggesting that CoCs are both bioavailable and toxic. Toxicity was also generally evident for Zone 3, although not as prevalent as several stations were not toxic and impacts on one of the three endpoints (amphipod survival) was not generally observed. Reduced effects were observed for the remaining zones where toxicity at some stations was occasionally observed, including the reference location (Zone 7).

*Field Effects.* Field effects parameters, summarized in Table 6.6-2, include bivalve condition indices, benthic community structure and tissue concentrations of fecal pollution indicators.

The overall ranking for field effects suggests intermediate probability of adverse effects for Zones 1, 2 and 3A. Low or baseline adverse field effects were observed for the remaining zones, while the reference location zone (Zone 7) also exhibited a low probability of adverse field effects.

*Avian Predators.* The food web modeling for avian aquatic predators evaluated the likelihood of adverse effects on gulls and heron from consumption of CoC-contaminated prey.

Despite the conservative assumptions employed, only a low probability of adverse effects (HQ < 10) was apparent for most zones, including Zones 3A to 6, and reference Zone 7 (Table 6.6-2). Only Zone 2 exhibited station-CoC-prey receptor pairings which were assessed as a high risk probability (HQ > 20). A baseline probability of adverse effects (HQ < 1) was observed for Zones 1 and 3.

### 6.6.3. Risk Synthesis

The individual Exposure and Effects WoE underlying indicator measures were discussed in the previous sections and summarized in Table 6.6-1 and Table 6.6-2, respectively. As a framework for discussion of risks for various areas of the McAllister Point Landfill study area, the following definitions of ecological risks has been developed for the McAllister Point Landfill Marine ERA:

*Baseline* risk is defined as the probability of adverse exposure and/or ecological effects equivalent to that from contamination and other environmental conditions not associated with the site.

A *Low* probability of ecological risks suggests possible, but minimal impacts based on some of the exposure or effects-based weights of evidence, while

impacts are undetectable by the majority of exposure and effects-based weights of evidence. Conditions of low risk probability typically lack demonstrable exposure-response relationships.

An *Intermediate* probability of ecological risk occurs for site conditions falling between high and low probabilities of risk. As such, the intermediate risk probability condition is typically characterized by multiple exposure or effects weights of evidence suggesting that measurable exposure or effects, but not both, are occurring at the site. Typically, quantitative exposure-response relationships are lacking. Intermediate risk probability may also be indicated if the spatial extent of apparent impact is highly localized (e.g., a single station), or if the impact occurs for periods of very limited duration.

Conditions indicating *High* probability of ecological risk occurs when numerous weights of evidence suggest pronounced contaminant exposure and effects, the spatial extent of apparent impact is great, the impact is likely to be persistent over long periods of time, and the available data support demonstrable exposure-response relationships.

As can be seen in the above definitions, a key element to the interpretation of ecological risk in this assessment is the extent to which adverse exposure and effects occur concurrently. Where such concurrence exists, there is strong evidence for a completed exposure pathway between the CoCs and the receptors of concern.

An overall evaluation of exposure and effects WoE is needed to facilitate the risk characterization, just as WoE-specific indicator data were evaluated to determine and carry forward information about each WoE into the summaries of exposure and effects data in Tables 6.6-1 and 6.6-2. The following approach was used to maintain overall consistency with the evaluation method used for the primary WoE:

<i>Baseline Adverse E/E Probability (B):</i>	Baseline (-) ranking for all indicators, or low (+) ranking observed for only one indicator;
<i>Low Adverse E/E Probability (L):</i>	Low (+) ranking observed for two or more indicators, or intermediate (++) ranking for only one indicator;
<i>Intermediate Adverse E/E Probability (I):</i>	Intermediate (++) ranking observed for two or more indicators, or high (+++) ranking for one indicator;
<i>High Adverse E/E Probability (H):</i>	Intermediate (++) or greater ranking observed for two or more indicators.

Following the derivation of overall exposure and effects ranking for each zone by the above criteria, the joint probability of exposure and effects is used to presume the probability of risk for each exposure zone, as follows:

- *Baseline Risk:* No greater than Baseline (B) ranking for Exposure and Effects WoE summaries;
- *Low Risk:* No greater than Low (L) ranking for Exposure and Effects WoE summaries;
- *Intermediate Risk:* Intermediate (I) ranking for both Exposure and Effects WoE summaries, or High (H) ranking for one WoE summary and no greater than Low (L) ranking for the other WoE summary; and
- *High Risk:* High (H) risk ranking for one Exposure and Effects WoE summary and Intermediate (I) or High (H) ranking for the other WoE summary.

As discussed previously for the individual WoE ranking, this approach is based on best professional judgement and the risk manager is encouraged to evaluate

alternative ranking approaches as it might relate to the general outcome of the risk assessment.

Eight EEZs were identified for the McAllister Point ERA, including:

1) Landfill Intertidal North; 2) Landfill Intertidal Middle; 3) Landfill Intertidal South; 4) Zone 3A; 5) Landfill Subtidal - Nearfield; 6) Landfill Subtidal - Farfield; 7) "Southern Depositional Area"; and 8) the Reference Site. Each of these zones appears to provide a unique habitat for target species, as well as considerable differences in CoC exposure, effects and risks, as discussed below:

*Zone 1: Landfill Intertidal North EEZ.* The exposure and effects WoE summary suggest a high adverse exposure condition but a low adverse effects probability (Table 6.6-3). CoC concentrations in sediment and porewater for Zone 1 stations did not generally exceed sediment benchmarks. In addition, exposure-response relationships between toxicity measures and CoC concentrations were not generally observed although in one instance, SEM metals were elevated and was shown to exhibit exposure-response relationships explaining observed toxicity in *Ampelisca*. Exposure-response relationships were not observed based on comparisons with the sea urchin fertilization test, nor were macrobenthic community structure responses discernable. There was indication of recent sources of fecal pollution in sediments and of the area, possibly originating from Gomes Brook which discharges north of the landfill (or from shorebirds inhabiting the intertidal), such that alternate CoC sources are possibly impacting this area. Low enrichment of CoCs in aquatic biota were evident, but this did not pose a risk to avian predators consuming these organisms. The sediment erosion event did not appear to increase CoC bioavailability for this zone.

Based on the above data, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, blue mussels and fish living in Zone 1 is presumed to be intermediate.

*Zone 2: Landfill Intertidal Middle.* The exposure and effects WoE summary suggest a high adverse exposure condition and an intermediate adverse effects probability (Table 6.6-3). Sediment-based Hazard Quotients reveal high CoC concentrations in this zone, particularly for PCBs and metals. SEM metals are high, and measured pore water copper and zinc concentrations exceeded the corresponding EPA Acute Water Quality Criteria for these metals. In general, sediment and tissue fecal pollution indicators did not indicate any significant contribution of alternate pollution sources to the area. Results of mussel tissue concentration comparisons of site vs. reference confirm CoC bioavailability of most metals (particularly lead at Station NSB-3) while similar comparisons for organics did not show evidence of enrichment. Avian predators were at high risk from consumption of prey in this zone. Clear, unambiguous exposure-response relationships between high SEM metals and high amphipod toxicity were observed. Porewater concentrations for zinc were more than twice the Water Quality Acute Criteria at Station NSB-5. An increased number of pollution-tolerant species were apparent at Station NSB-5 relative to northern zones; although this trend may be in part related to a habitat change between Stations NSB-4 and NSB-5 which would favor these macrobenthos. Bivalves had elevated tissue residues, which translated into intermediate risks to avian predators. The sediment erosion event resulted in increased CoC bioavailability in this zone.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, blue mussels and fish living in Zone 2 is presumed to be high.

*Zone 3: Landfill Intertidal South.* As with Zone 1, the exposure and effects WoE summary suggest a high adverse exposure condition but a low adverse effects probability (Table 6.6-3). Sediment-based Hazard Quotients generally high adverse exposure conditions, but the associated porewater concentrations for metals were only occasionally above Saltwater Chronic values. SEM bioavailability was high at one of

two zone stations (NSB-7) but baseline at the other station (NSB-6), hence intermediate overall. Sediment and tissue fecal pollution indicators did suggest recent sewage-related contaminants and thus the possible contribution of alternate pollution sources to the area. Tissue Concentration Ratios (TCRs) were high for mussels tissues relative to the reference location but were low for cunner. Avian predators were generally not at risk from consumption of biota inhabiting this zone. Slight toxicity to *Ampelisca* was observed at both stations sampled, but generally no toxicity was observed in porewater or elutriate tests with *Arbacia*. No effects on mussel condition were noted, but benthic community indicators did suggest species shifts in favor of pollution-tolerant forms at NSB-6, but this could be due to the availability of finer-grained sediments.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, blue mussels and fish living in Zone 3 is presumed to be intermediate.

*Zone 3A.* For this zone, the exposure and effects WoE summary suggest both intermediate adverse exposure and adverse effects probabilities (Table 6.6-3). Sediment Hazard Quotients suggest CoCs for two of eight sampling events had exceeded the ER-M by greater than two-fold, two were greater than ER-M and four were greater than the ER-L benchmark. Porewater metals at the one sampled station did not generally exceed criteria, and SEM metals were typically not bioavailable. Some indication of recent fecal pollution to the area was evident, but the limited data for bivalve TCRs suggest CoCs are not being concentrated in tissues to levels greatly above the reference condition. Accordingly, risk to avian predators was low for this zone. Benthic community analyses conducted at one station in this zone (MCL-12) did not suggest adverse effects. Toxicity to *Ampelisca* was not generally apparent, but there were indications of CoC toxicity to *Arbacia* fertilization. In this case, however, there exists uncertainty because of a lack of definitive exposure-response relationships for the porewater test (where matching CoC-toxicity data were available) and possible

sediment interference for the elutriate test. Still, the toxicity results, overall, suggest the probability of adverse CoC exposure, although the magnitude of this exposure is unclear. Also, the limited geographical extent and substrate character (e.g. hard pebble/shell cover) indicates reduced potential for widespread exposure or CoC remobilization to target receptors in the area.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, hard clams, lobster and fish living in Zone 3A is presumed to be intermediate.

*Zone 4: Landfill Subtidal - Nearfield.* The exposure WoE summary for this zone suggests intermediate adverse exposure conditions but a baseline adverse effects probability (Table 6.6-3). Sediment concentrations for stations in this zone exceeded ER-L benchmarks, but did not generally exceed ER-M benchmarks. Porewater concentrations were generally below criteria values, and SEM metals were not typically bioavailable. Some indication of possible alternate CoC sources were suggested from levels of fecal indicators in sediment and tissue residues. CoCs in tissue residues (particularly copper in lobster hepatopancreas) were high relative to reference values. Avian predators were not generally observed to be at risk from prey consumption in this zone. Toxicity was generally not apparent, and no indication of altered benthic community structure could be discerned.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, hard clams, lobster and fish living in Zone 4 is presumed to be intermediate.

*Zone 5: Landfill Subtidal - Farfield.* The exposure WoE summary for this zone suggests a low probability of adverse exposure and a baseline adverse effects probability (Table 6.6-3). Data available for evaluation of risk for this zone consisted

entirely of sediment and tissue data collected by TRC (1994). Sediment Hazard Quotients generally exceeded ER-L values. Tissue data for hard clams were slightly elevated relative to reference. SEM metals were not bioavailable. Avian predators were observed to be at low risk from ingestion of prey in this zone.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, hard clams, lobster and fish living in Zone 5 is presumed to be low.

*Zone 6: "Southern Depositional Area".* The exposure WoE summary for this zone suggests an intermediate probability of adverse exposure but a low adverse effects probability (Table 6.6-3). Extensive sampling in this region occurred during Phase I with largely confirmatory sampling during Phases II and III. Stations in this zone exhibited CoC concentrations which generally exceeded ER-L values. Porewater metals were not generally above WQC criteria, but SEM metals were generally bioavailable. Low levels of fecal pollution indicators were observed in sediments and biota. Tissue concentrations of CoCs in lobster at two sampled locations were highly elevated relative to reference, while hard clams were also enriched in CoCs, but to a lesser extent. However, the nature of CoCs were such that risks to avian predators consuming these biota were low. There did exist evidence of high toxicity to sea urchins during porewater fertilization tests for Station D3, sampled in Phase 1, but this observation was not confirmed in repeat sampling during Phase II. Also, definitive exposure-response relationships were not observed, partly because the observed toxicity was generally not high.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, hard clams, lobster and fish living in Zone 6 is presumed to be intermediate.

*Zone 7: Reference.* The exposure WoE summary for this zone suggests a baseline probability of adverse exposure and a low adverse effects probability (Table 6.6-3). CoC concentrations were generally below sediment benchmarks, and porewater metals were typically below criteria. Toxicity was observed for *Arbacia*; however, high un-ionized ammonia concentrations due to decomposition of organic matter contained in the eelgrass habitat appear responsible. Sediment fecal pollution indicators suggest recent sources of contamination at the deep station, possibly originating from Carr Creek on Conanicut Island. Macrobenthos species numbers and abundance were low relative to landfill zones. Tissue residues were also low, and associated impacts on avian predators from consumption of reference location mussels, hard clams and fish were also low.

Based on the above information, the probability of landfill-related CoC risk to infaunal benthic communities, shore birds, hard clams, lobster and fish living in the reference zone is presumed to be low.

## 6.7. UNCERTAINTY

The weight of evidence in this assessment is dependent upon analyses of exposure, effects, and risk characterization findings. Uncertainties discussed in the exposure phase of this assessment (Section 4.3) included:

- Adequacy of CoC selection and behavioral characterization;
- Adequacy of fate and transport evaluations, including station selection, spatial (horizontal) and vertical (sediment layering) patterns, and sample representativeness;

- Adequacy of characterization of temporal/spatial variability in CoC distribution; and
- Reliability of exposure point estimation methods, including SEM/AVS calculations, porewater extraction techniques, etc.

Uncertainties discussed in the effects assessment phase (Section 5.5) included:

- Adequacy of toxicity data, including comparability among test species and methods,
- Adequacy of biological investigations, including the appropriateness of the benthic community structure and condition endpoints measured, data analysis techniques, data availability limitations, taxonomic identification and inference as to the relative sensitivity of various species to pollutants,
- Lack of chemical concentration benchmarks for tissue residues,
- Adequacy and availability of national criteria as benchmarks, and
- Appropriateness of the selected bioassay species as representative of the indigenous community.

These exposure and effects uncertainties compound one another as exposure and effects data are integrated in the risk characterization. In addition to these uncertainties, there are additional uncertainties which are unique to the risk characterization, including:

- Limited toxicological data for target receptor species,
- Incomplete knowledge of community ecology including natural history (e.g., size of feeding range and site use) of many species, species sensitivities to contaminants and trophic level transfers, and natural changes and variability in biological/ecological systems, and

- Adequacy of bioaccumulation and toxicological models.

Tissue residues can be used as an indication of exposure; however, their importance in ecological risk assessments is currently limited since evidence linking ecological effects directly with contaminant concentrations in tissue is generally lacking. In addition, more complete understanding of bioaccumulation and trophic transfer is required to evaluate the role of tissue residues in the status of natural resources, and to provide data for evaluating risks to human health associated with seafood consumption.

The utility of field effects indicators including community structure measurements, such as the relative abundance of pollutant tolerant species, has considerable uncertainty with regard to ecological significance. For instance, it is unclear whether an increase in the pollution tolerant group is occurring at the detriment of other groups, or whether shifts in relative abundances adversely impact food web dynamics. In addition, the seasonal and temporal sensitivity to pollutants has not been assessed, and leads to uncertainty given that, for example, seasonal rainfall will affect groundwater leachate generation, or various life stages present at different times may have differential chemical sensitivity. There are seasonal changes in redox potential as well as the concentrations of sediment organic carbon and acid volatile sulfides. Samples collected during Phase II of the present assessment were not collected over multiple seasons, nor was the time of collection representative of annual minima. Each of these factors has the potential to effect both the toxicity and bioaccumulation of the CoCs.

The application of organic (BSAF) and inorganic (BAF) bioaccumulation models have several uncertainties. The BSAF model relies on an empirical assumption that porewater concentrations are in equilibrium with sediment concentrations. This may not be the case, especially at sites such as McAllister Landfill where CoC releases could (and likely are) episodic and variable tidally, daily, seasonally and over the life of the

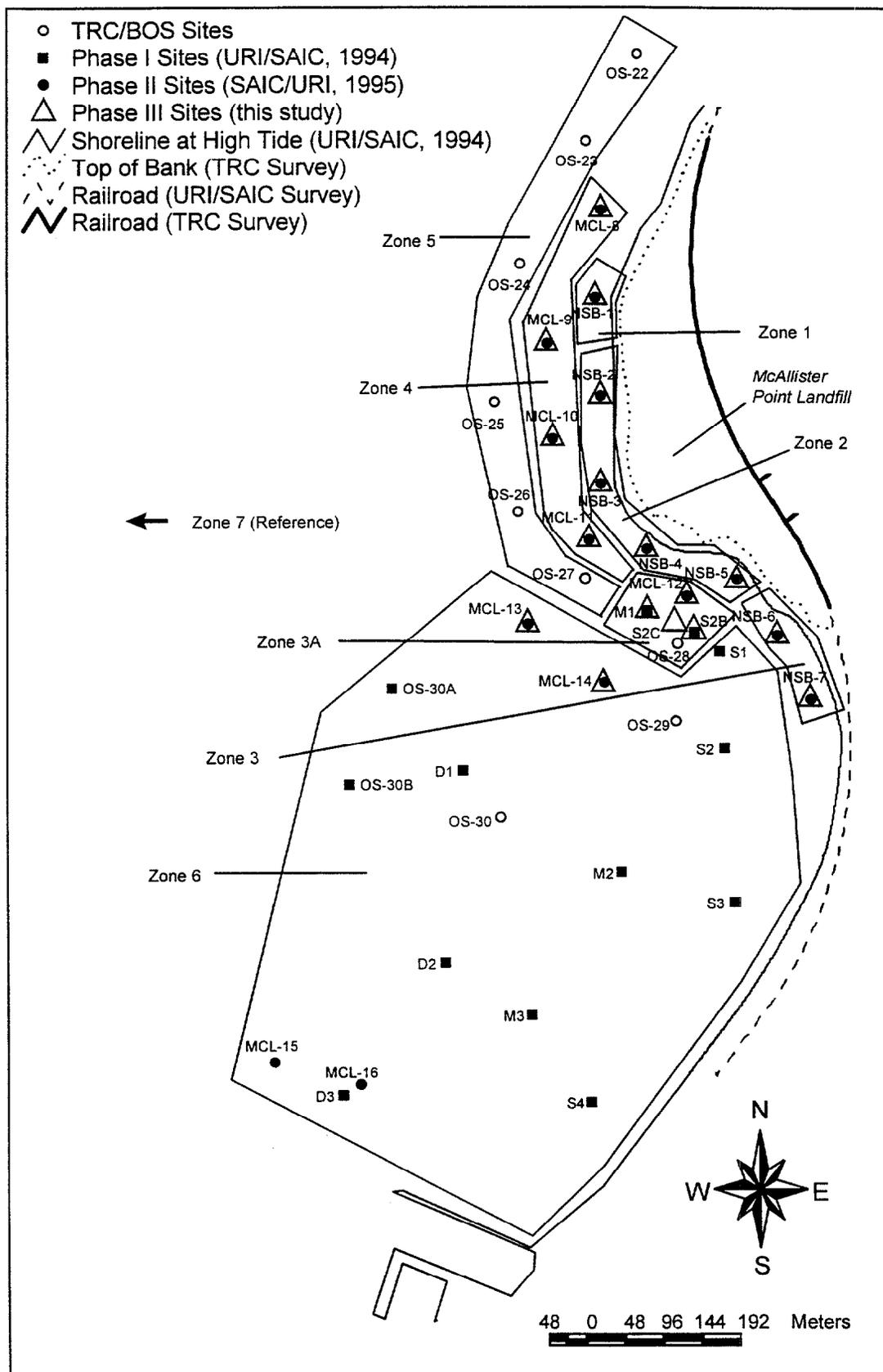
landfill. Uncertainty with BAF models (e.g. species-specific bioaccumulation patterns for various metals) is highly site-specific and may vary among species. To complete the analyses of tissue-based Hazard Quotients in this study, it was necessary to apply these models to generate the necessary reference benchmark concentrations. Thus, the resultant risk characterization carries uncertainty in this regard.

Uncertainties associated with the calculated Hazard Quotients and Hazard Indices exist because they do not necessarily reflect all chemicals or activities of chemical mixtures. In one portion of the assessment, an additive approach to HQs was taken in order to integrate multiple contaminant effects, since information is very limited on the toxicity of simultaneous exposure to mixtures of contaminants. However, this estimation does not incorporate potential synergistic interactions among chemicals; the sum of toxicities of individual chemicals may underestimate risk in some cases. Chemicals which were not measured always represent an exposure risk uncertainty. On the other hand, because a number of conservative indicators are used (e.g. ER-Ls), the estimates of risk are more likely to be overestimates than underestimates of true risks.

Given that Risk Characterization is a synthesis of findings from the Exposure and Effects Characterizations, it follows that uncertainties associated with these components of the Risk Assessment can be nullifying, additive or even compounded. A prime example is in the application of Hazard Quotients and derived Indices, where the numerator and denominator each represent point concentrations with an unknown departure from the "true" concentration. Toxicity-exposure relationships suffer the same uncertainty; separate error in estimates of survival and exposure concentration, for example, can compound or obscure true dose-response relationships or falsely suggest others which are misleading or unfounded.

The weight of evidence approach to characterization of risk is effective in reducing uncertainty because the probability that multiple exposure and effects indicators could spuriously suggest risk decreases as the number of indicators in agreement increases. However, this approach reduces uncertainty with respect to the location and magnitude of risk. It does not specifically address the ultimate source of this risk (i.e., the landfill vs. other CoC sources). This uncertainty has been addressed in the present study through analysis of spatial trends in CoCs, exposure pathways, and other endpoints (e.g. fecal pollution indicators) which might suggest alternative CoC sources.

Figure 6.0-1. Ecological Exposure Zones (EEZs) for the McAllister Point Landfill Marine Ecological Risk Assessment.



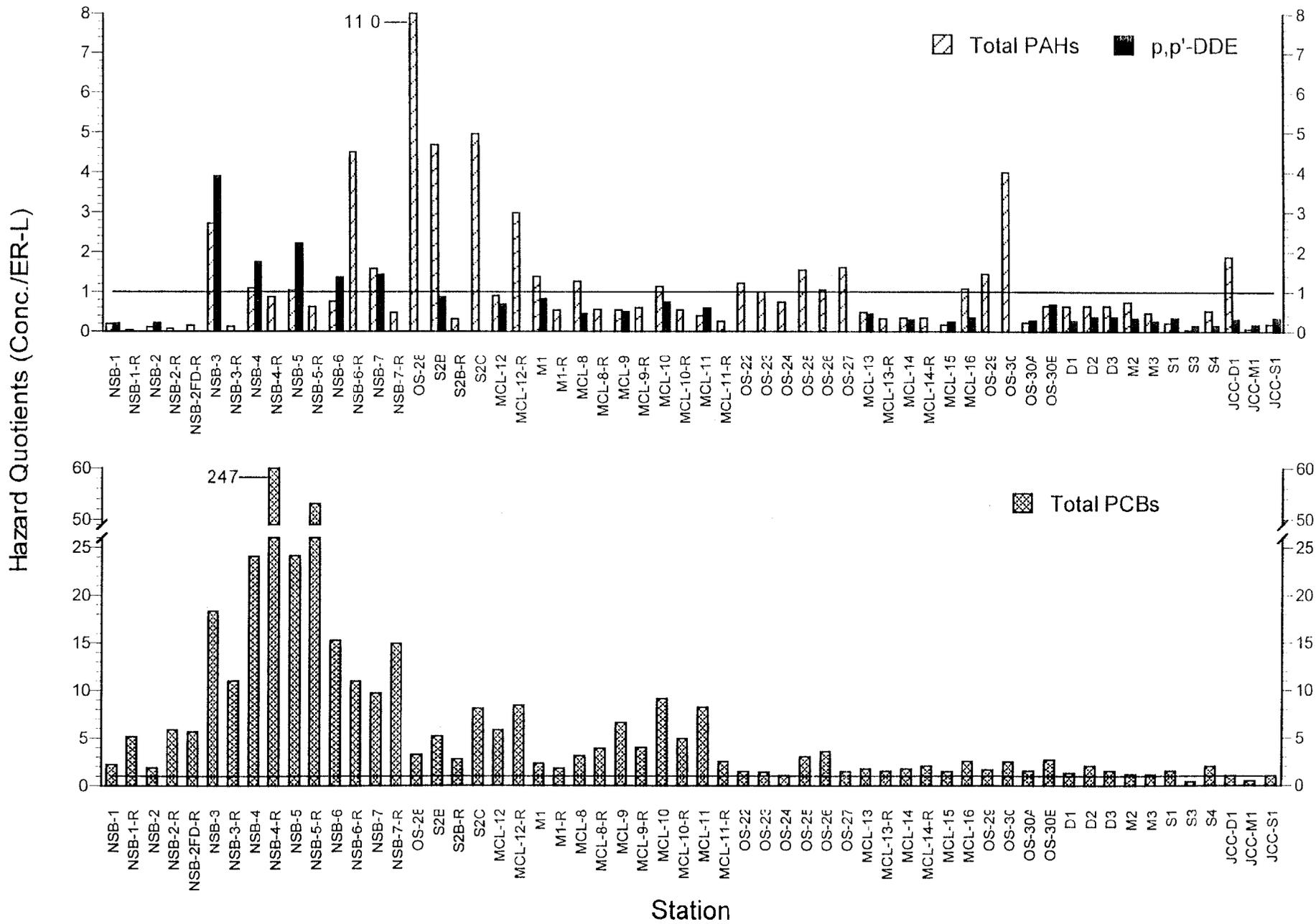


Figure 6.1-1. ER-L Hazard Quotients for organic contaminants in surface sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Solid line indicates HQ=1. Benchmark=NOAA ER-L values (Long *et al.*, 1995). See Figure 6.0-1 for station locations and sampling phases.

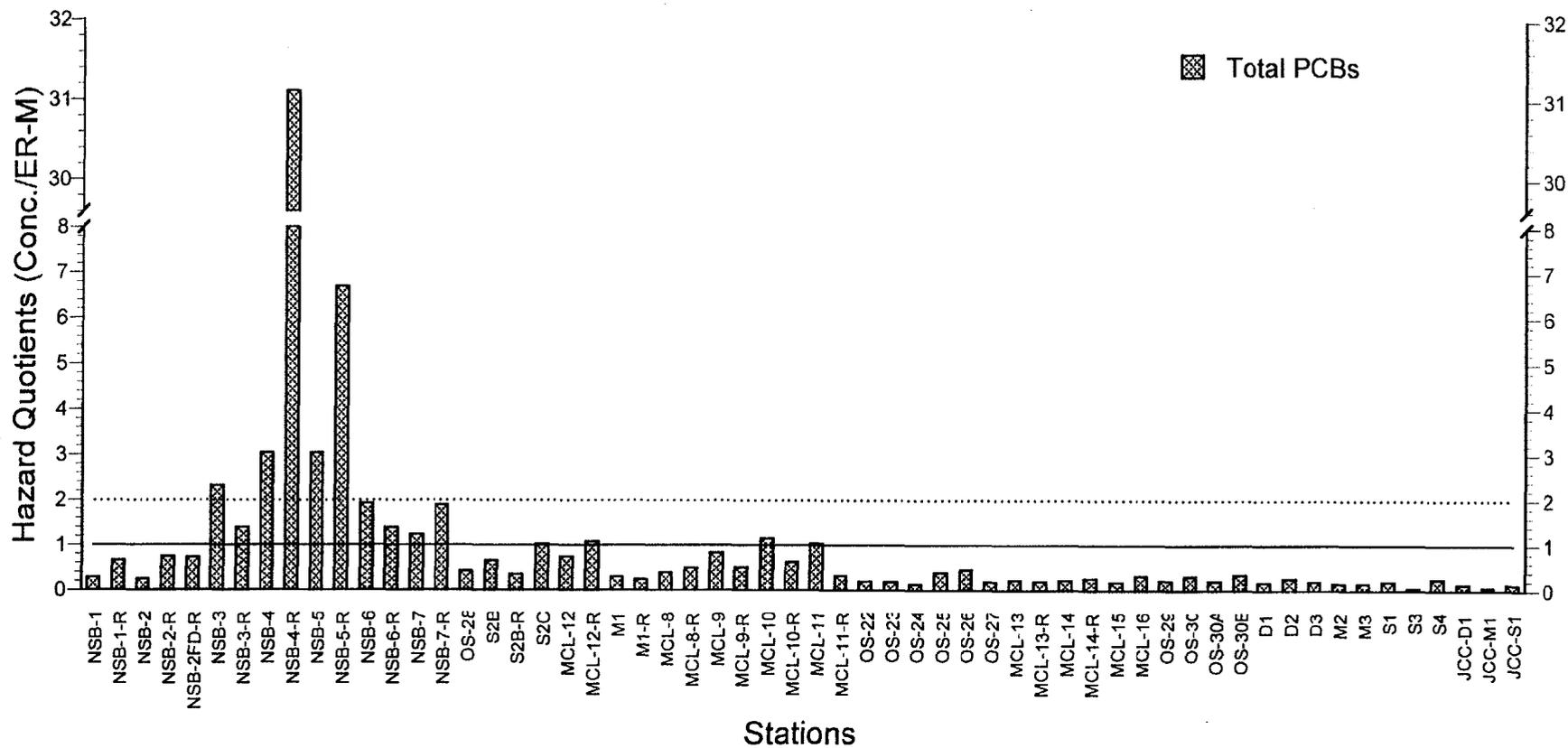


Figure 6.1-2. ER-M Hazard Quotients for Total PCBs in surface sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Solid line indicates HQ=1; dotted line indicates HQ=2. Benchmark=NOAA ER-M value (Long *et al.*, 1995). See Figure 6.0-1 for station locations and sampling phases.

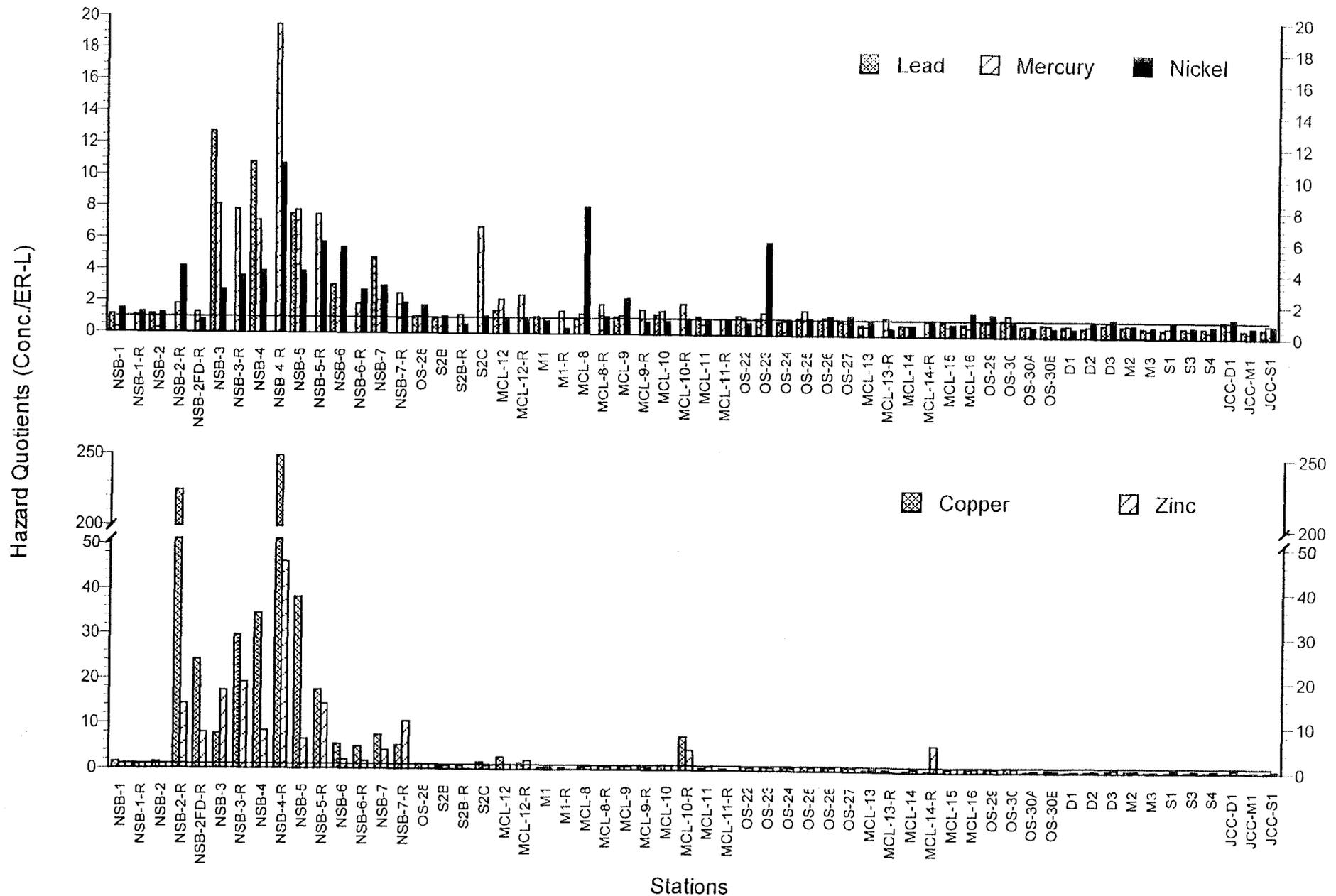


Figure 6.1-3. ER-L Hazard Quotients for metal contaminants in surface sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Solid line indicates HQ=1. Benchmark=NOAA ER-L values (Long *et al.*, 1995). See Figure 6.0-1 for station locations and sampling phases.

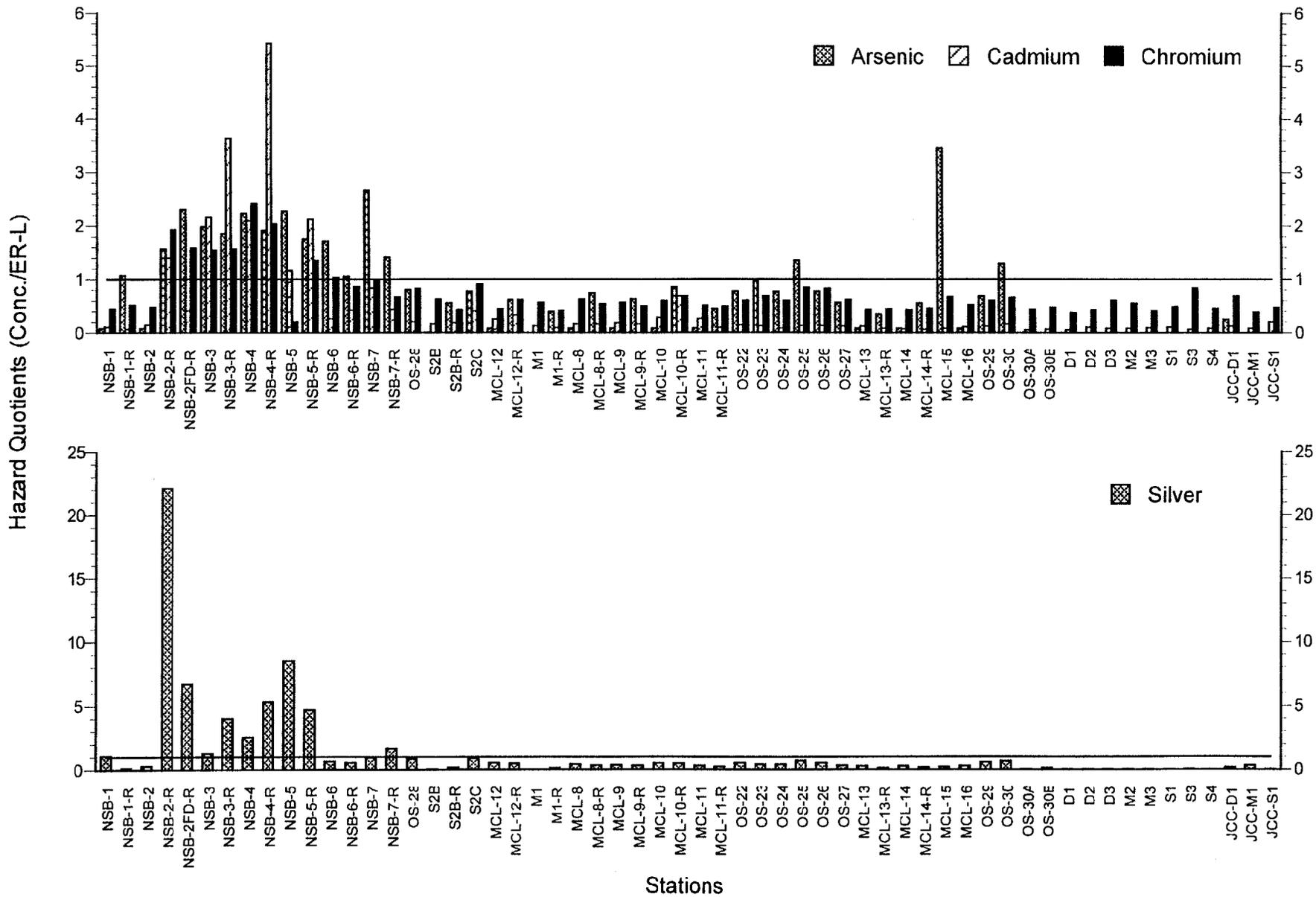


Figure 6.1-3 (continued). ER-L Hazard Quotients for metal contaminants in surface sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Solid line indicates HQ=1. Benchmark=NOAA ER-L values (Long *et al.*, 1995). See Figure 6.0-1 for station locations and sampling phases.

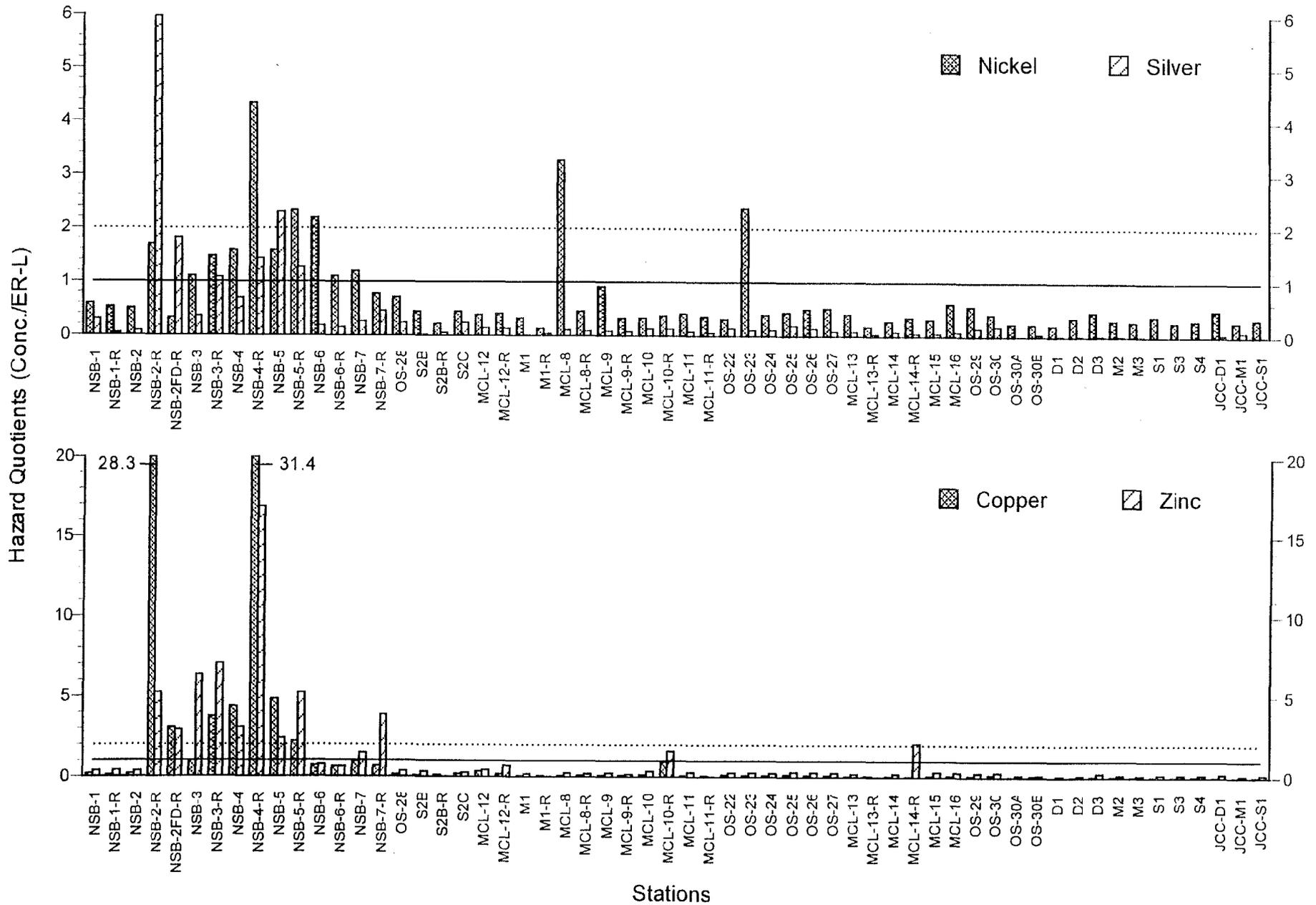


Figure 6.1-4. ER-M Hazard Quotients for metal contaminants in surface sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Solid line indicates HQ=1; dotted line indicates HQ=2. Benchmark=NOAA ER-M values (Long *et al.*, 1995). See Figure 6.0-1 for station locations and sampling phases.

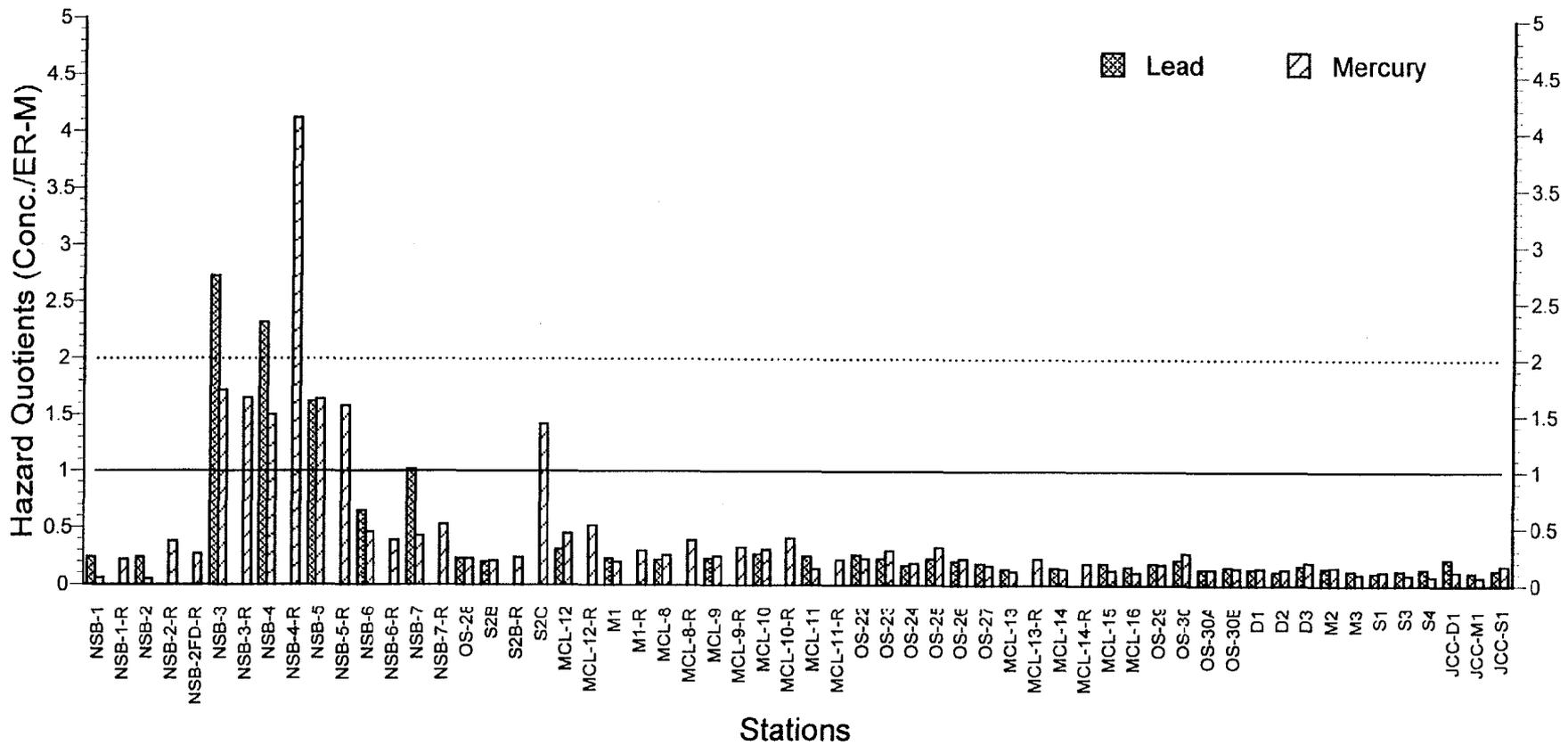


Figure 6.1-4 (continued). ER-M Hazard Quotients for metal contaminants in surface sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Solid line indicates HQ=1; dotted line indicates HQ=2. Benchmark=NOAA ER-M value (Long *et al.*, 1995). See Figure 6.0-1 for station locations and sampling phases.

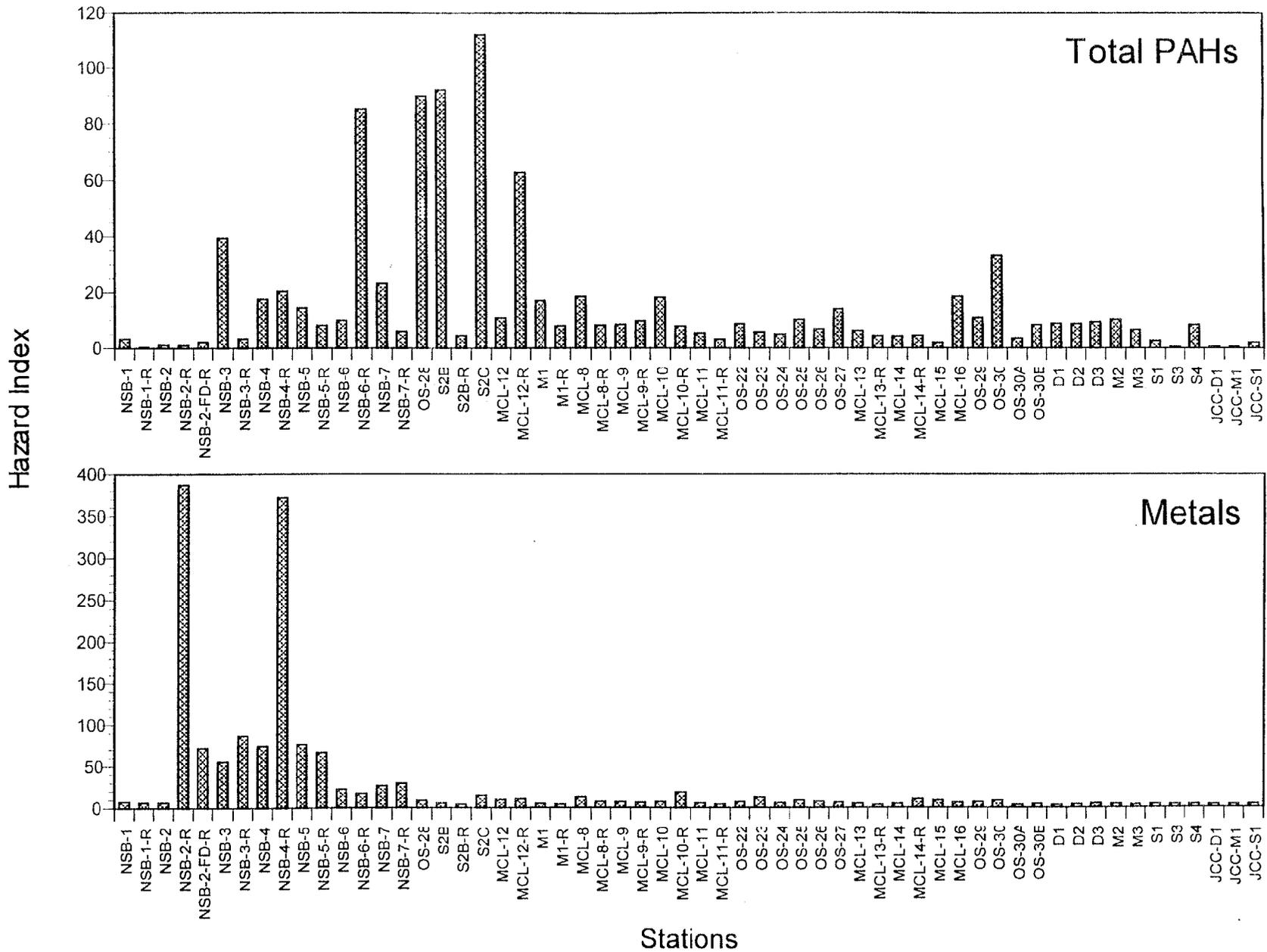
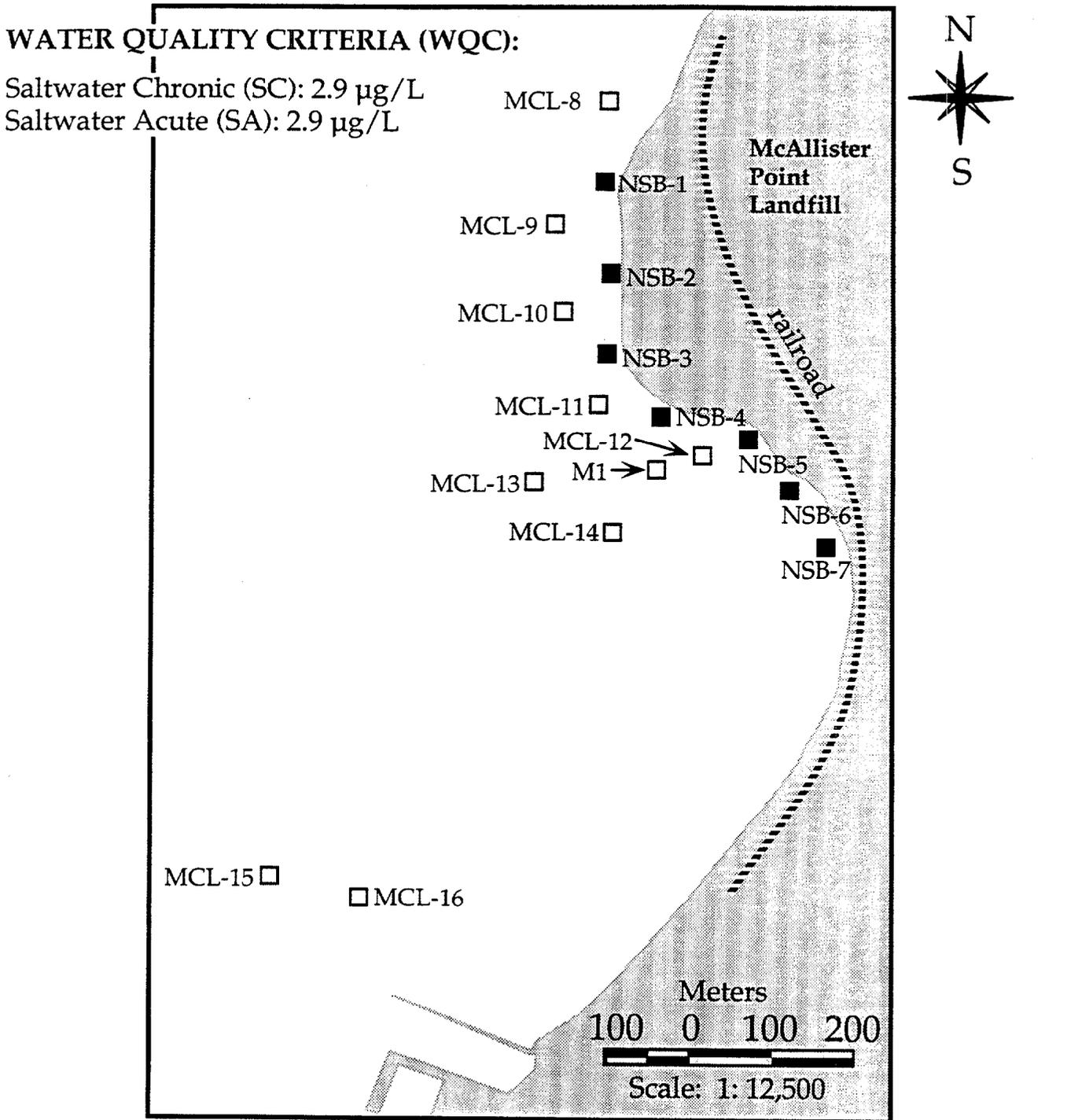


Figure 6.1-5. Hazard Indices (HIs) for Total PAHs and metals in surface sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. HIs based on ER-L HQs. See Figure 6.0-1 for station locations and sampling phases.

Figure 6.1- 6. Concentrations of copper in porewater relative to EPA water quality criteria in the McAllister Point Landfill study area.



- concentration less than WQC-SC
- ▨ concentration greater than WQC-SC but less than WQC-SA
- concentration greater than WQC-SA

Figure 6.1- 7. Concentrations of nickel in porewater relative to EPA water quality criteria in the McAllister Point Landfill study area.

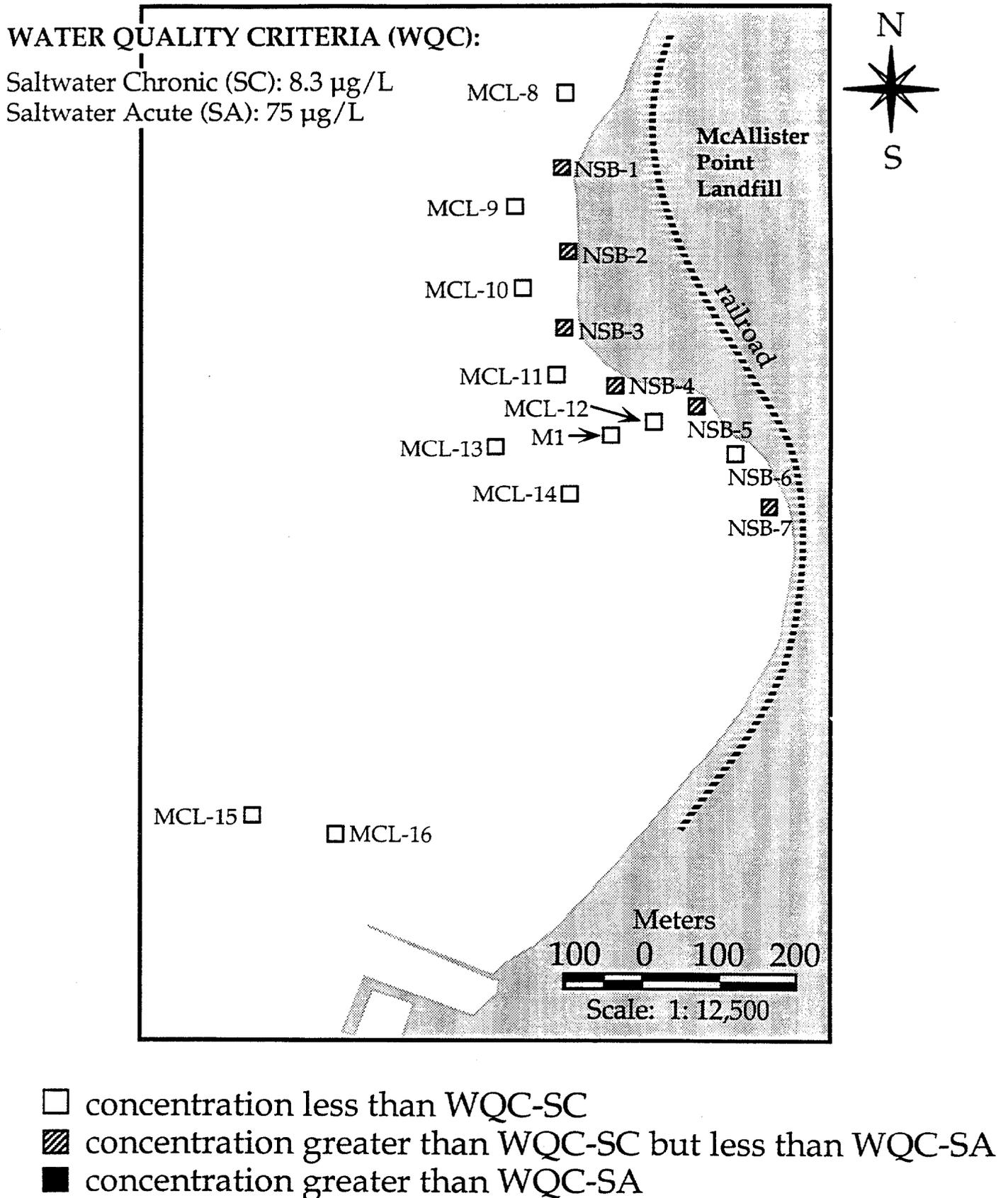
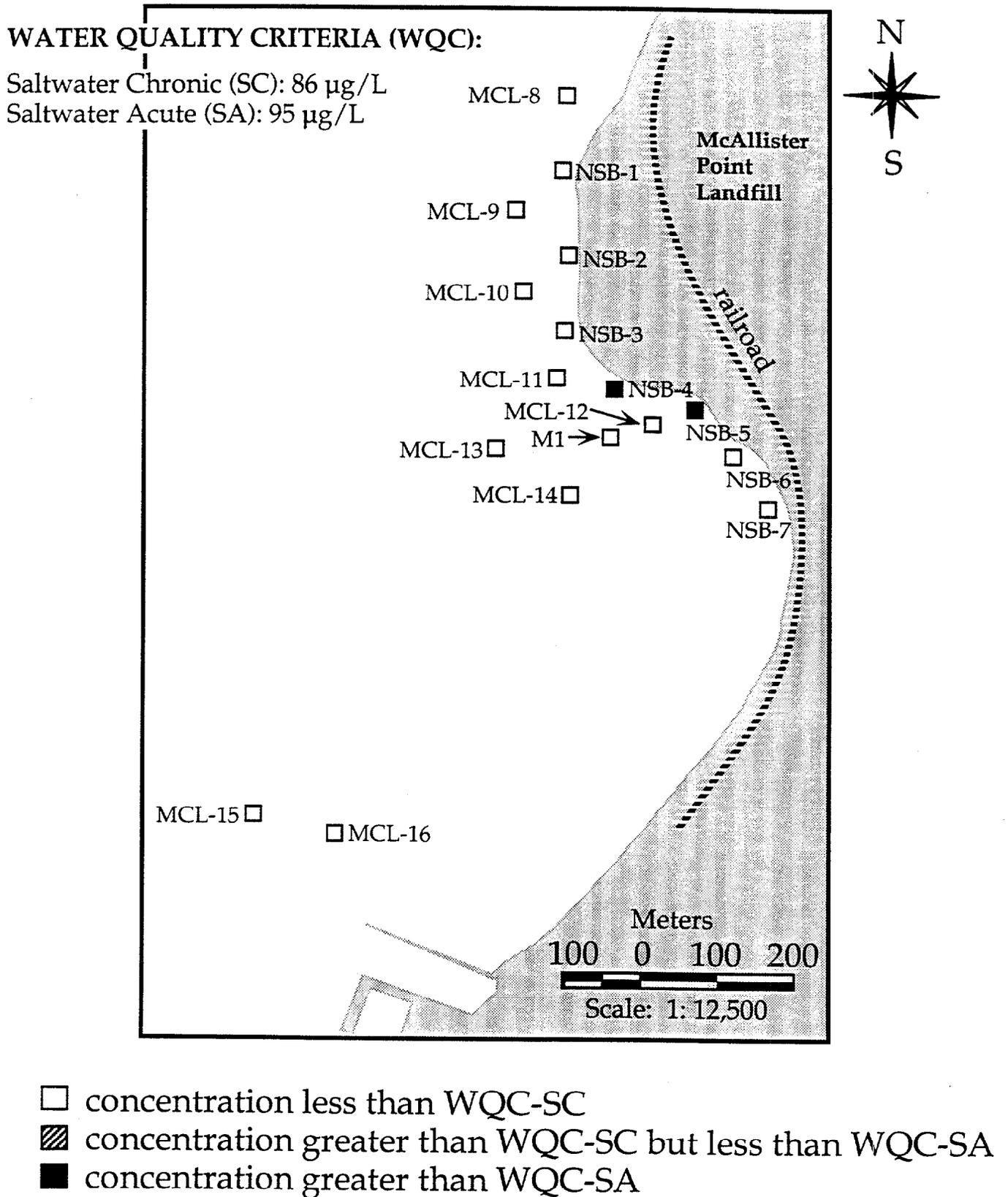


Figure 6.1- 8. Concentrations of zinc in porewater relative to EPA water quality criteria in the McAllister Point Landfill study area.



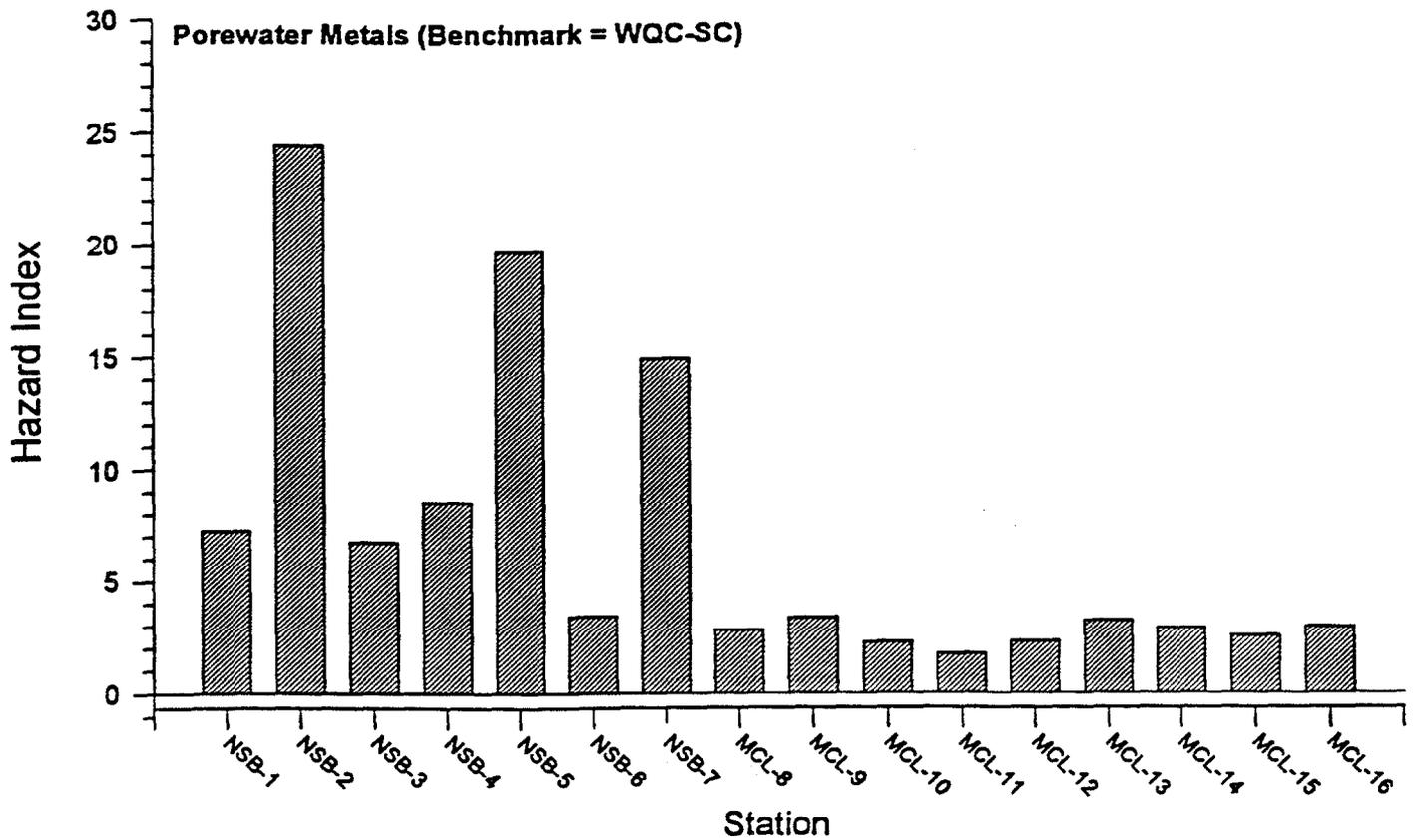


Figure 6.1- 9. Hazard Indices for metals in porewater from the McAllister Point Landfill study area. Criteria = EPA Water Quality Criteria Salt Water Chronic Values (WQC-SC)

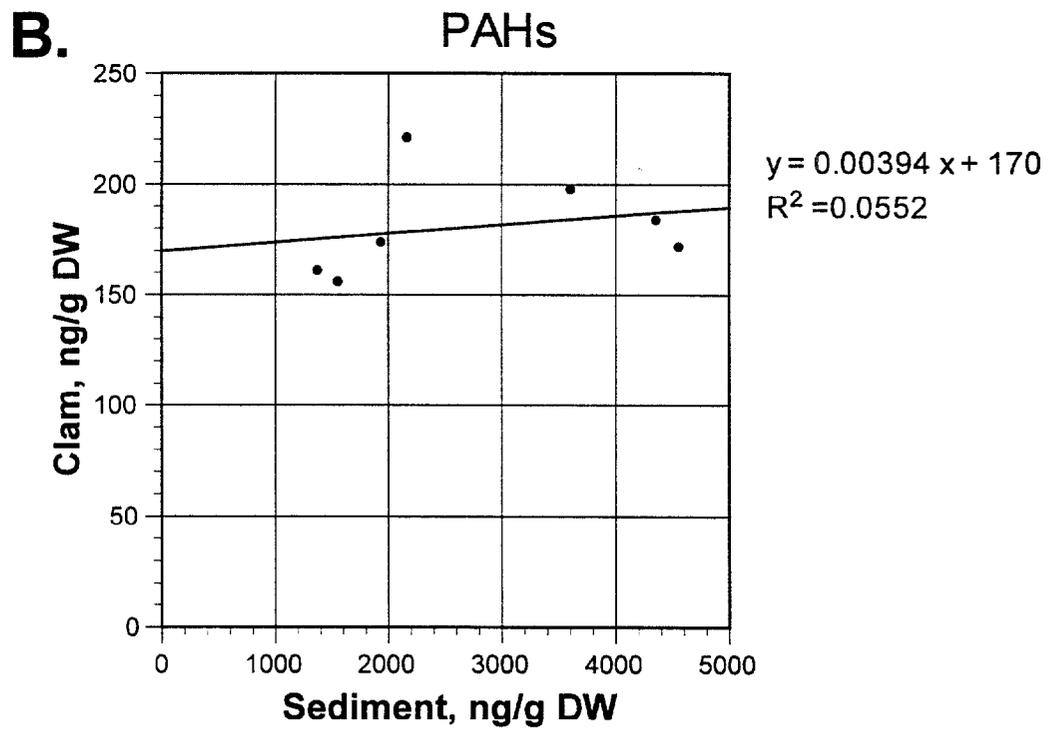
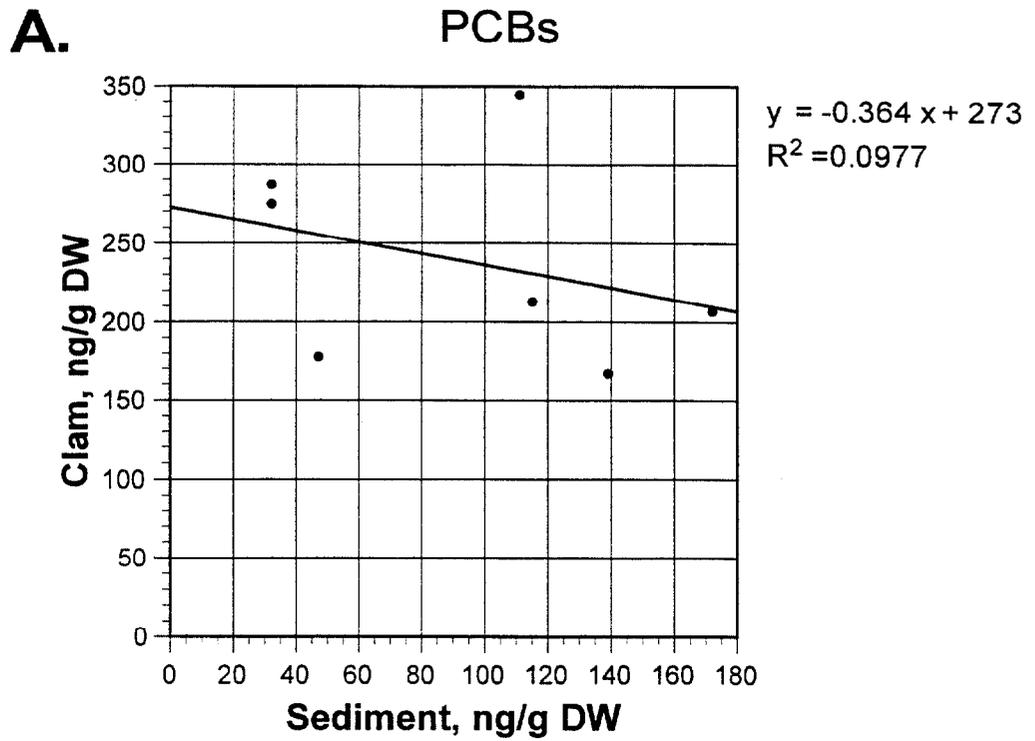


Figure 6.3-1. Organic contaminants in hard clams vs. surface sediments from seven McAllister Point Landfill study area stations. Surface sediments depths are 0-2 cm. A) Total PCBs, B) Total PAHs, C) p,p'-DDE, and D) TBT.

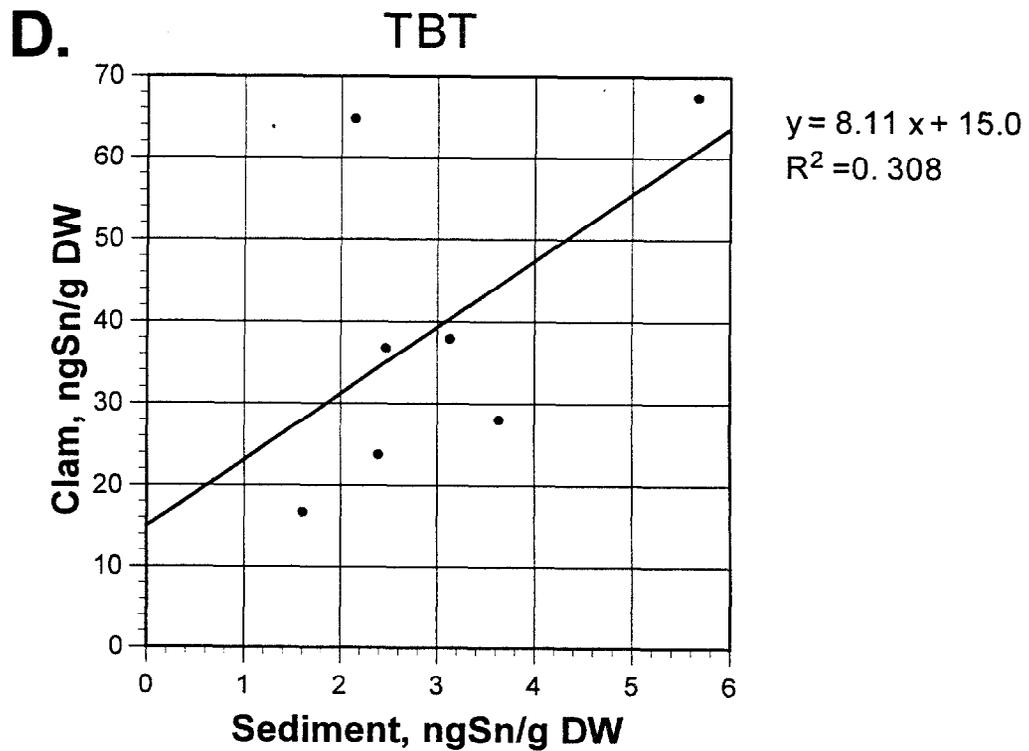
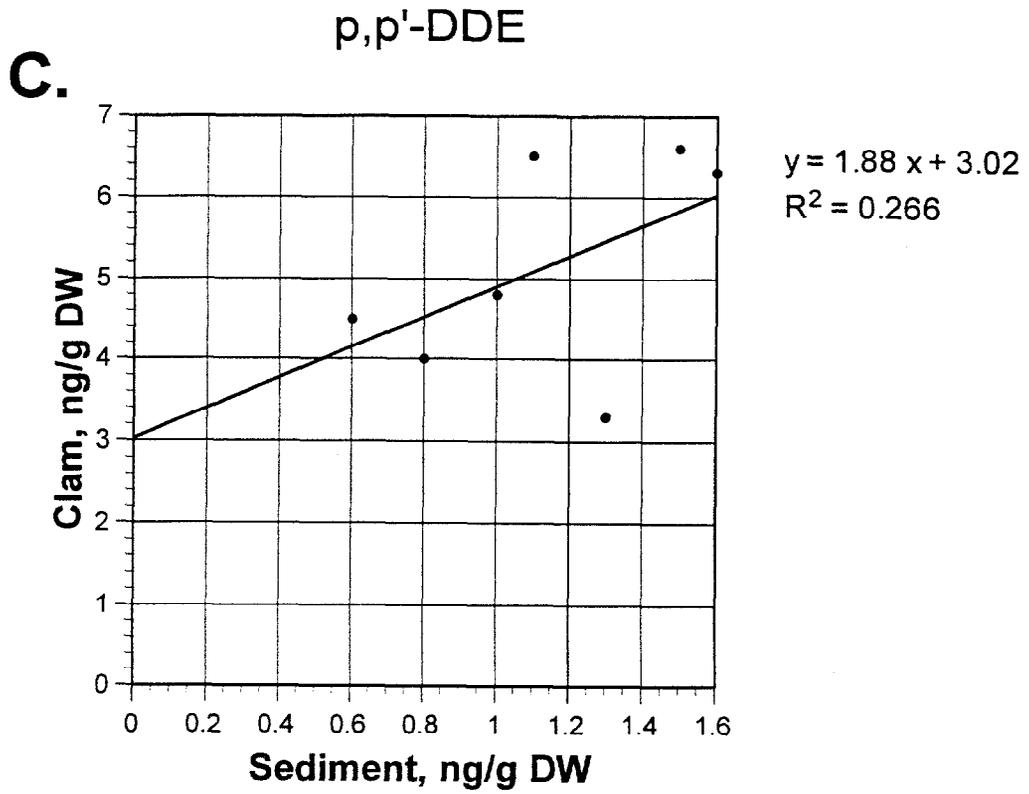
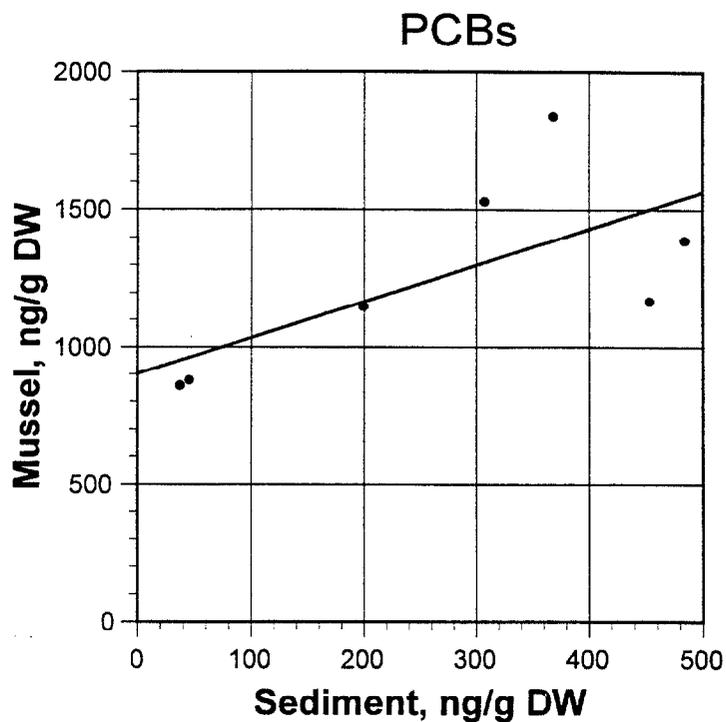


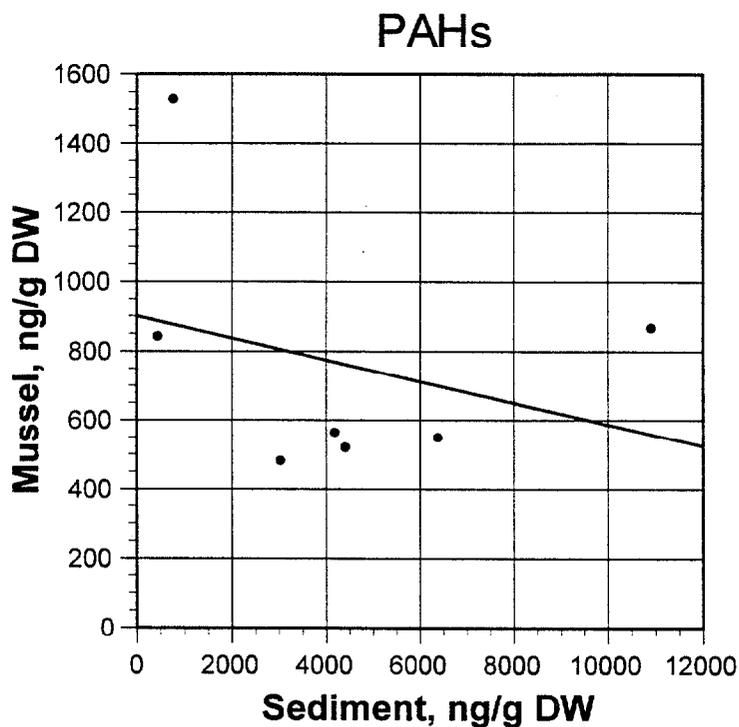
Figure 6.3-1 (continued). Organic contaminants in hard clams vs. surface sediments from seven McAllister Point Landfill study area stations. Surface sediments depths are 0-2 cm. A) Total PCBs, B) Total PAHs, C) p,p'-DDE, and D) TBT.

**A.**



$$y = 1.32x + 902$$
$$R^2 = 0.466$$

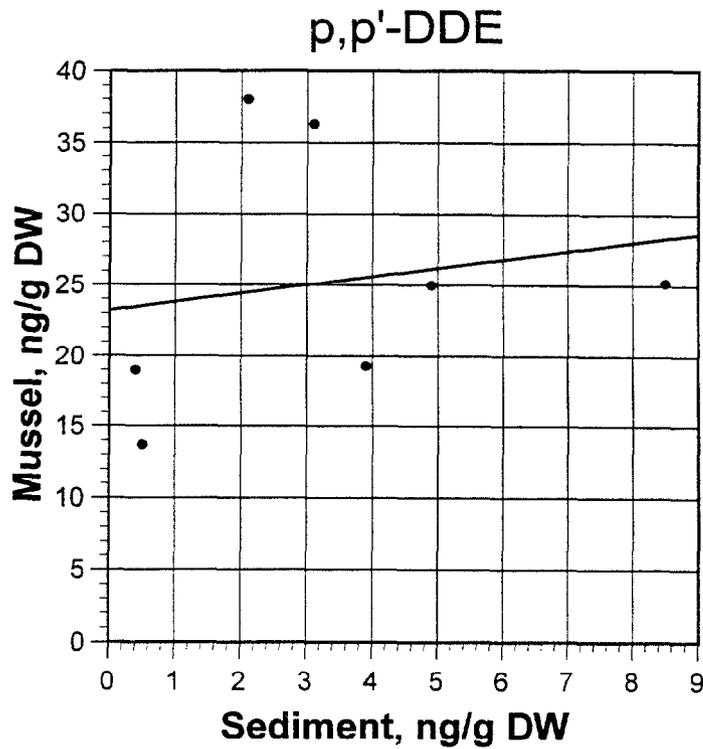
**B.**



$$y = -0.0313x + 900$$
$$R^2 = 0.0913$$

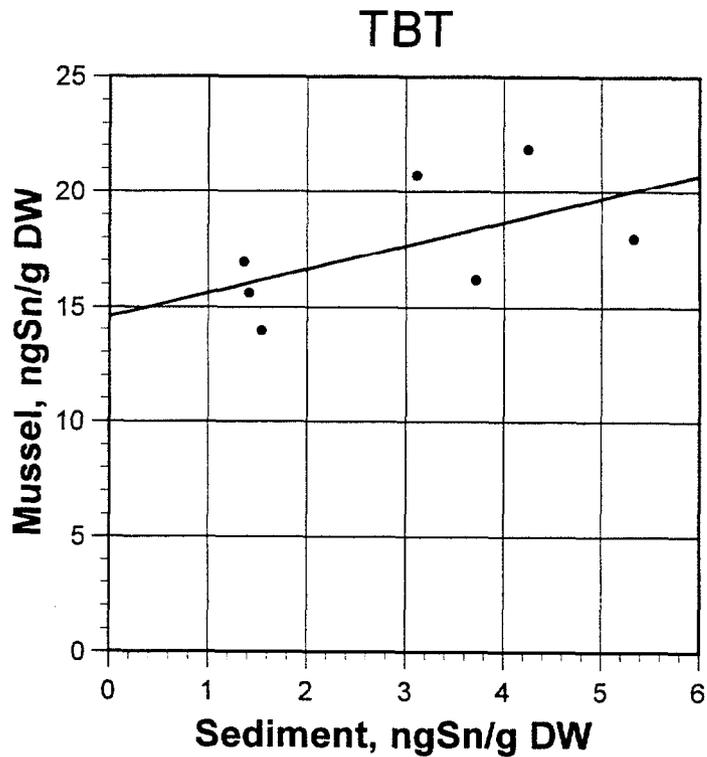
Figure 6.3-2. Organic contaminants in blue mussels vs. surface sediments from seven McAllister Point Landfill study area stations. Surface sediments depths are 0-2 cm. A) Total PCBs, B) Total PAHs, C) p,p'-DDE, and D) TBT.

C.



$$y = 0.603x + 23.2$$
$$R^2 = 0.0352$$

D.



$$y = 1.02x + 14.6$$
$$R^2 = 0.323$$

Figure 6.3-2 (continued). Organic contaminants in blue mussels vs. surface sediments from seven McAllister Point Landfill study area stations. Surface sediments depths are 0-2 cm. A) Total PCBs, B) Total PAHs, C) p,p'-DDE, and D) TBT.

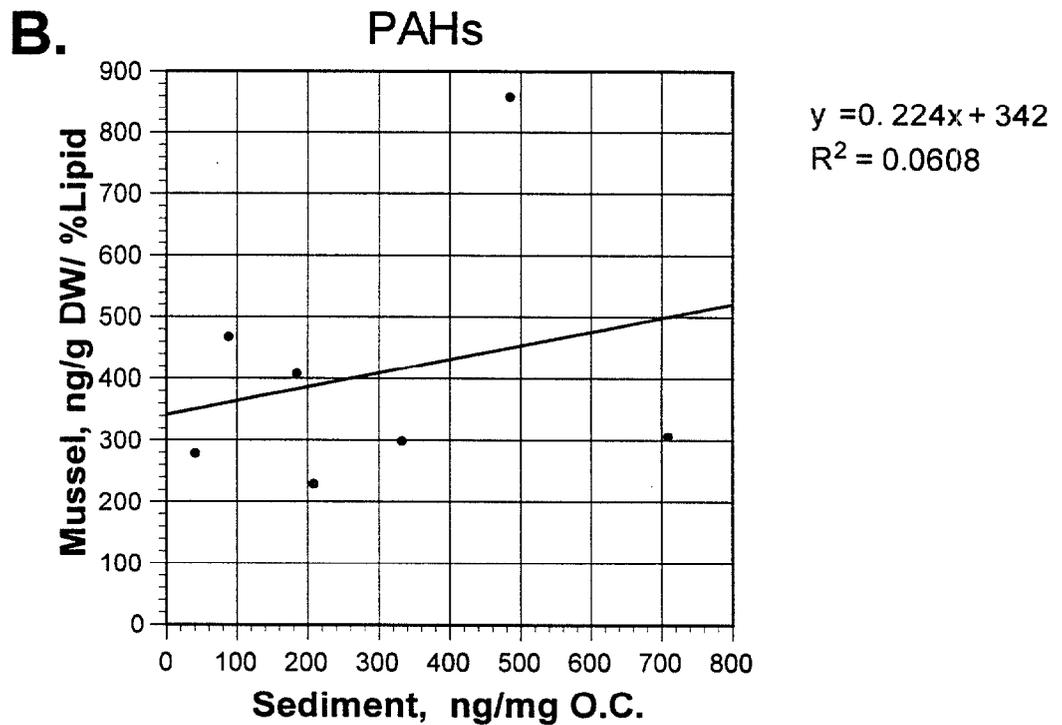
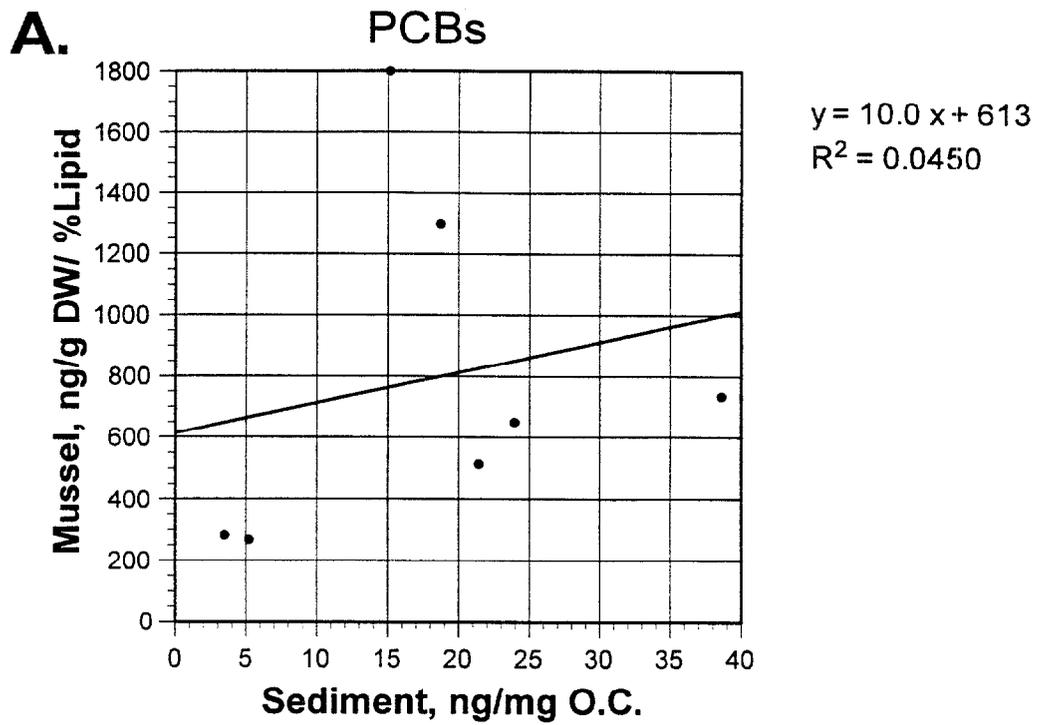


Figure 6.3-3. Organic contaminants (normalized to lipids) in blue mussels vs. surface sediments from seven McAllister Point Landfill study area stations. Surface sediments depths are 0-2 cm. A) Total PCBs, B) Total PAHs, C) p,p'-DDE, and D) TBT.

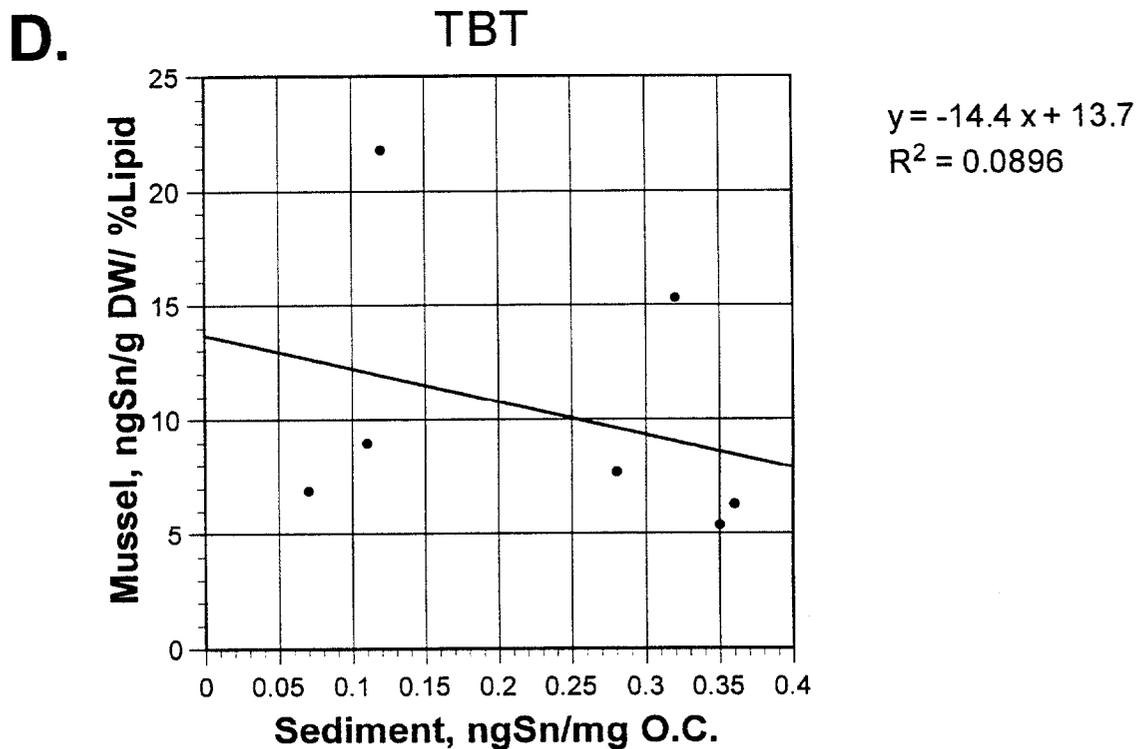
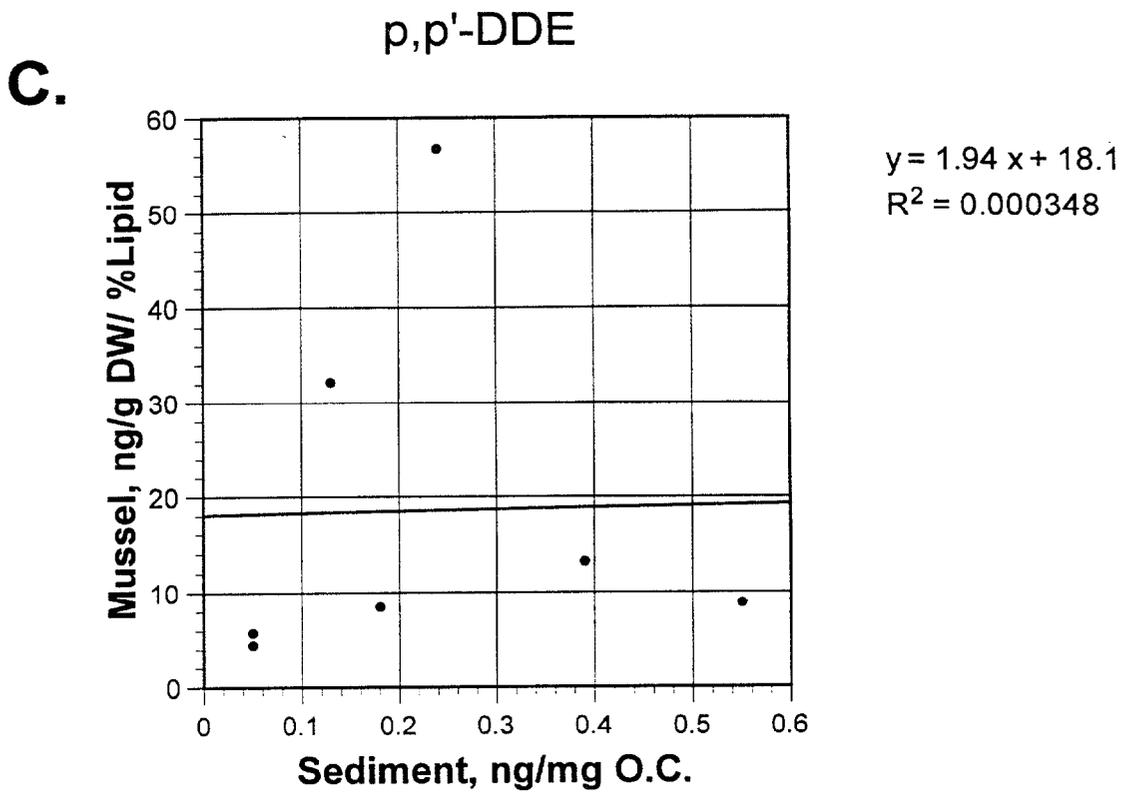


Figure 6.3-3 (continued). Organic contaminants (normalized to lipids) in blue mussels vs. surface sediments from seven McAllister Point Landfill study area stations. Surface sediments depths are 0-2 cm. A) Total PCBs, B) Total PAHs, C) p,p'-DDE, and D) TBT.

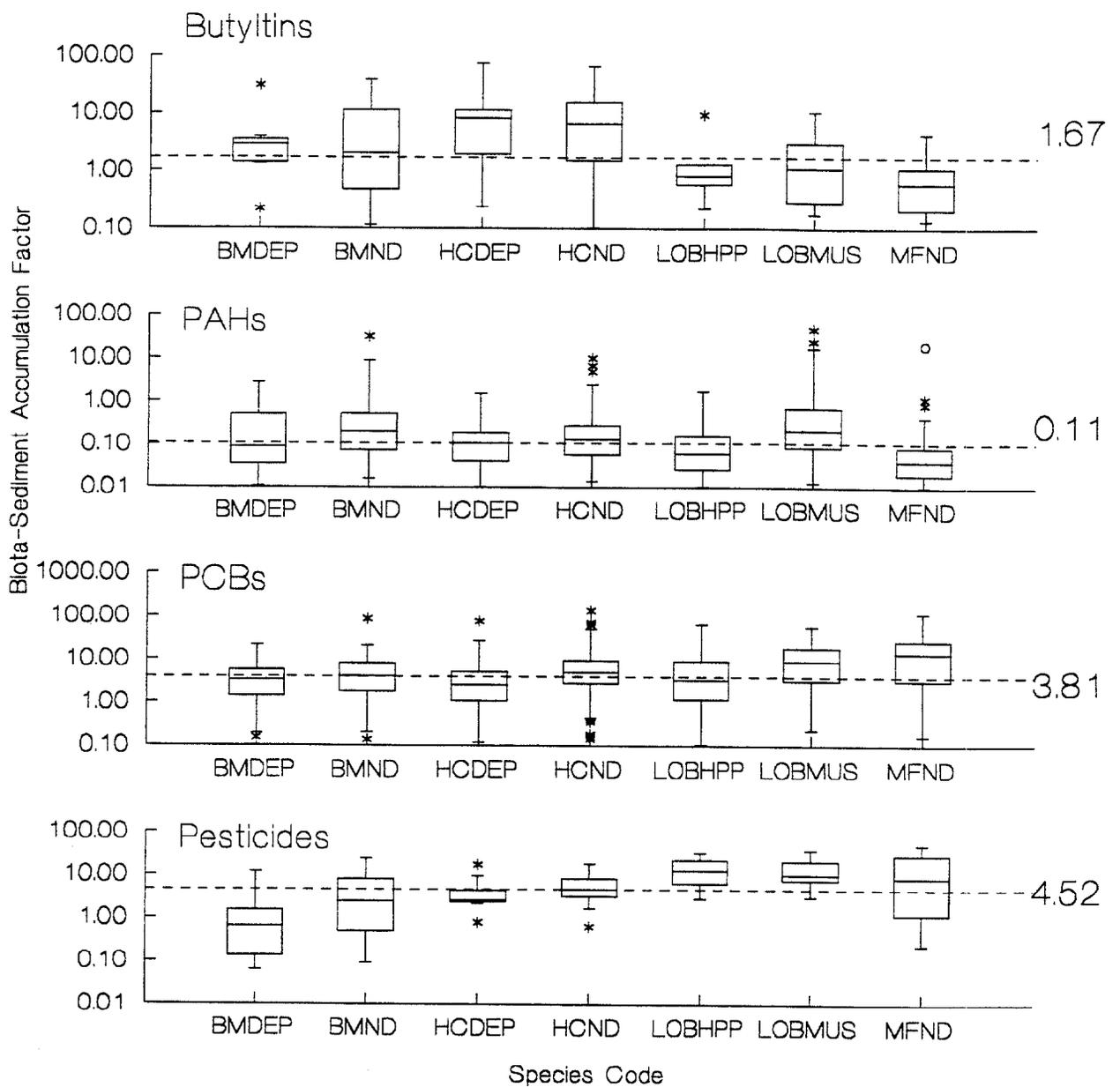


Figure 6.3-4. Box plots of Biota Sediment Accumulation Factors for organic contaminants in McAllister Point Landfill Marine ERA target receptors. The dashed line and number indicate the mean value for species groups. Codes: HC=hard clam; BM=blue mussel; LOBMUS=lobster muscle; LOBHPP=lobster hepatopancreas; MF=marine fish (cunner); DEP=depurated; ND=non-depurated. Refer to Section 6.3.1 in the text for explanations of box plot symbols.

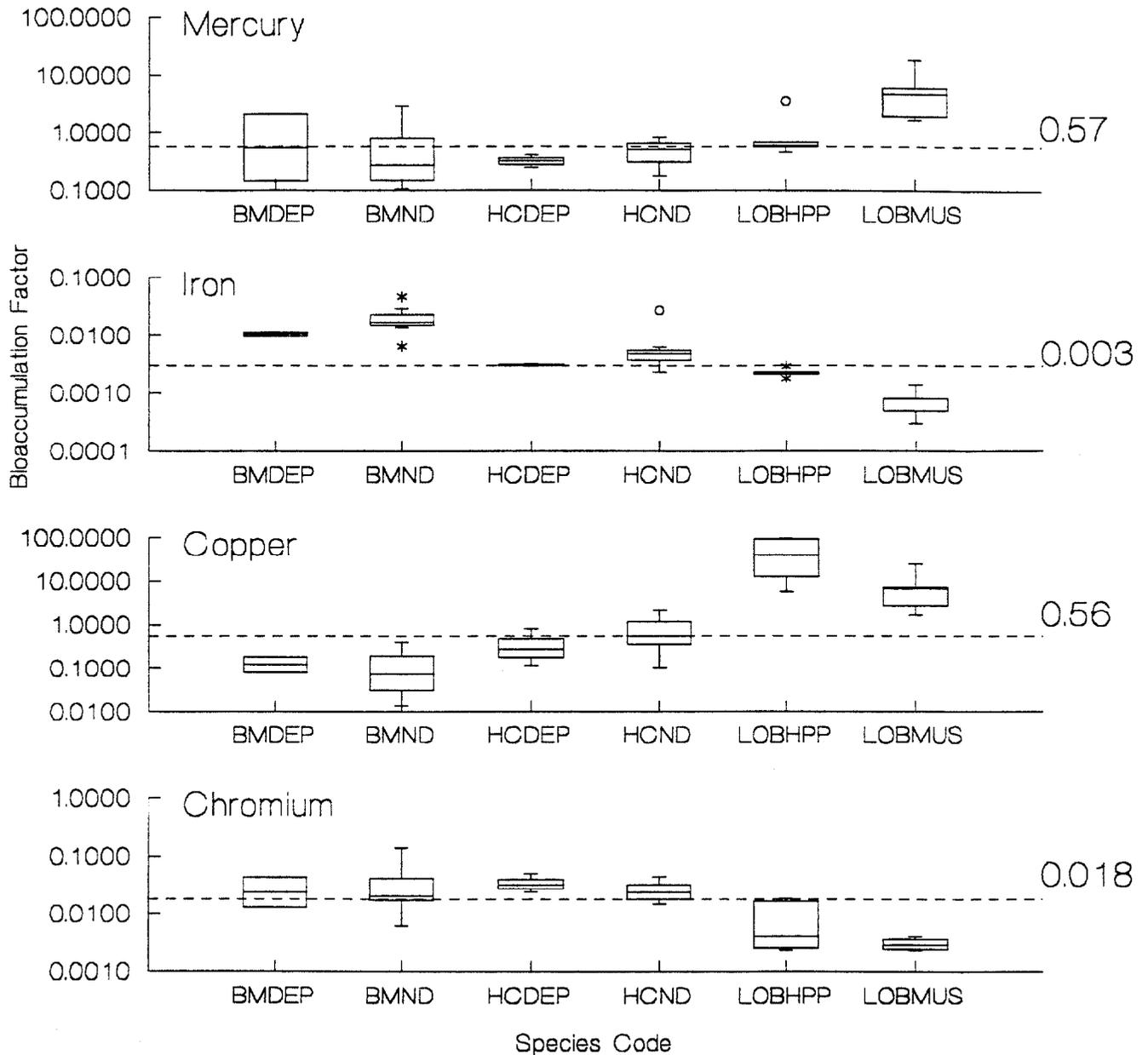


Figure 6.3-5. Box plots of Bioaccumulation Factors for metals in McAllister Point Landfill Marine ERA target receptors. The dashed line and number indicate the mean value for species groups. Codes: HC=hard clam; BM=blue mussel; LOBMUS=lobster muscle; LOBHPP=lobster hepatopancreas; MF=marine fish (cunner); DEP=depurated; ND=non-depurated. Refer to Section 6.3.1 in the text for explanations of box plot symbols.

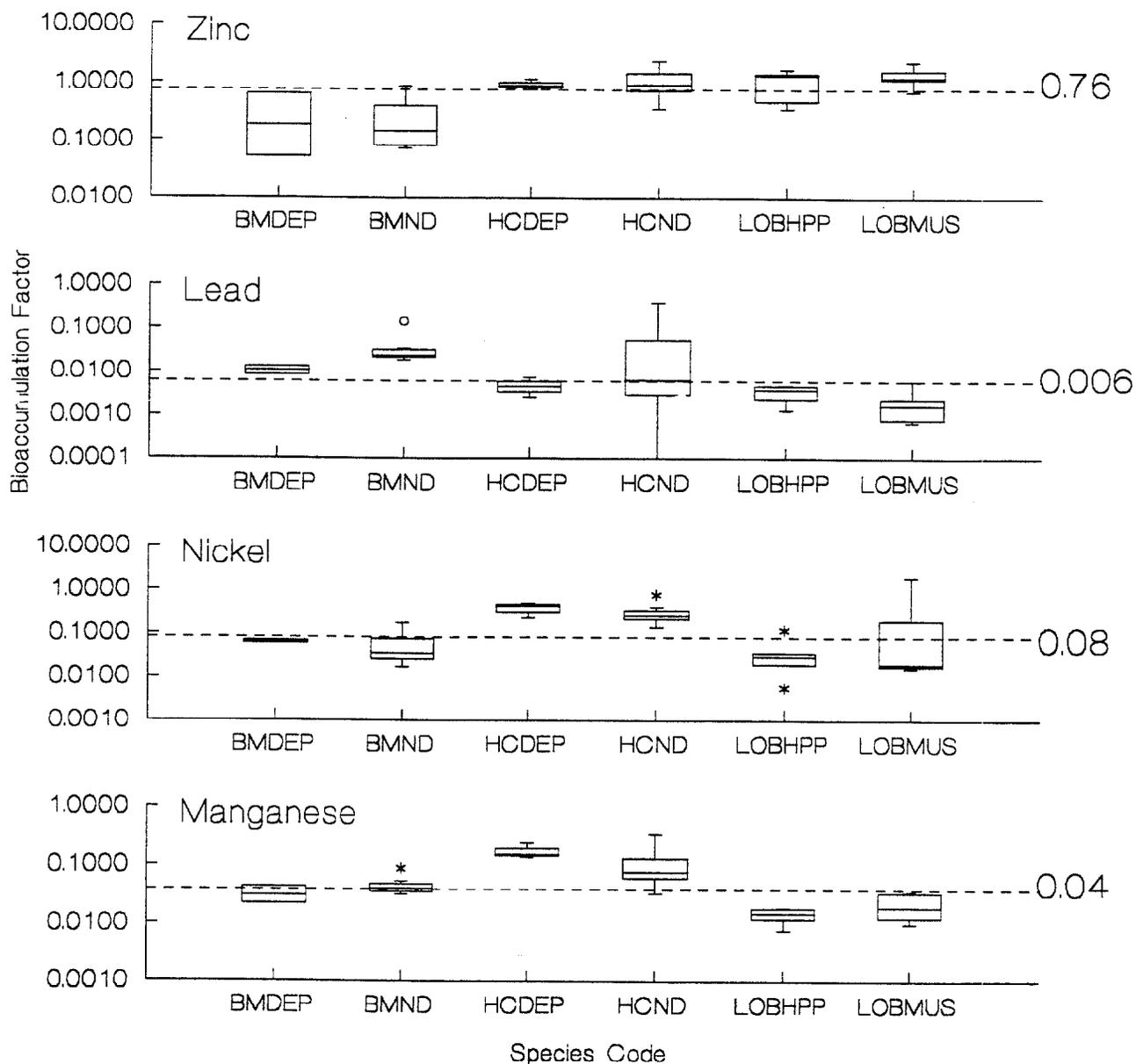


Figure 6.3-5 (continued). Box plots of Bioaccumulation Factors for organic contaminants in McAllister Point Landfill Marine ERA target receptors. The dashed line and number indicate the mean value for species groups. Codes: HC=hard clam; BM=blue mussel; LOBMUS=lobster muscle; LOBHPP=lobster hepatopancreas; MF=marine fish (cunner); DEP=depurated; ND=non-depurated. Refer to Section 6.3.1 in the text for explanations of box plot symbols.

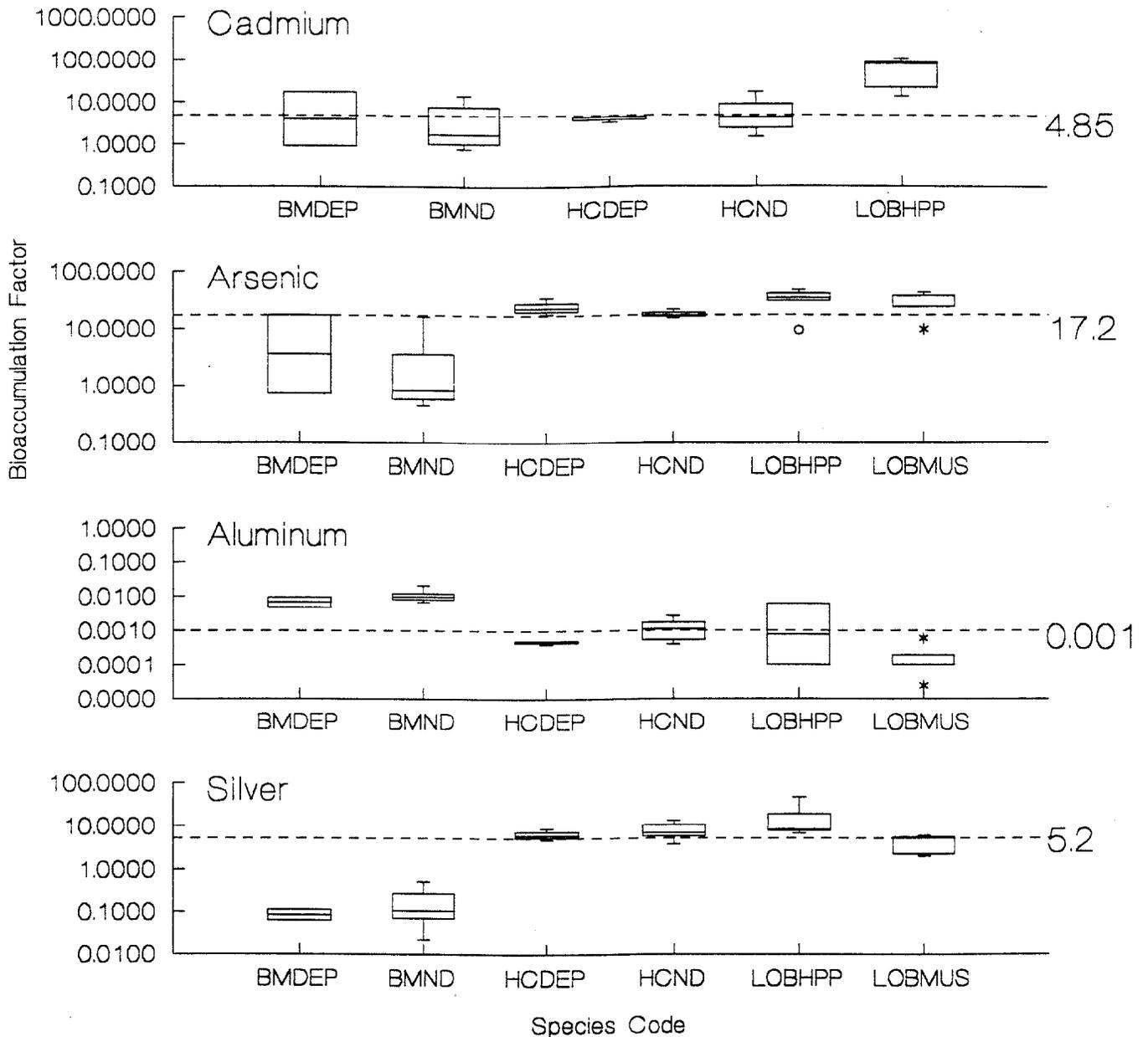


Figure 6.3-5 (continued). Box plots of Bioaccumulation Factors for organic contaminants in McAllister Point Landfill Marine ERA target receptors. The dashed line and number indicate the mean value for species groups. Codes: HC=hard clam; BM=blue mussel; LOBMUS=lobster muscle; LOBHPP=lobster hepatopancreas; MF=marine fish (cunner); DEP=deperated; ND=non-deperated. Refer to Section 6.3.1 in the text for explanations of box plot symbols.

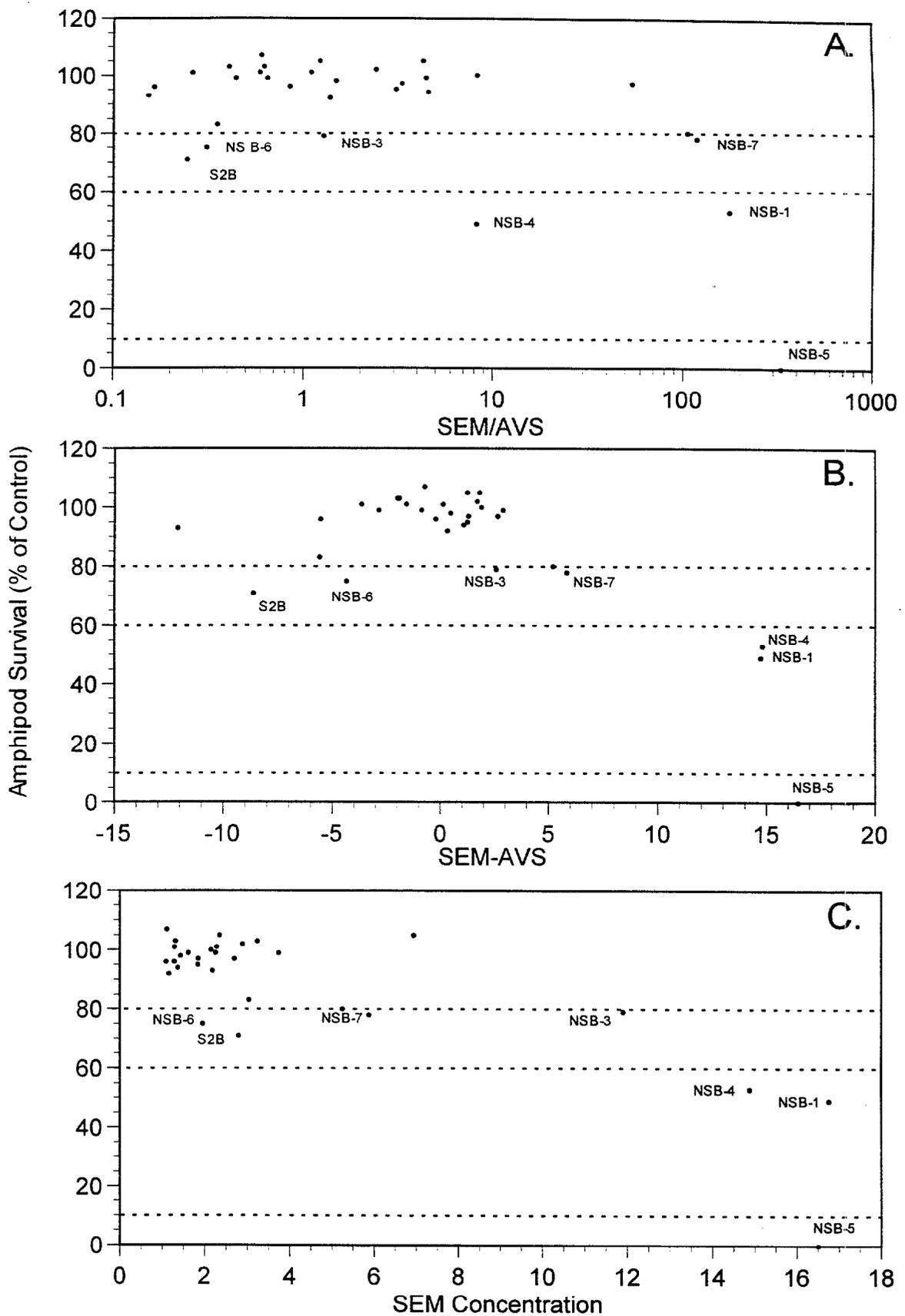


Figure 6.4-1. Amphipod survival versus A) SEM/AVS, B) SEM-AVS, and C) SEM concentration ( $\mu\text{mol/g}$ ) in whole sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. The dashed lines indicate threshold values for low (<80%), intermediate (60%), and high (<10%) impact on amphipod survival.

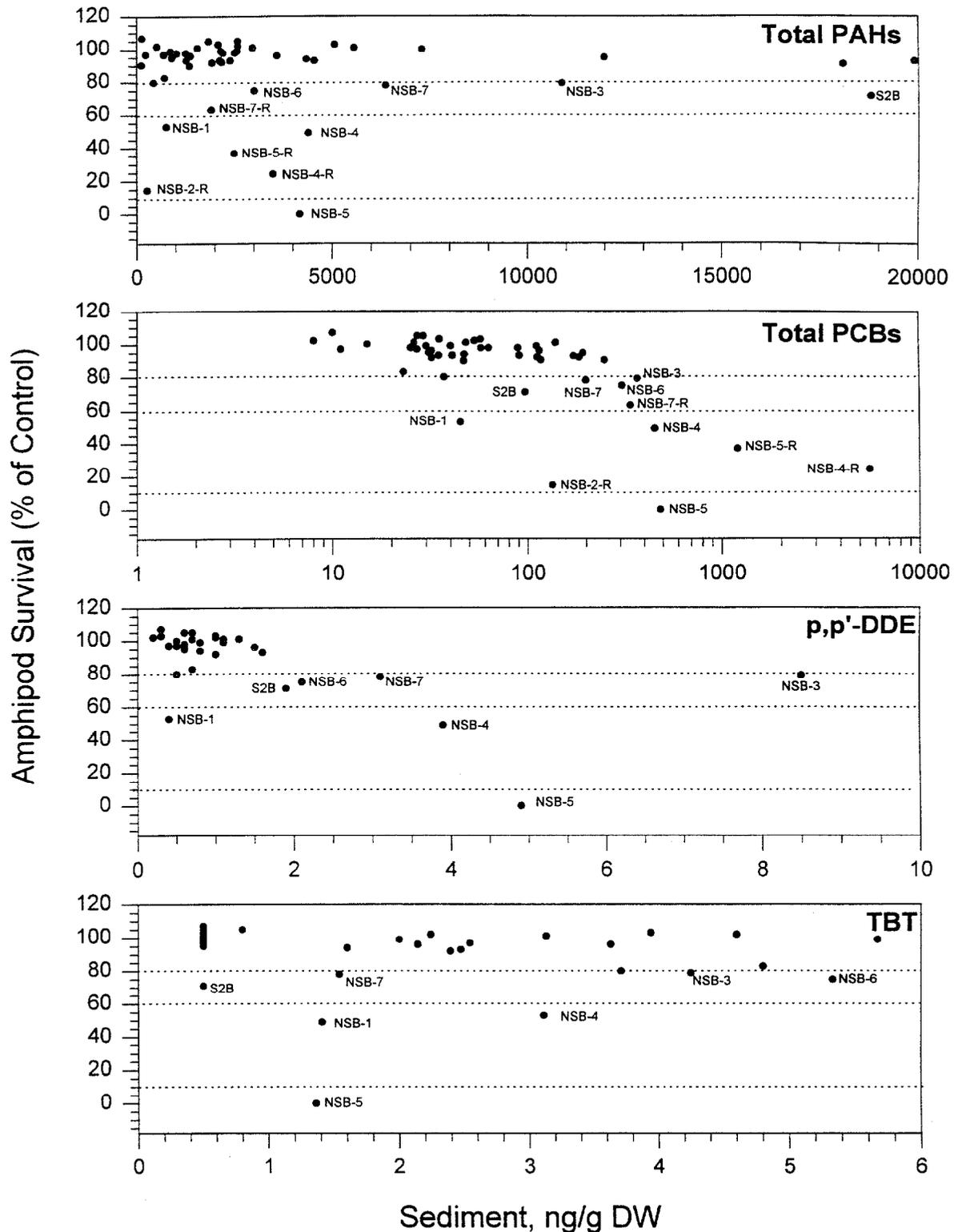


Figure 6.4-2. Amphipod survival vs. organic contaminants in sediments from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Dashed lines indicate threshold values for low (<80%), intermediate (<60%), and high (<10%) impact on amphipod survival.

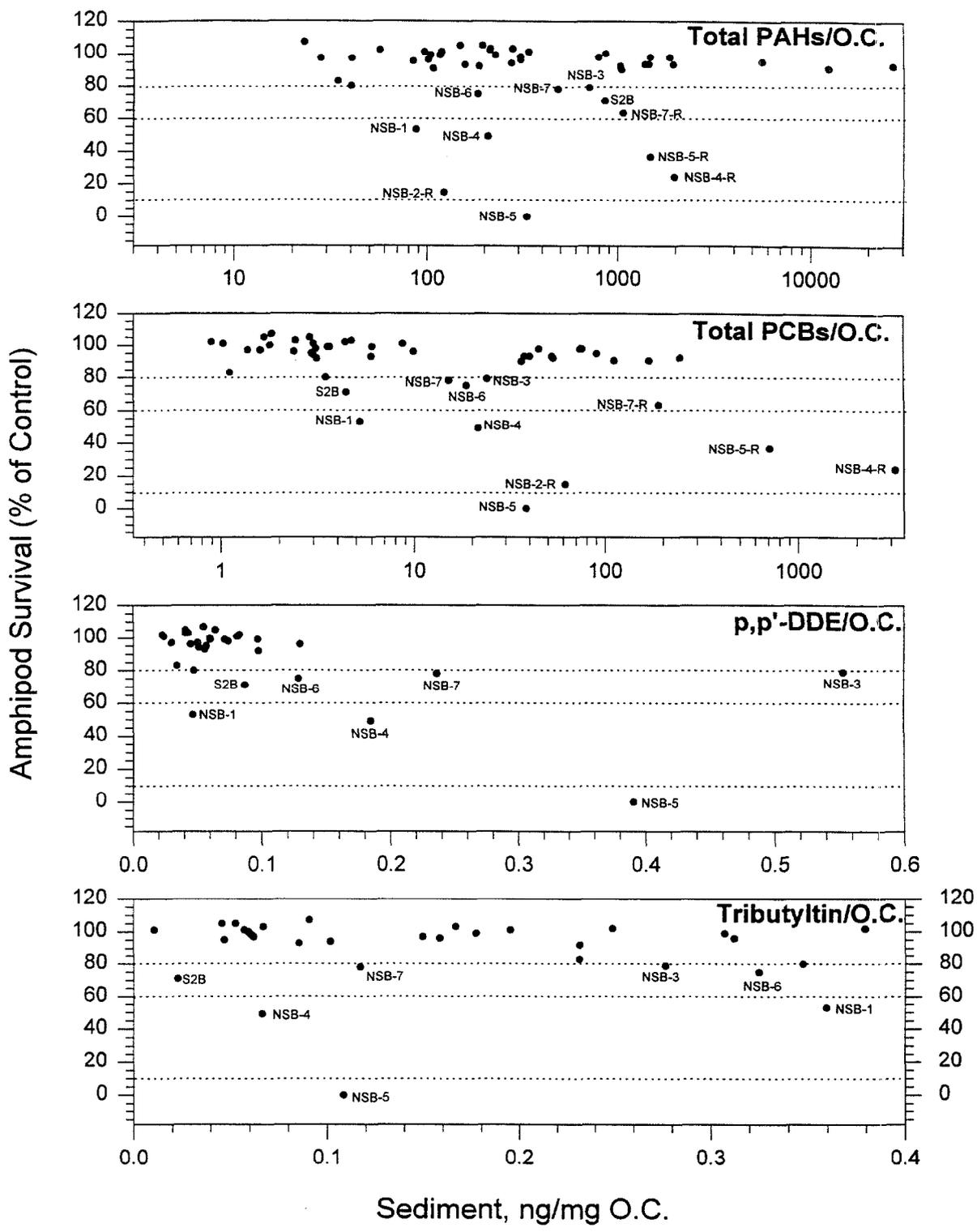


Figure 6.4-3. Amphipod survival vs. organic contaminants (normalized to TOC) in sediments from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Dashed lines indicate threshold values for low (<80%), intermediate (<60%), and high (<10%) impact on amphipod survival. O.C.=organic carbon.

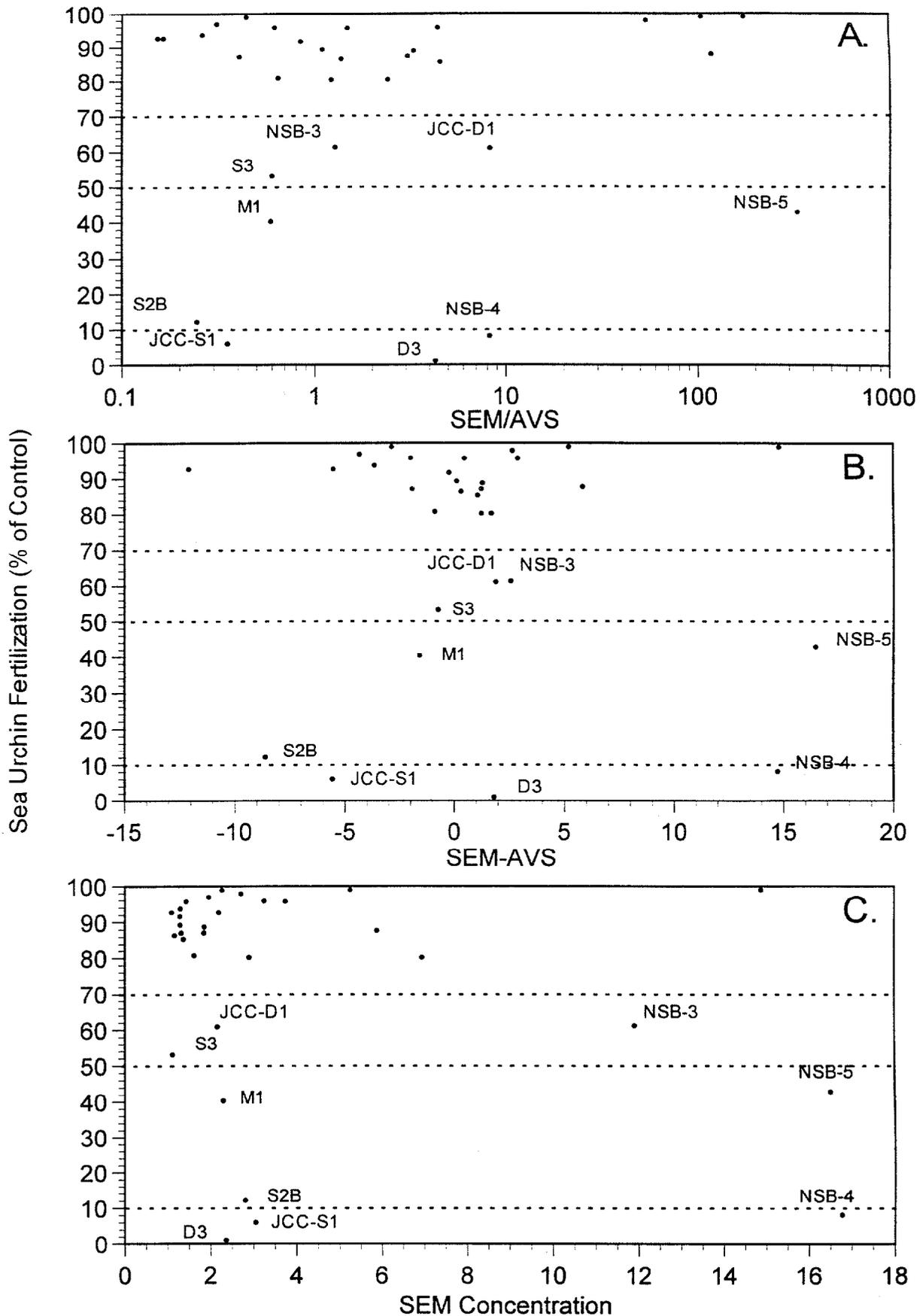


Figure 6.4-4. Sea urchin fertilization vs. A) SEM/AVS, B) SEM-AVS, and C) SEM concentration ( $\mu\text{mol/g}$ ) in porewaters from sediments collected from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. The dashed lines indicate threshold values for low (<70%), intermediate (<50%), and high (<10%) impact on sea urchin fertilization.

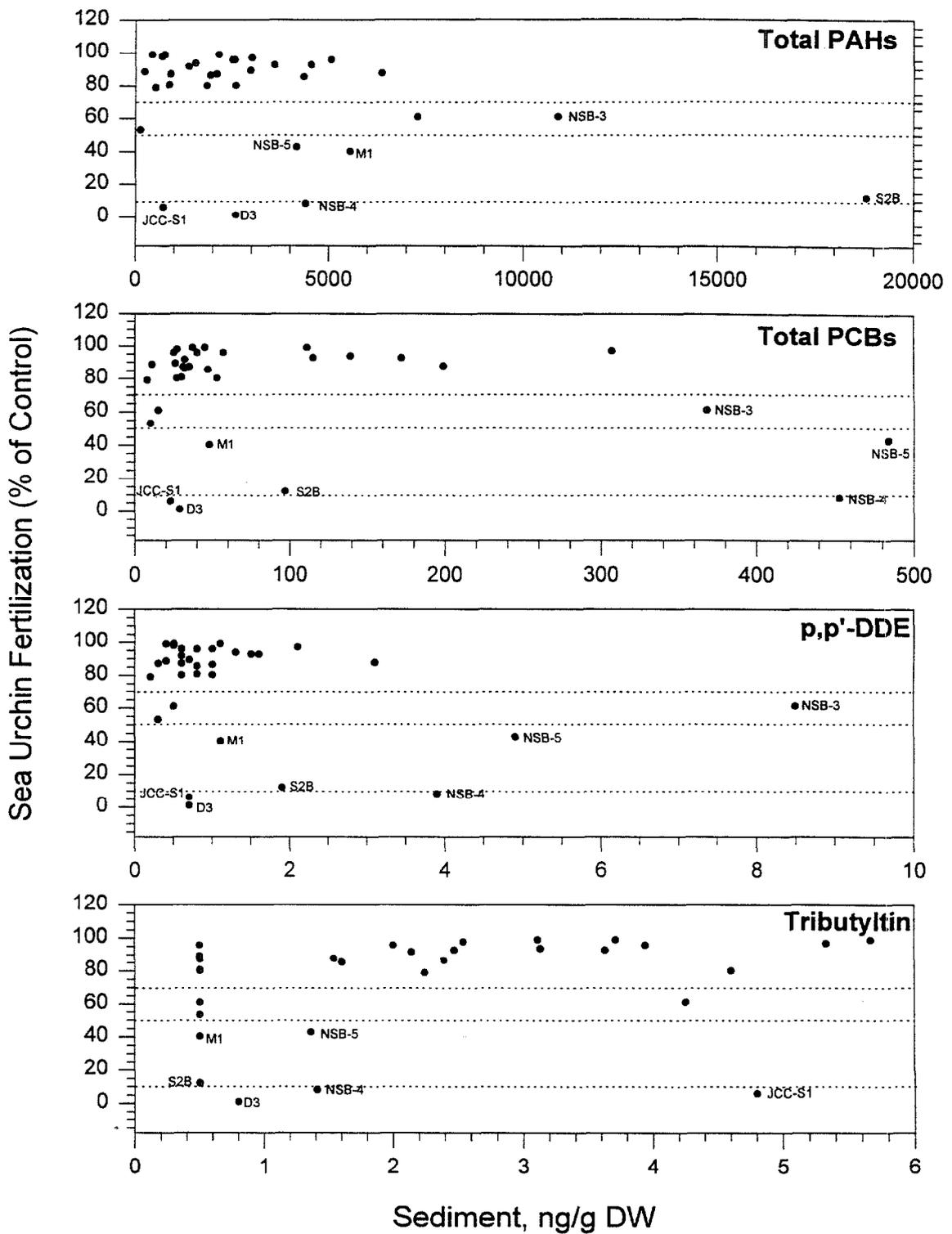


Figure 6.4-5. Sea urchin fertilization vs. organic contaminants in sediment porewaters from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Dashed lines indicate threshold values for low (<70%), intermediate (<50%), and high (<10%) impact on sea urchin fertilization.

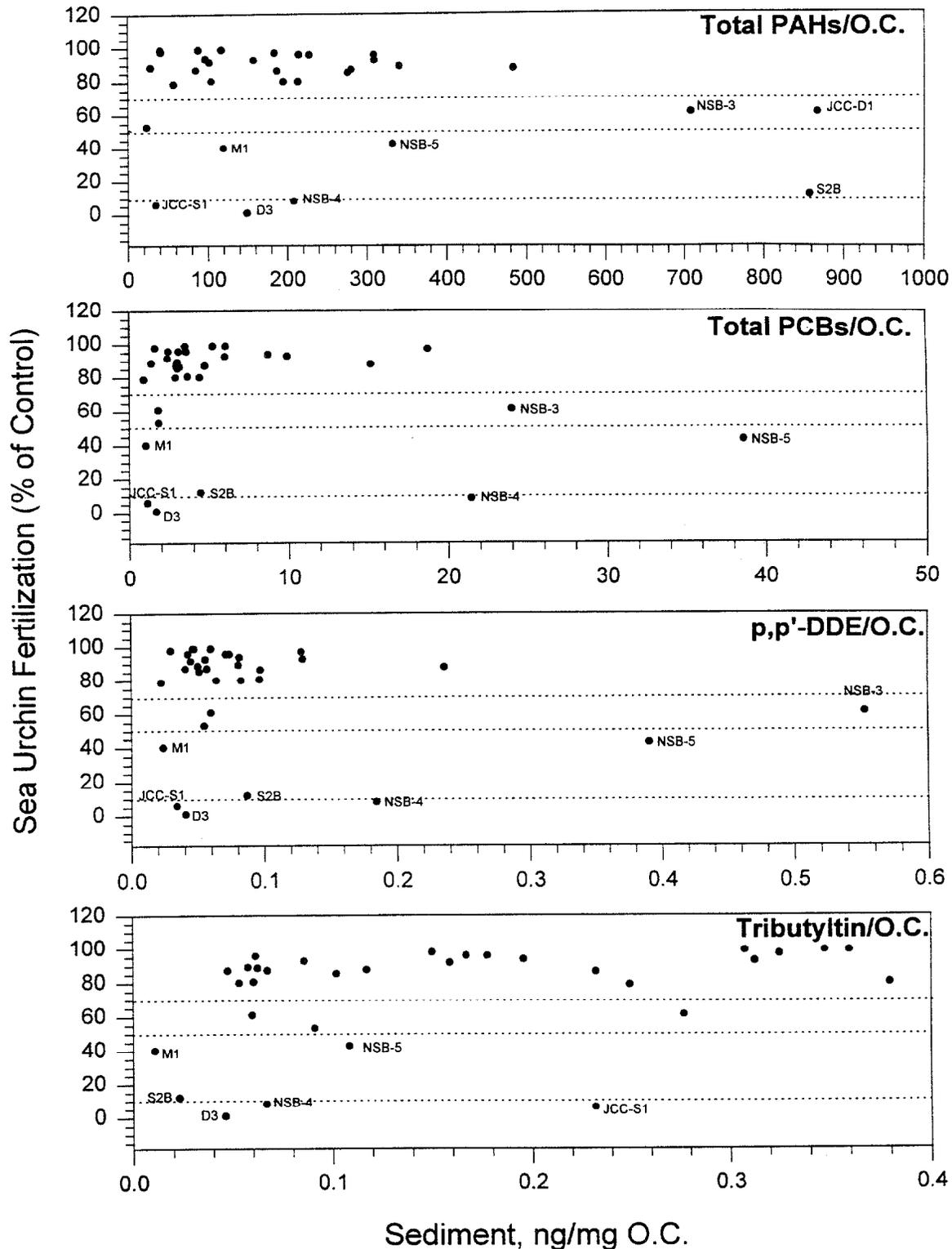


Figure 6.4-6. Sea urchin fertilization vs. organic contaminants (normalized to TOC) in sediment porewaters from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. Dashed lines indicate threshold values for low (<70%), intermediate (<50%), and high (<10%) impact on sea urchin fertilization.

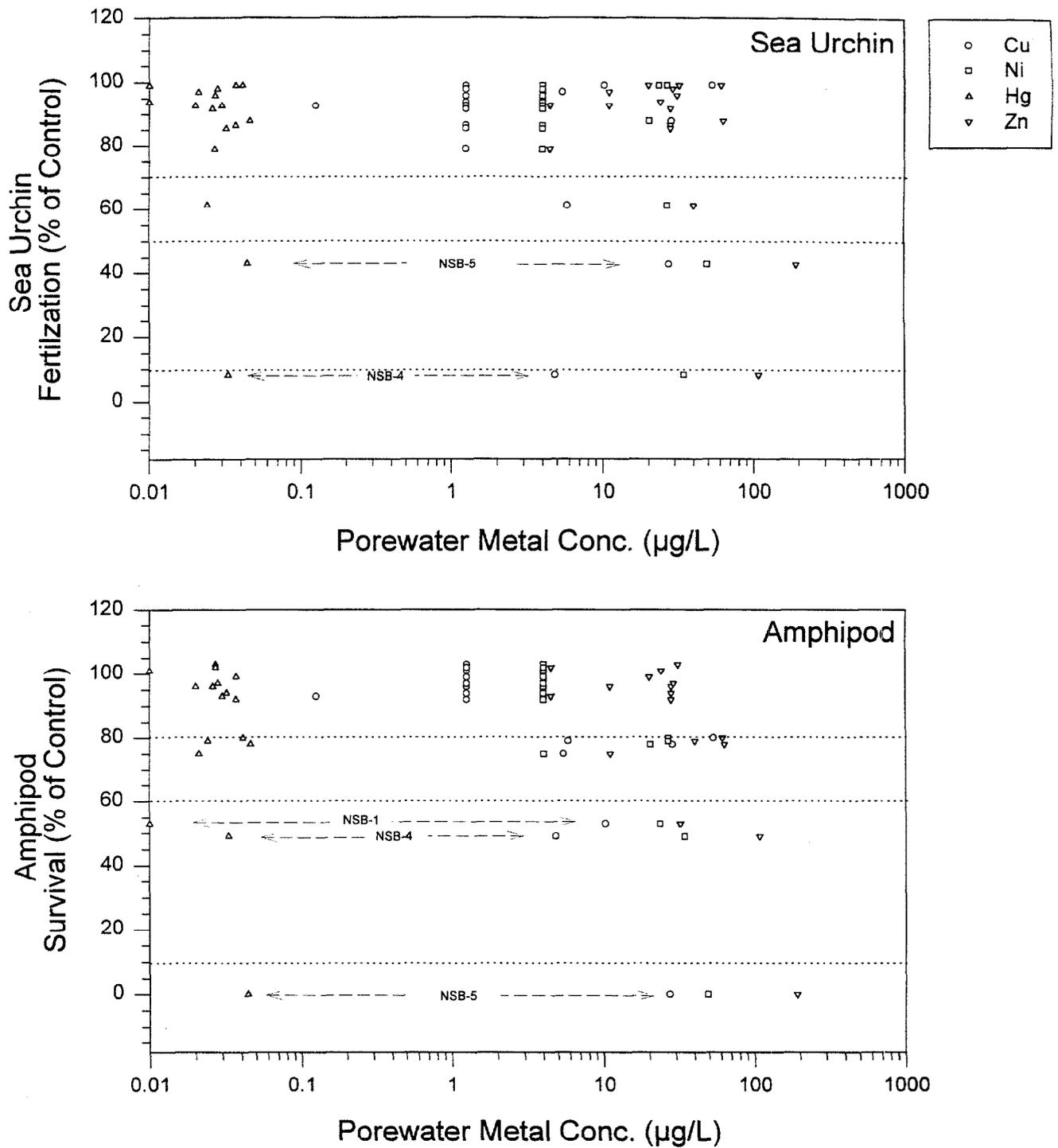


Figure 6.4-7. Sea urchin fertilization and amphipod survival vs. porewater metal concentrations in sediments from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. The dashed lines indicate the adverse effects threshold values for sea urchin fertilization and amphipod survival.

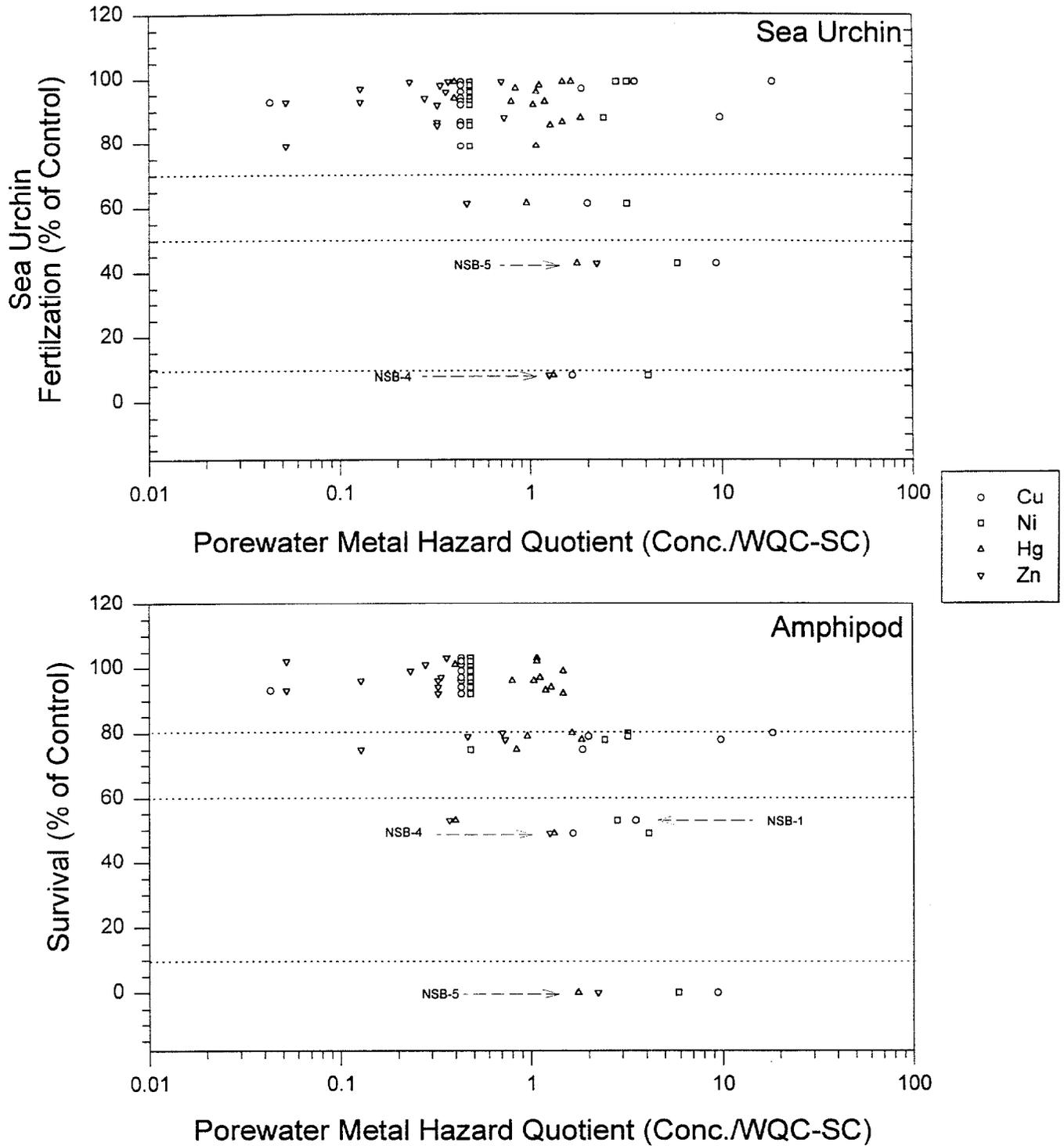


Figure 6.4-8. Sea urchin fertilization and amphipod survival vs. porewater metal Hazard Quotients in sediments from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. The dashed lines indicate the adverse effects threshold values for sea urchin fertilization and amphipod survival.

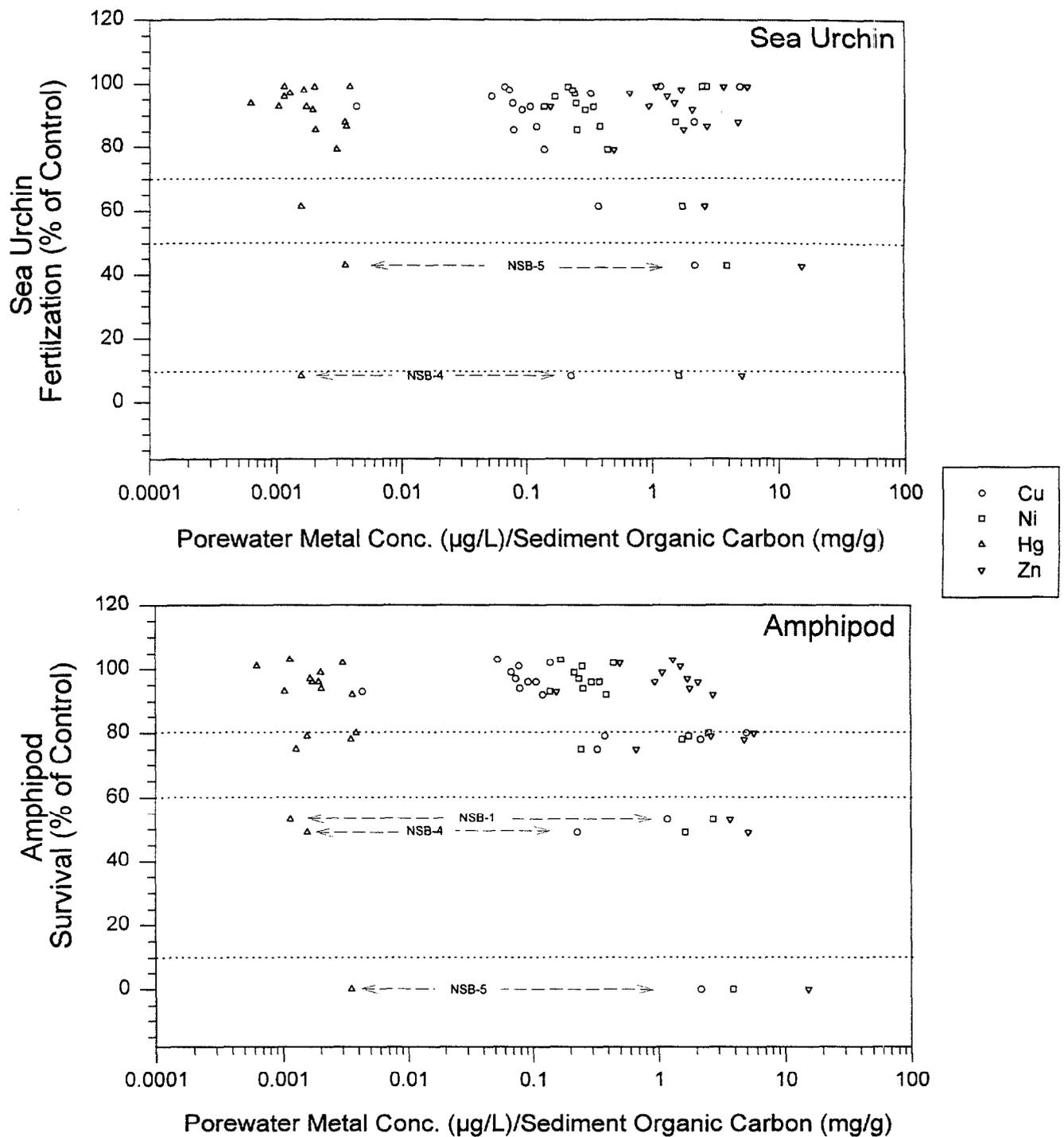


Figure 6.4-9. Sea urchin fertilization and amphipod survival vs. carbon-normalized porewater metal concentrations in sediments from the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location. The dashed lines indicate the adverse effects threshold values for sea urchin fertilization and amphipod survival.

## Metal

A.

Condition Index

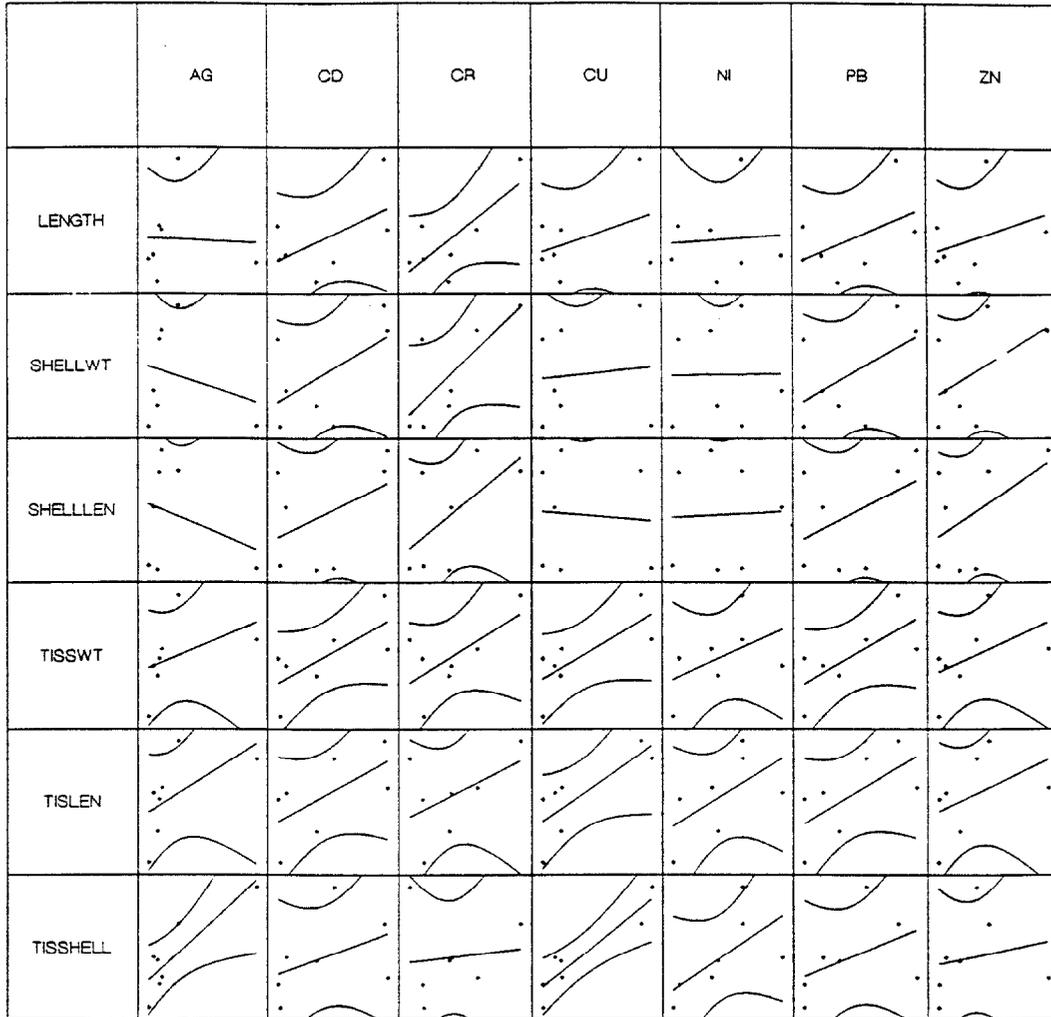


Figure 6.5-1. Blue mussel Condition Indices vs. sediment A) metal and B) organic concentrations at the McAllister Point Landfill study area stations. Lines indicate regression fit  $\pm$  95% confidence limits. Codes: LENGTH=shell length; SHELLWT=shell weight; SHELLEN=shell weight to length ratio; TISSWT=dry tissue weight; TISLEN=dry tissue weight to shell length ratio; TISSHELL=dry tissue weight to shell weight ratio.

## Organic Compound

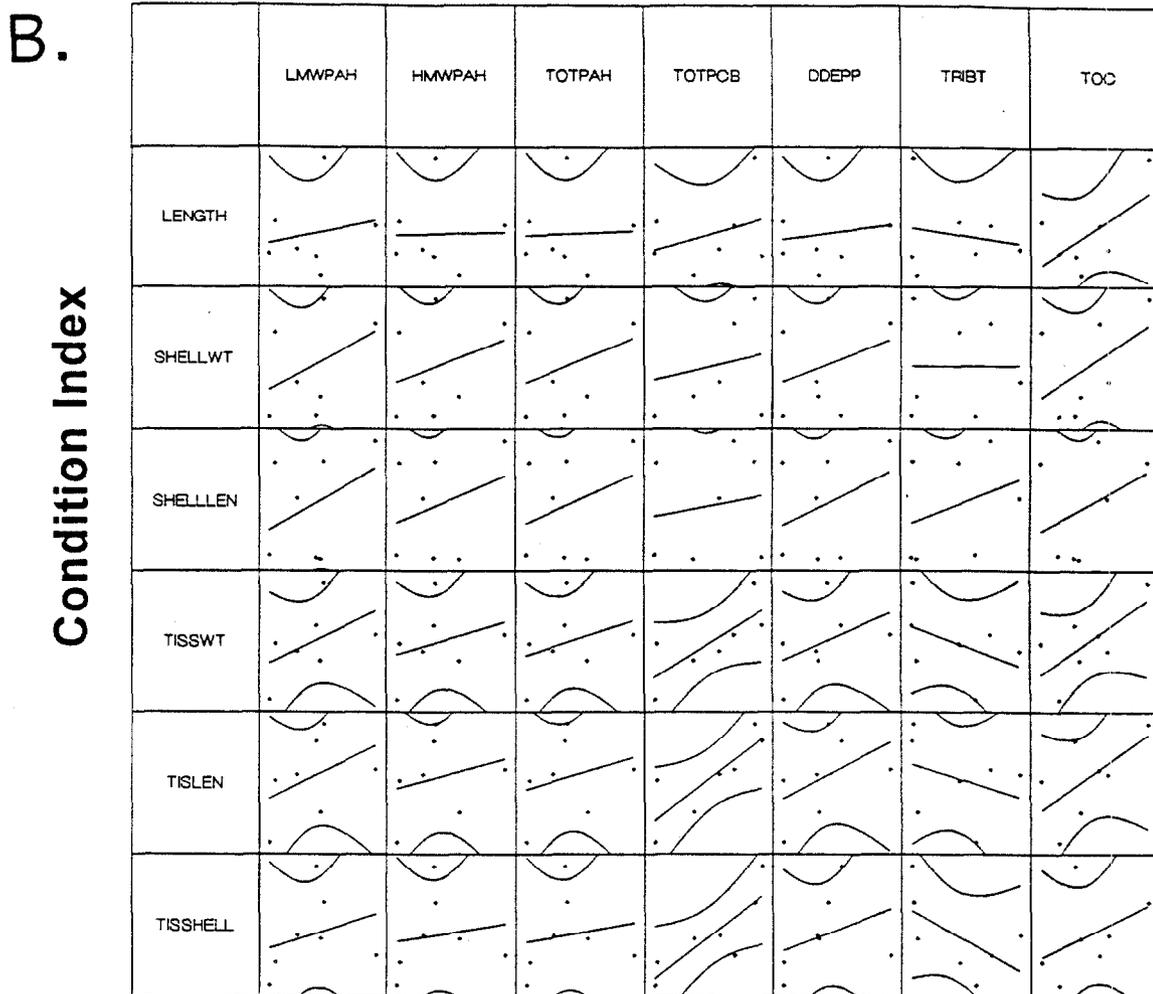


Figure 6.5-1 (continued). Blue mussel Condition Indices vs. sediment A) metal and B) organic concentrations at the McAllister Point Landfill study area stations. Lines indicate regression fit  $\pm$  95% confidence limits. Codes: LENGTH=shell length; SHELLWT=shell weight; SHELLLEN=shell weight to length ratio; TISSWT=dry tissue weight; TISLEN=dry tissue weight to shell length ratio; TISSHELL=dry tissue weight to shell weight ratio; LMWPAH=Low Molecular Weight PAHs; HMWPAH=High Molecular Weight PAHs; TOTPAH=Total PAHs; TOTPCB=Total PCBs; DDEPP=p,p'-DDE; TRIBT=Tributyltin; TOC=Total Organic Carbon.

## Metal

A.

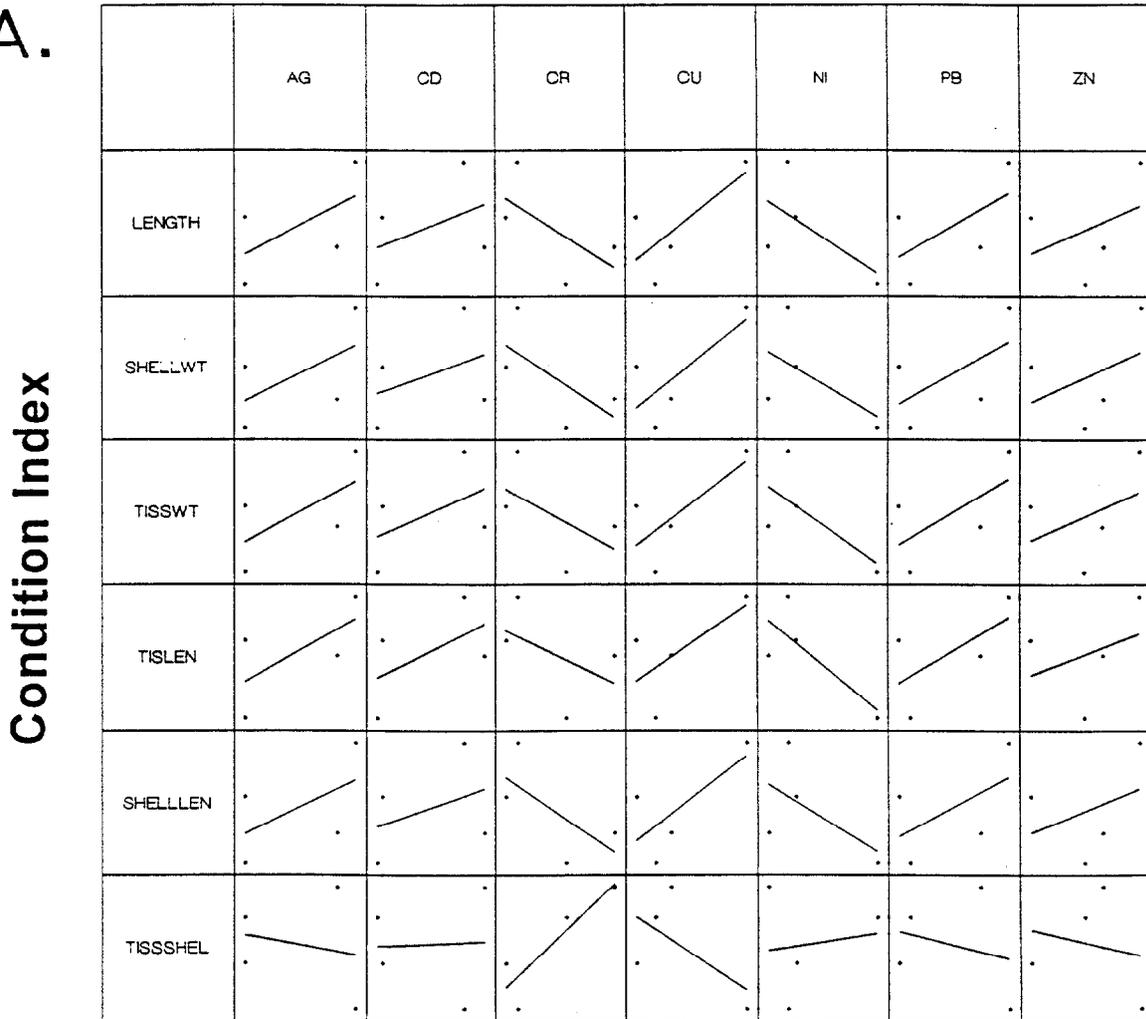


Figure 6.5-2. Hard clam Condition Indices vs. sediment A) metal and B) organic concentrations at the McAllister Point Landfill study area stations. Lines indicate regression fit  $\pm$  95% confidence limits. Codes: LENGTH=shell length; SHELLWT=shell weight; SHELLEN=shell weight to length ratio; TISSWT=dry tissue weight; TISLEN=dry tissue weight to shell length ratio; TISSHEL=dry tissue weight to shell weight ratio.

## Organic Compound

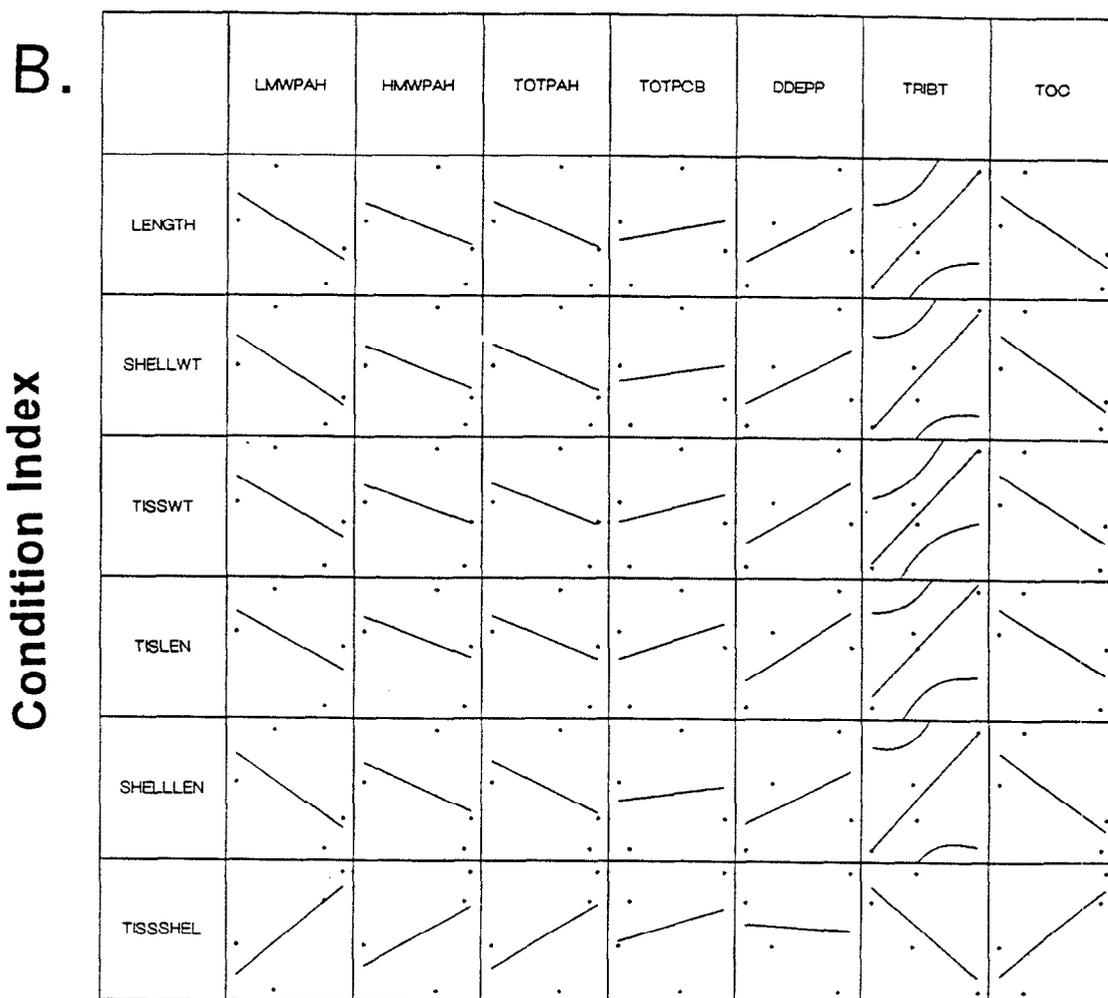


Figure 6.5-2 (continued). Hard clam Condition Indices vs. sediment A) metal and B) organic concentrations at the McAllister Point Landfill study area stations. Lines indicate regression fit  $\pm$  95% confidence limits. Codes: LENGTH=shell length; SHELLWT=shell weight; SHELLEN=shell weight to length ratio; TISSWT=dry tissue weight; TISLEN=dry tissue weight to shell length ratio; TISSHELL=dry tissue weight to shell weight ratio; LMWPAH=Low Molecular Weight PAHs; HMWPAH=High Molecular Weight PAHs; TOTPAH=Total PAHs; TOTPCB=Total PCBs; DDEPP=p,p'-DDE; TRIBT=Tributyltin; TOC=Total Organic Carbon.

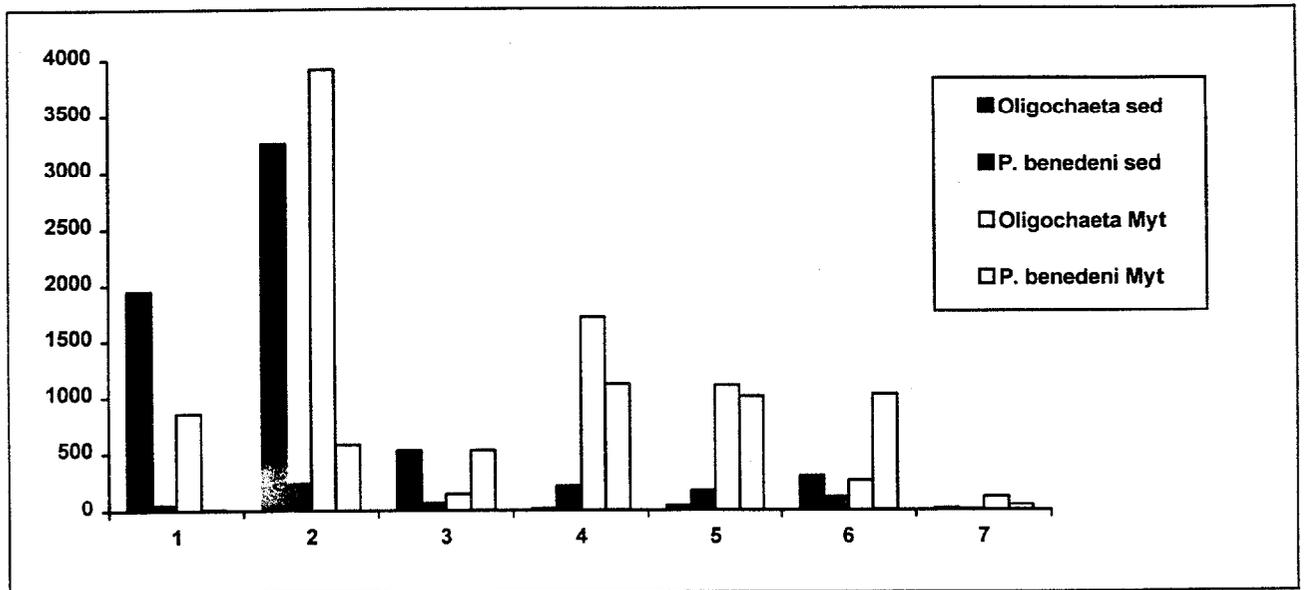
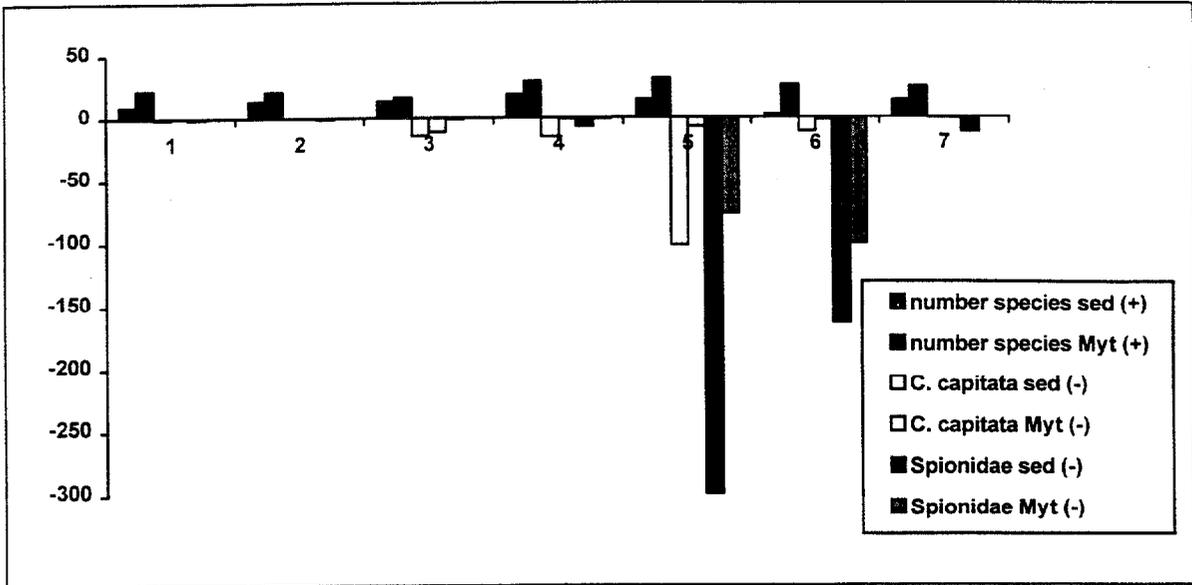


Figure 6.5.3 . Intertidal benthic community condition measures for the McAllister Point Landfill study area. Two samples were taken at each station, a *Mytilus* sample including a cluster of blue mussels embedded in sediment. Top graph: Possible indicators of high quality are shown as positive values. Possible indicators of reduced habitat are shown as negative values. Bottom graph: The density of the oligochaete, *Peloscolex benedeni* and non-*P. benedeni* oligochaetes are shown for single "sediment" and "Mytilus" samples.

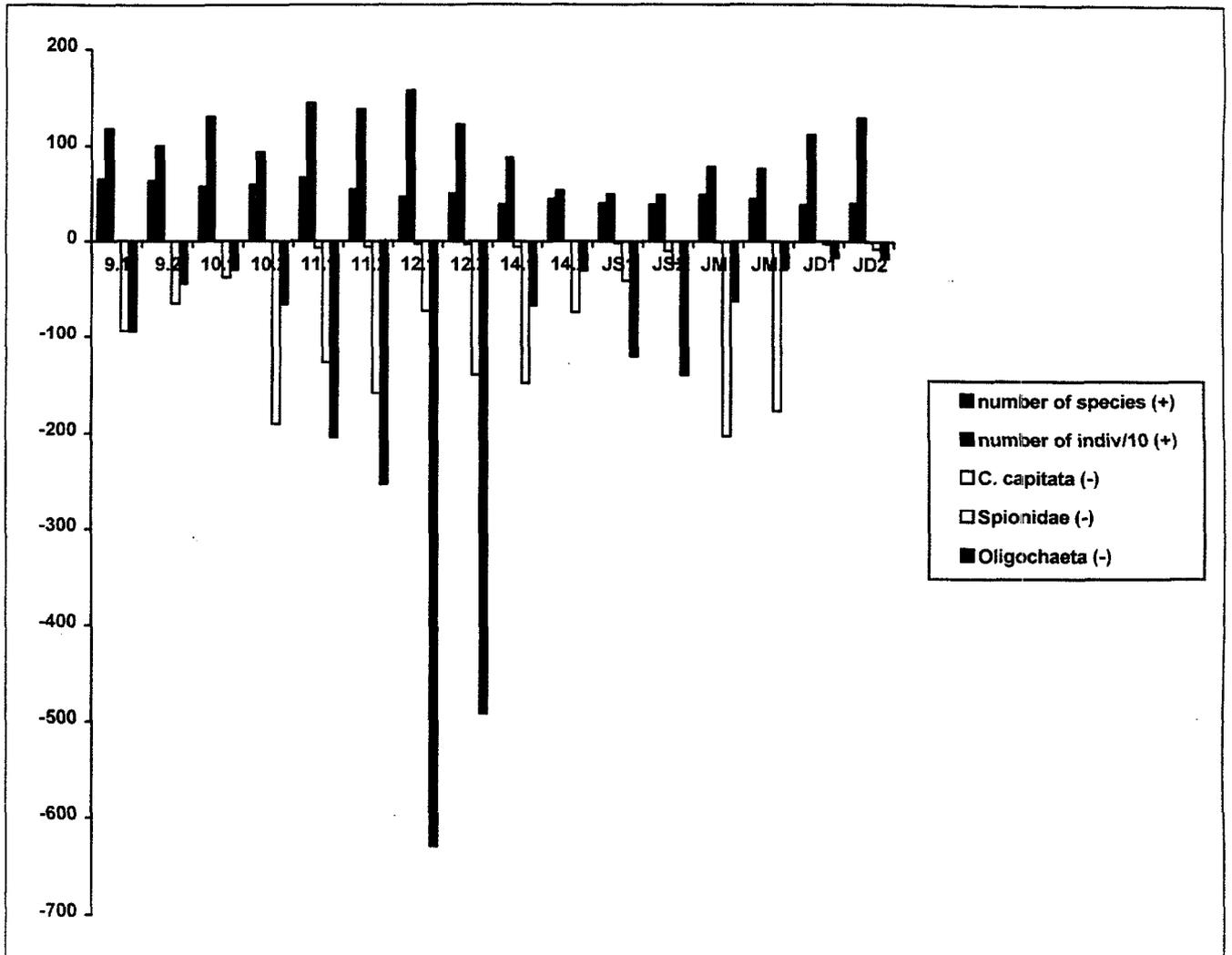


Figure 6.5.4. Subtidal benthic community condition measures (duplicate sample) for the McAllister Point Landfill study area and Jamestown Cranston Cove reference location. Possible indicators of high quality (number of species and individuals per sample) are shown as positive values; possible indicators of reduced habitat quality (numbers of *Capitella capitata*, combined *Spionidae*, and *Oligochaeta* spp.) are shown as negative values. Stations 9-12 characterize a north-south transect offshore of McAllister Point Landfill. Station 14 is offshore of Station 12. "J" stations represent shallow (S), mid (M), and deep (D) stations at the Jamestown Cranston Cove (JCC) reference location.



Table 6.1-2. Results of Simultaneously Extracted Metal (SEM) and Acid Volatile Sulfide (AVS) measurements in surface sediments and qualitative evaluation of divalent metal bioavailability for the McAllister Point Landfill study area.

Zone	Station	Depth (cm)	AVS <sup>1</sup>	SEM <sup>1</sup>		SEM/AVS		SEM-AVS		Risk
			( $\mu$ Mole/g dry)	( $\mu$ Mole/g dry)	Flag <sup>2</sup>	Ratio	Flag <sup>2</sup>	( $\mu$ Mole/g dry)	Flag <sup>2</sup>	Ranking <sup>3</sup>
1	NSB-1	0-6	0.11	14.9	+	175	+	14.8	+	+++
2	NSB-2	0-6	0.10	5.25	+	105	+	5.20	+	+++
	NSB-3	0-6	9.33	11.9	+	1.28	+	2.57	-	++
	NSB-4	0-6	2.05	16.8	+	8.20	+	14.7	+	+++
	NSB-5	0-6	0.10	16.5	+	330	+	16.5	+	+++
3	NSB-6	0-6	6.31	1.96	-	0.31	-	-4.35	-	-
	NSB-7	0-6	0.10	5.88	+	118	+	5.83	+	+++
3A	OS-28	0-2	28.2	2.08	-	0.07	-	-26.1	-	-
	S2B	0-2	11.4	2.80	-	0.25	-	-8.63	-	-
	MCL-12	0-2	6.64	1.09	-	0.16	-	0.93	-	-
	M1	0-2	3.88	2.29	-	0.59	+	-1.59	-	+
4	MCL-8	0-2	5.24	3.24	-	0.62	+	-1.99	-	+
	MCL-9	0-2	5.13	2.26	-	0.44	-	1.82	-	-
	MCL-10	0-2	14.3	2.19	-	0.15	-	2.03	-	-
	MCL-11	0-2	4.96	1.29	-	0.26	-	1.03	-	-
5	OS-22	0-2	1.36	0.38	-	0.28	-	-0.98	-	-
	OS-23	0-2	3.99	1.58	-	0.40	-	-2.41	-	-
	OS-24	0-2	2.89	1.25	-	0.43	-	-1.64	-	-
	OS-25	0-2	11.5	1.68	-	0.15	-	-9.84	-	-
	OS-26	0-2	9.19	1.20	-	0.13	-	-7.99	-	-
	OS-27	0-2	14.4	1.98	-	0.14	-	-12.4	-	-
6	MCL-13	0-2	0.84	1.16	-	1.38	+	-0.22	-	+
	MCL-14	0-2	1.52	1.28	-	0.85	+	0.44	-	+
	MCL-15	0-2	0.10	n/a		n/c		n/c		
	MCL-16	0-2	0.30	1.36	-	4.54	+	1.06	-	+
	OS-29	0-2	2.74	1.32	-	0.48	-	-1.42	-	-
	OS-30	0-2	11.6	1.42	-	0.12	-	-10.2	-	-
	OS-30A	0-2	0.60	1.84	-	3.07	+	1.24	-	+
	OS-30B	0-2	1.20	2.89	-	2.41	+	1.69	-	+
	D1	0-2	0.97	1.43	-	1.48	+	0.47	-	+
	D2	0-2	0.85	3.74	-	4.43	+	2.90	-	+
	D3	0-2	0.55	2.35	-	4.28	+	1.80	-	+
	M2	0-2	1.18	1.29	-	1.10	+	0.11	-	+
	M3	0-2	1.03	6.95	+	1.22	+	1.24	-	++
	S1	0-2	2.50	1.61	-	0.65	+	-0.89	-	+
	S3	0-2	1.85	1.11	-	0.60	+	-0.74	-	+
	S4	0-2	3.23	1.31	-	0.41	-	-1.92	-	-
7	JCC-D1	0-2	0.26	2.14	-	8.22	+	-6.09	-	+
	JCC-M1	0-2	0.56	1.85	-	3.30	+	1.29	-	+
	JCC-M1 <sup>4</sup>	0-2	1.00	1.49	-	1.50	+	-0.01	-	+
	JCC-S1	0-2	8.64	3.04	-	0.35	-	-5.59	-	-

"n/a"=not available; "n/c"=not calculated if concentrations of SEM or AVS are not available or are below the method detection limit.

See Figure 6.0-1 for location of sampling stations. "JCC" indicates Jamestown Cranston Cove reference location.

1 - Mean of two replicates per station.

2 - SEM Codes: SEM Conc. > 5  $\mu$ mol/g = "+"; SEM/AVS > 0.5 = "+"; SEM-AVS > 5  $\mu$ mol/g = "+".

3 - Overall Risk Ranking: "-" = no exposure, "+" = exposure seen in one indicator,

"++" = exposure seen in two indicators, "+++" = exposure in all indicators.

4 - Sampled during Phase II (SAIC/URI, 1996).

Table 6.2-1a. Tissue Concentration Ratios (TCRs) for McAllister Point Landfill Marine Ecological Risk Assessment by Zone and Station.<sup>1,2</sup>

Zone	Station	Species	PAHs												PCBs	PST	BT	Metals							Ranking <sup>3</sup>					
			2-Methylnaphthalene	Acenaphthene	Acenaphthylene	Anthracene	Benzo(a)anthracene	Benzo(e)pyrene	Chrysene	Dibenz(a,h)anthracene	Fluoranthene	Fluorene	HMW PAHs	LMW PAHs	Naphthalene	Phenanthrene	Pyrene	Total PAHs	Total PCBs	p,p'-DDE	Tributyltin	Arsenic	Cadmium	Chromium		Copper	Lead	Mercury	Nickel	Silver
1	NSB-1	BM		++	+	+	+	+	++	++	++	++	++	+	++	++	++	+	+	++	+	+	+	+	+	+	+	+	+	++
		MF	+	+	+	+	+	+	++	++	++	++	++	+	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++
2	NSB-2	BM		+	+	+	+	+	++	++	++	++	+	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++	
		MF		+	+	+	+	+	++	++	++	++	+	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++
	NSB-3	BM		+	+	+	+	+	++	++	++	++	+	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++	
		MF		+	+	+	+	+	++	++	++	++	+	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++
	NSB-4	BM		+	+	+	+	+	++	++	++	++	+	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++	
		MF	+	+	+	+	+	+	++	++	++	++	+	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++
	NSB-5	BM		+	+	+	+	+	++	++	++	++	+	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++	
		MF	+	+	+	+	+	+	++	++	++	++	+	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++
3	NSB-6	BM		+	+	+	+	+	++	++	++	++	+	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++	
		MF	+	+	+	+	+	+	++	++	++	++	+	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++
	NSB-7	BM		++	++	++	++	++	+++	+++	+++	+++	+	+++	+++	+++	+++	+	+	+++	+	+	+	+	+	+	+	+	+++	
		MF	+	+	+	+	+	+	++	++	++	++	+	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++
3A	OS-28†	HC		+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
		M1	+	+	++	++	+	+	+	+	++	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
4	MCL-9	HC	+	+	+	+	+	+	++	++	++	++	+	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++	
		HPP	+	+	+	+	+	+	++	++	++	++	+	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	++
	MCL-10	MUS	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
		HC	+	+	++	++	+	+	++	++	++	++	+	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	++	
	MCL-11	HPP	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
		MUS	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
5	OS-22	HC		+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
		OS-23		+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
	OS-24	HC		+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
		OS-25		+	+	++	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
	OS-26	HC		+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
		OS-27		+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
6	MCL-13	HC	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
		HPP	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
	MCL-14	MUS	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+++	
		HC	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
	MCL-16	HPP	++	+	+	+	+	+	++	++	++	++	+	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	+++	
		MUS	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
	S2	HC	+	+	++	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
		HC	-	+	+	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	

PAH=Polycyclic Aromatic Hydrocarbons; LMW=Low Molecular Weight (PAHs); HMW=High Molecular Weight (PAHs); PCB=Polychlorinated Biphenyls; PST=pesticides; BT=butyltins.

BM=blue mussel; HC=hard clam; HPP=lobster hepatopancreas; MUS=lobster muscle; MF=marine fish (cunner).

1 - Phase I = Samples collected by URI/SAIC (1994); Phase II = SAIC/URI (1995); TRC = TRC (1994). See Figure 6.0-1.

2 - Risk Codes: TCR<=1 - "-"; TCR>1 - "+"; TCR>10 - "++"; TCR>40 - "+++".

3 - Exposure Ranking: "+++ = intermediate (++) or higher exposure observed for two or more indicators, one of which indicates high (+++) exposure;

++ = intermediate (++) exposure observed for two or more indicators or high (+++) exposure for one indicator; "+" = low (+) exposure observed for two or more indicators

or intermediate (++) exposure for one indicator; "-" = low (+) exposure observed for only one indicator or no exposure for all indicators. See text in Section 6.0-2.

Table 6.2-1b. Tissue Concentration Ratios (TCRs) for McAllister Point Landfill Marine Ecological Risk Assessment by Species.<sup>1,2</sup>

Zone	Station	Species	PAHs													PCBs	PST	BT	Metals									Ranking <sup>3</sup>					
			2-Methylnaphthalene	Acenaphthene	Acenaphthylene	Anthracene	Benzo(a)anthracene	Benzo(e)pyrene	Chrysene	Dibenz(a,h)anthracene	Fluoranthene	Fluorene	HMW PAHs	LMW PAHs	Naphthalene				Phenanthrene	Pyrene	Total PAHs	Total PCBs	p,p'-DDE	Tributyltin	Arsenic	Cadmium	Chromium		Copper	Lead	Mercury	Nickel	Silver
1	NSB-1	BM	++	+	+	+	+	+	++	++	++	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	+	+	+	++		
2	NSB-2	BM	+	+	+	+	+	+	++	++	++	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	+	+	+	++		
2	NSB-3	BM	+	+	+	+	+	+	++	++	++	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	+	+	+	++		
2	NSB-4	BM	+	+	+	+	+	+	++	++	++	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	+	+	+	++		
2	NSB-5	BM	+	+	+	+	+	+	++	++	++	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	+	+	+	++		
3	NSB-6	BM	+	+	+	+	+	+	++	++	++	++	++	++	++	++	+	+	++	+	+	+	+	+	+	+	+	+	+	+	++		
3	NSB-7	BM	++	++	++	++	++	++	+++	+++	+++	+++	+++	+++	+++	+++	+	+	+++	+	+	+	+	+	+	+	+	+	+	+	+++		
3A	M1	HC	+	+	++	++	+	+	+	+	++	++	++	++	++	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++		
4	MCL-9	HC	+	-	+	+	+	+	++	+	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++		
4	MCL-10	HC	+	+	++	++	+	+	++	+	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++		
4	MCL-11	HC	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
3A	MCL-12	HC	+	+	++	+	+	+	+	+	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++		
6	MCL-13	HC	+	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
6	MCL-14	HC	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
6	MCL-16	HC	+	-	+	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
5	OS-22	HC	-	-	+	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
5	OS-23	HC	-	-	-	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
5	OS-24	HC	-	-	-	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
5	OS-25	HC	-	+	+	+	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
5	OS-26	HC	-	-	-	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
5	OS-27	HC	-	+	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
3A	OS-28	HC	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
6	S2	HC	-	+	+	++	+	+	+	+	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++	
4	MCL-9	HPP	+	+	+	-	+	+	+	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+++	
4	MCL-10	HPP	+	+	-	-	-	-	-	-	-	+	-	+	-	+	-	+	-	+	-	+	-	+	-	+	-	+	-	+	+	++	
6	MCL-13	HPP	+	+	-	+	-	+	+	-	+	+	+	+	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++
6	MCL-14	HPP	++	+	+	+	+	+	+	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+++
4	MCL-9	MUS	+	+	-	+	+	-	-	-	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	++
4	MCL-10	MUS	+	+	+	+	+	-	-	-	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
6	MCL-13	MUS	+	+	+	+	+	+	-	-	+	+++	+	++	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+++
6	MCL-14	MUS	+	+	+	+	++	+	+	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
1	NSB-1	MF	+	+	-	-	+	+	+	+	+	-	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	
2	NSB-3	MF	-	+	+	+	+	+	+	+	-	-	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
2	NSB-4	MF	+	+	+	+	+	+	+	+	-	-	-	+	++	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
3	NSB-6	MF	+	+	+	+	-	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+

PAH=Polycyclic Aromatic Hydrocarbons; LMW=Low Molecular Weight (PAHs); HMW=High Molecular Weight (PAHs); PCB=Polychlorinated Biphenyls; PST=pesticides; BT=butyltins.

BM=blue mussel; HC=hard clam; HPP=lobster hepatopancreas; MUS=lobster muscle; MF=marine fish (cunner).

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Table 6.3-1. Documentation of Avian Aquatic Receptor Exposure Factors for the McAllister Point Landfill Marine ERA.

Receptor Group	Body Weight, BW (kg)	Food Consumption Rate, FCR (kg total diet <sup>1</sup> /kg bird/day)	On-site Feeding Area: Foraging Area Ratio (a/fa)	Migration Factor, MF	Feeding Fraction, FF (kg prey/kg total diet)	Exposure Factor <sup>2</sup> , EF (kg prey/kg bird/day)
Herring gull	1.00 (EPA, 1993)	0.50 Estimated using allometric equation specific for seabirds: $FCR = 0.495BW^{0.704}$ (Nagy et al., 1987)	1.00 Assumes receptor feeds exclusively at site.	0.60 Spring/Fall Feb. - Aug. NW Atlantic populations (Burger, 1982)	1.00 Target receptors: Cunner Deployed blue mussels Indigenous blue mussels Lobster <i>Mercenaria mercenaria</i> <i>Pitar morrhuana</i>	0.30
Great Blue Heron	2.23 (EPA, 1993)	0.42 Estimated using allometric equation specific for herons: $\text{Log FCR} = 0.966 \cdot \text{logBW} - 0.64$ (Kushlan, 1978, cited in EPA, 1993)	1.00 Assumes receptor feeds exclusively at site.	0.66 Spring/Fall Mar. - Oct. Northern U.S. (Palmer, 1962)	1.00 Target receptors: Cunner Deployed blue mussels Indigenous blue mussels Lobster <i>Mercenaria mercenaria</i> <i>Pitar morrhuana</i>	0.28

1 - Units are dry weight.

2 -  $EF = FCR \cdot a/fa \cdot MF \cdot FF$

Table 6.3-2. Documentation of Toxicity Reference Values (TRVs) used for calculation of risks to Avian Aquatic Receptors of Concern (RoC) consuming prey in the McAllister Point Landfill study area.

Chemical Class	Target Analyte	RECEPTOR		TEST SPECIES DATA						TOXICITY REFERENCE VALUES (TRVs)				
		RoC <sup>D</sup>	BW <sup>1</sup> (kg)	Test Species	BW (Kg)	Endpoint	Endpoint Value <sup>2</sup>	Reference	Safety Factor <sup>3</sup>	Test NOAEL <sup>B</sup>	RoC	FCR <sup>5</sup>	Food Factor <sup>6</sup>	RoC TRV-EPC <sup>C,7</sup>
MET	Arsenic <sup>E</sup>	Gull	1.00	Mallard duck	1.000	Chronic NOAEL	5.14	Opresko et al. 1995	1	5.14	5.14	0.61	0.61	8.42
		Heron	2.23	Mallard duck	1.000	Chronic NOAEL	5.14	Opresko et al. 1995	1	5.14	3.93	1.07	0.48	8.18
	Cadmium <sup>F</sup>	Gull	1.00	Mallard duck	1.000	Chronic NOAEL	1.15	Opresko et al. 1995	1	1.15	1.15	0.61	0.61	1.89
		Heron	2.23	Mallard duck	1.000	Chronic NOAEL	1.15	Opresko et al. 1995	1	1.15	0.88	1.07	0.48	1.83
	Chromium <sup>G</sup>	Gull	1.00	Black duck	1.250	Chronic NOAEL	1.00	Opresko et al. 1995	1	1.00	1.08	0.61	0.61	1.77
		Heron	2.23	Black duck	1.250	Chronic NOAEL	1.00	Opresko et al. 1995	1	1.00	0.82	1.07	0.48	1.72
	Copper <sup>H</sup>	Gull	1.00	Chicken, 1-70 days old	0.534	Chronic NOAEL	47.0	Opresko et al. 1995	1	47.0	38.1	0.61	0.61	62.6
		Heron	2.23	Chicken, 1-70 days old	0.534	Chronic NOAEL	47.0	Opresko et al. 1995	1	47.0	29.2	1.07	0.48	60.7
	Lead <sup>I</sup>	Gull	1.00	American kestrel	0.130	Chronic NOAEL	3.85	Opresko et al. 1995	1	3.85	1.95	0.61	0.61	3.20
		Heron	2.23	American kestrel	0.130	Chronic NOAEL	3.85	Opresko et al. 1995	1	3.85	1.49	1.07	0.48	3.11
	Mercury <sup>J</sup>	Gull	1.00	Japanese Quail	0.150	Chronic NOAEL	0.45	Opresko et al. 1995	1	0.45	0.24	0.61	0.61	0.39
		Heron	2.23	Japanese Quail	0.150	Chronic NOAEL	0.45	Opresko et al. 1995	1	0.45	0.18	1.07	0.48	0.38
	Nickel <sup>K</sup>	Gull	1.00	Mallard duck, 1-90 days old	0.782	Chronic NOAEL	77.4	Opresko et al. 1995	1	77.4	71.3	0.61	0.61	117
		Heron	2.23	Mallard duck, 1-90 days old	0.782	Chronic NOAEL	77.4	Opresko et al. 1995	1	77.4	54.6	1.07	0.48	114
	Silver	Gull	1.00	Mallard duck (juvenile)	0.600	4 wk NOAEL	8.30	Van Vleet 1982	10	0.83	0.70	0.61	0.61	1.15
Heron		2.23	Mallard duck (juvenile)	0.600	4 wk NOAEL	8.30	Van Vleet 1982	10	0.83	0.54	1.07	0.48	1.11	
Zinc <sup>L</sup>	Gull	1.00	White Leghorn Hens	1.935	Chronic NOAEL	14.5	Opresko et al. 1995	1	14.5	18.1	0.61	0.61	29.6	
	Heron	2.23	White Leghorn Hens	1.935	Chronic NOAEL	14.5	Opresko et al. 1995	1	14.5	13.8	1.07	0.48	28.8	
PAH	1,6,7-Trimethylnaphthalene			No Data										
	1-Methylnaphthalene			No Data										
	1-Methylphenanthrene			No Data										
	2,6-Dimethylnaphthalene			No Data										
	2-Methylnaphthalene	Gull	1.00	Mallard duck	1.000	7 mo LOAEL	600	See Naphthalene	10	60	60.0	0.61	0.61	98.4
		Heron	2.23	Mallard duck	1.000	7 mo LOAEL	600	See Naphthalene	10	60	45.9	1.07	0.48	95.5
	Acenaphthene	Gull	1.00	Red-winged blackbird	0.065	Acute LD <sub>50</sub>	101	Schafer et al. 1983	80	1.26	0.51	0.61	0.61	0.83
		Heron	2.23	Red-winged blackbird	0.065	Acute LD <sub>50</sub>	101	Schafer et al. 1983	80	1.26	0.39	1.07	0.48	0.81
	Acenaphthylene	Gull	1.00	Red-winged blackbird	0.065	Acute LD <sub>50</sub>	101	See Acenaphthene	80	1.26	0.51	0.61	0.61	0.83
		Heron	2.23	Red-winged blackbird	0.065	Acute LD <sub>50</sub>	101	See Acenaphthene	80	1.26	0.39	1.07	0.48	0.81
	Anthracene	Gull	1.00	Red-winged blackbird	0.065	Acute LD <sub>50</sub>	111	Schafer et al. 1983	80	1.39	0.56	0.61	0.61	0.92
		Heron	2.23	Red-winged blackbird	0.065	Acute LD <sub>50</sub>	111	Schafer et al. 1983	80	1.39	0.43	1.07	0.48	0.89
	Benz[a]anthracene			No Data										
	Benzo[a]pyrene			No Data										
	Benzo[b]fluoranthene			No Data										
	Benzo[e]pyrene			No Data										
	Benzo(g,h,i)perylene			No Data										
	Benzo[k]fluoranthene			No Data										
	Biphenyl			No Data										
	Chrysene			No Data										
	Dibenz[a,h]anthracene			No Data										
	Fluoranthene			No Data										
	Fluorene	Gull	1.00	Red-winged blackbird	0.065	Acute LD <sub>50</sub>	101	Schafer et al. 1983	80	1.26	0.51	0.61	0.61	0.83
		Heron	2.23	Red-winged blackbird	0.065	Acute LD <sub>50</sub>	101	Schafer et al. 1983	80	1.26	0.39	1.07	0.48	0.81
	Indeno[1,2,3-cd]pyrene			No Data										
Naphthalene	Gull	1.00	Mallard duck	1.000	7 mo LOAEL	600	Eisler 1987	10	60	60.0	0.61	0.61	98.4	
	Heron	2.23	Mallard duck	1.000	7 mo LOAEL	600	Eisler 1987	10	60	45.9	1.07	0.48	95.5	
Perylene			No Data											
Phenanthrene	Gull	1.00	Mallard duck	1.000	7 mo LOAEL	600	Eisler 1987	10	60	60.0	0.61	0.61	98.4	
	Heron	2.23	Mallard duck	1.000	7 mo LOAEL	600	Eisler 1987	10	60	45.9	1.07	0.48	95.5	
Pyrene			No Data											
PCB	Total PCBs (c)	Gull	1.00	Ring-necked pheasant	1.000	Chronic NOAEL	0.18	Opresko et al. 1995*	1	0.18	0.18	0.61	0.61	0.30
		Heron	2.23	Ring-necked pheasant	1.000	Chronic NOAEL	0.18	Opresko et al. 1995*	1	0.18	0.14	1.07	0.48	0.29
PST	Aldrin	Gull	1.00	Ring-necked pheasant (juv)	0.800	7 wk NOAEL	0.05	Hall et al. 1971	10	0.005	0.005	0.61	0.61	0.008
		Heron	2.23	Ring-necked pheasant (juv)	0.800	7 wk NOAEL	0.05	Hall et al. 1971	10	0.005	0.004	1.07	0.48	0.007
	Hexachlorobenzene			No Data										
		Mirex			No Data									
		o,p'-DDE			No Data									
p,p'-DDE	Gull	1.00	Mallard duck	1.000	1.5 yr. NOAEL	1.36	Heath et al. 1972	1	1.36	1.36	0.61	0.61	2.23	
	Heron	2.23	Mallard duck	1.000	1.5 yr. NOAEL	1.36	Heath et al. 1972	1	1.36	1.04	1.07	0.48	2.17	
BT	Monobutyltin			No Data										
				No Data										
Dibutyltin	Gull	1.00	Japanese Quail	0.150	Chronic NOAEL	6.80	Van Vleet 1982	1	6.8	3.61	0.61	0.61	5.93	
	Heron	2.23	Japanese Quail	0.150	Chronic NOAEL	6.80	Van Vleet 1982	1	6.8	2.77	1.07	0.48	5.75	

1 - body weight, 2 - (mg CoC/kg-bw/day); 3 - Conversion factor for non-Chronic NOAEL data; 4 - test species NOAELx(bw test/bw Roc)<sup>0.75</sup>, see Section 6.3; 5 - Food Consumption Rate (kg prey/day); see Section 6.3; 6 - FCR/BW<sub>RoC</sub> body weight; 7 - RoC NOAEL/EPC=Exposure Point Concentration. A) Based on Arochlor 1254 toxicity; B) NOAEL = No Observable Effect Level (mg CoC/kg-RoC/day); C) NOAEL level for CoC consumption in food (mg CoC/kg prey dry weight); D) Receptor of Concern; E) assumed to be in the form of sodium arsenite; F) assumed to be in the form of cadmium chloride; G) assumed to be in the form of Cr(+3); H) assumed to be in the form of copper oxide; I) assumed to be in the form of metal; J) assumed to be in the form of mercuric chloride; K) assumed to be in the form of nickel sulfate; L) assumed to be in the form of zinc sulfate.

Table 6.3-3. Qualitative summary of CoC risks to Avian Aquatic Receptors consuming prey in the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location.

A. Herring Gull HQ (Benchmark = TRV-EPC).<sup>1</sup>

Zone	Station	Species	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Silver	Zinc	2-Methylnaphthalene	Acenaphthene	Acenaphthylene	Anthracene	Fluorene	Naphthalene	Phenanthrene	Total PCBs	p,p'-DDE	Tributyltin	Overall Ranking <sup>2</sup>	
1	NSB-1	BM	+	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	NSB-1	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-1-R	BM	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-1-R	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2	NSB-2	BM	+	+	+	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	++
	NSB-2-R	BM	-	-	-	+	+++	-	-	-	++	-	-	-	-	-	-	-	-	-	-	-	+++
	NSB-2-R	MF	-	+	+	+++	+++	-	-	+++	-	-	-	-	-	-	-	-	-	-	-	-	+++
	NSB-2-FD-R	BM	+	-	+	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-2-FD-R	MF	+	-	+	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-3	BM	+	+	+	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-3	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-3-R	BM	+	+	+	+	+	-	-	-	-	++	-	-	-	-	-	-	-	-	-	-	+
	NSB-3-R	MF	-	+	+	+	+	+	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-4	BM	+	+	-	-	+	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+
	NSB-4	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-4-R	BM	+	+	+	++	++	+	-	-	-	+++	-	-	-	-	-	-	-	-	-	-	+++
	NSB-4-R	MF	-	+	+	+++	+	+	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+++
	NSB-5	BM	+	-	+	-	+	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+
NSB-5-R	BM	+	+	+	-	+	-	-	-	-	++	-	-	-	-	-	-	-	-	-	-	++	
NSB-5-R	MF	-	+	-	+	+	+	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+	
3	NSB-6	BM	+	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-6	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-6-R	BM	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-6-R	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-7	BM	+	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-7-R	BM	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
NSB-7-R	MF	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-	
3A	OS-28	HC	+	-	++	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-12	HC	+	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	M1	HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
4	MCL-9	HC	+	+	-	-	+	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-9	HPP	+	+	-	++	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-9	MUS	+	-	-	+	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-10	HC	+	-	-	-	+	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-10	HPP	+	+	-	+	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-10	MUS	+	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
MCL-11	HC	+	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+	
5	OS-22	HC	+	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	OS-23	HC	+	-	+++	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	++
	OS-24	HC	+	-	+++	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	++
	OS-25	HC	+	-	+++	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	++
	OS-26	HC	+	-	++	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	OS-27	HC	+	-	+	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
6	MCL-13	HC	+	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+
	MCL-13	HPP	+	+	-	++	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-13	MUS	+	-	-	+	-	+	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-14	HC	+	-	-	+	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-14	HPP	+	+	-	++	-	-	-	++	+	-	-	-	-	-	-	-	-	-	-	-	++
	MCL-14	MUS	+	-	-	+	-	+	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-16	HC	+	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+
S2	HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
7	JCC-D1	HPP	+	+	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	++	-	-	+
	JCC-D1	MUS	+	-	-	-	-	+	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	JCC-M1	HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	JCC-S1	HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

TRV = Toxicity Reference Value (Table 6.3-2). EPC = Exposure Point Concentration (Prey Species Concentration).  
 HQ = Hazard Quotient = Prey EPC/TRV-EPC.; Ranking: HQ>1 = "+", HQ>10 = "++", HQ>20 = "+++". Raw data in Appendix A-2-4.  
 1 - Residue concentration predicted from bioaccumulation model for Phase III ("R") stations (see Section 6.3).  
 2 - Overall Ranking: see text in Section 6.0-2. Raw HQ data in Appendix A-2-4.  
 BM=blue mussel; HC=hard clam; MF=marine fish (cunner); HPP=lobster hepatopancreas; MUS=lobster muscle.

Table 6.3-3 (continued). Qualitative summary of CoC risks to Avian Aquatic Receptors consuming prey in the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location.

B. Herring Gull HQ (Benchmark = TRV-Dose).<sup>1,2</sup>

Zone	Station	Species	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Silver	Zinc	2-Methylnaphthalene	Acenaphthene	Acenaphthylene	Anthracene	Fluorene	Naphthalene	Phenanthrene	Total PCBs	p,p'-DDE	Tributyltin	Overall Ranking <sup>3</sup>	
1	NSB-1	BM	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
	NSB-1	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
	NSB-1-R	BM	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
	NSB-1-R	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
2	NSB-2	BM	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	
	NSB-2-R	BM	-	-	-	+	++	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+	
	NSB-2-R	MF	-	-	-	++	++	-	-	++	-	-	-	-	-	-	-	-	-	-	-	++	
	NSB-2-FD-R	BM	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	NSB-2-FD-R	MF	-	-	-	+	+	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+	
	NSB-3	BM	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	
	NSB-3	MF	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	NSB-3-R	BM	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	NSB-3-R	MF	-	+	-	+	+	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+	
	NSB-4	BM	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	NSB-4	MF	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	NSB-4-R	BM	-	+	-	+	+	-	-	-	-	++	-	-	-	-	-	-	-	+++	-	-	+++
	NSB-4-R	MF	-	+	-	++	+	+	-	-	+	-	-	-	-	-	-	-	-	+++	-	-	+++
	NSB-5	BM	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
NSB-5-R	BM	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+	
NSB-5-R	MF	-	+	-	+	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+	
3	NSB-6	BM	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	
	NSB-6	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	
	NSB-6-R	BM	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	
	NSB-6-R	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	
	NSB-7	BM	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	
	NSB-7-R	MF	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	
3A	OS-28	HC	-	-	+	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	
	MCL-12	HC	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+	
	M1	HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	
4	MCL-9	HC	-	-	-	-	+	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+	
	MCL-9	HPP	+	+	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+	
	MCL-9	MUS	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	MCL-10	HC	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	
	MCL-10	HPP	+	+	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+	
	MCL-10	MUS	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
5	MCL-11	HC	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	
	OS-22	HC	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	OS-23	HC	-	-	++	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	OS-24	HC	-	-	++	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	OS-25	HC	-	-	++	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	OS-26	HC	-	-	+	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	OS-27	HC	-	-	+	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
6	MCL-13	HC	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+	
	MCL-13	HPP	+	+	-	+	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+	
	MCL-13	MUS	+	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	MCL-14	HC	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+	
	MCL-14	HPP	+	+	-	+	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+	
	MCL-14	MUS	+	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+	
	MCL-16	HC	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
7	JCC-D1	HPP	+	+	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	JCC-D1	MUS	+	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
	JCC-M1	HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	
	JCC-S1	HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	

HQ = Hazard Quotient = Prey Dose/TRV-Dose; Ranking: HQ>1 = "+", HQ>10 = "++", HQ>20 = "+++".

1 - Prey Dose = prey EPC \* EF (Table 6.3-1); TRV = Toxicity Reference Value (Table 6.3-2).

2 - Residue concentration predicted from bioaccumulation model for Phase III ("R") stations (see Section 6.3).

3 - Overall Ranking: see text in Section 6.0-2. Raw HQ data in Appendix A-2-4.

BM=blue mussel; HC=hard clam; MF=marine fish (cunner); HPP=lobster hepatopancreas; MUS=lobster muscle.

Table 6.3-3 (continued). Qualitative summary of CoC risks to Avian Aquatic Receptors consuming prey in the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location.

C. Great Blue Heron HQ (Benchmark = TRV-EPC).<sup>1</sup>

Zone	Station	Species	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Silver	Zinc	2-Methylnaphthalene	Acenaphthene	Acenaphthylene	Anthracene	Fluorene	Naphthalene	Phenanthrene	Total PCBs	p,p'-DDE	Tributyltin	Overall Ranking <sup>2</sup>	
1	NSB-1	BM	+	-	-	-	-	-	-	-	+	.	.	.	.	.	.	.	+	.	.	+	
	NSB-1	MF	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.
	NSB-1-R	BM	-	-	-	-	-	-	-	-	-	.	.	.	.	.	.	.	.	+	.	.	.
	NSB-1-R	MF	-	-	-	-	-	-	-	-	-	.	.	.	.	.	.	.	.	+	.	.	.
2	NSB-2	BM	+	+	+	-	+	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	++
	NSB-2-R	BM	+	+	+	+	+++	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+++
	NSB-2-R	MF	-	+	+	+++	+++	-	-	-	+++	.	.	.	.	.	.	.	.	+	.	.	+++
	NSB-2-FD-R	BM	+	-	+	+	+	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+
	NSB-2-FD-R	MF	+	-	-	+	+	-	-	-	++	.	.	.	.	.	.	.	.	+	.	.	+
	NSB-3	BM	+	+	+	-	+	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+
	NSB-3	MF	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.
	NSB-3-R	BM	+	+	+	+	+	-	-	-	++	.	.	.	.	.	.	.	.	+	.	.	+
	NSB-3-R	MF	-	+	-	+	+	+	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+
	NSB-4	BM	+	+	-	-	+	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+
	NSB-4	MF	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	+	.	.	.
	NSB-4-R	BM	+	+	+	++	++	+	-	-	+++	.	.	.	.	.	.	.	.	+++	.	.	+++
	NSB-4-R	MF	-	+	+	+++	+	+	-	-	+	.	.	.	.	.	.	.	.	+++	.	.	+++
	NSB-5	BM	+	-	+	-	+	-	-	-	+	.	.	.	.	.	.	.	.	++	.	.	+
NSB-5-R	BM	+	+	-	-	+	+	-	-	+	.	.	.	.	.	.	.	.	++	.	.	++	
NSB-5-R	MF	-	+	-	+	+	+	-	-	+	.	.	.	.	.	.	.	.	++	.	.	+	
3	NSB-6	BM	+	-	-	-	+	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+
	NSB-6	MF	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	++	.	.	+
	NSB-6-R	BM	-	-	-	-	-	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+
	NSB-6-R	MF	-	-	-	-	-	-	-	-	.	.	.	.	.	.	.	.	.	+	.	.	.
	NSB-7	BM	+	-	-	-	+	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+
	NSB-7-R	MF	-	-	-	-	-	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+
3A	OS-28	HC	+	-	++	-	-	-	-	-	+	.	.	.	.	.	.	.	.	.	.	.	+
	MCL-12	HC	+	-	-	-	-	-	-	+	+	.	.	.	.	.	.	.	.	.	.	.	+
	M1	HC	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
4	MCL-9	HC	+	+	-	-	+	-	-	+	+	.	.	.	.	.	.	.	.	+	.	.	+
	MCL-9	HPP	+	+	-	++	-	-	-	+	+	.	.	.	.	.	.	.	.	+	.	.	+
	MCL-9	MUS	+	-	-	+	-	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+
	MCL-10	HC	+	-	-	-	+	-	-	+	+	.	.	.	.	.	.	.	.	+	.	.	+
	MCL-10	HPP	+	+	-	+	-	-	-	+	+	.	.	.	.	.	.	.	.	+	.	.	+
	MCL-10	MUS	+	-	-	+	-	-	-	-	+	.	.	.	.	.	.	.	.	+	.	.	+
MCL-11	HC	+	-	-	-	-	-	-	-	+	+	.	.	.	.	.	.	.	+	.	.	+	
5	OS-22	HC	+	-	-	-	-	-	-	-	+	.	.	.	.	.	.	.	.	.	.	.	+
	OS-23	HC	+	-	+++	-	+	-	-	-	+	.	.	.	.	.	.	.	.	.	.	.	++
	OS-24	HC	+	-	+++	-	+	-	-	-	+	.	.	.	.	.	.	.	.	.	.	.	++
	OS-25	HC	+	-	+++	-	+	-	-	-	+	.	.	.	.	.	.	.	.	.	.	.	++
	OS-26	HC	+	-	++	-	+	-	-	-	+	.	.	.	.	.	.	.	.	.	.	.	+
	OS-27	HC	+	-	+	-	+	-	-	-	+	.	.	.	.	.	.	.	.	.	.	.	+
6	MCL-13	HC	+	-	-	-	-	-	-	+	+	.	.	.	.	.	.	.	.	+	.	.	+
	MCL-13	HPP	+	+	-	++	-	-	-	+	+	.	.	.	.	.	.	.	.	+	.	.	+
	MCL-13	MUS	+	-	-	+	-	+	-	+	+	.	.	.	.	.	.	.	.	+	.	.	+
	MCL-14	HC	+	-	-	-	-	-	-	+	+	.	.	.	.	.	.	.	.	+	.	.	+
	MCL-14	HPP	+	+	-	++	-	-	-	++	+	.	.	.	.	.	.	.	.	+	.	.	++
	MCL-14	MUS	+	-	-	+	-	+	-	+	+	.	.	.	.	.	.	.	.	+	.	.	+
MCL-16	HC	+	-	-	-	-	-	-	+	+	.	.	.	.	.	.	.	.	+	.	.	+	
S2	HC	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
7	JCC-D1	HPP	+	+	-	-	-	-	-	+	+	.	.	.	.	.	.	.	.	++	.	.	+
	JCC-D1	MUS	+	-	-	-	-	+	-	-	+	.	.	.	.	.	.	.	.	.	.	.	+
	JCC-M1	HC	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.
	JCC-S1	HC	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.

TRV = Toxicity Reference Value (Table 6.3-2). EPC = Exposure Point Concentration (Prey Species Concentration).  
 HQ = Hazard Quotient = Prey EPC/TRV-EPC.; Ranking: HQ>1 = "+", HQ>10 = "++", HQ>20 = "+++". Raw data in Appendix A-2-4.  
 1 - Residue concentration predicted from bioaccumulation model for Phase III ("R") stations (see Section 6.3).  
 2 - Overall Ranking: see text in Section 6.0-2. Raw HQ data in Appendix A-2-4.  
 BM=blue mussel; HC=hard clam; MF=marine fish (cunner); HPP=lobster hepatopancreas; MUS=lobster muscle.

Table 6.3-3 (continued). Qualitative summary of CoC risks to Avian Aquatic Receptors consuming prey in the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location.

D. Great Blue Heron HQ (Benchmark = TRV-Dose).<sup>1,2</sup>

Zone	Station	Species	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Silver	Zinc	2-Methylnaphthalene	Acenaphthene	Acenaphthylene	Anthracene	Fluorene	Naphthalene	Phenanthrene	Total PCBs	p,p'-DDE	Tributyltin	Overall Ranking <sup>3</sup>	
1	NSB-1	BM	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-1	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-1-R	BM	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-1-R	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2	NSB-2	BM	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-2-R	BM	-	-	+	+	+++	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-2-R	MF	-	+	-	++	++	-	-	++	-	-	-	-	-	-	-	-	-	-	-	-	++
	NSB-2-FD-R	BM	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-2-FD-R	MF	-	-	-	+	+	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-3	BM	+	-	-	-	+	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+
	NSB-3	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-3-R	BM	-	+	-	-	+	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+
	NSB-3-R	MF	-	+	-	+	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-4	BM	-	-	-	-	+	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+
	NSB-4	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-4-R	BM	-	+	+	+	+	-	-	-	-	++	-	-	-	-	-	-	-	-	-	-	+++
	NSB-4-R	MF	-	+	-	+++	+	+	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+++
	NSB-5	BM	-	-	-	-	+	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+
NSB-5-R	BM	-	-	-	-	+	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+	
NSB-5-R	MF	-	+	-	+	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+	
3	NSB-6	BM	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-6	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-6-R	BM	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
	NSB-6-R	MF	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NSB-7	BM	-	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+
	NSB-7-R	MF	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
3A	OS-28	HC	+	-	+	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-12	HC	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	M1	HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
4	MCL-9	HC	-	+	-	-	+	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-9	HPP	+	+	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-9	MUS	+	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-10	HC	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-10	HPP	+	+	-	+	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-10	MUS	+	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	MCL-11	HC	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	+
5	OS-22	HC	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
	OS-23	HC	-	-	++	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
	OS-24	HC	+	-	++	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
	OS-25	HC	+	-	++	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
	OS-26	HC	-	-	+	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
	OS-27	HC	+	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
	6	MCL-13	HC	-	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-
MCL-13		HPP	+	+	-	+	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+
MCL-13		MUS	+	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+
MCL-14		HC	-	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+
MCL-14		HPP	+	+	-	+	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+
MCL-14		MUS	+	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+
MCL-16		HC	-	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+
S2		HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
7	JCC-D1	HPP	+	+	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	JCC-D1	MUS	+	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+
	JCC-M1	HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	JCC-S1	HC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

HQ = Hazard Quotient = Prey Dose/TRV-Dose; Ranking: HQ>1 = "+", HQ>10 = "++", HQ>20 = "+++".

1 - Prey Dose = prey EPC \* EF (Table 6.3-1); TRV = Toxicity Reference Value (Table 6.3-2).

2 - Residue concentration predicted from bioaccumulation model for Phase III ("R") stations (see Section 6.3).

3 - Overall Ranking: see text in Section 6.0-2. Raw HQ data in Appendix A-2-4.

BM=blue mussel; HC=hard clam; MF=marine fish (cunner); HPP=lobster hepatopancreas; MUS=lobster muscle.

Table 6.3-4. Overall qualitative summary of CoC risks to Avian Aquatic Receptors consuming prey in the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location.

A. Herring Gull

Zone	Station	Blue Mussels			Hard Clams			Lobster Hepatopancreas			Lobster Muscle			Cunner			Effects Ranking <sup>3</sup>
		HQ-EPC <sup>1</sup>	HQ-Dose <sup>2</sup>	Ranking <sup>3</sup>	HQ-EPC	HQ-Dose	Ranking	HQ-EPC	HQ-Dose	Ranking	HQ-EPC	HQ-Dose	Ranking	HQ-EPC	HQ-Dose	Ranking	
1	NSB-1	+	.	.													
	NSB-1-R	.	.	.									.	.	.	.	
2	NSB-2	+	+	++									+++	++	+++	+++	
	NSB-2-R	+++	+	++									+++	++	+++	+++	
	NSB-2-FD-R	+	+	+									+	+	+	+	
	NSB-3	+	+	+									.	.	.	.	
	NSB-3-R	+	+	+									+	+	+	+	
	NSB-4	+	+	+									+	+	+	+	
	NSB-4-R	+++	+++	+++									+++	+++	+++	+++	
	NSB-5	+	+	+													
	NSB-5-R	++	+	+									+	+	+	+	
3	NSB-6	+	+	+									.	.	.	.	
	NSB-6-R	+	.	.									.	.	.	.	
	NSB-7	+	.	.									.	.	.	.	
	NSB-7-R	+	.	.									.	.	.	.	
3A	OS-28				+	+	+									+	
	MCL-12				+	+	+									+	
	M1				.	.	.									.	
4	MCL-9				+	+	+	+	+	+	+					+	
	MCL-10				+	+	+	+	+	+	+	.	.			+	
	MCL-11				+	.	.	.	.	.	.					.	
5	OS-22				++	.	.									+	
	OS-23				++	+	+									+	
	OS-24				++	+	+									+	
	OS-25				++	+	+									+	
	OS-26				+	+	+									+	
	OS-27				+	+	+									+	
6	MCL-13				+	+	+	+	+	+	+					+	
	MCL-14				+	+	+	++	+	+	+	+				+	
	MCL-16				+	.	.	.	.	.	.					.	
	S2				.	.	.	.	.	.	.					.	
7	JCC-D1				.	.	.	+	+	+	+	+				+	
	JCC-M1				.	.	.	.	.	.	.					.	
	JCC-S1				.	.	.	.	.	.	.					.	

1 - HQ-EPC = EPC/TRV-EPC; see Table 6.3-3.

2 - HQ-Dose = Dose/TRV-Dose; see Table 6.3-3.

3 - Species/indicator-specific rankings and Effects Ranking: "+++" = intermediate (++) or higher effects observed for two or more indicators/species, one of which indicates high (+++) effects; ++ = intermediate (++) effects observed for two or more indicators/species or high (++) effects for one indicator/species; "+" = low (+) effects observed for two or more indicators/species or intermediate (++) effects for one indicator/species; "." = low (+) effects observed for only one indicator/species or no effects for all indicators/species. Effects rankings for single-species stations equal species-specific ranking. See text in Section 6.3.

Table 6.3-4 (continued). Overall qualitative summary of CoC risks to Avian Aquatic Receptors consuming prey in the McAllister Point Landfill study area and Jamestown Cranston Cove (JCC) reference location.

**B. Great Blue Heron**

Zone	Station	Blue Mussels			Hard Clams			Lobster Hepatopancreas			Lobster Muscle			Cunner			Effects Ranking <sup>3</sup>
		HQ-EPC <sup>1</sup>	HQ-Dose <sup>2</sup>	Ranking <sup>3</sup>	HQ-EPC	HQ-Dose	Ranking	HQ-EPC	HQ-Dose	Ranking	HQ-EPC	HQ-Dose	Ranking	HQ-EPC	HQ-Dose	Ranking	
1	NSB-1	+	-	-													
	NSB-1-R	-	-	-													
2	NSB-2	+	+	+													
	NSB-2-R	+++	++	+++									+++	++	+++	+++	+++
	NSB-2-FD-R	+	+	+													
	NSB-3	+	+	+													
	NSB-3-R	+	+	+													
	NSB-4	+	+	+													
	NSB-4-R	+++	+++	+++										+++	+++	+++	+++
	NSB-5	+	+	+													
	NSB-5-R	++	+	+										+	+	+	+
3	NSB-6	+	+	+										+			
	NSB-6-R	+	-	-													
3A	NSB-7	+	+	+													
	NSB-7-R	+	-	-													
4	OS-28				+	+	+										+
	MCL-12 M1				-	-	-										
5	MCL-9				+	+	+	+	+	+	+	+	+				+
	MCL-10				+	+	+	+	+	+	+	+	+				+
	MCL-11				+	+	+										+
6	OS-22				+	+	+										+
	OS-23				++	+	+										+
	OS-24				++	+	+										+
	OS-25				++	+	+										+
	OS-26				+	+	+										+
	OS-27				+	+	+										+
7	MCL-13				+	+	+	+	+	+	+	+	+				+
	MCL-14				+	+	+	++	+	+	+	+	+				+
	MCL-16				+	+	+										+
	S2				-	-	-										
7	JCC-D1							+	+	+	+	+	+				+
	JCC-M1				-	-	-										-
	JCC-S1				-	-	-										-

1 - HQ-EPC = EPC/TRV-EPC; see Table 6.3-3.

2 - HQ-Dose = Dose/TRV-Dose; see Table 6.3-3.

3 - Species/indicator-specific rankings and Effects Ranking: "+++" = intermediate (++) or higher effects observed for two or more indicators/species, one of which indicates high (+++) effects; ++ = intermediate (++) effects observed for two or more indicators/species or high (+++) effects for one indicator/species; "+" = low (+) effects observed for two or more indicators/species or intermediate (++) effects for one indicator/species; "-" = low (+) effects observed for only one indicator/species or no effects for all indicators/species. Effects rankings for single-species stations equal species-specific ranking. See text in Section 6.3.

Table 6.6-1. Summary of Exposure-based Weights of Evidence for the McAllister Point Landfill Marine Ecological Risk Assessment.<sup>1</sup>

Zone <sup>6</sup>	Station	Sediment Hazard Quotients <sup>2</sup>		Porewater Hazard Quotients <sup>3</sup>				SEM/AVS <sup>4</sup>		Fecal Pollution Indicators <sup>5</sup>		Tissue Concentration Ratios <sup>6</sup>						
		Sediment Hazard Quotients	Zone Exposure Ranking <sup>7</sup>	Copper <sup>3A</sup>	Zinc	Mercury	Nickel	Zone Exposure Ranking <sup>7</sup>	SEM Bioavailability	Zone Exposure Ranking <sup>7</sup>	Sediment Fecal Pollution Indicator Ranking	Zone Exposure Ranking <sup>7</sup>	Blue Mussel	Hard Clam	Lobster Hepatopancreas	Lobster Muscle	Cumner	Zone Exposure Ranking <sup>7</sup>
1	NSB-1	+	+	++	-	+	+	+++	+++	+++	+++	++					+	+
	NSB-1-R	+	+	++	-	+	+	+++	+++	+++	+++	++						+
2	NSB-2	+	+++	++	-	+	+	+++	+++		++	++						++
	NSB-2-R	+++																
	NSB-2-FD-R	+++																
	NSB-3	+++		+	-	+	+		++	+++	++							
	NSB-3-R	+++																
	NSB-4	+++		+	++	+	+		+++			++						
	NSB-4-R	+++																
NSB-5	+++		++	+++	+	+		+++	-		++							
NSB-5-R	+++																	
3	NSB-6	+++	+++	+	-	+	-	-	++		+	++						+++
	NSB-6-R	++																
	NSB-7	++		++	-	+	+	+++		+		+++						
	NSB-7-R	+++																
3A	OS-28†	++	+++								++		+					+
	S2B	+++																
	S2B-R	-																
	S2C	+++																
	MCL-12	+		-	-	+	-			++								
	MCL-12-R	++																
	M1	+							+				++					
M1-R	+																	
4	MCL-8	++	+	-	-	+	-	+	-		++							+++
	MCL-8-R	+																
	MCL-9	+		-	-	+	-	-				++	+++	++				
	MCL-9-R	+																
	MCL-10	+		-	-	+	-	-				++	++	+				
	MCL-10-R	+																
	MCL-11	+		-	-	+	-	-		++			+					
	MCL-11-R	-																
5	OS-22†	+	+										+					+
	OS-23†	++											+					
	OS-24†	-											+					
	OS-25†	+											++					
	OS-26†	+											+					
	OS-27†	+											+					
6	MCL-13	+	+	-	-	+	-	+	+	++	++	++	++	+++				+++
	MCL-13-R	-																
	MCL-14	-		-	-	+	-		+				+	+++	+			
	MCL-14-R	++																
	MCL-15	+		-	-	+	-			++		+						
	MCL-16	+		-	-	+	-					+						
	OS-29†	+																
	OS-30†	+																
	OS-30A	-																
	OS-30B	-																
	D1	+																
	D2	+																
	D3	+																
	M2	+																
	M3	+								++								
	S1	-																
	S2	-											++					
	S3	-																
	S4	+																
	7	JCC-D1	-	-	-	-	+	-	-	+								
JCC-M1		-							+									
JCC-S1		-							-									

† - TRC (1994) data.

SEM = Simultaneously Extractable Metals; AVS = Acid Volatile Sulfides; WQC = EPA Water Quality Criteria.

1 - Phase I = Samples collected by UR/SAIC (1994); Phase II = SAIC/URI (1995); Phase III = resampling (1996); TRC = TRC (1994).

2 - Sediment Hazard Quotient analyte-specific rankings: see Table 6.1-1.

3 - Porewater Hazard Quotients: Analytes included for which WQC are available and CoCs were above detection. See Appendix A-2-3.

Rankings: < WQC-Chronic = -; WQC-Saltwater Chronic to Saltwater Acute = "+"; > WQC-Saltwater Acute = "++"; "+++ = > 2 x WQC-Saltwater Acute.

3A - No WQC-SA value for copper: "-" = < WQC-SC; "+" = > WQC-SC; "++" = > 2 X WQC-SC.

4 - SEM Bioavailability Ranking (see Table 6.1-2).

5 - Sediment Fecal Pollution Indicator Ranking: see Table 4.2-4.

6 - Site vs. Reference Tissue Concentration Ratios (TCRs; Table 6.2-1).

7 - Zone Exposure Ranking: "+++ = intermediate (++) or higher exposure observed for two or more indicators, one of which indicates high (+++) exposure; "++ = intermediate (++) exposure observed for two or more indicators or high (+++) exposure for one indicator; "+" = low (+) exposure observed for two or more indicators or intermediate (++) exposure for one indicator; "-" = low (+) exposure observed for only one indicator or no exposure for all indicators. See text in Section 6.0-2. Exposure rankings for stations for which only one indicator observation was available are equal to the indicator observation ranking.



Table 6.6-3. Overall Summary of Exposure and Effects-based Weights of Evidence and Characterization of Risk for the McAllister Point Landfill Marine Ecological Risk Assessment.

Zone	WEIGHTS OF EVIDENCE										Overall Risk Probability Ranking <sup>10</sup>
	EXPOSURE						EFFECTS				
	Sediment Hazard Quotients <sup>1</sup>	Porewater Hazard Quotients <sup>2</sup>	SEM and AVS <sup>3</sup>	Fecal Pollution Indicators <sup>4</sup>	Tissue Conc. Ratio <sup>5</sup>	Rank <sup>9</sup>	Laboratory Toxicity <sup>6</sup>	Field Effects <sup>7</sup>	Avian Predators <sup>8</sup>	Rank <sup>9</sup>	
1	+	+	+++	+++	+	H	+	++	-	L	Intermediate
2	+++	+++	+++	++	++	H	+++	++	+++	H	High
3	+++	+	++	+	+++	H	+	+	-	L	Intermediate
3A	+++	-	-	++	+	I	+++	++	+	I	Intermediate
4	+	+	-	++	+++	I	+	-	+	L	Intermediate
5	+	-	-	-	+	L	-	-	+	B	Low
6	+	+	+	++	+++	I	++	-	+	L	Intermediate
7	-	-	+	-	-	B	++	+	+	L	Low

1- Sediment Hazard Quotient Zone Exposure Ranking: see Table 6.6-1.

2- Porewater Hazard Quotient Zone Exposure Ranking: see Table 6.6-1.

3- SEM and AVS Zone Exposure Ranking: see Table 6.6-1.

4- Sediment Fecal Pollution Indicators Zone Exposure Ranking: see Table 6.6-1.

5- Tissue Concentration Ratios Zone Exposure Ranking: see Table 6.6-1.

6- Laboratory Toxicity Zone Effects Ranking: see Table 6.6-2.

7- Field Effects Ranking: Based on results of Condition Index, Benthic Community Structure, and Tissue Fecal Pollution Indicators: see Table 6.6-2.

8- Avian Predator Zone Effects Ranking: see Table 6.6-2.

9- Overall Zone Exposure/Effects (E/E) Risk Probability Ranking (see text Section 7.1):

B = Baseline Risk; L = Low Risk Probability; I = Intermediate Risk Probability; H = High Risk Probability.

B = Low (+) E/E ranking observed for only one indicator or baseline E/E ranking observed for all indicators;

L = Intermediate (++) E/E ranking observed for only one indicator or low (+) E/E ranking observed for two or more indicators;

I = High (+++) E/E ranking observed for only one indicator or intermediate (++) E/E ranking observed for two or more indicators;

H = High (+++) E/E ranking observed for two or more indicators;

10- Overall Zone Risk Probability Ranking (see text Section 7.1):

Baseline = No greater than Baseline (B) ranking for E/E WoE summaries;

Low = No greater than Low (L) ranking for E/E WoE summaries;

Intermediate = Intermediate (I) ranking for both E/E WoE summaries, or High (H) ranking for one WoE and no greater than Low (L) ranking for the other WoE summary;

High = High (H) ranking for one WoE summary and Intermediate (I) or greater ranking for the other WoE summary.

## 7.0. SUMMARY AND CONCLUSIONS

This section summarizes the results of the marine ERA conducted for the McAllister Point Landfill, located at the Naval Education and Training Center (NETC) - Newport, RI. The U.S. EPA's ERA Framework and applicable EPA Region I guidance were used to generate and interpret the data required to complete this risk assessment. The objectives of this study were to:

- Assess ecological risks to the offshore environments of McAllister Point and Narragansett Bay from chemical stressors associated with the McAllister Point Landfill;
- Develop information sufficient to support risk management decisions regarding site-specific remedial options; and
- Support communication to the public of the nature and extent of ecological risks associated with the McAllister Point Landfill.

The following sections present and discuss the findings of this Marine Ecological Risk Assessment (ERA), including Problem Formulation, Site Characterization, Exposure and Ecological Effects Assessments, Characterization of Ecological Risks, Risk Synthesis and Uncertainty Analysis.

### 7.1. SYNTHESIS OF STUDY FINDINGS

The findings of exposure and effects indicators within the each overall WoE are evaluated jointly in order to interpret the *overall probability* of adverse ecological exposure/effects (E/E) by zone. The synthesis of risk by Ecological Exposure Zone (EEZ; Table 6.6-3) is supported by the information presented in Exposure (Table 6.6.1)

and Effects (Table 6.6.2) summary tables, as well as equally important evaluations of the strength of exposure-response relationships and/or presence of confounding factors which could artificially mask or enhance perceived risks. The zones correspond to various geographic- and depth-related characteristics of the region as well as CoC exposure and effects. The risk summary table includes exposure information (e.g., chemical concentrations in sediments, porewater and tissues) and effects-based measures (e.g., toxicity, field effects, and possible effects on avian aquatic predators related to consumption of contaminated prey).

The classification of risk for the McAllister Point Landfill Marine ERA are grouped into the following categories: high, intermediate, low and baseline. Definitions of each category are presented in Section 6.0; a summary of risk rankings by zone is presented below.

*High Risk Probability Zones.* As described in Section 6.0, high ecological risk is suggested by numerous weights of evidence indicating probable exposure and effects, as well as demonstrable exposure-response relationships. In the present investigation, Zone 2 is categorized as the high risk zone for the study area. The conclusion of high risk observed in Zone 2 is supported by numerous weights of evidence suggesting high CoC-related exposure (e.g. high sediment Hazard Quotients) and effects (e.g. high toxicity, altered benthic community structure), as well as the existence of plausible exposure-response relationships (i.e. metals).

*Intermediate Risk Probability Zones.* Intermediate ecological risks are typically associated with multiple exposure- or effects-based weights of evidence occurring, but generally not both. However, quantitative exposure-response relationships are typically lacking. Intermediate risk probability may also be indicated by highly localized apparent impact, or impact of very limited duration. Zones which demonstrate intermediate risk probability include Zone 1, Zone 3, Zone 3A, Zone 4 and Zone 6. It is difficult to

delineate the spatial extent of impacts at Zone 3A, but the area would appear somewhat isolated given lack of exposure and effects indicators observed at proximal stations to this zone. A potential factor mitigating risk in Zone 3A is the fact that sediments are covered by a hard pebble and shell layer which must be penetrated to access the finer grained sediment. Thus, resuspension and transport of CoCs away from this zone would appear limited somewhat by the geology of the environment.

*Low Risk Probability Zones.* A low risk probability was indicated for the remainder of the McAllister Point Landfill study area which includes Zone 5 and reference Zone 7. Thus, the probability of landfill-related risk to ecological receptors associated with Zone 5 is comparable to the probability of ecological risks associated with reference Zone 7.

*Baseline Risk Probability Zones.* None of the zones met the definition for baseline risk.

In most cases, the overall Exposure WoE for the each zone was the same or greater as the Effects WoE, which is expected when the exposure pathway being evaluated has been properly evaluated, i.e. sediment or sediment-associated CoCs are measured and found to be causing the adverse exposure which results in adverse effects. The one instance where this did not occur was for reference Zone 7, where ammonia toxicity was believed to have contributed to the observed response, hence CoCs were likely not the primary cause of the observed effects. This finding helps to substantiate the presumed risks and reduce associated uncertainty.

## **7.2. OTHER POTENTIAL SOURCES OF STRESS AND CoCs**

The present day routes of CoC transport are most likely coming from erosion and resuspension of in place sediment contaminants. Prior to capping, surface water runoff and seep water percolating out of and through the landfill above grade may have also been more important sources of CoCs than in present day, although the investigation of the relative contributions of the two sources has not been completed as of this writing.

Contamination from other sources may potentially enter the landfill region through creeks and culverts to the north and south of the site. Although this study was not designed to directly measure these sources, circumstantial evidence from fecal pollution indicator data does support this possibility. Such as source, however, clearly does not explain the bulk of contamination found in the landfill intertidal zone or nearshore, subtidal environments. Hence, it is concluded that indigenous biological communities in the immediate vicinity of the landfill are at risk primarily due to landfill-related stressors.

## **7.3. LIMITATIONS OF THE ASSESSMENT**

The conclusions drawn in this assessment are based on an extensive database of sediment and tissue chemistry, biological indicators, and toxicity evaluations, with broad spatial and temporal coverage. The data are internally consistent and supportive, and of high quality, meeting and exceeding, for example, detection limits as specified by the NOAA Status and Trends Program. Therefore, the values can be interpreted with confidence for comparisons to commonly accepted guidelines, such as ER-L values (Long *et al.*, 1995).

The assessment of ecological risk is a process of minimizing uncertainty with regard to characterization of exposure and effects, and the integration of these data as cause-effect relationships. The risk conclusions reached in this study are based on weight of evidence; those areas exhibiting more numerous lines of evidence for or against adverse impact are associated with less uncertainty in the conclusion. The present study provides extensive weight of evidence for the intertidal and nearshore subtidal zones of the McAllister Point Landfill study area upon which the risk conclusions were based. Somewhat more limited information was available for Zone 5 (e.g. no toxicity data), as the data was derived from a previous (TRC/BOS) study.

The apparent similarity of chemical bioaccumulation among species as predicted by equilibrium partitioning suggests that this pathway is well understood. Similar models for metals bioaccumulation are not presently available, and accordingly there exists greater uncertainty with regard to the extrapolation of data from the target species (e.g., cunner) to other species that are of concern (e.g., winter flounder) in the ecosystem. This uncertainty is reflected in the greater variance observed among metals BAF values, relative to that observed for organic chemicals. Differences observed between species can, in some cases, be related to transport mechanisms for the metals. In general, however, the variance in bioaccumulation factors among species for most metals and the organics is constrained to approximately 2-5 fold for metals, and two-fold for organics. Hence the models would appear to apply to other target receptors (e.g. winter flounder) not sampled in this investigation.

The present study, taken in consideration with prior investigations, yields an extensive data set comprising primarily spatial coverage. However, uncertainty exists in that seasonal effects were not specifically considered in the present study. The Phase III of this investigation determined that a sediment erosion event at the site modified CoC exposure and effects of landfill-related CoCs in certain zones of the study

area. It is unknown as to what effect future erosional events, if any, might have on CoC bioavailability and associated risks.

The present study was conducted under a comprehensive Work/Quality Assurance Plan, and data validation has been performed and found to meet the study requirements. Potential errors in the study design and protocols were minimized through peer review and evaluation. Data collection activities were reasonably complete, but perhaps limited by less than desirable abundances of fish and bivalves, particularly at the reference site. However, the available site tissue residue data for various species does suggest that trends in chemical composition and bioaccumulation are similar among species. This finding reduces the uncertainty in extrapolation of exposure pathways and effects from target receptor species which were directly measured and other (e.g., winter flounder) whose present abundances did not permit collection.

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