Risk Assessment of the Potential Release of PCBs and Other Contaminants from Sunken Navy Ships in the Deep Ocean: Ex-AGERHOLM Case Study

SPAWAR
Systems Center
San Diego

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FINAL REPORT

PEO SHIPS
Program Executive Office Ships

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Prepared by:
SPAWAR Systems Center San Diego
Battelle Ocean Sciences
MEC Analytical Systems, Inc.

PEO SHIPS

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1.0. Executive Summary

1.1. Introduction

This report assesses the ecological and human health risks of polychlorinated biphenyls (PCBs) that could be released from former Navy vessels sunken in the deep ocean during training and weapons testing conducted as part of the Navy's deep water sinking exercise (SINKEX) program. This document presents the ecological risks determined from a detailed site-specific study of the ex-AGERHOLM; a destroyer (DD-862) sunk in 916 m (2750 ft) of water about 120 nm off the coast of San Diego, CA, in 1982. The site-specific assessment followed the guidelines recommended by the "Ocean Testing Manual (Green Book)" (USEPA/USACE 1991) and the "Framework for Ecological Risk Assessment" (USEPA 1992, 1998a, b) to assess ecological impacts to deep-sea benthic, epibenthic, and pelagic receptors at the site. In addition, concentrations of PCBs measured in commercially harvested sablefish collected from the ex-AGERHOLM site and reference sites located about 4 nm from the ex-AGERHOLM were used to assess the ecological risks from trophic transfer of PCBs (Supplement I) and human health risks from seafood consumption (Supplement II). The technical data and information presented in this report were developed to meet the regulatory requirements identified by the U.S. EPA for conducting SINKEX missions in deep water off the continental shelf.

Prior to 1989, the Navy's SINKEX program intentionally, and sometime inadvertently, sank target vessels – former U.S. Navy warships – during live fire training and weapons testing exercises. Most SINKEXs took place in ocean waters greater than 3,000 m (6,000 ft) in depth, although some ships inadvertently sank in depths of 600-1000 m (1,800-3,000 ft). In 1989 the Navy voluntarily suspended the SINKEX program and discontinued the donation of ships for reef building (REEFEX) when it was discovered that PCBs-in solid materials (PCBs-ISM) were likely to be found aboard the ships being used for these programs. The solid materials containing PCBs included electrical cable insulation, rubber products, felt gaskets, bulkhead insulation, paints, and other similar materials.

An initial evaluation of the problem, based on limited laboratory data, existing literature, and a numerical model of PCB releases in the deep ocean, concluded that PCBs left onboard target ships did not pose an adverse risk to the marine environment (Richter et. al., 1995). However, the U.S. Environmental Protection Agency (EPA) noted that the report contained no field data and did not consider the release of potentially toxic substances other than PCBs. The EPA stipulated that field data documenting acceptable risk from sunken SINKEX ships would be needed for the EPA to concur with continuation of the Navy's SINKEX program. In 1995, the Space and Naval Warfare Systems Center San Diego (SSC-SD, formerly the Naval Command Control and Ocean Surveillance Center) in cooperation with the EPA Office of Water developed a plan to evaluate the marine environment in the vicinity of a sunken U.S. Navy ship to determine whether potentially toxic materials from that ship (particularly PCBs) were being released, and whether any releases might pose an adverse risk to the environment. A PCB Leach Rate study and a field study of the environmental effects from sunken Navy vessels at SINKEX sites was proposed by

the Navy, reviewed by the EPA, and approved in an agreement signed by the EPA and the Navy in 1996.[1]

The field study consisted of a series of oceanographic cruises conducted between 1996 and 1999, first to locate a vessel, and then to collect site-specific data from the deep ocean environment adjacent to the sunken hulk and reference areas surrounding the site. The results of the study found no evidence of ecological risk to deep-sea benthic, epibenthic, and pelagic communities associated with PCBs-ISM onboard the ex-AGERHOLM. Furthermore, analysis of PCB concentrations in sablefish collected from the ex-AGERHOLM and reference sites showed negligible risk to ecological receptors from trophic transfer of PCBs (Supplement I) and negligible risk to human consumers of sablefish though the commercial market basket pathway (Supplement II). The approach, results, and conclusions of the field study are presented in this report.

1.2. Technical Approach

The objectives of the study were to determine if potential contaminants of concern (PCOC): (1) have been released from a representative sunken naval vessel, and if so, (2) whether they have adversely impacted the adjacent marine environment. A site conceptual model was developed to relate the possible source and release of contaminants from the sunken vessel to the biological receptors ultimately at risk in the deep-sea ecosystem and food chain (e.g., human health) (Figure 1-1). Site-specific data were collected to evaluate the exposure pathways, assess impacts to the assessment endpoints, and develop a risk-based assessment of potential impacts from releases of PCBs from SINKEX vessels. Additionally, the decision was made to consider polycyclic aromatic hydrocarbons (PAHs) and metals as PCOC in view of the possibility of their potential to cause toxicity and bioaccumulated in the food chain.

[1] Agreement Between the Department of the Navy and the United States Environmental Protection Agency: USE OF VESSELS CONTAINING POLYCHLORINATED BIPHENYLS AS TARGETS AND TEST PLATFORMS RESULTING IN THEIR SINKING, Signed 19 August 1996 by R.B. Pirie, Jr., ASN (I&E), Dept. of the Navy, and S.A. Herman, Asst. Administrator for Enforcement and Compliance Assurance, U.S. EPA.

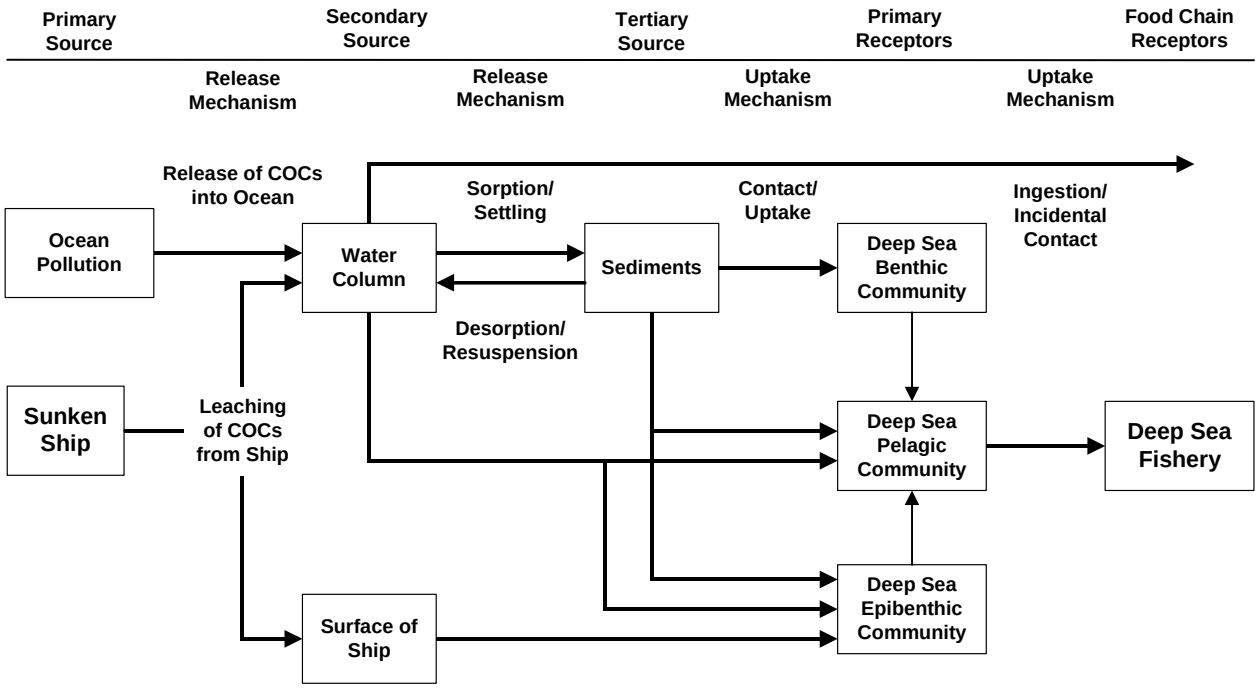

Figure 1-1. Site Conceptual Model depicting the pathways between stressors at sunken ship and ecological and human health receptors at risk.

1.2.1. The Study Site and General Assessment Approach

A focused attempt was made to locate a suitable hulk, which had been sunk under the SINKEX program, to meet the requirements of the field study. Considerable time and effort was expended in the search for a candidate vessel, until the summer of 1996, when the hulk of the ex-AGERHOLM was discovered. The ex-AGERHOLM (DD-862) was a WWII-era destroyer sunk during a weapons test in June 1982. The sunken vessel is located approximately 120 nautical miles off the southern California coast at a depth of 916 m (2,750 ft, Figure 1-2). While only a single sunken ship was located and studied in this effort, information obtained from ex-AGERHOLM site was assumed to be representative of the types of ships of that class, age, and degree of preparation used as expendable targets in the pre-1990 SINKEX program. Since 1990, SINKEX ships have been more extensively cleaned, particularly for PCBs. Therefore; the ex-AGERHOLM likely contained more PCBs-ISM than ships sunk after 1998. Consequently, more recent SINKEX vessels will likely pose less risk from PCBs-ISM.

The ex-AGERHOLM was sunk at a shallower depth (2,750 ft) rather than the required 6,000 ft in accordance with a Navy decision (CNO Memorandum Ser 04R/395370 dated 29 December 1982). The reason for this was that the vessel began to take on excessive water as it was being towed to a designated SINKEX site, and it sunk prematurely. For the purposes of this study, the site was considered a deep-water disposal site and evaluated within the framework of the "Evaluation of Dredged Material Proposed for Ocean Disposal, Testing Manual" (commonly referred to as the "Ocean Testing Manual" or the "Green Book"); jointly produced by the EPA and the U.S. Army Corps of Engineers (USACE) (USEPA/USACE, 1991). Additionally, the USEPA's "Framework for Ecological Risk Assessment" (ERA) (USEPA, 1992, 1998a, b) was used to provide guidance in structuring the study as a risk assessment.

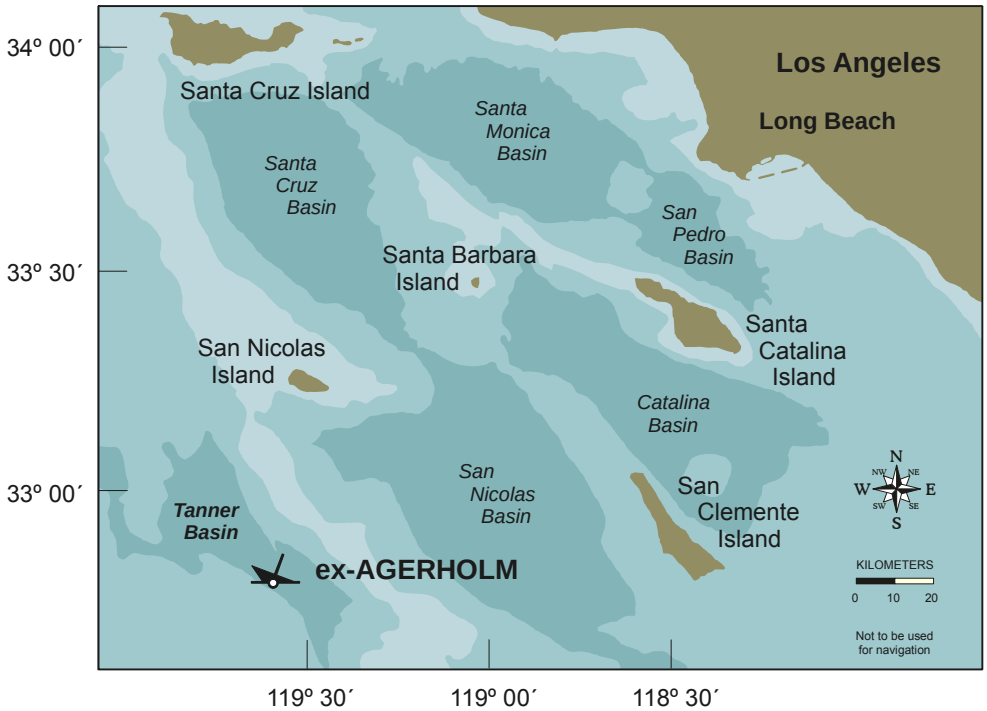

Figure 1-2. Geographic setting of sunken ship ex-AGERHOLM 120 nm west of San Diego.

The Risk Assessment Framework recommends the logical progression of study planning commencing with problem formulation, proceeding through an analysis phase consisting of an exposure and effects assessment, and culminating with a risk characterization phase. For the SINKEX Study, the problem formulation phase began with stressor/source characterization, progressing to the selection of assessment endpoints to represent those ecological attributes, which require protection, and culminating with the development of the site conceptual model (Figure 1-1). The site conceptual model describes the expected environmental pathways for exposure and effects to the assessment endpoints. The Ocean Testing Manual guides the risk assessor through a tiered-sampling approach based on a triad of tests: chemical analysis to measure contaminant concentrations in sediment, sediment toxicity bioassays, and sediment bioaccumulation analyses. The analysis plan developed for the SINKEX Study used the triad of tests as specific measures of exposure and effects to evaluate potential effects to the assessment endpoints identified in the conceptual model.

1.2.2. Potential Stressors of Concern

Polychlorinated biphenyls (PCBs) are a group of synthetic halogenated aromatic hydrocarbons that have been linked to adverse environmental and public health issues. As mentioned previously, the regulatory driver for this study was PCBs-ISM that had been discovered on Navy ships; thus PCBs are the primary contaminant of concern (PCOC). Total PCBs within this investigation were defined in two different ways. The first approach summed the 21 congeners used in the Ocean Testing Manual (referred to as the “Green Book 21”) using the value of zero for any nondetected congener (USEPA/USACE 1991). The second approach included the addition of 5 coplanar congeners considered important in environmental health assessments (referred to as the “SINKEX 26”) using half of the detection limit for any undetected congener.

Navy ships are comprised mostly of metal; therefore, it is possible that metal corrosion processes could also be responsible for any potential toxicity effects and/or bioaccumulation observed during the study of a SINKEX vessel. To address the possibility that effects not associated with PCBs (i.e., the primary stressor of concern) might be found, it was decided to sample for metals – the main constituent of Navy ship construction. Correspondingly, eight potential metal contaminants found in a typical Navy ship were considered secondary contaminants of concern (SCOCs): mercury (Hg), zinc (Zn), lead (Pb), chromium (Cr), copper (Cu), nickel (Ni), cadmium (Cd) and silver (Ag).

Another set of SCOCs was polycyclic aromatic hydrocarbons (PAHs). Many PAH compounds are normally present in fuel and lube oils. While the Navy attempted to empty all target ships (including the ex-AGERHOLM) of these liquids during preparation, it is possible that some PAH compounds remained after cleaning, leaving small volumes or residues at the bottom of storage tanks, operating machinery, and other equipment.

Finally, several other kinds of sediment properties were selected to aid in data interpretation related to the above stressors of concern, including acid volatile sulfides/simultaneously extracted metals (AVS/SEM), particle size, total organic carbon (TOC) and calcium carbonate ($CaCO_3$).

1.2.3. Assessment & Measurement Endpoints, and the Sampling and Analysis Design

As part of the Problem Formulation phase in this SINKEX ERA, and based on an evaluation of the exposure pathways to PCBs associated with the ship, assessment endpoints were identified to evaluate effects in the following deep ocean communities:

The growth and survival of:

- Benthic infauna living in the deep ocean sediment bed (Deep Sea Benthic Community);
- Epibenthic infauna living on top of the deep ocean sediment bed (Deep Sea Epibenthic Community); and
- Pelagic fish present in the deep ocean (Deep Sea Pelagic Community).

Integrating these assessment endpoints with the testing scheme recommended by the Ocean Testing Manual, an analysis plan was developed to use (1) sediment chemistry, (2) sediment acute and chronic toxicity bioassays, and (3) sediment bioaccumulation analyses as the three primary lines of evidence in the SINKEX study. The following laboratory tests were used to assess sediment toxicity:

1. 10-day acute toxicity test (survival and reburial, the latter test used only to validate the survival test) on surrogate amphipods (*Rhepoxynius abronius*), exposed to site sediments.
2. 28-day chronic toxicity test (growth and survival) on surrogate polychaetes (*Neanthes arenaceodentata*), exposed to site sediments.

To assess bioaccumulation of contaminants from the sediments, the following laboratory tests were used:

3. Tissue concentrations of PCBs, metals, and PAHs in surrogate clams (*Macoma nasuta*), after 28-day laboratory exposures to sediments from the site.

4. Tissue concentrations of PCBs, metals, and PAHs in surrogate polychaetes (*Nephtys caecoides*) after 28-day laboratory exposures to sediments from the site.

1.2.4. Use of a Decision Matrix in a Weight-of-Evidence Approach

A decision matrix (Table 1-1) was developed to establish the decision rules prior to commencement of the field studies for the final evaluation of ecological risk due to PCBs from the sunken ship ex-AGERHOLM. This decision matrix was the result of planning efforts conducted jointly with the EPA to ensure that the technical approach and final evaluation criteria were consistent with the U.S. EPA's framework for Ecological Risk Assessment and the Ocean Testing Manual's protocols for the testing of contaminated sediments prior to ocean disposal.

Table 1-1. Risk decision matrix for sediment data.

Outcome Number	Sediment Chemistry	Acute/Chronic Toxicity	Laboratory Bioaccumulation	Benthic Community	Indication of Risk
1	–	–	–	±	None
2	+	–	–	±	None
3	–	+	–	±	Potential Local
4	+	+	–	±	Probable Local
5	+	–	+	±	Potential Food Chain
6	+	+	+	±	Probable Local and Potential Food Chain

+ Significant unfavorable difference for Inner Ring for any endpoints measured.
– No significant unfavorable difference for Inner Ring for endpoints measured.
± No effect on decision.

1.2.5. Sampling Execution and Statistical Analysis

Three successful sampling efforts on the ex-AGERHOLM took place between September 1998 and November 1999. Eighteen sediment stations were sampled on two rings located 2-4 meters (Figure 1-3) from the vessel and approximately 1,000 meters (reference stations, Figure 1-4), respectively. The project design included precision sediment sampling (±2 m) through the use of a remotely operated vehicle (ROV), the development of an innovative sampling device capable of obtaining sediment with an intact (i.e., undisturbed) sediment surface, ultra-low chemical analyses of sediment and tissue samples, radiological dating of sediments, sedimentation rate sampling, toxicity and bioaccumulation testing utilizing two marine species as surrogates for deep-water infauna, bottom current measurements and bottom water chemistry. Stations on the same ring were considered to be field replicates for the ship or reference site.

In addition, fish traps were deployed to obtain samples of representative deep-ocean pelagic fish found at the ex-AGERHOLM site and reference sites located about 4 nm from the hulk (Figure 1-4). These data were used to evaluate the ecological risk of trophic transfer of PCBs (Supplement I) and assess human health risks associated with seafood consumption (Supplement II).

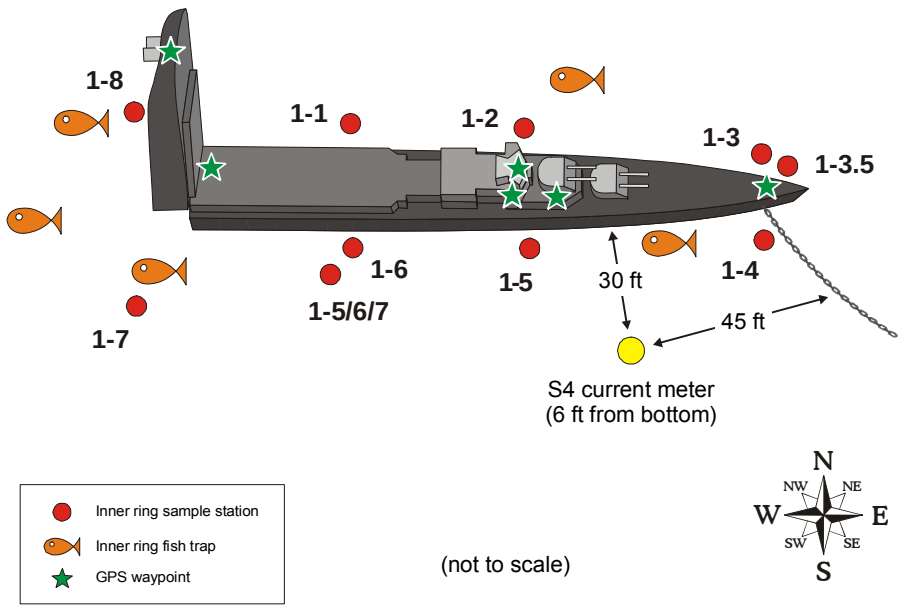

Figure 1-3. Sampling design for Inner Ring.

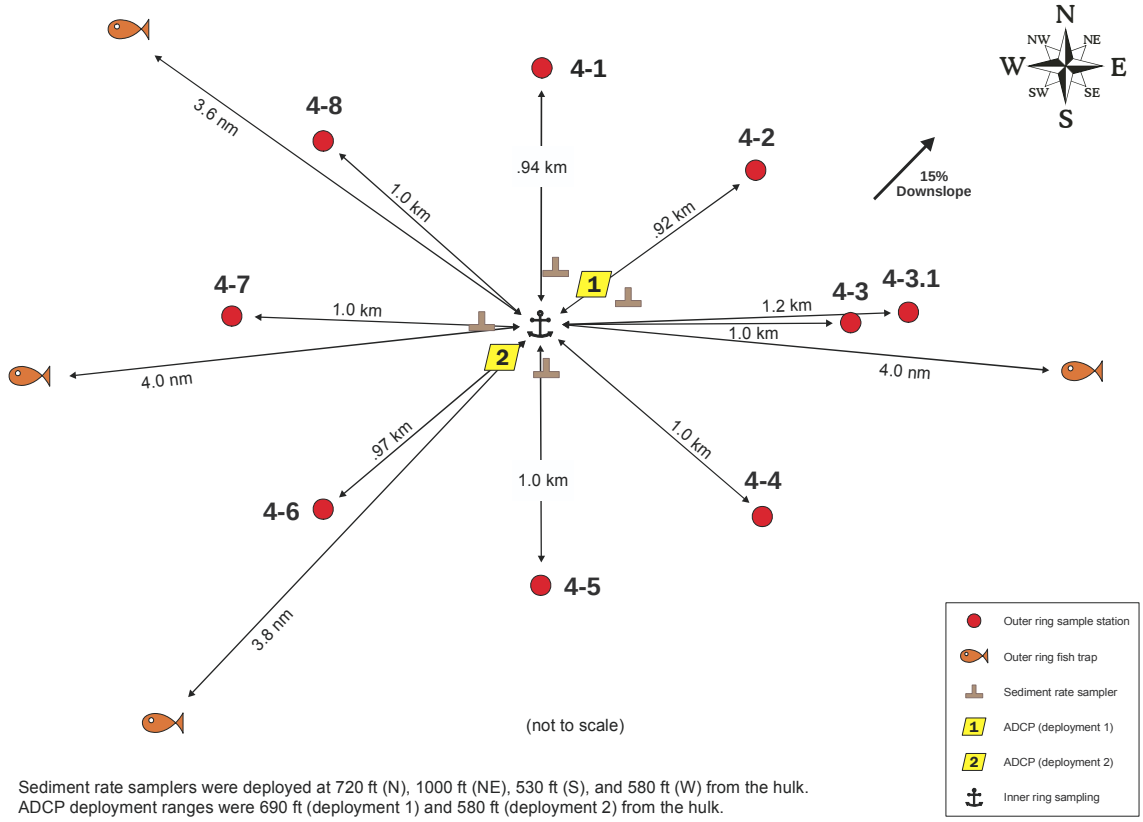

Figure 1-4. Sampling design for reference sites of Outer Ring sediment stations and fish sampling stations.

A two-way analysis of variance (ANOVA) was used to compare conditions at the sunken vessel site to the reference sites with sample date as the interaction variable. Because not all samples needed were collected during a single cruise and the individual cruises were about one year apart, time of sampling was included as a possible interaction term in the statistical analysis. The null hypothesis addressed by this investigation was:

> *There is no significant statistical difference ($p<0.05$) between the ex-AGERHOLM and reference sites for any of the biological and chemical measurements performed.*

where p is the probability that any differences are not due to chance alone.

1.3. Study Results

The preponderance of data indicates no significant elevation of PCBs in the sediment/or in tissue of benthic organisms since the sinking of the ex-AGERHOLM in 1982. Table 1-2 summarizes the overall results for the three "determining" lines of evidence, with a "NS" indicating that there was no statistical difference ($p<0.05$) found between the Inner Ring (ex-AGERHOLM site) and the Outer Ring (Reference site).

Table 1-2. Overall results from the three lines of evidence used in risk determination.

Line of Evidence	Overall Result	Results of Individual Tests			
PCB Chemistry	NS[1]	21GB Congeners[2]		26 SINKEX Congeners[3]	
		NS		NS	
Toxicity	NS	10-day *Rhepoxynius*		28-day *Neanthes*	
		Survival[4]	Reburial[5]	Survival	Growth[6]
		NS	NS	NS	NS
PCB Bioaccumulation	NS	28-day *Nephtys*		28-day *Macoma*	
		Tissue Concentration		Tissue Concentration	
		NS		NS	

Notes:

1) NS= Not statistically ($p<0.05$) significant

2) 21GB Congeners=Total Green Book PCBs are determined through the summation 21 specific PCB congeners, with non-detected congeners given a value of zero.

3) 26 SINKEX=The 21 Green Book Congeners + 5 additional coplanar PCBs with non-detected congeners given a value of ½ the detection limit.

4) 10% survival difference was statistically different, but did not meet Green Book species-specific requirements to show a 20% difference, based on inherent variability in response of this particular species.

5) Reburial testing used to validate the survival test (if no statistical difference, survival data can be used)

6) Large difference in Zero Time size of test organisms caused uncertainty in results, but Inner Ring test worms grew faster, relative to control worms, than those of the Outer Ring.

1.3.1. Primary Line of Evidence: PCB Chemistry in Sediments

Sediment chemistry was evaluated by comparing the sum of PCB congeners measured in samples from the Inner Ring to samples from the Outer Ring and determining whether sediment concentrations exceeded benchmark concentrations associated with causing ecological effects. "Effects range" values for various chemical contaminants have been developed by the National

Oceanographic and Atmospheric Administration's (NOAA) National Status and Trends Program (NS&T, Long and Morgan 1991, Long et al. 1995) and have been recommend for use for assessing ecological risk of contaminated sediments in estuarine and marine environments (USEPA 1996, Buchman 1999). The Effects Range-Low (ERL) is the concentration of a chemical below which adverse biological effects are rarely observed. Alternatively, a higher benchmark, the Effects Range-Median (ERM) is a chemical concentration, above which adverse biological effects frequently occur.

The mean PCB concentrations measured in the deep ocean sediments collected from each of the 10 Inner Ring and eight Outer Ring stations based on the sum of 21 Green Book congeners and 26 SINKEX congeners are shown in Figure 1-5 and Figure 1-6, respectively. The PCB concentrations measured in samples from the Inner Ring (mean 3.90 µg/kg, range 0.49-9.80 µg/kg for 21 Green Book and mean 5.54 µg/kg, range 0.36-14.25 µg/kg for 26 SINKEX) were about twice as high as PCB concentrations measured in samples from the Outer Ring (mean 1.52 µg/kg, range 0.41-4.28 µg/kg for 21 Green Book and mean 2.07 µg/kg, range 0.87-4.85 µg/kg for 26 SINKEX), but the differences were not statistically significant (21 Green Book congeners p=0.13 and 26 SINKEX congeners p=0.07). All sediment PCB concentrations were below the ERL (22.7 µg/kg).

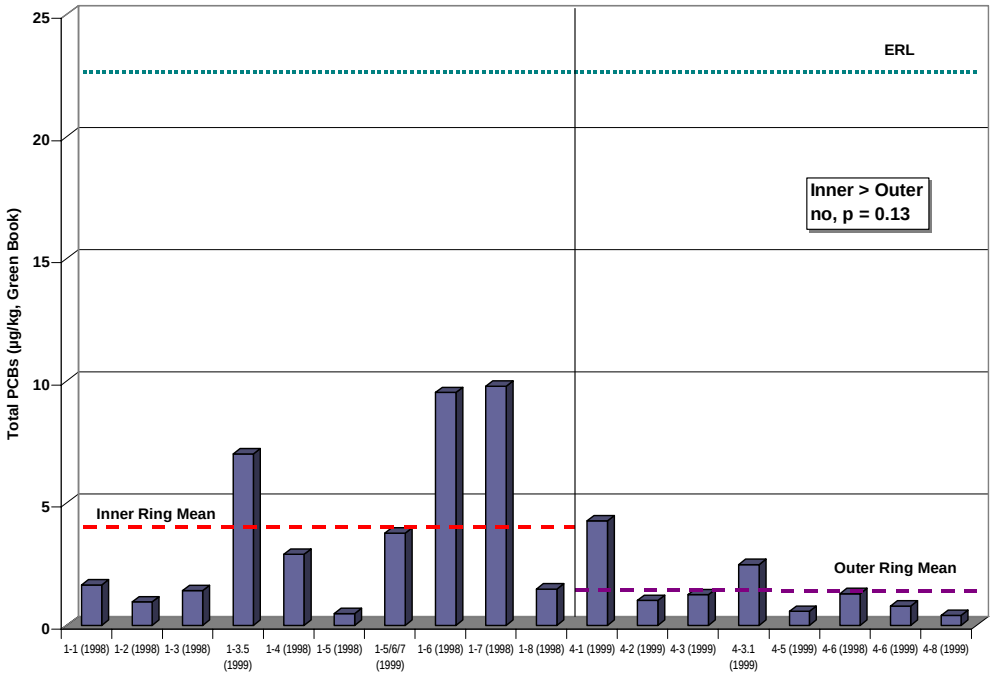

Figure 1-5. Total of 21 Green Book PCB Congeners measured at Inner and Outer Ring stations. Labels on x-axis denote sample location (see Figure 1-3) and year collected.

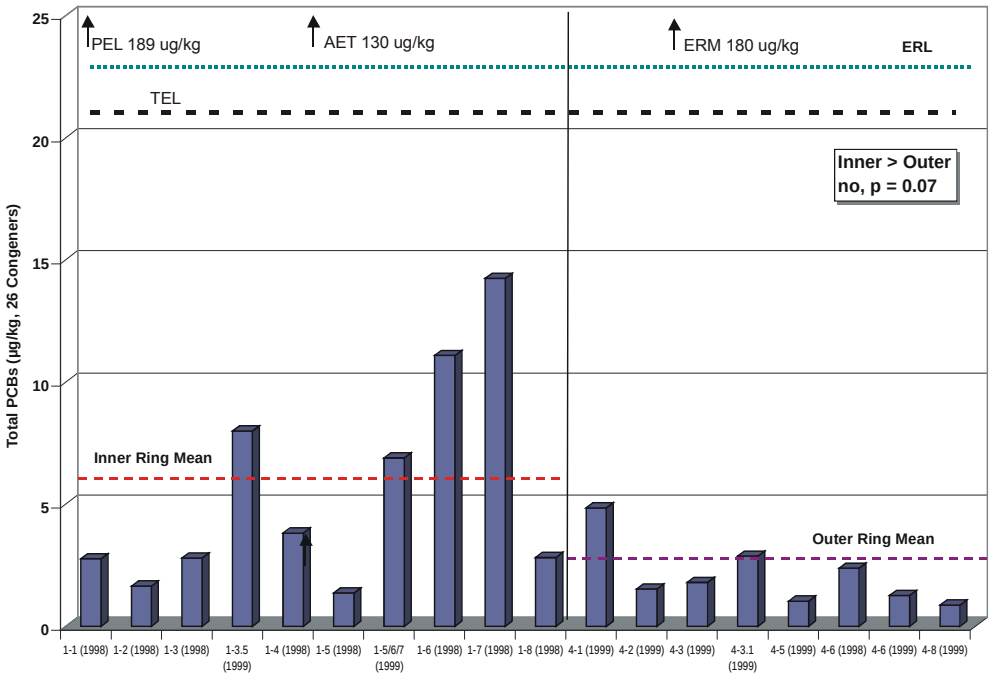

Figure 1-6. Total of 26 PCB Congeners measured at Inner and Outer Ring stations. Labels on x-axis denote sample location (see Figure 3) and year collected. One-half detection limit was used for nondetected congeners.

1.3.2. Primary Line of Evidence: Sediment Toxicity

Sediment toxicity was evaluated by determining whether there was statistically and biologically significant increase of toxicity in sediments from the Inner Ring compared to the Outer Ring. Statistical significance was evaluated using a two-way ANOVA with site and time of sampling as the interacting variables. Biological significance was defined by the Green Book as >20% reduction in survival relative to controls (USEPA/USACE 1991).

Results for the three sediment toxicity tests are shown in Figure 1-7, Figure 1-8, and Figure 1-9. Comparing the Inner Ring (ship site) to the Outer Ring (reference site), the acute 10-day sediment toxicity test using *Rhepoxynius* (amphipod) resulted in mean survival values of 83% and 93%, respectively. The reduction in is survival measured in sediments Inner Ring was less than the Green Book criteria of 20%, so the overall result was “not significant” (Figure 1-7). Reduced amphipod survival at Station 1-6 (65% survival) may have been caused by coarser grain size and less total organic carbon (TOC) present in the sediments from that station.

The *Neanthes* (worm) chronic 28-day test showed no evidence of toxicity. Mean survival for the Inner Ring (96%) was very close to that of the Outer Ring (98%) and was not statistically significant (p=0.4, Figure 1-8).

For the 28-day *Neanthes* growth test, mean absolute growth rates were greater for the Outer Ring (0.13 mg/day) than for the Inner Ring (0.1 mg/day, Figure 1-9), but a different interpretation resulted when growth rates were compared to each of their respective control groups. The Outer Ring tests began with a batch containing larger worms, and the growth of these larger worms was less (17% slower growth suppression) than that of the Inner Ring worms (30 % slower

growth), relative to their respective negative (i.e., native “unexposed” sediment) controls (see Relative Percent Difference, Figure 1-9).

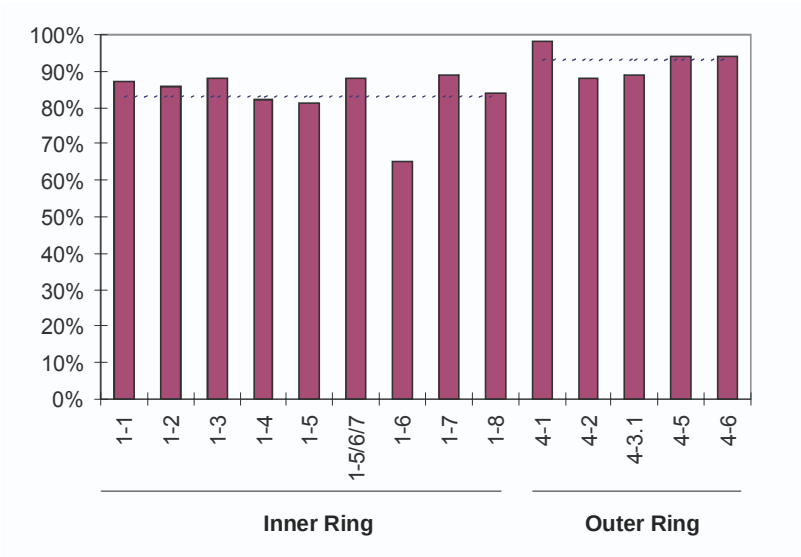

Figure 1-7. *Rhepoxynius abronius* survival.

Figure 1-8. *Neanthes arenaceodentata* survival.

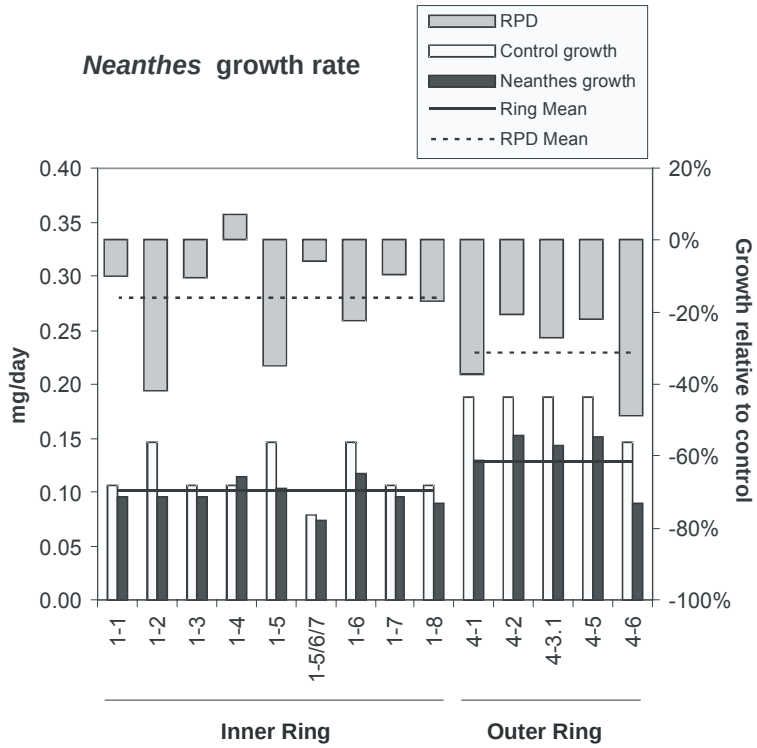

Figure 1-9. *Neanthes arenaceodentata* growth rates (RPD=Relative Percent Difference).

1.3.3. Primary Line of Evidence: PCB Bioaccumulation.

The potential for PCBs to accumulate in the food chain was evaluated by determining whether there was a statistically significant increase in the bioaccumulation of PCBs in organisms exposed to sediment from the Inner Ring compared to the Outer Ring. Statistical significance was evaluated using a two-way ANOVA with site and time of sampling as the interacting variables (USEPA/USACE 1991).

Figure 1-10 and Figure 1-11 present the PCB bioaccumulation test results for *Macoma* and *Nephtys*, respectively, for both the sum of 21 Green Book congeners and the sum of 26 SINKEX congeners. Comparisons between Inner and Outer Ring showed no statistical differences at the $p<0.05$ levels for *Macoma* (Figure 1-10) or for *Nephtys* (Figure 1-11). Station 1-8 was notably elevated, with the highest PCB concentrations for both *Macoma* and *Nephtys*, at nearly quadruple the next highest station in *Macoma.* This and other elevated results at the ship's stern are discussed in the following section.

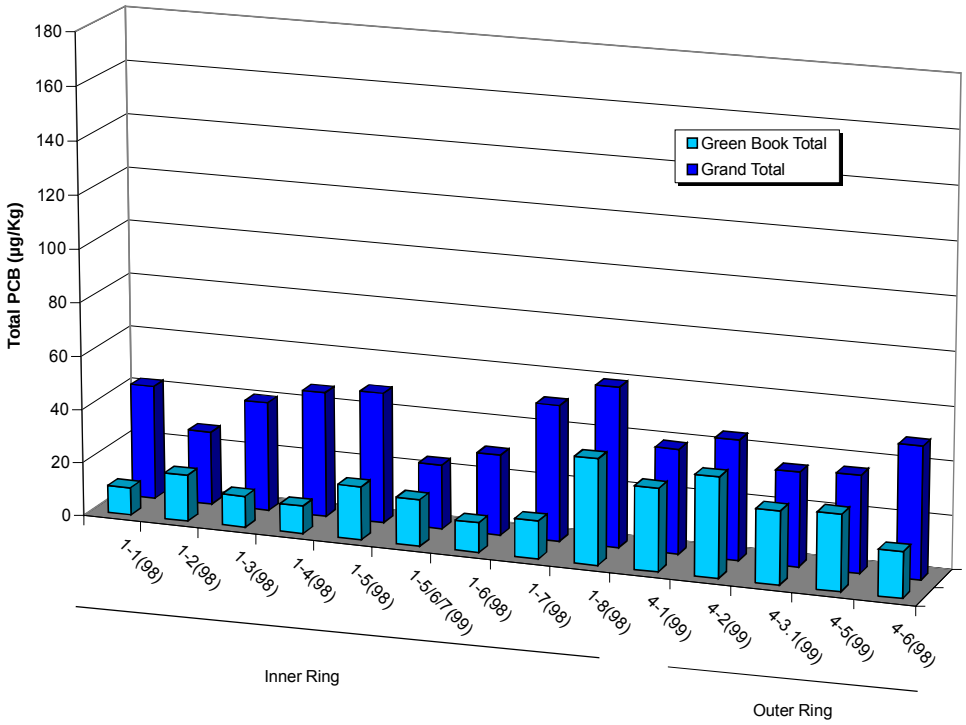

Figure 1-10. The sum of congeners measured in *Macoma nasuta* tissues (µg/kg dry weight). The sum of 21 Green Book congeners (Green Book Total) and the sum of 26 SINKEX congeners (Grand Total) are shown for each station. Labels on x-axis denote sample location (see Figure 1-3) and year collected.

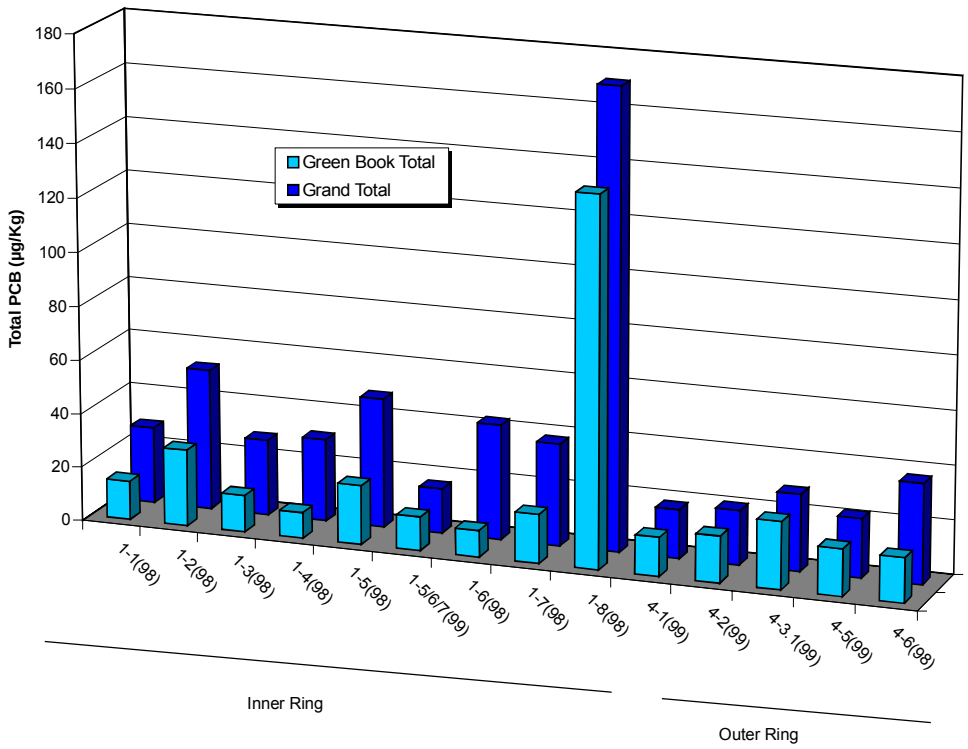

Figure 1-11. The sum of congeners measured in *Nephtys caecoides* tissues (µg/kg dry weight). The sum of 21 Green Book congeners (Green Book Total) and the sum of 26 SINKEX congeners (Grand Total) are shown for each station. Labels on x-axis denote sample location (see Figure 1-3) and year collected.

1.3.4. Supplementary Lines, Secondary COCs, and Individual Stations

Supplemental lines of evidence were evaluated to provide a more complete analysis of potential risks present at the ex-AGERHOLM site. These lines of evidence were collected and analyzed to:

- More completely characterize the relatively unknown environment at the deep ocean benthic site, using standard assessment techniques for benthic infaunal community structure;
- Provide additional information on potential sources of toxicity, other than PCBs (i.e., PAHs and metals);
- Obtain additional information on the distribution of Aroclors, congeners, and total PCBs, characterize core depth profiles of PCBs, and evaluate the spatial distribution of PCBs at the site;
- Evaluate the potential ecological effects from trophic transfer of PCBs, assessed using the sablefish data; and
- Assess human health risks from seafood consumption of sablefish.

Benthic Community: Results from the benthic community analysis were used to characterize the deep ocean environment at the site. Enumeration of 1,508 benthic specimens resulted in 240 identified taxa (exclusive of nematodes and calanoid copepods) collected in 23 boxcore samples, representing a fairly diverse deep ocean community (Appendix A). There were no statistically

significant differences in the "whole community" measures of diversity, richness, and abundance between the Inner (ex-AGERHOLM) and Outer (Reference site) Rings. There were differences in major taxonomic groups between the two sites, but these differences were correlated with differences in sediment (grain size and TOC) at the two sites.

Secondary COCs (SCOCs): The results for the SCOCs showed that cadmium, copper, nickel and silver were statistically higher and biologically available in samples from the Inner Ringer (Table 1-3).

Table 1-3. Matrix indicating significant Inner and Outer Ring differences for secondary (PAHs and metals) chemicals of concern ("+" indicates significantly higher concentrations from the Inner Ring [p≤0.05], "–" indicates rings not significantly different).

Secondary Chemicals of Concern	Sediment Chemistry	Macoma Bioaccumulation	Nephtys Bioaccumulation
Total PAH_{41}	–	–	–
Cadmium	+	+	+
Chromium	–	–	–
Copper	+	+	–
Lead	–	–	–
Mercury	–	–	–
Nickel	+	–	–
Silver	+	–	+
Zinc	–	–	–

Spatial Distribution of PCBs. The highest chemical concentrations and evidence of negative biological response were measured at stations that clustered near the large break in the hull at the rear of the ship (Figure 1-12, Table 1-4). These results were not correlated station-to-station or among lines of evidence (i.e., chemistry, toxicity, bioaccumulation). For example, Station 1-6 exhibited toxicity but no effects-inducing levels of PCBs or any other contaminant. Station 7 had higher chemistry levels for a number of constituents: chromium bioaccumulated from this station in the *Macoma* clam, PCBs measured by congeners and Aroclors (the latter measurement representing about 80% of the total Aroclor concentration detected for the sum of all stations), and PAHs. The concentration for this station was at least 6 times higher than any other station, but was still below effects-inducing levels found in the scientific literature. Sediment grain size characteristics of the ocean bottom appeared to be the most likely explanation for the toxicity observed at Station 1-6. Qualitatively, these various individual analytical results did not appear to be correlated. Quantitatively, attempts to use statistics to probe for correlations did not reveal any significant relationships.

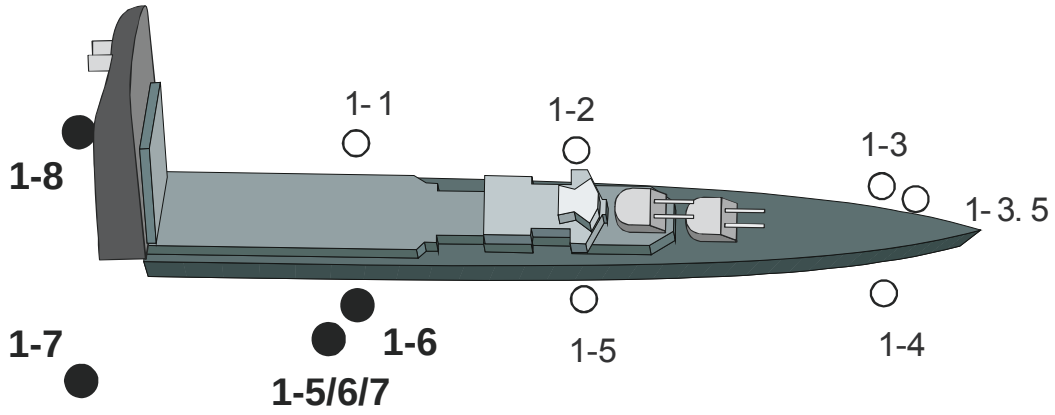

Figure 1-12. Inner Ring sediment sampling stations. Stations where elevated biological or chemical results were found to be clustered, in bold.

Table 1-4. Individual sampling stations where biological or chemical results were observed.

	Station	Line of Evidence	Specific Test or COC	Remarks
Primary	1-6	sediment toxicity	amphipod survival	grain size, TOC effect
	1-6&7	sediment PCBs	By congeners	below Effects Range-Low
	1-8	sediment PCB bioaccumulation	*Nephtys*	higher than clean Mussel Watch
Supplementary	1-7	sediment PCBs	by aroclor	above Effects Range-Low
	1-7	sediment PAHs	by 41 total PAHs	below effects level
	1-7	sediment chromium bioaccumulation	*Macoma*	higher than clean Mussel Watch
	1-5/6/7	sediment core PCBs	3 cm depth measurement (1900 ppb)	anomalous data presumably due to sediment mixing during/after impact

Principal Components Analysis (PCA) of the chemistry and texture analysis measured in the sediments identified the following grouping factors and their corresponding contributions to the total variability (% in parentheses) in all SINKEX measurements:

- Hg, Ni, Ag, TOC (31%);
- Grain Size (28%);
- Cd, Cu, total PCBs (21%); and
- $CaCO_3$, Al, Cr, Fe (21%).

Each of the four factor groupings accounted for between one-fifth and one-third of the total variability. Pearson's Correlation statistical tests were used to test for relationships between sediment and tissue concentrations, but none were found.

It is hypothesized that there were more contaminants released from the break in the ship and deposited into the sediments after the ship settled onto the ocean bottom, compared to the other parts of the hulk. The break in the hull, internal structures, and equipment at the rear of the ship may have caused an increase in exposed surface area of the inside of the hulk for leaching and/or particulate transfer of contaminants from shipboard materials into the environment. These results suggest that PCBs and other contaminants released from the ship were very localized and were confined to areas with the immediate vicinity of the ship.

Sablefish Assessment: The exposure assessment found that sablefish sampled from the ex-AGERHOLM (ship) site had statistically higher concentrations (by a factor of 1.4-1.5) of PCBs than sablefish sampled from reference locations about 4 nm away from the ship. Tissue residue benchmarks were developed to evaluate potential effects from exposure to Total PCBs. The benchmarks for Total PCB were based on the tissue screening value (TSV), bioaccumulation critical value (B_{CV}), and critical body residues (CBRs). These benchmarks are chemical residue thresholds at or below which adverse toxicological effects would not be expected. The initial comparison of sablefish data to tissue residue benchmarks showed that there was low risk of exposure to primary and secondary consumers in the deep-sea pelagic community because Total PCBs in sablefish from the ship site were significantly higher than reference and three samples from the ship sites exceeded the most conservative benchmark (TSV) used in the analyses. However, no sample exceeded any of the less conservative benchmarks, suggesting it was unlikely that exposure was be harmful (see Supplement I for details of this analysis).

Human Health Risk Assessment: The Navy collected samples of sablefish from the area around the ex-AGERHOLM in 1998 and 1999. PCB concentrations found in the samples were used to calculate both an increased cancer risk for people from eating these sablefish, and the risk of developing health effects other than cancer. These health risk assessments were based on typical exposures from commercial market basket fish. The risk values were all lower than, or safer than, levels which are considered acceptable by the United States Environmental Protection Agency. The fish around the ex-AGERHOLM are therefore considered safe to eat (see Supplement II for details of this of the human health risk assessment).

In summary, based on the sampling design, in which individual stations represented replicates for both the ship and reference sites, the contaminant and toxicity signals observed did not contribute to an overall statistical difference between the ship and reference sites.

1.3.5. Overall Risk Determination

Primary Lines of Evidence in Decision Matrix: Based on the negative outcomes from the three primary lines of evidence in comparing Inner Ring (ship site) to the Outer Ring (reference site), it was determined that there is no significant risk to the marine environment from PCBs-ISM onboard the ex-AGERHOLM.

Supplementary PCB Data: There are isolated individual spikes of sediment and tissue PCB concentrations, as well as toxicity, which were observed at individual sampling stations, but no correlation among these results with respect to specific location, lines of evidence (i.e. sediment chemistry, toxicity, bioaccumulation), and environmental compartment.

Secondary COCs Data: Based on the assessment of secondary COCs, there appear to be several metals, including Ni, Cd, Cu, and Ag, that have been released from the vessel and have accumulated in the sediment near the ship.

Nickel was present at sediment concentrations high enough to pose a probable risk to site biota, however the nickel benchmark criterion used in this determination is suspect. Two other metals (Cu and Cd) were measured at sediment concentrations high enough to pose a low but possible risk.

Three metals (Cd, Cu, and Ag) were shown in laboratory tests to bioaccumulate at higher levels in test organisms exposed to the ship sediments (relative to those exposed to reference site sediments). Copper concentrations accumulated in *Macoma* tissues were significantly greater at the Inner Ring (98.2 mg/kg) than the Outer Ring reference (60.4 mg/kg) and copper accumulated to levels much higher than observed in "clean" Mussel Watch stations (6.0-12.8 mg/kg).

Three other metals (lead, mercury, and zinc) bioaccumulated in both ship and reference sediment laboratory tests – to levels higher than the "clean" Mussel Watch stations – indicating possible elevated levels of these metals in ocean sediments.

Other Supplementary Data: Physical characteristics, especially grain size, which differed between the ship and reference sites, may explain overall differences in benthic community structure and toxicity between the two sites.

Supplementary Sablefish Analysis: The overall risk determined by the probability of exceeding benchmarks for Total PCBs showed very low risk of potentially harmful exposure to the deep-sea pelagic community and negligible risk to sablefish from critical body residues of Total PCB.

Human Health Risk Assessment: The human health risk assessment concluded that there was negligible risk of PCB exposure to human consumers of sablefish though the commercial market basket pathway.

1.4. Uncertainty

Uncertainty is a critical part of any scientific study, including risk assessment. The major sources of uncertainty in this study are summarized below:

1. Small number of samples: Because of the sampling difficulties, fewer samples than originally planned were collected, which resulted in a decrease of statistical power and less confidence in the conclusions that can be drawn from the data. The depth of the site, the difficulty in obtaining samples and the heterogeneous nature of the bottom substrate (rock and sediment) decreased the number of actual high-quality samples that could be obtained. These factors required a reduction in the number of sampling sites, thereby decreasing the total number of stations assessed and sediment samples retrieved.
2. Long time interval between field sampling events: Due to sampling and scheduling difficulties, samples were taken more than a year apart, which introduced the possibility of a time-dependent error that decreased the ability to statistically discriminate differences between ship and reference sites. Specifically, most inner ring ship stations were sampled in September 1998 and most of the outer ring reference stations were sampled in November 1999.

3. Inconsistencies in lab analyses caused by time interval (#2, above): Samples collected and analyzed over a year apart caused some inconsistencies in analytical procedures as itemized below:
   - In a few cases, samples were analyzed by different laboratories and personnel, using different methods and equipment. However, adequate quality assurance and quality control procedures were always used, and the precision and accuracy of the overwhelming majority of the results were never in question.
   - For the bioaccumulation tests, different-sized worms at the initiation of the two experiments (*Neanthes*) caused uncertainty in the comparison of growth rates.
4. Possible laboratory contamination: There were two instances of possible external contamination of laboratory animals/equipment, leading to difficulties in interpretation:
   - An increase in tissue PCBs in *Macoma* and *Nephtys* control specimens during bioaccumulation tests, possibly due to contaminated lab water or sediment; and
   - High initial (time zero) body burdens of PAHs in *Macoma* controls, most likely from test organisms obtained from a vendor, which were not discovered until detailed analysis of the data was accomplished.
5. Different sediment used for chemical and biological tests: The complications in sampling design forced a two-pronged requirement at every sediment station. One small, undisturbed sample of sediment for chemical analysis and several larger samples from the same location composited as one large volume of sediment for bioassays (i.e., toxicity testing and bioaccumulation) had to be retrieved from each station. Since the only effective sampling device for sediments proved to be the small boxcores manipulated by ROV, many samples were required to produce a composite sample large enough to support bioassay testing for a single station. The sediments used for bioaccumulation testing were obtained and treated differently than sediment chemistry samples, and due to the apparent heterogeneity of contaminants, created difficulties in relating site exposure to bioaccumulation and toxicity observed in the laboratory.

Only one study site used: During initial planning stages, a scientific review panel thought that it was necessary to conduct investigations at more than just one SINKEX site. Although considerable time, resources and effort were spent attempting to find more than one SINKEX vessel, only the ex-AGERHOLM was ultimately located and utilized as the lone study site.

1.5. Comparisons to Global PCB Data

It is useful to examine where the results of the SINKEX study fits into a global perspective on PCBs in sediments. Figure 1-13shows that even the highest PCB concentrations reported in the SINKEX study are considerably lower than the most conservative benchmark for adverse effects in any of the literature reviewed (Section 2.3 of this report). This supports the conclusion that the levels of PCBs measured in the sediments at the ex-AGERHOLM site are below the range of concentrations reported to cause effects from PCB exposure.

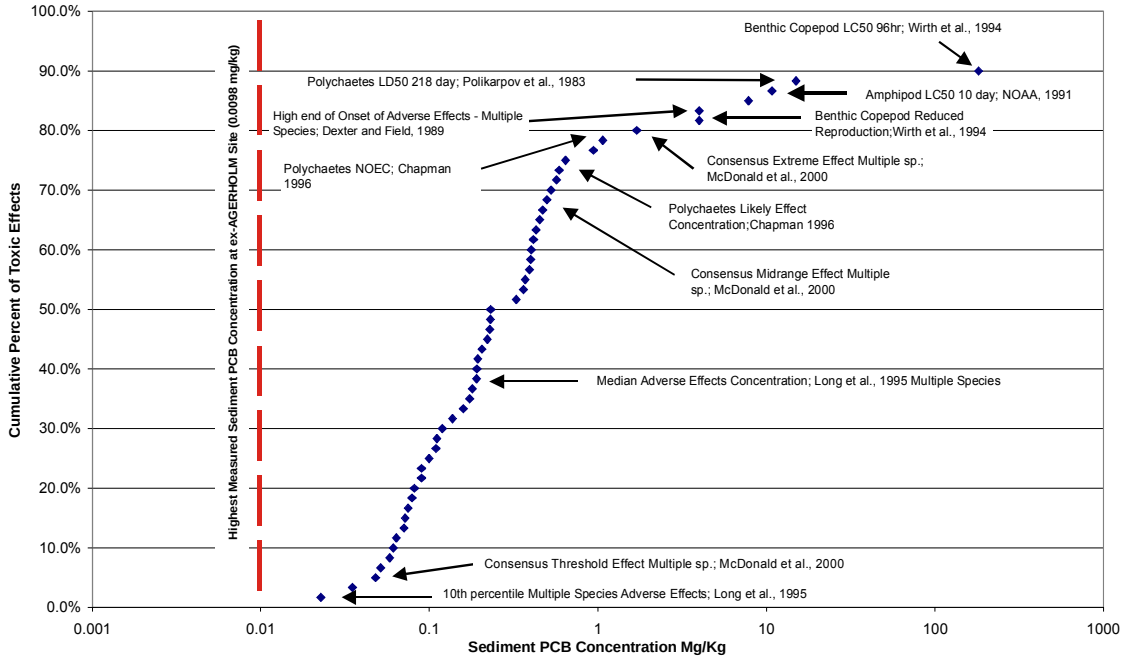

Figure 1-13. Sediment PCB toxicity reported from literature (PCBs as Aroclors or congener sums).

1.6. Conclusions

In conclusion, the results of the SINKEX study support a finding of no incremental risk to the marine environment associated with PCBs-ISM onboard the ex-AGERHOLM and there was no evidence of adverse effects. The conclusions from the study were based on a weight-of-evidence decision matrix developed prior to initiating the study. This technical approach is scientifically defensible and was the result of planning and consensus among Navy and U.S. EPA risk assessors and managers. The study also adheres to principles laid out by the U.S. EPA and the Army Corps of Engineers for ocean sediment testing and ecological risk assessment. A simple "study site" vs. "reference site" statistical testing design was used; however, the execution of deep ocean sampling over a period of one year required the use of a two-way ANOVA rather than a simple "t-test." No statistically and biological/ecologically significant differences were found between the study site and reference sites, for any of the three primary lines of evidence (sediment chemistry PCBs, sediment toxicity, and PCB bioaccumulation). No differences in overall ecological community structure measures were observed between the two sites, and the human health risk values from eating sable fish in commercial market basket exposure scenario were all lower than, or safer than, levels which are considered acceptable by the United States Environmental Protection Agency. Both results support the conclusion that decommissioned Navy vessels can safely be used for SINKEX exercises.

Physical characteristics of the sites, especially grain size, differed between the ship and reference sites, and were weakly correlated with differences observed among taxonomic sub-groupings of benthic community structure between the two sites. These physical differences may also explain the toxicity observed at Station 1-6. Isolated, elevated measurements for some COCs (PCBs, metals, PAHs) in sediment and tissue samples were observed for some stations, and these higher concentrations appeared to be clustered near the massive break in the ship structure. With the exception of three metals, the slightly elevated COCs were not statistically significant when

combined as replicates in the Inner versus Outer Ring comparisons. Nickel poses a probable localized risk, but using a benchmark in which there is low confidence, while copper and cadmium pose a possible, but low localized risk. The supplemental sablefish ecological risk assessment found very low risk of potentially harmful exposure to the deep-sea pelagic community and negligible risk to sablefish from critical body residues of Total PCBs. The human health risk assessment concluded there was negligible risk to the human consumers of sablefish though the commercial market basket pathway. Finally, when compared to global data on marine PCBs, the PCB concentrations in sediments found at the ex-AGERHOLM site are well below any observed effects levels.

1.7. References

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2.0. Problem Formulation

2.1. Introduction

In April 1995, The Naval Sea Systems Command (NAVSEA) tasked the Environmental Sciences Division of Space and Naval Warfare Systems Center – San Diego (SSC-SD) (formerly Naval Command, Control and Ocean Surveillance Center, Research, Development, Test & Evaluation Division - NRaD) to evaluate the marine environment in the vicinity of a sunken U.S. Navy ship to determine whether potentially toxic material from that ship was being released and whether any release(s) might pose an adverse risk to the environment. Ships sunk under the Navy's deep-water Sinking Exercise (SINKEX) program were thought to contain polychlorinated biphenyls in solid materials (PCBs-ISM).

In order to ensure operational readiness, the Navy routinely conducted training and weapons test exercises resulting in both the intentional and sometimes inadvertent sinking of Navy target vessels that were intended for reuse as target vessels. These sinkings normally took place in ocean waters greater than 3000 meters in depth, although some exercises were conducted in shallower waters. NAVSEA also asked that SSC-SD work cooperatively with the Navy Environmental Health Center (NEHC) to perform a human health risk assessment, if required.

The impetus for this study was the Navy's termination of SINKEXs pending discussions concerning regulatory oversight and the associated permit held under the Marine Protection, Research and Sanctuaries Act (MPRSA) for the SINKEX program. Prior to 1990, in addition to conducting deep-water sinking exercises, the Navy also donated unneeded vessels to local agencies for use in reef building. After initial discovery of PCBs aboard Navy ships in 1989, the Navy voluntarily suspended SINKEXs and discontinued the donation of ships for reef building until it was determined whether PCBs-ISM posed an unacceptable risk to the marine environment.

In 1994, NAVSEA requested SSC-SD to evaluate the risk of PCBs leaching from material located in these sunken ships using existing information. The result was a report based on available literature and a numerical model, which concluded that PCBs on these ships did not pose an adverse risk to the marine environment (Richter et. al., 1995). However, the U.S. Environmental Protection Agency (EPA) noted that there was no field data involved in the report and that there was no consideration of potentially toxic substances other than PCBs. The EPA indicated that field data would support continued EPA concurrence with the Navy SINKEX permit. Therefore, a field study of the environmental effects from sunken Navy vessels was proposed and approved in an agreement signed by the EPA and the Navy in 1996. [2]

While sinking exercises were unilaterally suspended by the Navy after 1990, in accordance with the Navy/EPA agreement, which was predicated on preliminary data from this study, a limited number of SINKEXs were authorized to proceed pending the final outcome of the sunken vessel study. Since the agreement was signed in 1996, the Navy has sunk approximately ten vessels each year. Previously, a ship was not considered suitable as a SINKEX hulk unless a ship survey

[2] Agreement Between the Department of the Navy and the United States Environmental Protection Agency: USE OF VESSELS CONTAINING POLYCHLORINATED BIPHENYLS AS TARGETS AND TEST PLATFORMS RESULTING IN THEIR SINKING, Signed 19 August 1996 by ASN(I&E) and EPA Assistant Administrator for Enforcement and Compliance Assurance.

determined that all onboard PCB concentrations were below 50 ppm in solid samples and less than 10 µg/100 cm^{-2} in wipe samples, used to measure surface contamination. These levels were the respective PCB disposal and cleanup thresholds found in the Code of Federal Regulations, Title 40, Part 761.62. The 1996 agreement removed the 50 ppm criterion for solid materials based on the preliminary findings of this study.

The study objectives were to determine if potential contaminants of concern: (1) have been released from a representative sunken naval vessel, and if so, (2) whether they have adversely impacted the adjacent marine environment. Specifically, the intent was to conduct a stressor-driven, retrospective, site-specific risk assessment that would evaluate PCB contamination of the ocean benthos that could possibly be related to one of these ships. Initially, the requirement to conduct a human health risk assessment initially seemed unnecessary because there was thought to be no possible link between a sunken vessel and human receptors. A focused attempt was made to locate a suitable hulk, which had been sunk under the SINKEX program to meet the requirements of the field study. For a number of reasons, considerable time and effort was expended in the search for such a vessel until discovery of the location of the hulk of the ex-AGERHOLM in the summer of 1996.

Figure 2-1. The USS AGERHOLM (DD 826) in 1978.

The ex-AGERHOLM (Figure 2-1) is a WWII-era destroyer hulk sunk in June 1982 and located approximately 120 nautical miles off the southern California coast at a depth of 2,750 feet (Figure 2-2). Because only a single sunken ship was located and subsequently studied in this effort, information obtained from this hulk is, of necessity, considered representative for any ship of that class, age, and degree of preparation, used as an expendable target in pre-1990 fleet

training/sinking exercises (SINKEX). This site-specific study is intended to assess the sunken vessel's effects upon the marine environment through standard environmental methods, including the following:

- chemical concentrations in "site" sediment, tissue, and water;
- laboratory studies performed on "site" sediments and surrogate organisms that measured acute and chronic toxicity; and
- laboratory studies performed on "site" sediments and surrogate organisms that estimated bioaccumulation potential.

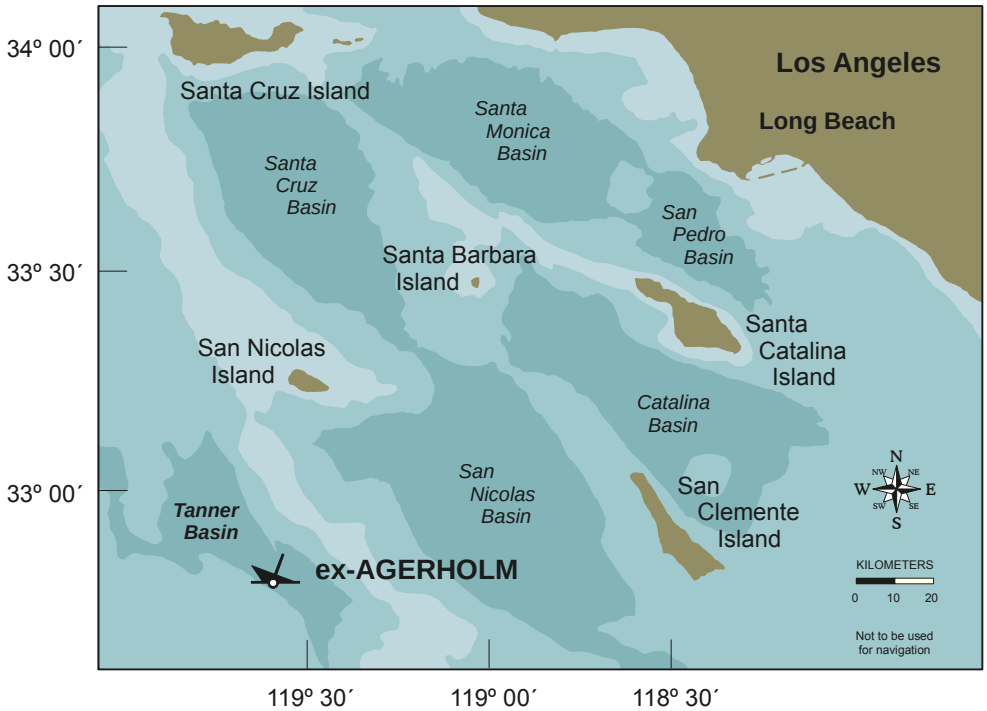


Figure 2-2. Position of the ex-AGERHOLM.

For the purposes of this study, the site was considered a deep-water site and evaluated within the framework of the "Evaluation of Dredged Material Proposed for Ocean Disposal, Testing Manual" (commonly referred to as the "Ocean Testing Manual" or the "Green Book"); jointly produced by the EPA and the U.S. Army Corps of Engineers (USACE) (USEPA/USACE, 1991). Additionally, the USEPA's "Framework for Ecological Risk Assessment" (ERA) (USEPA, 1992) was utilized to provide guidance in structuring the study as a risk assessment. Five oceanographic sampling efforts were subsequently conducted between July 1997 and November 1999 in support of this project.

2.2. Background

In April 1989, personnel at Puget Sound Naval Shipyard discovered PCBs-ISM (specifically in sound damping felt) while dismantling submarines. The Navy immediately launched a comprehensive monitoring program to survey inactive vessels that were slated for disposal via

scrapping or foreign military sales. The intent was to provide an understanding of the distribution of PCBs in Navy vessels. Years of survey work have indicated that PCBs were widely but randomly distributed in ships that had been constructed and/or repaired while PCBs-ISM were still in use on active ships (the 1979 PCB ban [Allen, 1990] affected only new construction vessels).

As mentioned previously, the initial effort to address this issue was a paper study submitted to NAVSEA 00T in 1994 by SSC-SD. This paper examined potential deep ocean effects of PCBs linked to the hypothetical sinking of a decommissioned naval vessel, which had been used as a target in a fleet readiness exercise. This preliminary study was based on a substantial literature search coupled with a mathematical model that predicted both hydrodynamic dispersal and sediment adsorption of PCBs. This model used estimated parameters for solubility, temperature, partitioning characteristics and a conservative current regime to predict PCB concentrations in water and sediment surrounding the immediate vicinity of the hulk. The report concluded that PCBs leaching from PCBs-ISM (i.e., impregnated in gaskets, insulation material and electrical cabling that was not removed prior to sinking) could slowly disseminate into the environment. The model also indicated that the slow rate of release, which would be partially dependent on the low ambient temperature/low solubility characteristics of PCBs in seawater, would result in increased water and sediment concentrations in the vicinity of such a hulk, but would remain well below one part-per-trillion in open ocean background concentrations. The principal transport and fate theory was that sorption by sediment organic material surrounding the hulk (assuming limited advection) was the primary fate of the PCBs due to the hydrophobic nature of PCBs. An additional conclusion in the study was that, although PCB sediment concentrations near the hulk would potentially be above background levels, they would pose no notable threat to benthic organisms. An explanation of this model and the conclusions derived from it are included in Section 2.3.3.6.

This preliminary study was considered a first step in a scientific inquiry of this issue by the principal regulatory body (EPA - Office of Water) and the decision was made by EPA-OW to require the Navy to conduct further study, including field sampling. The major purpose of the agreement specifying the study was, "to specify the manner in which the Navy is to proceed with certain activities…vital to the national defense while the Navy conducts its sunken vessel study…" A specific condition of the agreement was that the Navy proceed with a plan to collect empirical data from the site of a sunken Naval vessel.

The final step prior to initiation of the SINKEX study was to locate a suitable ship. This was a much more difficult task than initially anticipated. Very little historical or observational information had been collected during routine sinking exercises, and the best electronic navigation systems available at the time of sinking for Navy warships and the range support vessels were generally Loran C or Omega. Ships were usually sunk during or shortly after major fleet training exercises. Hulks used as targets during these exercises were typically damaged by submarine, ship or aircraft weapons, and some did not sink until a concerted effort was made to remove what had become a hazard to navigation. Since this often occurred after most Navy participants left the area, few personnel remained to record vessel conditions during sinking. A civilian towing company typically observed, and often assisted in the final sinking of the hulks.

The effort to find a hulk for study was defined by accepting a list of suitable target hulks that (1) would be representative of the type of ship used in SINKEXs, (2) would meet a certain level of confidence with respect to a priori knowledge about their geographic locations, and (3) would

meet other requirements of the study that related to knowledge of the type of source material onboard and the possibility of release and transport into the marine environment.

In any situation involving the location of an object in the deep ocean, it is imperative that good positional data be available (i.e., latitude, longitude, and depth). Unfortunately, that information was not adequately collected or archived for SINKEX vessels sunk prior to 1990. Extensive research was required to determine the historical details of a number of specific target exercises and subsequent sinkings. Based on discussions with the Atlantic and Pacific Fleet commands, 15 locations were determined for past sinking exercises undertaken between 1986 and 1991. Of the 15 sunken vessels, 10 were in the Atlantic Ocean off the southeastern United States and one was in the Atlantic off the coast of Morocco. The four others were sunk in the Pacific, one south of the Aleutians, two off of Hawaii, and one off of Mexico. Additional evaluation was necessary to determine the confidence one would have in the recorded positional information prior to committing considerable time and resources to a search effort. A final but necessary consideration included characterization of the selected site as to its suitability for deep-ocean survey and sampling.

Three primary targets were selected based on the following criteria: time on the bottom, confidence in positional data, depth, bottom topography, representativeness of vessel type to typical sinking operations, and information on probable PCB content. Two separate cruises were undertaken using state-of-the-art side-scan sonar technology to search deep ocean areas in sinking locations off the Southern California coast. The searches centered on the best-estimated positions of the ex-HIGBEE and the ex-BAUSELL, both WWII-era destroyers selected as primary candidate vessels for study. Neither vessel was located during these two cruises. The third candidate hulk, the ex-AGERHOLM, which was sunk in 1982, was located in July of 1996 by the research submarine USS DOLPHIN (AGSS 555). Although the ex-AGERHOLM rested on the ocean bottom in shallower water than that in which the other two candidates were sunk, it was deemed to be the best sampling opportunity. The crew of the DOLPHIN provided photography and videography to positively identify the hulk.

2.3. Integration of Available Information

Under the EPA's Framework for Ecological Risk Assessment (USEPA, 1992), "integration of available information" is the first major element of Problem Formulation (Figure 2-3). It consists of an a priori discussion of what is known about the site's stressor and exposure characteristics, the environment potentially at risk, and known ecological effects associated with the site and sources under investigation. It culminates with the development of a conceptual model and the selection of assessment endpoints. An analysis plan is then developed, in which measures of exposure and effects, which can be linked back to the assessment endpoints, are determined. The sampling and analysis design defines the data necessary to characterize risk.

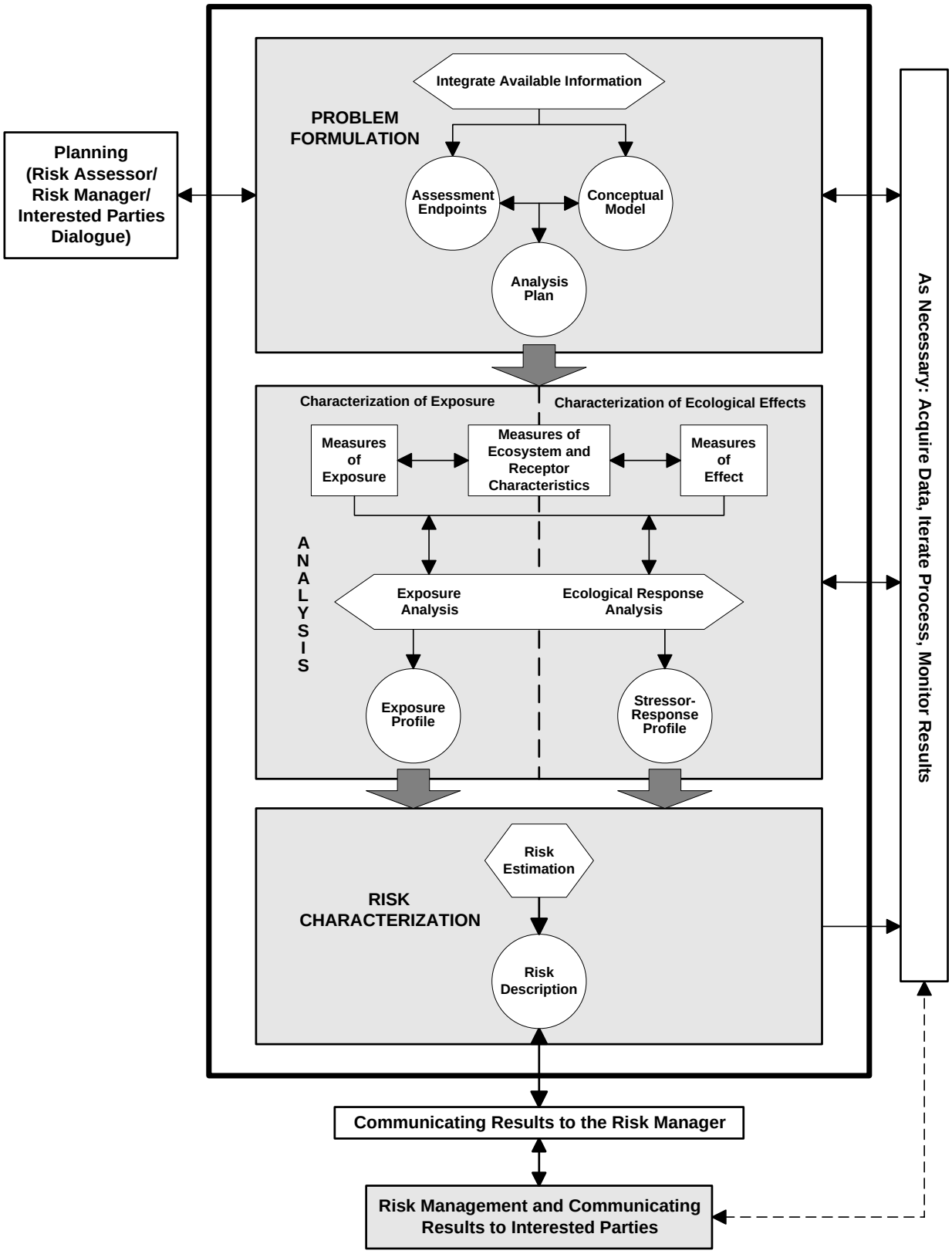


Figure 2-3. Framework for ecological risk assessment (USEPA, 1998).

2.3.1. Source Characteristics at SINKEX Site/Identification of Stressors

Chemical contaminants (PCBs, metals, and PAHs) were the drivers for this ecological risk assessment. Therefore, this ERA was considered to be stressor- or source-driven, as opposed to effects-driven. The goal is to determine if there are any unknown effects which have occurred, are occurring, or will occur due to the stressors/sources of concern. Consequently, the first step in Problem Formulation in this ERA was to characterize the stressors/sources of concern.

2.3.1.1. Chemicals of Potential Concern in SINKEX Hulks

Conceptually, all the materials that make up the ship eventually are buried, dissolved, disintegrated, and/or leach into the surrounding sediment and seawater. Thereafter, any dissolved materials (e.g. chemicals) may sorb to particles in the environment or be advected away from the site. Some leached materials enter the food chain, while others degrade on the ocean floor, others will be advected away from the site by the current. The regulatory driver for this study was the Ocean Disposal requirements under the Marine Protection, Research and Sanctuaries Act (MPRSA) and PCB-impregnated materials that had been discovered on Navy ships. PCBs are the primary contaminant of concern (PCOC) for this study.

Navy ships are comprised mostly of metal, so several metals were considered secondary contaminants of concern (SCOCs). Another SCOC was the class of chemicals referred to as polycyclic aromatic hydrocarbons (PAHs). Many of these compounds are known to exhibit toxicity and were present in fuel and lube oils used onboard ships. While the Navy attempted to empty target ships of these liquids, it is possible that some cleaning events were insufficient to remove all PAH compounds, leaving small volumes or residues at the bottom of storage tanks, operating machinery, and associated equipment at the bottom of storage tanks.

2.3.1.2. PCBs as Primary Chemical of Concern

Polychlorinated biphenyls (PCBs) are a group of synthetic halogenated aromatic hydrocarbons that have been unfavorably linked to environmental and public health issues. The industrial production of PCBs began in 1929 and continued in the United States until 1977 (Mearns et al., 1991). They were used as dielectric fluids in transformers and capacitors, as hydraulic fluids, as heat transfer and vacuum pump fluids, and in many other applications. In the USA, Germany, France, UK, Japan, Spain, and Italy, PCB production has been estimated at 1,054,800 tons during 1930-1980 (Yamada et al., 1997). Others have estimated the total global production at 1.2 million tons, with 31 percent of this quantity still present in the environment (Ali et al., 1997). In another report (Tanabe, 1988), the global PCB loading was estimated currently at 370×10^3 tons. Of this, almost all (360×10^3) was contained in coastal sediments and the world oceans. The seawater fraction was thought to contain over 60% of the global environmental load. The estimated active portion (still in use) was 780×10^3 tons (Tanabe, 1988). These estimates suggested that PCB pollution was not necessarily nearing an end.

PCBs are extremely stable compounds and very persistent in aquatic environments. Past investigations have suggested that PCBs are a potential hazard to marine life because of their great stability, persistence, lipophilicity, and because they are poorly metabolized by biological systems (Harrad et. al., 1994). A recent study has shown that PCBs may cause population level effects in fish. Specifically, non-migratory estuarine fish (*Fundulus heteroclitus*) in a superfund site (New Bedford Harbor, Massachusetts) developed tolerance to PCBs at sediment concentrations beginning at 200-400 ng/g (Nacci et al., 2002). Few references were found that

document a field study in which PCBs are associated with an adverse ecological effect in the marine environment, noting that bioaccumulation is not necessarily an adverse effect. However, with respect to marine mammal populations, the Marine Mammal Commission concluded that "there is good reason to be concerned that survival and reproduction in certain marine mammal populations may have been affected, and are being affected, by persistent contaminants, particularly organochlorines. The potential effects of contaminants may include morbidity and mortality from acute toxicity (although mortality has not yet specifically been shown in marine mammals), disruption of endocrine cycles and developmental processes causing reproductive failures or birth defects, suppression of immune system function, and metabolic disorders resulting in cancer or genetic abnormalities" (O'Shea et al.,1999).

The PCB molecular structure shown below is based on two covalently linked phenyl rings that are substituted at the periphery.

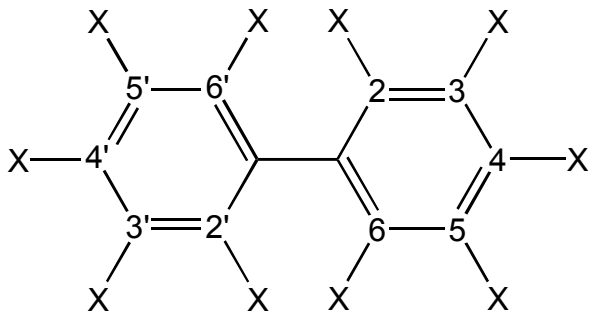


Figure 2-4. Polychlorinated biphenyl molecular structure, where each X is either hydrogen (H) or chlorine (Cl) substituted in various combinations at the numbered carbon positions on the rings. The 2, 6, 2', or 6' carbons are at the ortho positions; 3, 5, 3', or 5' carbons are at the meta positions; and 4 and 4' carbons are at the para positions. There are 209 possible isomers, commonly referred to as "congeners". See text for details.

The biphenyl rings are substituted with either hydrogen (H) or chlorine- (Cl) groups (denoted by X above) in various combinations, with a generic formula of $C_{12}H_aCl_b$ (where a+b=10). It is important to note that the prefix "poly" in polychlorinated biphenyl does not follow the standard chemical convention indicating a "polymer." PCBs are not polymers; rather "poly" simply represents multiple chloro-substituents on the biphenyl ring. The various combinations of hydrogen and chloro-substituents on a biphenyl ring yield 209 unique molecules or positional isomers, commonly referred to as congeners, excluding the unchlorinated, or fully H-substituted molecule, "biphenyl". By convention, these 209 congeners have unique BZ numbers (after Ballschmiter and Zell, 1980) assigned to them (BZ1-BZ209) and correspond to numbers adopted by IUPAC and CAS. There are differences in molecular naming conventions between IUPAC and BZ. All 209 PCB congeners can be grouped according to chlorination level. These are referred to as homolog groups (Cl-1, Cl-2, Cl-3…Cl-10); each group consisting of a distribution of same-chlorination level (and same molecular weight) congeners. Summation of all ten homolog groups corresponds to total PCBs (tPCBs) for any given sample.

Analyses of PCBs are typically reported as individual congeners, tPCBs, Aroclor (or Aroclor equivalents) and in some instances, homologs. Commercially prepared PCBs were originally

marketed under the trade name Aroclor. These were the most common PCBs used and are complex mixtures produced by bulk chlorination. Aroclors are generally described by a 4-digit numerical notation, in which the last two digits indicate the weight percent chlorine (e.g., Aroclor 1254 is 54% chlorine).

PCBs are largely hydrophobic compounds, meaning that they are not readily water-soluble. However, even with transport limited by low aqueous solubilities because of their environmental stability and persistence, PCBs can readily bioaccumulate in the lipid-rich tissues of marine and other organisms (i.e., they are lipophilic).

PCBs are a primary ecological concern because they are ubiquitous in the natural environment and their tendancy to bioaccumulate and biomagnify up the food chain and as a result, many biological organisms have measurable amounts of PCBs in their tissues. Their chemical and physical stability have led to their well-known and well-studied persistence in the environment. In the marine environment, PCBs tend to biomagnify in food chains, particularly in birds and mammals. The EPA has regulated them based on numerous studies showing their "estimated" effects on human health. Background levels for seawater in the open ocean tend to be in the low or sub parts-per trillion (pptr) range; while sediments, which tend to accumulate PCBs, can often have levels two to three orders of magnitude higher.

2.3.1.2.1. Documentation of PCBs on Navy Ships

The Naval Sea Systems Command conducted an extensive shipboard sampling program for PCBs in the 1990s. It was driven by the unexpected discovery of PCBs found impregnated in solid shipboard materials. The following excerpt from the Navy-EPA PCB agreement explains:

> "…In 1989, the Navy discovered that wool felt used as acoustical damping material on submarines and as gasket material on all vessels may contain viscous PCBs by weight at concentrations of 50 to 300,000 ppm. The felt material was procured from 1948 to the later 1970s under specifications that required a fire retardant and was used during both new construction and repair of vessels. The Navy promptly notified EPA of its discovery and removed the material from the Navy Supply System. Subsequent investigations by the Navy since 1990 have determined that solid PCBs are also present in some commercial items, such as paint, rubber mounts, rubber and plastic cable insulation, and tape, used on vessels…"

Table 2-1 lists shipboard materials measured to have contained PCBs in at least some samples.

Table 2-1. Shipboard materials known to have contained PCBs.

Electrical Cable Insulation	Rubber Gaskets
Felt Gaskets	Adhesives
Foundation mounts	Paints
Caulking	Rubber isolation mounts
Thermal insulation materials:	Pipe hangers
fiberglass	Plasticizers
felt	Light ballasts
foam	Tapes
cork	Solid surface contamination

Adopted from "Agreement Between The Department of the Navy and The United States Environmental Protection Agency Washington, D.C.: Use of Naval Vessels Containing Polychlorinated Biphenyls as Targets and Test Platforms Resulting in Their Sinking"

2.3.1.2.2. Documentation of PCBs on the Study Vessel

The USS AGERHOLM (DD-826) was built for use by the U.S. Navy, by Bath Iron Works in the mid-1940s and later refurbished in the 1960s (Figure 2-1). The ship was originally commissioned in 1946 as a GEARING-class destroyer. It was decommissioned in 1978 and intentionally sunk in 1982.

At the time the ex-AGERHOLM was sunk, no document existed that detailed its most recent material composition. This observation is typical for any ship that has undergone regular maintenance, repairs and modifications over the years. The Navy and its contractors design a class of ship, but various shipyards that have their own standard procedures, practices, and construction materials, build individual ships. Even ships of the same class built at the same shipyard may differ slightly in composition. Furthermore, the ships' crew performed much of the repair work and maintenance, resulting in even a greater heterogeneity of "source" materials (i.e., added at a later date).

John J. McMullen and Associates (JJMA) performed source characterization studies to estimate quantities of PCB-containing materials on the ex-AGERHOLM prior to sinking (JJMA, 1998). JJMA utilized a number of different resources; including ship design specifications, construction and maintenance records, and other Naval historical, inventory and administrative databases. JJMA also executed a sampling and analysis program to measure PCBs and metals in shipboard materials on suitable surrogate vessels. All of the available information was compiled, validated, and analyzed to estimate the mass of PCBs and metals on the ex-AGERHOLM at the time of its sinking.

PCB sources associated with sunken Navy vessels are primarily located in equipment/ components that contain residual PCBs and PCB-contaminated non-liquid construction materials within the vessels such as electrical cables, bulkhead insulation, rubber, felt gaskets, heating and ventilation (HVAC) gaskets, sealants and heat resistant paints. PCB residuals and PCB-contaminated materials are considered as the most likely sources of PCBs that might originate from a hulk, if such materials are present.

In the NAVSEA-directed program to measure PCBs prior to disposal or transfer of ships from the U.S. Navy, PCBs were measured in solid materials as total PCBs or as total Aroclors, but not

all 209 individual congeners, because such an effort would have been cost-prohibitive. Estimates of PCBs thought to have been onboard the ex-AGERHOLM in 1982 range from 12 lbs to 80 lbs. The breakdown by shipboard material type is shown in Table 2-2.

Table 2-2. Initial Load Estimates of PCBs (in lbs) on the ex-AGERHOLM prior to SINKEX 1982.

Shipboard Materials	Low Estimate	High Estimate	Average Estimate
HVAC gaskets (felt, rubber & paper composite)	0.1	41.7	20.9
electrical cable insulation	9.3	20.2	14.7
paint	2.4	9.7	6.1
oils/greases	0.02	8.3	4.2
rubber products	0.1	0.1	0.1
Foam insulation	0.1	0.1	0.1
Totals	**12**	**80**	**52**

Table adopted directly from 1 Dec 98 Draft JJMA Report title "Weight Estimates for PCBs and Selected Metals Sunk on ex-AGERHOLM (DD-826) for the Deep Water Sunken Ship Study. Note that authors used same values for all three estimates in the case of rubber products and foam insulation.

2.3.1.3. Secondary Chemicals of Concern

Metals and PAHs are the secondary chemicals of concern (SCOCs) because of their potential to cause toxicity related effects. There are no other chemicals investigated as potential contaminants being released from the SINKEX site.

2.3.1.3.1. Metals

Individual heavy metals (Ag, As, Cd, Cr, Cu, Hg, Ni, Pb, Zn) are considered ecologically important, as they are potentially toxic at various concentrations in the marine environment (Rainbow and Furness, 1990; Krenkel, 1975). Certain metals have also been observed to bioaccumulate in marine and other organisms. The following seven metals represented a large mass fraction of a typical ship: mercury (Hg), zinc (Zn), lead (Pb), chromium (Cr), copper (Cu), nickel (Ni), and cadmium (Cd) (JJMA, 1998). Metal sources from a sunken ship would be primarily from the low-carbon structural steel (Ni, Cu, Cr), paints (Cu, Zn, Cr, Pb), electrical cabling (Cu), ballast (Pb), electrical switching equipment (Hg), piping systems (Cu, Ni), and sacrificial anodes (Zn) left onboard sunken hulks. Because there was concern that silver (Ag) could also be found, while it was not assessed by JJMA, it was added to the suite of metals for testing. Finally, aluminum (Al) and iron (Fe) were added to provide background comparisons to aid in the data interpretation (i.e., metals typically found in high concentrations in uncontaminated soils and sediments). Arsenic was not expected to be present on Navy ships. Consequently, a total of 10 metals were selected for examination, 8 for their shipborne contributions and 2 for background levels and normalization/standardization purposes.

The total mass of heavy metals estimated to have sunk with the ex-AGERHOLM is between 278.6 and 294.5 tons (Table 2-3). This estimate is probably very conservative (i.e., high), because removal of certain equipment from the ship was not recorded and JJMA used worst-case assumptions that all equipment remained on board.

Table 2-3. Estimates of Heavy Metals (in tons) on ex-AGERHOLM prior to SINKEX.

Metal	Low Estimate	High Estimate	Average Estimate
Cadmium (Cd)	0.02	0.02	0.02
Chromium (Cr)	6.97	7.69	7.33
Copper (Cu)	202.6	208.4	205.5
Lead (Pb)	30.1	31.5	30.8
Nickel (Ni)	21.2	22.7	21.95
Zinc (Zn)	17.7	24.2	20.95
Mercury (Hg)	0.02	0.02	0.02
Totals	278.6	294.5	286.6

Table adopted directly from 1 Dec 98 Draft JJMA Report title "Weight Estimates for PCBs and Selected Metals Sunk on ex-AGERHOLM (DD-826) for the Deep Water Sunken Ship Study. Note that authors used same values for all three estimates in the case of rubber products and insulation.

2.3.1.3.2. PAHs

PAHs are reported as individual compounds and total PAH (tPAH). PAHs are petroleum-based products, some of which are known carcinogens (Amdur et. al., 1991). Fuel and oils are the most common sources of PAHs. PAHs do not bioaccumulate in aquatic biota as readily as PCBs and some metals, because many species (especially fish) metabolize these compounds (National Research Council of Canada, 1983). The primary source of PAHs within a sunken Navy vessel would be residual fuels that have not been completely removed. According to OPNAVINST 6240.31, at the time the ex-AGERHOLM was sunk, residual fuels and oil products had previously been removed from the ship. Consequently, there was no attempt by JJMA to estimate an initial PAH load for the ex-AGERHOLM.

2.3.2. Site Description and Ecosystems Potentially at Risk

The EPA-Navy SINKEX Agreement mentioned above was based on the proposed future sinking of ships at depths greater than 6,000 ft. Little was known about the ocean currents, sedimentation rates of particulate matter, sediment properties, bottom water and ecological community at the ex-AGERHOLM site (at 2,750 ft depth) prior to field collections performed for this ERA. The following discussion represents our general understanding of the deep-ocean pelagic and benthic environments, as well as ecosystems potentially at risk from contaminants leaching off of a sunken hulk.

2.3.2.1. Deep Ocean Pelagic Environment

Pelagic biomass trapped in nets typically decreases by two to three orders of magnitude from the surface to the near bottom (Rowe et al., 1974; Weikert, 1990), where a sudden increase in biomass is typically observed (Wishner, 1980). Angel (1990), summarizing much previous biological data, characterizes abyssal benthic communities as rich in novel species with many shallow water orders represented, although occurring typically at 1%-2% of surface water biomass concentrations. The bottom, acting as a trap for sinking and resuspended particles, is organically rich and supports a higher level of metabolic activity than the water immediately above. The flux of organic material from the surface to the benthos is generally proportional to the overlying primary productivity and inversely proportional to the depth of the water column.

2.3.2.2. Deep Ocean Benthic Environment

Benthic macrofaunal biomass ranges from 0.1 g to 10 g m^{-2} in most of the regions where open ocean SINKEXs might occur and is dominated by filter and deposit feeding organisms (primarily ophiuroids – brittle and basket stars). Reworking of the sediment through digging, burrowing, fecal production, and tube building by benthic organisms tends to mix the near-surface sediments. Tube construction, excavation mounds and ingested fine-grained material may alter sediment erosion rates and susceptibility to resuspension (Parsons et al., 1984). Opportunistic, mobile scavengers such as amphipods and fish have been photographed feeding on carcasses (Dayton and Hessler, 1972; Isaacs and Schwartzlose, 1975), though the quantitative importance of these organisms and amount of carbon delivered to the deep-sea bed as large particles has not been determined.

Biomass for deep-ocean benthos is relatively small when compared with typical biomass found for shallow coastal regions. Benthic and epibenthic organisms occurring at depths similar to the study site (i.e., 2,700 ft.) feed mainly on settled detritus and carrion. Plants are non-existent due to the lack of sunlight or photo-energy sources, and food for larger predatory vertebrates (e.g., sablefish) is presumed to be less available than in a littoral environment.

2.3.2.3. Ecosystems Potentially at Risk from Sunken Ship Contaminants

A sunken hulk has "habitat" characteristics (e.g., large surface area and shelter) that could attract organisms living at depth. As time passes after a SINKEX vessel lands on the ocean floor, it may be colonized, resulting in increased biomass and biodiversity of resident biota on and near the hulk, thereby creating a reef effect. However, the processes that create reefs from hulks in the deep oceans may be quite different from those observed in shallow waters. Under this scenario, the sunken vessel may undergo a natural evolution. Initially, the vessel would be barren and provide refuge and cover for nektonic (swimming) organisms such as small fish. Over time (years to decades), filter and detritus feeders, such as echinoderms (brittle stars, basket stars and sea cucumbers), as well as benthic burrowing worms and crabs would migrate to the hulk. As biomass around the hulk increased, predatory organisms, including cephalopods and fishes, would begin to feed on smaller organisms, thus developing a food chain. Colonizing organisms may be exposed to contaminants emanating from the hulk via the water column, sediment, or food chain exposures.

While other communities may be present, the benthic infaunal community was selected as the most important ecological community at risk from contaminants related to the hulk. Infaunal communities are important indicators of environmental quality because the small organisms comprising these communities live on and in direct contact with the sediments, many of these organisms feed on sediment particulates, and they are an important prey for larger organisms. Changes in the infaunal community may reflect impacts from physical disruption of the habitat, alteration of trophic and biological relationships, and/or chemical contaminants. As mentioned previously, at least two types of impacts from the hulk have the potential to affect infaunal communities: reef effects (the physical presence of large hard surface structure) and contaminant effects (the release of chemical contaminants). Reef effects result from the physical structure of the sunken vessel functioning as a large artificial reef located on a sloped bottom generally devoid of exposed hard-bottom habitat. Both natural and artificial reef structures significantly affect adjacent soft-bottom communities by altering bottom boundary currents (affects food supply and changes in grain size) and providing habitat for predators that forage on the infauna

near the reef. Consequently, reef effects are evident as differing gradients in the physical environment (e.g., grain size and TOC), but also as a natural response in the alteration of the demersal, epibenthic, infaunal, and other deep ocean communities, regardless of whether the reef is natural or artificial. Contaminant effects can either enhance or degrade infaunal communities dependent upon the type and concentration of contaminant. The challenge is to distinguish reef effects from possible contaminant effects (Fabi et.al., 1999).

"Healthy" communities are typically characterized by high diversity (i.e., many species), moderate abundance (i.e., not dominated by a few opportunistic and/or tolerant species), and an ability to adjust to a range of typical, natural environmental conditions. High abundances of a few tolerant species may indicate environmental stress, such as exposure to high levels of organic matter or toxic chemicals. If an impact is significant, community composition would most likely be altered, possibly leading to changes in community function. If impacts are severe, the community may be completely lost (e.g., Pearson and Rosenberg 1978).

Changes in infaunal community composition normally occur along natural environmental gradients (e.g., water depth, grain size, food sources) and gradients caused by human development (e.g., chemical contaminant concentrations, physical disruption of habitat, nutrients). The primary measures used to evaluate infaunal communities are the number of species (taxa) per sample (i.e., species richness), abundance, and derived indices of diversity (e.g., Shannon-Wiener Diversity, Margalef's Species Richness, Evenness, and Dominance).

There is little consensus among biologists regarding the suitability of which community measures and/or derived measures are most useful for describing community properties or for documenting pollutant impacts (Tetra Tech 1985). This controversy continues at the present time with new approaches and indices being developed as measures of community response (Gray 1981; Gray et al. 1990; Warwick 1993; Warwick and Clarke 1994; Bergen et al. 1998). However, despite the controversy of specific methods, measures such as species diversity and richness have proven very useful for assessing community structure in time and space. Generally, a greater number of species represents a healthier and more stable environment, and models of community stress suggest decreasing species richness as one of the first indications of an impacted community. Other indications of altered or stressed communities include decreases in diversity indices, high abundance of tolerant species, and changes in species composition reflecting changes in community function.

2.3.3. Exposure Characteristics

The next steps in this Problem Formulation phase of the ERA is to discuss what was known about **Exposure** and **Effects** related to the site of concern and its contaminant sources after the sinking of the SINKEX vessel, in order to provide a point of departure for developing the SINKEX study design. Consequently, this section on Exposure Characteristics addresses physical behavior of PCBs in the marine environment and fate and effects phenomena such as degradation. Following the physical fate and effects summaries, the results of a modeling approach to PCB release from sunken Navy vessels in the deep sea is presented and discussed.

2.3.3.1. Solubility and Persistence

Physical chemistry data on PCBs vary considerably. For the sake of brevity and consistency in this report, we will denote solubilities of PCBs in mass per unit volume (e.g., $\mu g\ L^{-1}$) on a weight basis (e.g., µg/kg as ppb), realizing that this is an approximation that ignores minor deviations in

water density from 1.0 g/mL. With some exceptions (tetra and deca-chlorobiphenyls), PCB aqueous solubilities generally decrease with increasing level of chlorination (higher molecular weight congeners). For example, distilled water solubility varies from 5.9 ppm for mono-chlorinated biphenyl congeners, 0.3 ppm for di-chlorobiphenyls, to 6 ppb for hepta-chlorobiphenyls, whereas deca-chlorobiphenyl is 15 ppb (Alford-Stevens, 1986). Solubility of different PCB positional isomers within a homolog group (same number of chlorines, different ring positions) can vary considerably (Dexter & Pavlou, 1978) and solubility in artificial seawater was found to be about five times lower than the corresponding values in distilled water. In addition, solubility was found to increase exponentially with temperature in the range 4 ºC to 80 ºC for each of six PCBs with one representative congener in each homolog group (one, four, five, six, eight, nine and ten chlorines) investigated, as well as for biphenyl. These values can be used to determine solubilities within 10% error for all congeners represented, except for decachlorobiphenyl (16% error) (Dickhut, et al., 1986). Available literature data on solubilities of Aroclors in distilled water are plotted in Figure 2-5. Curve fitting of the data leads to predicted solubilities of approximately 34 ppb, 16 ppb and 6 ppb, respectively, for Aroclors 1254, 1260 and 1268. Figure 2-6 shows the variation of solubility with temperature of selected congeners containing 4, 5, 6, 8, 9, and 10 chlorines attached to the biphenyl ring (Dickhut et al., 1986; Shiu et al., 1997). Extrapolation of these data can be used to estimate solubilities at deep-sea temperatures of 4 ºC. Depending on the isomer, the estimated solubilities for Aroclor 1260 in seawater at that temperature range between 0.2 ppb and 1.2 ppb.

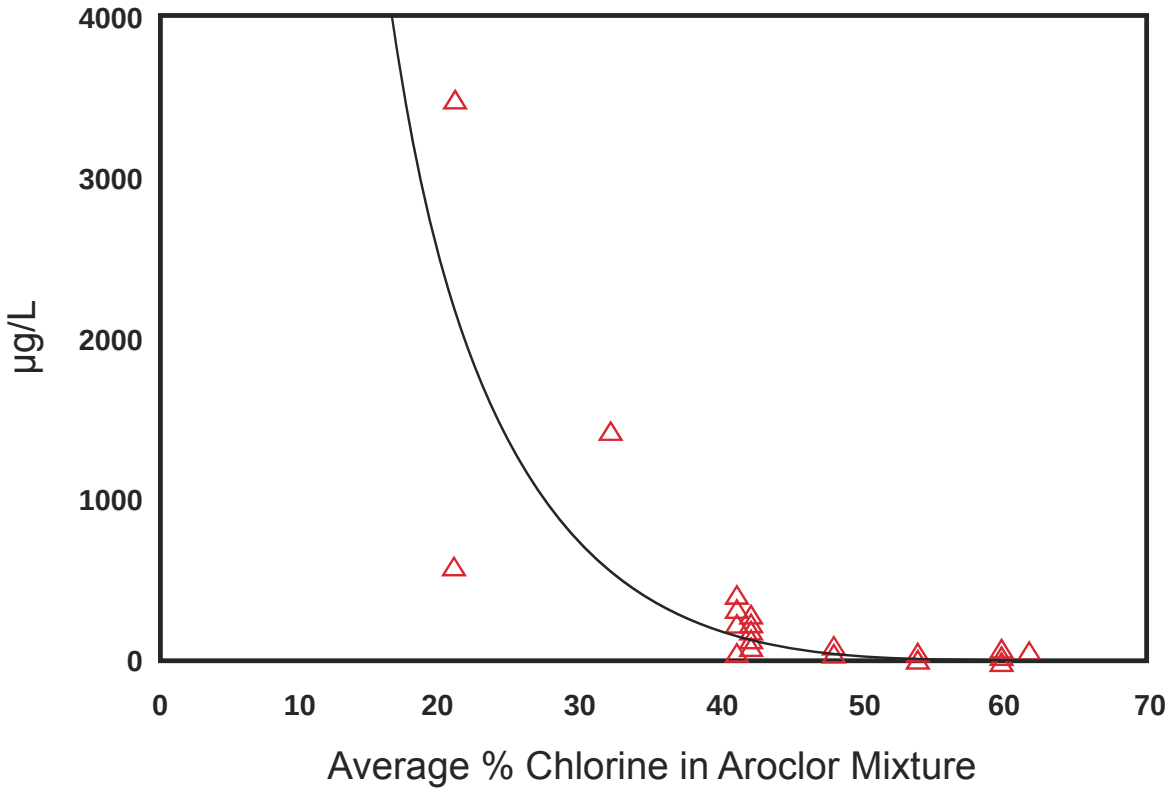


Figure 2-5. Solubility of Aroclors in distilled water.

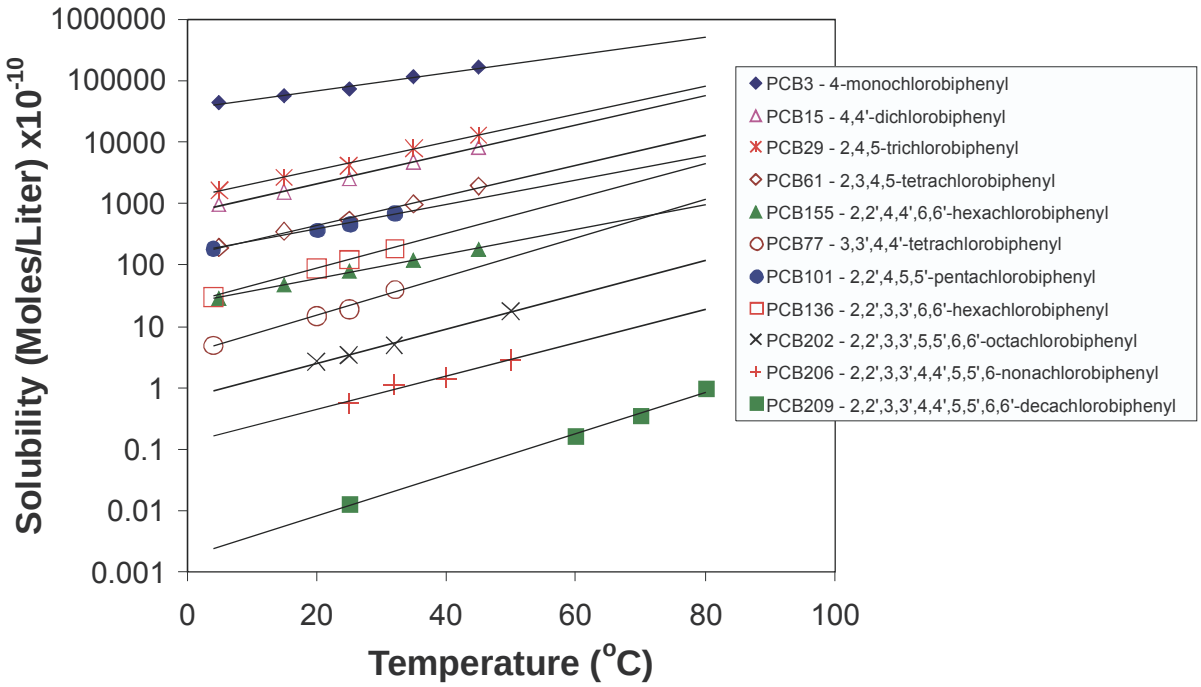


Figure 2-6. PCB Congener solubility in distilled water from two separate temperature dependence studies. PCB congeners 3, 15, 29,61,and 155 were plotted using data from Shiu, et al., 1997, and PCB congeners 77, 101, 136, 202, 206, and 209 were plotted using data from Dickhut, et al., 1986.

2.3.3.2. Partitioning and Sorption

The degree and rate at which solvated PCBs undergo sorption processes in seawater is critical in assessing the environmental risk they pose. Adsorption and desorption rates depend on the PCB mixture and substrata to a great extent. PCBs have octanol/water partition coefficients in the range of 10^5 to 10^7; the partition coefficient is an indicator of a compound's tendency to adsorb onto sediments, bioaccumulate in organisms (typically in fat or the liver), and biomagnify when moving through the food chain (Chiou et al., 1977). Adsorption and desorption of Aroclor 1254 on polymeric materials commonly used in laboratories have been studied. Adsorption of PCBs was found to increase with the lipophilicity of the polymer and to be irreversible on soft polymers (Cseh et al., 1989). PCBs tend to quickly bind to sediment, once released into an aqueous environment. In a study using clays and natural lake sediments Di Toro and Horzempa (1982) found hexachlorobiphenyl reaching equilibrium adsorption onto kaolinite and montmorillonite within 3 hours. Sediment/water partition coefficients were found to be 103 and nearly 104 respectively. Results indicated that sediment-adsorbed PCB fractions may be comprised of both reversibly and permanently bound components, though the bulk remained bound to the sediment. In a separate study examining rates at which equilibrium conditions were reached; clays, sand, silica gel and soil were tested for adsorption of PCBs from distilled water. Equilibrium was reached in 24 hours between saturated PCB solution and added adsorbent when the mixtures were shaken. (Haque et al., 1974).

The organic carbon normalized partition coefficient (K_{OC}) values for PCB isomers with an equal number of chlorine atoms are strongly dependent on the number of chlorine atoms in the ortho position (2, 2’, 6, or 6’ in Fig 2-4). Additionally, the importance of the organic carbon fraction in

the sediment as a factor that controls sorption of hydrophobic contaminants by surface sediments has been confirmed. Grain size has also been identified as a major source of remaining variation, by as much as a factor of 10 (i.e., the smaller the grain size, the more sorption occurs). The desorption rate constants for marine sediments are larger than those for soils and freshwater sediments (Booij et al., 1997). However, it has recently been reported that the traditional organic carbon normalized solid-water distribution model severely underestimates observed K_{OC} values for a wide range of compound classes including PCBs (Bucheli and Gustafsson, 2001). Failure to account for the presence of soot-like subfractions of the bulk particulate organic matter was suggested as a causative factor. The difference between observed and predicted K_{OC} values was generally relatively small in surface water but much greater in sediment. Inclusion of the soot factor in the solid-water partitioning model led to better agreement with field observed distributions. A consequence of the overestimated dissolved exposures (the ratio of observed and predicted Koc valuesthat is inversely correlated with the observed and predicted dissolved concentrations) from the traditional model, results in a lower mobility of compounds such as PCBs than currently expected.

In addition to sediment partitioning, the behavior of PCBs in pore water has also been investigated. The effect of aeration on the partitioning of 2,2′,4,4′ tetrachlorobiphenyl to anoxic pore water dissolved organic material (DOM) was recently reported from three estuarine sites (Pedersen et al., 1999). Pore water DOM is derived from the decomposition of aquatic and terrestrial organisms and from the byproducts of microbial metabolism. As a result, DOM is largely composed of a heterogeneous mixture of organic molecules including humic acids, fulvic acids, hydrophobic acids, carbohydrates, amino acids, carboxylic acids, and hydrocarbons. In general, the association of hydrophobic organic compounds (HOCs) with DOM results in higher pore water concentrations, and reduced bioavailability of HOCs. Only freely dissolved HOCs appear to be taken up by benthic organisms through ventilation of sediment interstitial water and contaminated overlying water. The results supported the findings of others that the ability of humic substances to bind HOCs such as 2,2′,4,4′ tetrachlorobiphenyl increases with DOM hydrocarbon content. However, upon exposure of anoxic sediment pore water to dissolved oxygen, DOM bound HOC's are released into the aqueous phase and their mobility is increased. Aeration of anoxic sediments may also result in increased HOC bioavailability to water column organisms and benthic invertebrates via the diffusion rout of exposure (Pedersen et al., 1999). To a limited extent, aeration of sediment pore water may occur in a deepwater site such as that of the ex-AGERHOLM via sediment scouring due to currents and bioturbation from infaunal activity. Hence some increase in availability of sediment bound PCBs may occur due to aeration.

2.3.3.3. Biodegradation and Transformation

In general it may be assumed that the photo degradation rate of PCBs in water is about a tenth of the photo-degradation rate in the atmosphere (Sinkkonen and Paasivirta, 2000). Most UV light is absorbed in the top two meters of natural fresh and coastal marine waters. The photo-degradation rates of organic compounds in solution usually show only a weak temperature dependency. A 10 ºC increase in temperature accelerates the reaction only by a factor between 1.15 and 1.5. Regarding biodegradation in general, an increase in temperature by 10 ºC may result in a concomitant increase in the degradation rate by a factor of 2.2, and decreased degradation may result with a decrease in temperature. Others have suggested a factor of 2.5-3 for each 10 ºC rise in temperature (Sinkkonen and Paasivirta, 2000). Estimates of biodegradation half-life times for PCBs in sediments and soils vary from several years to decades. Anaerobic dechlorination has

been observed in a large number of sediments. In Hudson River sediments, penta and tetra chlorinated PCBs have shown half-lives averaging about ten years. A half-life of nine years has been estimated for PCB 105, 126, 156, and 169 in anaerobic sediment (Sinkkonen and Paasivirta, 2000).

Biodegradation of PCBs has been shown to be inversely proportional to their chlorine content, and degree of chlorination at the ortho positions. In addition, PCBs that are chlorinated preferentially on one of the phenyl rings are degraded more quickly, and it is the less chlorinated ring that is attacked first (Furukawa et al., 1978). Mono-, di- and trichlorinated species can be significantly biodegraded or biotransformed, as well as volatilized. Studies on aerobic microbial metabolism of PCBs have shown that PCBs containing fewer than 5 chlorines per molecule are extensively degraded, while heavier molecules tend to persist in the environment (Rochlind et al., 1986). Since biodegradability of PCBs is a function of the number of C-H bonds available for hydroxylation, fewer chlorine atoms allow for more adjacent unchlorinated carbons, and result in higher rates of bio-oxidative activity by microorganisms (Tabak et al., 1981). A recent study (Williams and May, 1997) has shown that microbial aerobic degradation of Hudson River sediments spiked with Aroclor 1242 can occur at temperatures as low as 4 ºC within six weeks. Extensive degradation of most of the dichlorobiphenyls, particularly 2,2′-dichlorobiphenyl and 2,4-dichlorobiphenyl, occurred within five months. Degradation at such low temperatures, although in river sediments, suggests that microbial aerobic processes may also be possible at least at the sediment surface at the ex-AGERHOLM site.

PCBs with five or more chlorine atoms per molecule tend to adsorb to suspended materials and accumulate in the organic components of sediments due to low aqueous solubility, increased hydrophobic characteristics leading to high octanol/water partition coefficients, and resistance to biodegradation. Evidence is accumulating rapidly; however, that bacterial dechlorination of the more chlorinated congeners does occur in anaerobic, reducing sediments. Dechlorination of heavily contaminated, anaerobic sediments in the upper Hudson River, NY (primarily contaminated with Aroclor 1242); Silver Lake, MA (Aroclor 1260); Waukegan Harbor, IL (Aroclor 1248); and Sheboygan Harbor, WI was detected when sediment samples showed a congener composition quite different from that found in the original contaminants (Brown et al., 1987). At first thought to be the result of differential transport and partitioning, the role of microorganisms in anaerobic dechlorination was confirmed by seeding anaerobic sediments, freshly spiked with Aroclor 1242, with Hudson River sediment microbes and observing rapid dechlorination while keeping the sediments anaerobic (Quensen et al., 1988). Dechlorination was most extensive in sediments with the highest spiked PCB concentrations (700 ppm), but did not occur at all in sterilized sediments. It was reported that 53% of the total chlorine was removed in 16 weeks and the proportion of mono- to dichlorobiphenyls increased from 9% to 88%. Chlorine groups were not removed from the ortho positions on the molecules. The 2-chlorobiphenyl concentration increased from 0% to 63% in the sediments spiked with the highest concentration of Aroclor 1242 (Quensen et al., 1988). Dechlorination of deeper sediments (15-17.5 cm) in New Bedford Harbor, MA, heavily contaminated (≤6-7% dry weight) with Aroclors 1242 and 1254 has also been observed (Lake et al., 1992). Half-lives for different congeners have been found to be on the order of 10 to 20 years, though the rate and extent of degradation was considered to be highly site specific, dependent upon factors such as initial PCB concentrations, depth, temperature, other contaminant species, and nutrients present. Microbial metabolism of PCBs is thought to be due to the microbial community's evolved ability to "detoxify" aromatic

compounds. Partial dechlorination by anaerobes sets the stage for further biodegradation by aerobic bacteria if oxygen levels subsequently increase (Hooper et al., 1990).

Anaerobic slurries of estuarine sediments from Baltimore Harbor spiked with 800 ppm Aroclor 1260 induced extensive meta-dechlorination and moderate ortho-dechlorination in all incubated cultures, except in the case of sterilized controls after six months (Wu et al., 1998). Both meta- and ortho-dechlorination were stimulated by the addition of single PCB congeners (2,3,4,5-tetrachlorobiphenyl and 2,3,5,6-tetrachlorobiphenyl to the Aroclor spiked sediment. After 181 days, 45-65 percent of the meta chlorines and 9 to 18 percent of the ortho chlorines had been removed depending upon congener supplementation. Maximal chlorine removal appears to require the complementary action of two or more dechlorination processes. Anaerobic PCB dechlorination has the potential to reduce the toxicity of PCBs and convert highly persistent congeners, frequently the more extensively chlorinated congeners, into forms that are more amenable to aerobic degradation. It was observed in general that PCB dechlorination is more stable (more extensive and with shorter lag times) when sediments are stored anaerobically at room temperature (20-22 ºC), rather then at 4 ºC (Wu et al., 1998).

Recent studies conducted with sediment slurries from estuarine river systems (Tansui and the Erjen River, China) have appeared in the literature describing anaerobic PCB degradation under such conditions (Cham-en-Kuo et al., 1999). Tetra chlorobiphenyl, and four coplanar congeners were added to anoxic sediment slurries at a concentration of 10 milligrams per liter. During a two year incubation in sulfate-containing marine sediments, biphenyl (unchlorinated) was present, while all other chlorinated congeners, except for 3,3′,4,4′,5,5′-hexachlorobiphenyl were dechlorinated in sediment slurries collected from both rivers. Microbial communities in sediment from the Erjin River also promoted meta-dechlorination activity, but only after removal of all chlorines in para positions. The rates of PCB dechlorination in anaerobic environments were slow, with major changes usually occurring only over a period of months or years. Most PCB dechlorination studies have been conducted in freshwater sediments; therefore understanding the PCB biotransformation potential in marine and estuarine sediments under anaerobic conditions is only based on a limited number of reports (Cham-en-Kuo et al., 1999). Recently, it has been reported that reductive dechlorination in estuarine sediments (Sado Estuary, Portugal) has resulted in increased proportions of lower-chlorinated PCB congeners to the higher chlorinated congeners at increased depths (Gil and Vale, 2001) . A half-life of approximately 10 years was estimated in the study, which is similar to half-lives estimated in other studies cited in this section.

Regarding the biodegradability of the three most toxic coplanar congeners, #77 (3,3′, 4,4′-tetrachlorobiphenyl) is the most biodegradable, #126 (3,3′, 4,4′, 5-pentachlorobiphenyl) is persistent but moderately biodegradable, while #169 (3,3′, 4,4′, 5,5′-hexachlorobiphenyl) is nearly metabolically stable (Tanabe, 1988). However, these three coplanar congeners have rarely been reported in environmental samples (McFarland and Clarke, 1989).

2.3.3.4. Diffusion and Current Transport in the Deep Ocean

The physical environment of the bathyal and abyssal sea bed greater than 3,000 m in depth[3] has been recently illuminated by the high energy benthic boundary layer experiment (HEBBLE) in the north Atlantic (Nowell and Hollister, 1985). HEBBLE data and other studies indicate that a turbulent, relatively homogeneous nepheloid layer, 10 m-100 m thick exists over much of the abyssal plains (Richards, 1990). This layer is isolated from the overlying water column to some extent by temperature-induced density gradients. Particle residence times within the layer are estimated at hundreds of days. Currents, typically 1-5cm sec^{-1}, arise from semidiurnal, inertial and lower frequency (50-100 day) oscillations caused by tides and meso-scale (100s km) eddies that extend throughout the water column. Water speeds from 20-40cm sec^{-1} have been measured over several days, resulting in local erosion or deposition of sediment eroded from distant sites (Gross and Nowell, 1990). Low frequency (>2 days) eddy currents with much higher velocities, approaching several knots (50-150 cm sec^{-1}) have been observed at 3,000-4,000 m depth in the vicinity of the Gulf Stream, and at lower velocities throughout the North Atlantic basin (Schmitz, 1984). Low frequency, eddy currents often exceeding 10 cm sec-1 have been observed in a topographically featureless area of the North Pacific at 6,200 m (Imawaki and Takano, 1982). These currents, if they extend to the bottom, are strong enough to erode unconsolidated sediment particles in excess of 0.5 mm diameter (Komar, 1976; Bruun, 1966). However, these conditions are not necessarily predictive of currents in the vicinity of the ex-AGERHOLM, which lies in much shallower water and on the slope of a basin.

2.3.3.5. The Sinking and Subsequent Release of PCBs from the Barge IRVING WHALE.

PCB sources, physical characteristics, and fate and effects processes have been discussed in the previous sections, and are related to the ex-AGERHOLM where possible. We have not found a comparable deep-water site where a vessel with known PCB source data has sunk. A shallow-water site does exist and will be discussed in the following text. This site is located in 67 m of water (220 ft) in the Gulf of Saint Lawrence. This is where the barge IRVING WHALE (82.3 m long, 17.7 m wide) sunk at this site in 1970. It was estimated that the barge contained approximately 6,800 liters (1,500 gallons) of Aroclor 1242 in a heat-transfer system. Analysis of sediments immediately beside the barge exhibited concentrations as high as 890 ppm as Aroclor 1242. No PCBs have been found in sediments beyond 2.5 kilometers from the site. On August 8, 1996 the barge was lifted and removed from this site. As of 1999, it is estimated that 150 kg of PCBs remain in the sediments at the sinking site, and that 5,700 kg were lost from the barge and dispersed over time. The 150 kg that remains in situ is estimated to be less than 1% of the total PCB sediment load currently in the Gulf of Saint Lawrence. A risk assessment performed for this site suggests that there is no risk to snowcrab populations, human health, or the Gulf ecosystem. Monitoring of snow crab tissue indicated that in all but one sample, PCBs were below detection limits (0.01 μg/g wet wt.). Two digestive gland samples contained PCBs at levels (2 and 2.7 ppm) equal to or greater than the health consumption guideline of 2 ppm; Fisheries and Oceans Canada and Environment Canada, 1997; Environment Canada, 1997; Gilbert and Walsh, 1996).

The nature of the PCB source (liquid Aroclor 1242) and the environment where the barge sunk are considerably different from the ex-AGERHOLM site in several ways. Aroclor 1242 is less

[3] While the SINKEX study ended up with a hulk that is lying in less than 1,000 m depth, SINKEXs in general are intended to be conducted in depths greater than 2,000m and many have been done in depths greater than 3,000m.

chlorinated than Aroclors expected to be on the ex-AGERHOLM such as 1254,1260, and 1268. Aroclor 1242 was released as a liquid from the barges' heating exchange system in a relatively fast time frame, as opposed to the ex-AGERHOLM's theoretical slow release from PCBs-ISM. The sediments at the IRVING WHALE site are very sandy and coarse (87% sand, 11% gravel, and 2% clay), while sediments at the ex-AGERHOLM site are mostly fine silt and clay. The water current at the IRVING WHALE site is approximately 25 cm per second, while 99.6% of the bottom currents measured at the ex-AGERHOLM site (discussed in Section 3.2.1.1) are less than 15 cm per second. From a source and physical environment perspective, the IRVING WHALE site is quite different than the ex-AGERHOLM site[4].

2.3.3.6. Fate and Transport/SSC SD Modeling Assessment

Prior to the current SINKEX study, estimates of the amount of PCBs delivered to the deep seabed by a SINKEX hulk and the probable environmental processes these compounds undergo were used in a simple finite-difference, one-dimensional model to predict PCB transport and adsorption onto the sediment. Results of the model were used to evaluate the water and sediment PCB concentrations to which benthic organisms might be exposed. These data, coupled with PCB uptake and toxicity measurements made on marine organisms living at shallower depths, provided the groundwork for further analysis of the environmental risk posed by PCBs to the benthos. The model was updated with new information in 2005 and used to predict water and sediment PCB concentrations around a ship. A new total mass of 86 lbs (39.1 kg) was used for PCBs in PCB-containing materials aboard ship by including an estimate for PCBs associated with bulkhead insulation. Water and sediment PCB concentrations were predicted, assuming either a 0.1 cm sec^{-1} current (1% of current outside of vessel). In both cases, PCBs were assumed on average to spread equally in all directions.

2.3.3.6.1. Release into Aquatic Environment

Load assumptions: The model was developed to predict water and sediment PCB concentrations around a typical sunken ship. Vessel release for ex-AGERHOLM was calculated from shipboard solid material specific leach rates[5] (George et. al., 2006, in preparation), which were applied to source estimates of quantities and concentrations of each material summarized earlier in Section 2.3.1.2.2 (JJMA, 1998). Shipboard solid material specific leaching was evaluated in the leach rate study under temperatures and pressures expected for deep ocean conditions, and the materials tested were complimentary to this SINKEX ERA: HVAC felt gaskets, electrical cable insulation, paint, rubber products, bulkhead insulation, and foam rubber insulation. Oils and greases were not evaluated in the leach rate study, but were approximated for modeling purposes, using Aroclor 1254 control dissolution rates reported in the leach rate study. The source term loading estimate for ex-AGERHOLM performed by John J. McMullen and Associates (JJMA, 1998) provided quantities and concentrations for all of the PCB-containing shipboard materials above, with the exception of bulkhead insulation, for which information was

[4] Interesting information on the IRVING WHALE is available on the internet at http://collections.ic.gc.ca/island/whale.htm, http://www.ns.ec.gc.ca/whale2/pcb.html, and http://www.ns.ec.gc.ca/whale2/index.html

[5] Due to the complexity of the various PCB-related studies, the leach rate data is referenced here and used in the 2005 update to the 1994 modeling, but they are appropriately described in detail later (Section 3.2.1.3), since they are integral to a larger data set based on the post-1994 sampling and analytical efforts described as the SINKEX Project.

unavailable. The JJMA results were used directly as reported for HVAC felt gaskets, electrical cable insulation, paint, rubber products, and foam rubber insulation. Bulkhead insulation was estimated separately using well-known quantities of bulkhead insulation onboard another vessel, the ex-ORISKANY, with a total of 104,126 lbs. This amount of bulkhead insulation was scaled to approximate the amount onboard ex-AGERHOLM by the ratio of their displacements (3,500:27,100). This resulted in an estimated 13,448 lbs of bulkhead insulation, which was assumed to have the same concentration as ex-ORISKANY (537 ppm at 95% UCL), thus contributing 7.2 lbs of PCBs to the 86 lb total PCB load indicated above. The estimated source loadings by material are shown in Table 2-4. These quantities and the corresponding shipboard solid specific leach rates were used to calculate the total vessel release rates from all materials, tabulated by homolog as a function of time in the mass loading input file used by the model (Table 2-5).

Table 2-4. Revised Initial Load Estimates used in the 2005 modeling effort for PCBs (in lbs) on ex-AGERHOLM prior to 1982.

Shipboard Materials	High Estimate
HVAC gaskets (felt only)	40.0
electrical cable insulation	20.2
paint	9.7
oils/greases*	8.3
rubber products	0.1
foam insulation	0.1
bulkhead insulation	7.2
Totals	**86**

Table for the revised modeling effort in 2005, for only those materials tested in the leach rate study, uses data from 1 Dec 98 Draft JJMA Report title "Weight Estimates for PCBs and Selected Metals Sunk on ex-AGERHOLM (DD-826) for the Deep Water Sunken Ship Study". Bulkhead insulation, not estimated in the JJMA report, was instead estimated as described in 2.3.3.6.1. Oils/grease release rates were not determined in the leach rate study, but were approximated using Aroclor 1254 control dissolution rates.

Leaching, transport and partitioning assumptions: Considering that most ships are sunk in depths over 3,000 m and the average depth from the 12 known events was about 3,900 m, a bathyal/abyssal ocean environment with a benthic boundary layer was assumed. The ex-AGERHOLM was modeled as an elliptical volume 119 m by 11 m and 6.8 m high. The dimensions were chosen to match the estimated volume of the real ship. The benthic boundary layer was modeled to be 6.8 m thick. Transport model predictions, first made from a 1992 model, were replaced with predictions based on measured 10 cm sec-1 current speeds, PCB release data from ship materials in the laboratory, and a partitioning model similar to one employed to predict PCB dispersal from ships sunk as artificial reefs in shallow water (SSC-SD et al., 2006).

The model predicts the transport and abiotic fate of specific PCB homologs, due to their distinct chemical characteristic. PCB homologs are first released into water internal to the ship, based on Navy leach rate studies (George et al., 2006). PCBs then slowly leak outside of the ship, and are advected away in a current. The ship is assumed to have no internal structure or adsorptive sediment. Outside of the ship, PCBs approach new equilibrium concentrations in water, sediment, total suspended solids (TSS), and dissolved organic carbon (DOC) to a distance of 1.2 km in all directions from the ship.

PCBs are released from solid materials within the internal ship volume at 1-minute intervals and mix instantaneously into the interior water. PCB mass is assumed to be released out of the ship at the same rate as the release inside, ranging from 1 to 100 mg total PCB day^{-1}. Table 2-5 lists PCB homolog release rates as measured from typical shipboard materials and weighted by the ex-AGERHOLM's expected mass load. Shipboard solid-specific leach rates were measured over 405 days (George et al., 2006), effectively limiting the model results to the empirical leach rate timeframe.

Table 2-5. PCB homolog release rates for ex-AGERHOLM over an initial 405 days following sinking. Release is based on empirically determined leach rates for each shipboard solid at deep ocean temperature (4°C) (George, et al., 2006), and estimated PCB concentrations and quantities of shipboard solids onboard ex-AGERHOLM (JJMA Report, 1998).

	Sum of All Material Contributions Averaged over each Interval ex-AGERHOLM Max est conc Total Vessel Release Rate (g PCB/day)										
interval	**Cl1-all**	**Cl2-all**	**Cl3-all**	**Cl4-all**	**Cl5-all**	**Cl6-all**	**Cl7-all**	**Cl8-all**	**Cl9-all**	**Cl10-all**	**tPCBs-all**
0.003	0.0E+00	3.0E+00	1.7E+01	3.1E+01	7.7E+01	9.7E+01	1.6E+02	0.0E+00	0.0E+00	0.0E+00	3.9E+02
1	0.0E+00	0.0E+00	1.4E-04	2.9E-04	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	4.4E-04
6	6.5E-04	1.9E-03	7.4E-04	1.5E-02	1.8E-04	0.0E+00	1.2E-02	0.0E+00	0.0E+00	0.0E+00	3.0E-02
14	4.3E-04	1.5E-03	1.6E-03	2.9E-02	2.5E-02	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	5.8E-02
21	5.1E-04	1.5E-03	2.2E-03	5.5E-02	5.0E-02	2.2E-04	0.0E+00	0.0E+00	0.0E+00	0.0E+00	1.1E-01
28	3.7E-04	1.1E-03	2.8E-03	7.2E-02	8.0E-02	6.6E-04	0.0E+00	0.0E+00	0.0E+00	0.0E+00	1.6E-01
35	2.1E-04	7.7E-04	4.3E-03	6.1E-02	8.3E-02	1.1E-02	5.8E-06	0.0E+00	0.0E+00	0.0E+00	1.6E-01
42	2.4E-04	8.1E-04	1.3E-03	5.2E-02	7.2E-02	1.4E-02	7.7E-06	0.0E+00	0.0E+00	0.0E+00	1.4E-01
42	2.1E-04	6.7E-04	1.3E-03	4.9E-02	7.0E-02	2.1E-02	2.5E-05	3.9E-06	1.2E-05	0.0E+00	1.4E-01
49	2.0E-04	6.3E-04	2.0E-03	5.2E-02	8.2E-02	2.2E-02	4.2E-06	0.0E+00	0.0E+00	0.0E+00	1.6E-01
42	2.3E-04	7.8E-04	2.9E-03	5.1E-02	8.7E-02	1.7E-02	1.9E-05	2.1E-05	1.4E-06	0.0E+00	1.6E-01
42	1.1E-03	6.7E-04	5.6E-03	5.7E-02	1.0E-01	2.9E-02	3.8E-05	3.0E-05	0.0E+00	0.0E+00	2.0E-01
83	1.7E-04	5.1E-04	1.9E-03	3.5E-02	4.9E-02	4.0E-04	2.6E-05	2.7E-05	0.0E+00	0.0E+00	8.7E-02

Outside of the ship, PCB concentrations are calculated in concentric bins (elliptical annuli) 6 m wide, expanding away from the ship and extending from the bottom to 6.8 m above the bottom. Partitioning from water to other media is assumed to follow first–order kinetics. As shown in Figure 2-7, 99% of equilibrium concentrations are assumed reached within 24 hours (Di Toro and Horzempa, 1982). As a conservative approach, no PCB degradation is assumed to occur. The model time step of 1 minute allows 0.32% equilibrium concentrations to be reached between water DOC and TSS and sediment. A width of 6 m was chosen to match the distance a 10 cm sec-1 current travels in the 1-minute model time step. The model extends outwards 200 bins or 1,200 m from the ship. The model assumes advective "plug flow," through which the entire volume of a bin (water, TSS, and is moved to the next bin with each time step). Sediment is not transported between bins, and thus acts as a sink for PCBs in each bin.

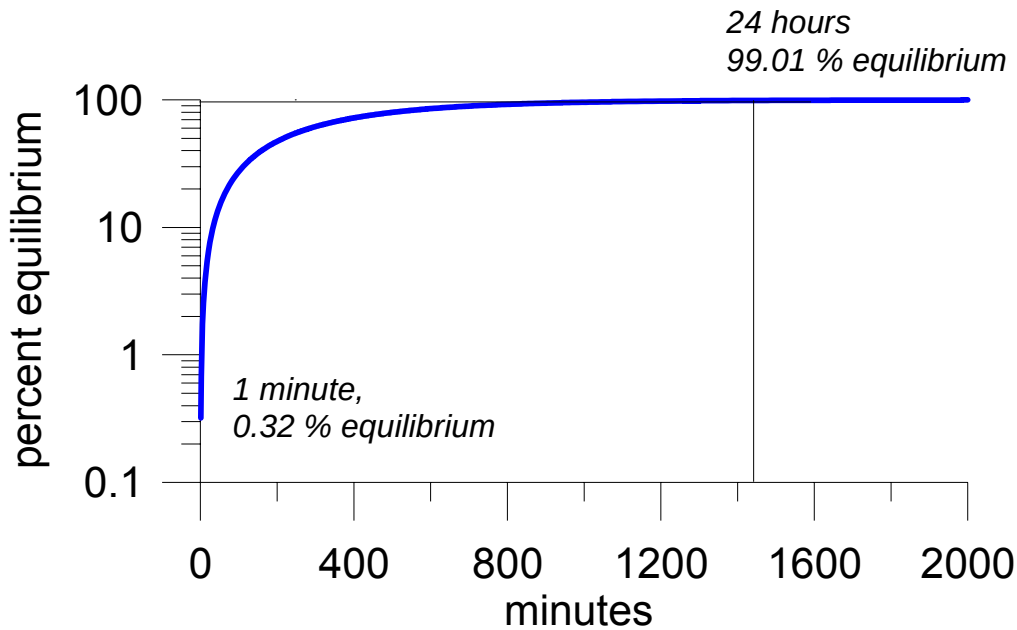


Figure 2-7. Equilibrium curve for PCBs released from the ex-AGERHOLM and 1-minute model time step.

Several assumptions are made about the matrices in which PCB homolog concentrations are calculated. Water is assumed to have a density of 1 g ml-1. Sediment is assumed to contain 1% total organic carbon (TOC), denoted in the model as fraction of organic carbon, f_{oc}; in the case of sediment, this is equal to 0.01. Sediment and adsorbed PCBs are assumed mixed (bioturbated) to a depth of 10 cm. Sediment is assumed to have a density of 1.5 g ml-1. DOC is assumed to occur at 0.6 mg liter-1 in the water column and be composed of 100% TOC. TSS is assumed to occur at 10 mg liter-1 and be composed of 15% TOC.

Equilibrium homolog concentrations are defined as homolog water concentration times the bulk partition coefficient of the matrix. (Fetter, 1999; Maidment, 1993). The bulk partition coefficient is the product of the water-organic carbon partitioning coefficient K_{oc} and the fraction organic carbon f_{oc}. K_{oc} values specific for each homolog are listed below in Table 2-6. The bulk partition coefficient is also the slope of the initial sediment adsorption isotherm. (Weber et al., 1990; Thibodeaux, 1996) Desorption of PCBs from sediment is assumed to follow a shallower isotherm than adsorption and preserve a residual adsorbed fraction which cannot be reversibly desorbed (Di Toro and Horzempa, 1982).

Table 2-6. Homolog water-organic carbon partitioning coefficients used in TDM.

PCB homolog	K_{oc}
Monochloro	$4.61*10^3$
Dichloro	$1.14*10^4$
Trichloro	$4.22*10^4$
Tetrachloro	$4.51*10^4$
Pentachloro	$8.61*10^4$
Hexachloro	$1.2*10^6$

PCB homolog	K_{oc}
Heptachloro	$2.19*10^6$
Octachloro	$2.85*10^6$
Nonachloro	$9.24*10^6$
Decachloro	$8.72*10^7$

In the model, PCBs are released only down current from the ship, but the currents are assumed to flow equally in all directions over time. As a result, PCB dispersal radially symmetric around the ship; the PCB load is distributed in all directions in each one-minute time step. Within each minute time step, water, DOC, TSS and sediment can either adsorb or desorb PCBs, depending on their adsorbed concentrations and the equilibrium water concentration. Plug flow moves water and entrained DOC and TSS into the next outer bin in each time step. In the second and subsequent bins, water PCB concentrations are diluted by the larger volume of the next bin. The density of DOC and TSS in the water column (g ml-1 water) is assumed to be constant; hence there is more DOC and TSS in the next, larger, outer bin than in the previous bin. In order to conserve PCB mass associated with this DOC and TSS transport between bins, potential adsorption or desorption is calculated for only the fraction of DOC and TSS advected from the previous inner bin into the current bin. The current bin contains this DOC and TSS from the previous bin, plus an additional amount of initially "clean" (PCB-free) DOC and TSS fraction to maintain the same density. The new PCB mass adsorbed to DOC or TSS from the previous bin is divided by the mass of all the DOC or TSS in the current bin – effectively diluting it. This approach is logical if one assumes that currents first advect particles in one direction, then another, making the average concentration equal to the incoming load averaged over all DOC or TSS mass at that range from the ship.

2.3.3.6.2. Modeling Results

Total PCB concentrations in water, sediment, TSS and DOC were highest next to the ship and are shown in Figure 2-8. The concentrations vary with time, reflecting the varying release rates measured from shipboard materials and listed in Table 2-1 and Table 2-5. In this and subsequent figures, the ten PCB homolog concentrations were summed to yield total PCB concentrations, for each type of matrix. The individual homologs must be modeled separately to account for their different K_{oc} values, which express themselves in the degree of partitioning from water to the other abiotic media. Highest concentrations occur immediately on sinking; water concentrations peak at 4e-14 (0.04 parts per trillion), sediment concentrations peak at 1.5 e-11, TSS concentrations peak at 3e-9, and DOC concentrations peak at 2e-8 (20 parts per billion). The concentration differences in these media mirror the increasing percent organic carbon assumed. The final lower concentrations at 405 days are assumed to represent steady-state conditions, though release rates would most likely continue to decrease. Note that the sediment concentration remains stable due to the permanently bound PCB fraction argued by DiToro and Horzempa (1982).

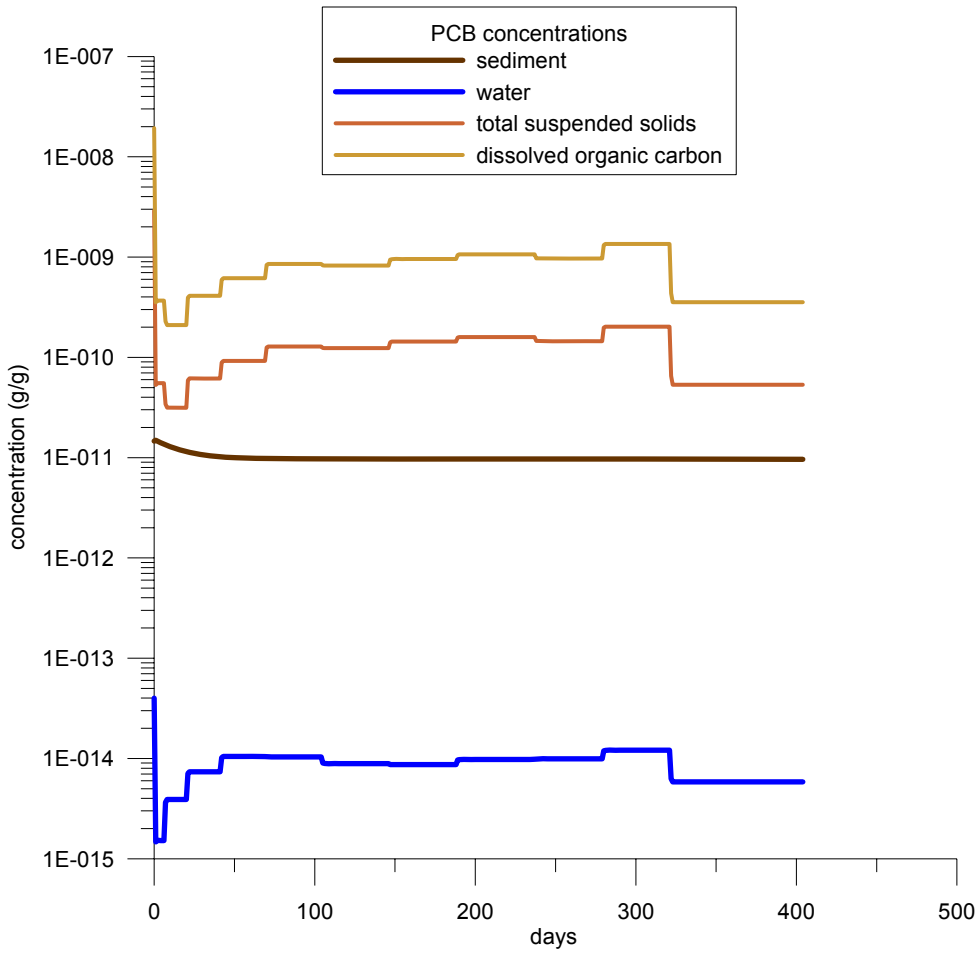


Figure 2-8. Total PCB concentrations adjacent to the ex-AGERHOLM.

PCB mass retained in the water, sediment, TSS and DOC within the model domain are shown in Figure 2-9. The largest fraction (0.3 g) is permanently bound to the sediment. The vast majority of PCBs is advected out of the model domain via water, TSS and DOC. Assuming the release rate during the final interval (322-405 days) is maintained instead of decreasing as the data suggest (0.087 g day^{-1}, Table 2-5), the remaining calculated PCB mass of 38.9 kg would require approximately 1,240 years to escape the ship.

Total PCB concentrations in the different media as a function of distance from the ship are shown in Figure 2-10. In this plot, the concentrations are averaged through time, hence are lower than the initial concentrations associated with the initial release rates, but higher than those resulting from the lower, long-term release rates measured between 322 and 405 days in the laboratory. Reduction in concentrations is primarily due to volume dilution, and secondarily – to partitioning from the water to sediment TSS and DOC in the 200-minute transit time within the model domain.

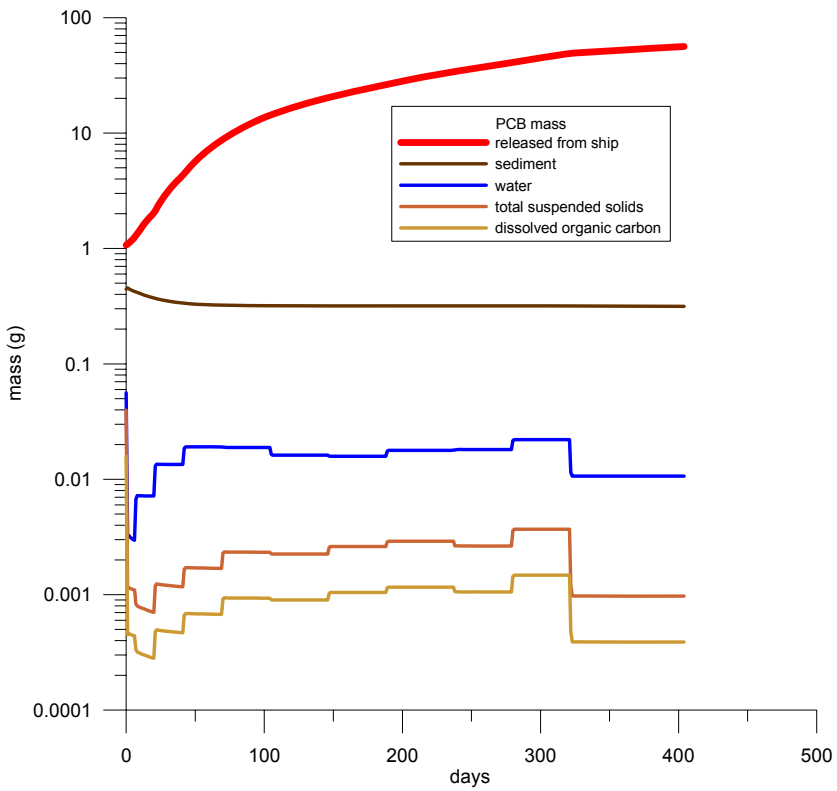


Figure 2-9. Total PCB mass in various abiotic compartments.

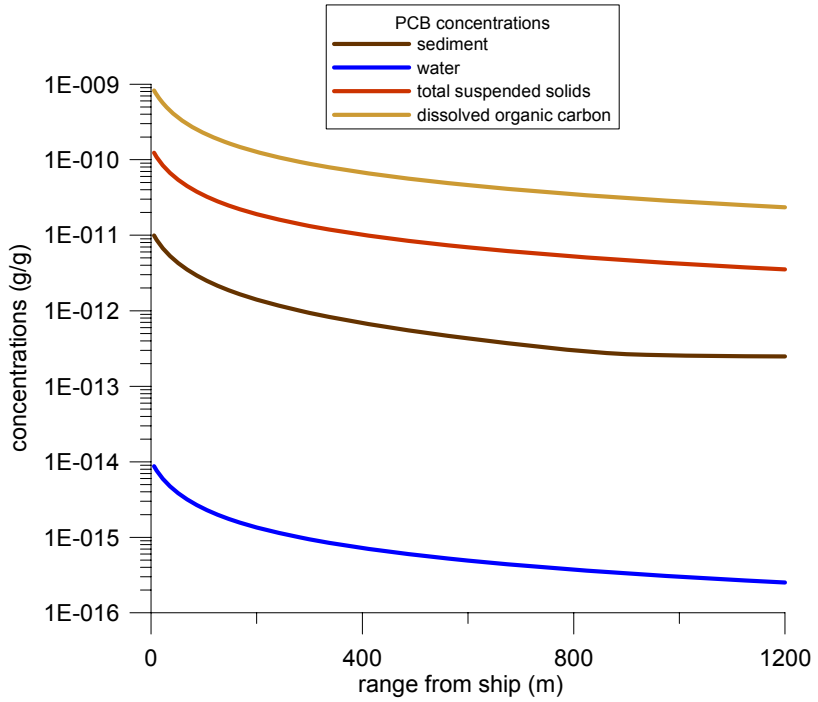


Figure 2-10. Mean total PCB concentrations around the ship. Concentrations are average values at specific distance from the ship averaged over the 405-day release period.

2.3.4. PCB Levels and Effects in the Deep Ocean

The following sections describe PCB concentrations in open ocean waters and sediments. This background information is followed by discussions of aquatic toxicity and sediment toxicity, bioaccumulation, biomagnification and trophic transfer, PCB tissue burdens and critical concentrations, and the concept of toxic equivalency factors and relevance to marine species.

2.3.4.1. Background PCB Concentrations in Open Ocean Waters

PCB concentrations in the open oceans and Mediterranean Sea are typically on the order of a few pptr or less and usually higher at the surface than at depth. Harvey et al. (1973) measured PCBs at 35 and 27 parts per trillion (pptr) at the surface in the North Atlantic and Sargasso Sea respectively. Values at 1,000 m to 1,500 m ranged from 1 pptr to 1.5 pptr. A year later (Harvey et al. 1974) much lower surface values were found (0.8 pptr and 1 pptr respectively) which they attributed to loss through a downward transport in the water column of particulate material (particle flux). Deep-water PCB concentrations remained around 0.1 pptr to 2 pptr. Knap et al. (1986) found only 0.1 pptr PCBs at the surface in the Sargasso Sea. Schulz et al. (1988) found PCBs at 4 pptr to 50 pptr in surface waters of the North Atlantic, but lower values (0.01 pptr) at depth. Tanabe and Tatsukawa (1983) found low concentrations (<1 pptr) in the Pacific and Antarctic Oceans that tended to be uniform with water depth. In the Mediterranean, Geyer et al. (1984) found PCBs at 1 to 2 pptr at the surface and 0.1 to 2 ppt in deep water. Burns and Villeneuve (1987) reported total PCBs as Aroclor 1254 at 14 pptr in the Mediterranean at a depth of 1,500 m.

Some recent data has been reported from the Catalan sea in the Western Mediterranean. Individual PCB congeners were determined in 15 samples of suspended particulate matter. The extracted congeners reported were No. 28, 52, 44, 70, 101, 118, 153, 138, 187, 128, 180, and 170. Also, six dissolved-phase samples were taken and analyzed. Concentrations of total PCBs (sum of 12 congeners) ranging from 28-63 pg/L in the dissolved phase were one order of magnitude higher than in the corresponding particulate phase 1.7-16.6 (Dachs et al., 1997). Open ocean total PCBs at 1,500 m were less than 1 pg/L. In western Mediterranean surficial seawater at 15 meters in the open sea, the PCB concentration was to 8.1-55.6 pg/L. No general trends of PCB congener distribution with depth were observed. It was noted that a conversion to Aroclor 1242 and 1254 equivalents can be obtained by multiplying the sum of the 12 congeners by 2.94 and 2.75, respectively (Dachs et al., 1997). PCBs on particulate samples collected at 1,000 m in the Gibralter Strait ranged from 0.01 to 1.7 pg/L for the congeners 28, 52, 101, 118, 153, 138, and 180 (Marti et al., 2001). In the western Mediterranean Sea, PCB concentrations in coastal and open seawater have shown a slight decreasing trend in recent years (Tolosa et al., 1997). In general, recently reported PCB concentrations in water contrast sharply with those measured during the 1970s in Atlantic waters, although they are in the same range as the levels detected during 1988 in the North Sea and North Atlantic. Even though PCBs exhibit high particle/dissolved concentration ratios, the relatively low levels of suspended particles in Mediterranean waters (0.4-1 ppm) suggest the dissolved phase of the open ocean as the dominant reservoir of PCBs.

Recent data from the North Atlantic Ocean near Iceland indicate that PCB congener values in surface and deep water are extremely low ranging in the femtogram (10^{-15} gm) per liter level. Use of a filtration /extraction system capable of sampling up to 2,000 liters of water has made such measurements possible. The sum total of 23 congeners ranged from 10-1048 fg/L in

solution, and 286-11,241 fg/L in suspension (Schulz-Bull et al., 1998). The PCB concentrations decreased from the surface to the bottom at each station. The lowest deepwater samples ranged from 10-254 fg/L. The coplanar congeners 77, 126, and 169 were below their detection limits (<2 fg/L) in all samples. Suspended particulate PCB concentrations were positively correlated with particulate organic carbon suggesting that suspended organic material is a carrier for PCBs in the water column.

Data from deep ocean samples from the Sea of Japan have recently been collected and indicate very low total PCB (sum of 30 congeners) concentrations. Values in solution at 1,500 and 3,000 meters depth were low at 0.2-0.3 pg/L. At 2,000-3,000 meters depth total PCB values in solution ranged from 183-604 fg/L (Kannan et al., 1998). Surface water total PCB concentrations ranged from 0.1-1.2 pg/L. Principal complement analysis showed that surface water is characterized by lower chlorinated PCBs than deep water which is characterized by higher chlorinated PCBs.

Some data are available from sites in the general area of the ex-AGERHOLM sinking several years prior to the 1982 sinking. In a 1976 study, surface water at five Southern California Bight stations ranged from 3-9.6 ng/L (pptr). At depths of 500-1,500m PCBs were 2.3-10.3 ng/L (assumed to represent total PCBs). The overall-range of these values is 0.2-9.6 ng/L for surface water, and 2.3-10.3 ng/L for deepwater (Risebrough et al, 1976). Data where latitudinal and longitudinal locations were given are plotted in Figure 2-11. These locations are relatively close to the site of the ex-AGERHOLM, which is located at 119° 35.6′W and 32° 45.4′N.

Figure 2-11. Total PCBs as Aroclor 1254 in surface water from the western boundary of the Southern California Bight (data are from Risebrough et al., 1976).

Nearly all authors cited agree that transport of PCBs to deep ocean water and sediments typically occurs via settling of detrital particles and fecal pellets. Fecal pellets sink at rates ranging from

20-260 meters per day for copepods and 150-850 meters per day for euphausiids, speed sufficiently high to ensure fast transport of PCBs to the sediment (Tolosa et al., 1997). In another study, coastal areas deposition of fecal matter also played a focal role in transport of PCBs. The filter feeding activity and subsequent release of fecal matter by mussels (*Macoma edulis*) increased gross sedimentation of carbon to the benthos by 45 percent when compared to areas with no mussels. By selectively feeding on particles rich in organic carbon the mussels also concentrated associated contaminants and thereby increased gross sedimentation of PCBs by 50 percent. This suggests that mussel biodeposition will increase the availability of PCBs to benthic deposit feeders living in or in the vicinity of mussel beds (Bjork et al., 2000). Protozoan grazing of bacteria can produce organic material, which can sorb PCBs more efficiently than background seawater dissolved organic carbon (Kujawinski et al., 2001). A recent report documents the transport of PCBs via small (0.1-0.5 cm diameter) plastic resin pellets in Japanese coastal waters. Apparent adsorption coefficients for these pellets ranged from 105 to 106. Since plastic resin pellets (from industrial raw materials used in the plastics industry) are widely distributed in the ocean worldwide, they may serve as a general transport medium of PCBs (Mato et al., 2001).

Sediment particle traps moored 3,200 m below the surface of the Sargasso Sea and 1000 m above the bottom yielded a daily PCB flux associated with particulate settling ranging from 1.2 to 10.8 ng m^{-2}, corresponding to a yearly flux of 1.6 µg m^{-2} into the sediments (Knap et al., 1986). Particulate PCB concentrations in this study ranged from 50-350 ppb dry weight. Burns and Villeneuve (1987) measured a yearly Mediterranean PCB flux rate of 13 µg m^{-2}. Elder and Fowler (1977) measured a yearly PCB flux of 80 µg m^{-2} to 125 µg m^{-2} in the Mediterranean, but only 1.4 µg m^{-2} to 4.1 µg m^{-2} in the Atlantic. Some data concerning PCBs and suspended particulate matter have recently been reported from the Western Mediterranean Sea. Residence times of the suspended particulate matter were estimated at every depositional area ranging from 1.46-8.05 years for PCBs. Concurrently, sediment settling velocities of suspended particulate matter ranged from 0.37-0.78 meters per day for PCBs (Dachs et al., 1997). Tolosa et al. (1997) have estimated the residence time of PCBs in the upper water column in the Mediterranean as probably on the order of 2-5 years.

2.3.4.2. Background PCB Concentrations in Open Ocean Sediments

Deep ocean sediments generally contain PCBs at concentrations in the low ppb range. Note that this level is three to four orders of magnitude higher than deep-water concentrations, and is reasonable, considering the partitioning coefficient measured by Di Toro and Horzempa (1982). PCBs were detected in all 94 sediment samples taken from 19 stations in the Gulf of Maine, ranging from trace concentrations to 130 ppb dry weight (Larsen et al., 1985). In the open Mediterranean Sea, flocculent layers immediately above the sediments contained two orders of magnitude more PCBs than the sediment (98 ppb vs. 0.2-1.9 ppb). A particle/water partition coefficient for PCBs was estimated at 10^6 to 10^7 ml/g. PCBs in the sediment that were more highly chlorinated resembled Aroclor 1260. The flocculent layer directly above the sediment contained PCB congeners that resembled Aroclor 1242 and 1254 signatures (Burns and Villeneuve, 1987). A review article reported PCBs at 0.2 to 9 ppb dry weight in Mediterranean sediments (Geyer et al., 1984). In the open western Mediterranean Sea sediment PCB values range from 0.8-33 ng/g dry weight as Aroclor 1260. Basford and Eleftheriou (1988) measured PCBs at 1.1 to 3.7 ppb dry weight in North Sea sediments. Measurement methods for PCBs have varied in literature. The limited inter-comparison data that are available generally supports comparability between total PCBs quantified by packed or capillary columns (Tolosa et al., 1997

Figure 2-12 summarizes numerous studies where PCBs were detected in marine sediments. Some data are available for deep benthic samples. Data primarily from shallower samples are presented since there is a scarcity of deep benthic data. Data are generally reported as dry weight, although some wet weight values were also found in the literature. These values were used as reported and not converted to dry weight because no suitable conversion factors exist and no grain sizes or moisture content are known. Because no information was found concerning sediment PCB concentrations near the site where the ex-AGERHOLM was sunk, available data from the Santa Barbara basin is presented. Additional background data summarized in a NOAA Technical Memorandum (Mearns et al., 1991) from the Southern California Bight, and specifically from some of the offshore islands are also presented in Figure 2-12. The offshore island data represent PCB measurements, which are geographically close to the ex-AGERHOLM site, but not in areas and depths identical to it. Some sediment data from the Southern California coastal shelf, near Orange County approximately 50-60km offshore, and at 600m depth has been reported (Thompson et al., 1984). Total PCB concentrations in this area ranged from 2-7 ppb in surface sediments (0-2 cm).

Additional data for sediment samples from 281 estuarine sites in the Gulf of Mexico were collected in 1993-1994 and analyzed as part of the United States Environmental Protection Agency, Environmental Monitoring and Assessment Program (EMAP) (Wade et al., 1993). Less than 4 percent of sediments studied in EMAP exceeded 20 ng/g for the sum of 20 polychlorinated biphenyls. These data are referenced as part of this report because of the wide scope of the EMAP program, although it is recognized that the sample locations are estuarine and not deep-ocean sites. The program was designed to assess the state of the environment comprehensively and randomly on a regional scale. On a national scale, a coastal sediment database (COSED), has been constructed using data from the NOAA status and trends program, EMAP, and other monitoring and assessment programs. Ten samples from the COSED database were found to exceed 20 ng/g Total PCBs. Only one exceeded the high benchmark for PCBs, >80 ng/g, of Daskalakis and O'Connor (1995). A summary of sediment PCB concentrations, collection sites, and references is provided in Appendix B.

Reductions in PCB concentrations in the Dutch coastal zone sediments have recently been reported. The largest reduction, 80 percent, in the median concentration of PCBs, occurred in surface sediment in the open sea, approximately 20 kilometers offshore, between 1986 and 1996 (Laane et al., 1999). Three main reasons for the observed decline were postulated. These include the decrease in loading from various sources; the sedimentation and mixing of less polluted suspended matter into the active sediment layer; and the washing out of previously deposited more heavily polluted particles. These data are mentioned because atmospheric deposition of PCBs has decreased with bans and restricted PCB use, and could impact ocean sediments in the event of scouring and replacement of sediment with cleaner source material. A cautionary statement regarding sediment PCB concentrations and carbon content normalization has been recently reported (Olsson et al., 2000). The authors stated that carbon normalization gave misleading information for temporal studies in the Baltic Sea, probably due to the different origins of carbon in the Baltic Sea, where a TOC mixture from two different sources exists (recent biological processes and carbon from glacial inputs).

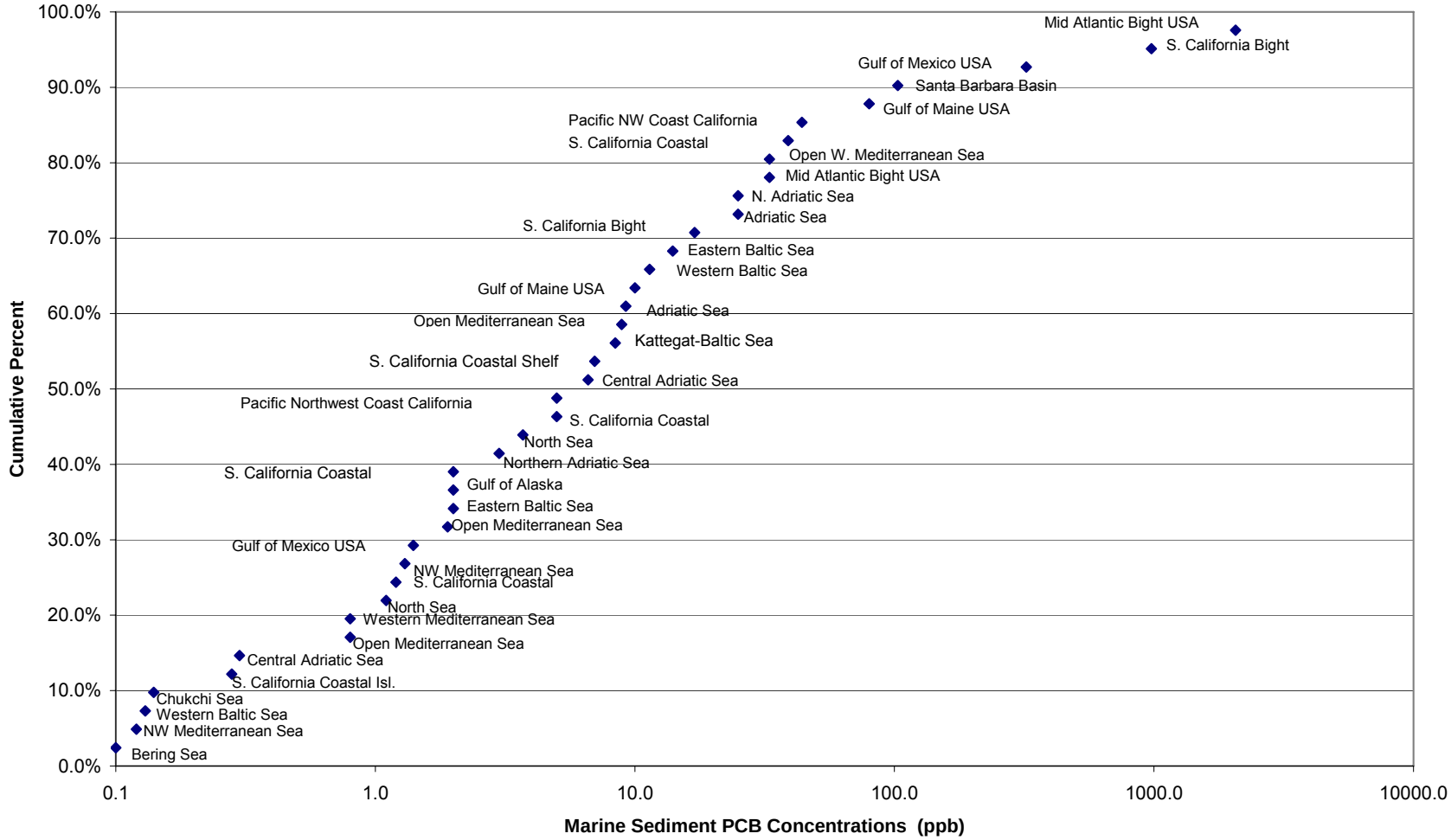


Figure 2-12. Marine surface sediment PCB concentrations.

2.3.4.3. PCB Toxicity

2.3.4.3.1. General Observations

In a general overview of PCB toxicity, McFarland and Clarke (1989) note that PCBs are not acutely toxic to aquatic biota in the natural environment. Any toxic effects of aquatic environmental PCB contamination appear to most likely be sublethal and chronic. Commercial Aroclors are dominated by mixtures of tetra-, penta-, hexa-, and hepta-chlorobiphenyl homolog groups. These are generally not acutely toxic to fish and aquatic invertebrates at typical concentrations found in the environment (Suedel et al., 1997). In laboratory tests acute toxicity (LD_{50}) of commercial PCB formulations increases with increasing chlorine content in the order of Aroclors 1221, 1232, 1242, 1248, and 1254. Aroclors 1260, 1262, and 1268 are less toxic than 1254. Long-term toxicity of PCBs has also been noted where Aroclor 1254 and Kanechlor 500 (commercial PCB product marketed in Japan) are the most toxic formulations (Tanabe, 1988).

New toxicity information from a 1996 cancer study of four commercial mixtures (Aroclors) strengthens the case that all PCB mixtures can cause cancer, although different mixtures have different potencies. Bioaccumulated PCBs, which have been transformed from original commercial mixtures, are of greatest concern because they appear to be more toxic than commercial PCBs and more persistent in the body (Cogliano, 1998). The PCB fraction that adsorbs to sediment or soil tends to be higher in chlorine content and persistence than the original Aroclor mixture. This fraction also tends to be less susceptible to metabolism and elimination, and therefore more persistent and possibly toxic. A new approach developed by EPA (USEPA, 1996a) uses toxicity studies of commercial mixtures to develop a range of cancer potency estimates and then considers the effect of environmental processes to choose appropriate values for representative classes of environmental PCB mixtures (Cogliano 1998). Mechanistic studies have demonstrated tumor promoting activity in liver or lung in mice and rats exposed to Aroclor 1254 and several specific congeners. Representing those with four to six chlorines were tetrachloro-PCBs 47, 49, 52, and 77; pentachloro-PCBs 105-118 and 126; and hexachloro-PCB 153 (Cogliano, 1998).

The following discussion concerns PCB toxicity in water, sediments, and tissues. In order to specifically assess bioavailability of PCBs in sediments, concentrations in sediment should be normalized to organic carbon content (Di Toro, et al., 1991). However, this is rarely done. Many toxicity studies have been conducted at PCB concentrations far above those reported for deep ocean sediments and water values. To our knowledge, data concerning the toxicity of PCBs, specifically addressing deep ocean species, are not available. In some instances, natural sediments were used in the presence of other measured or unmeasured toxicants, complicating interpretation and confounding the results. Often, studies are performed with reference only to total PCB or specific Aroclor concentrations. Since toxicity of individual congeners varies, accurate assessment of toxicity may not be possible from such studies (de Boer, 1988).

Regarding PCB congeners in general, few of the 209 possible congeners are considered to exhibit toxicity in spite of public perceptions about the dangers of PCBs (Bright et al., 1995). The coplanar congeners 77, 126 and 169 exhibit “dioxin-like toxicity” because of their demonstrated affinity for the aryl hydroxylase acceptor (Safe, 1990; McFarland and Clarke, 1989). However, the toxicity relationships developed for these congeners are based on mammalian toxicity tests. Such toxicity equivalency factors are not likely to apply to organisms lacking the aryl hydroxylase enzyme system and far removed from mammals phylogenetically.

The potential for toxicity to non-mammalian species is probably not fully described by classification of PCB congeners according to the type of mammalian microsomal enzyme induction (McFarland and Clarke, 1989). Additionally, coplanar congeners are usually present in much lower concentrations environmentally than nonplanar congeners (Bergen et al., 1996). If potential toxicity, environmental prevalence, and relative abundance in animal tissues are used as criteria, the number of environmentally threatening congeners is reduced to perhaps 36 compounds. Of these, 25 account for 50-75% of total PCBs in fish, invertebrate, bird, and mammalian tissue samples (McFarland and Clarke, 1989). Section 2.5.6.3.1 lists the compounds.

2.3.4.3.2. Water Column Toxicity

The following literature summary reviews laboratory PCB toxicity testing resulting from water column exposure within a phylogenetically broad range of marine species, and is meant to provide an introduction to order-of-magnitude concentrations that result in an observed toxicological response. This is often manifested through impaired growth or reproductive success, which could result in population effects. Since several types of toxicity tests, of variable duration, with several species, and different life stages have been reported, the range of effects in terms of PCB concentrations is broad, and may vary greatly within phyla (Figure 2-13). The discussion of water toxicity presented below focuses on information where PCB toxicity was not seen, or resistence was seen, in contrast to the effects data summarized in the figure.

As a general rule, PCB concentrations between 1 and 10 ppb cause a noticeable decrease in the total biomass as well as cellular size in phytoplankton cultures. Resistance to PCB toxicity has been demonstrated by diatoms exposed to 10-30 ppb PCB for 30 days (Cosper et al., 1987). Resistance was retained for 2 years by the culture, despite no additional PCB exposure. Differences in tolerance to PCBs were noted between oceanic and estuarine isolates; estuarine isolates are typically more resistant to PCB exposure.

Although the fathead minnow (*Pimephales promelas*) is a freshwater species, some recent data have appeared which show a lack of toxicity upon exposure to di-ortho substituted PCB congeners. The effects of five di- ortho polychlorinated biphenyl congeners numbers 52, 101, 138, 153, and 180 on survival, growth, and reproduction in the fathead minnow were determined (Suedel et al., 1997). Previous studies have shown that di-ortho substituted PCB congeners such as numbers 105,118,128,138, and 153 are inactive in fish. PCB congeners that are ortho substituted have much lower binding affinities with the arylhydroxylase (Ah) receptor, which correlate with a reduction in their ability to induce benzo(a)pyrene hydroxylase, and ethoxyresorufin-O-deethy-lase (EROD) activity. Therefore the potential for induction of carcinogenic processes is reduced. Fathead minnows were exposed to one of two nominal concentrations, 2.5 micrograms per liter and 25.0 micrograms per liter for each congener for a total of 13 weeks under flow-through conditions. At termination, survival was 92.6-100 percent in all PCB congener treatments. The PCB congeners tested had no significant sublethal effects on reproductive success within the concentration range examined. *Pimephales promelas* accumulated substantial amounts of all PCBs, with tissue concentrations ranging from 13-183 mg per kilogram wet weight at termination in PCB exposed fish (Suedel et al., 1997). These tissue residues are one to several orders of magnitude greater than those reported for these congeners in aquatic biota previously collected in industrial waterways of the great Lakes.

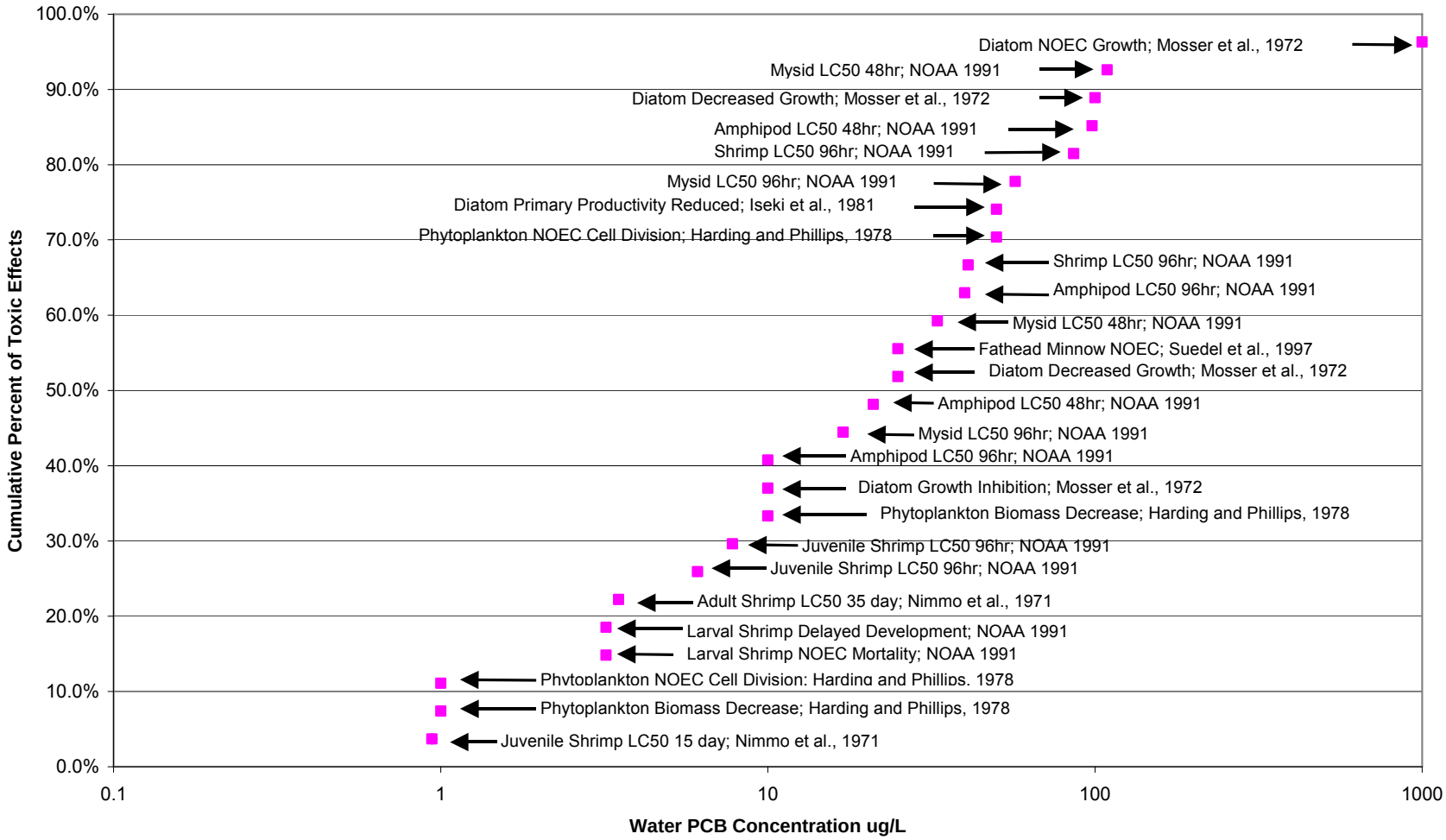


Figure 2-13. Aquatic (water column) PCB toxicity (the PCB concentration reported may be as an Aroclor equivalent, or the sum of several congeners).

2.3.4.3.3. Sediment Toxicity

Similar to aquatic toxicity testing, several types of sediment toxicity tests have been reported in the literature, using several species at different life stages, and varying test durations. Correspondingly, the range of PCB concentrations causing effects relating to sediment exposure is broad, as is the case with aquatic toxicity literature values. The studies, summarized in Figure 2-14, used natural or spiked sediments in a laboratory setting, with results generally reported on a dry weight basis, and are not normalized to total organic carbon content. Other studies have reported results on a TOC normalized basis and are discussed below.

In a study using spiked sediments, no polychaete mortality was noted at 10 or 20 days at 0.28-9.35 ppm dry wt total PCBs (low doses), or at 27.4 ppm total PCBs (high dose). All concentrations were normalized to % TOC (Murdoch et al., 1997). With natural sediments, no significant mortality was seen at 0.36-31.8 ppm dry wt (TOC normalized).

An estimated probable effects concentration in sediment of 0.592 ppm dry weight, and 30.4 ppm, on an organic carbon basis, was determined from the literature and weight of evidence for total PCBs in Southern California Bight sediments with polychaetes (Chapman, 1996). However, Total PCB NOEC values of 1.07 ppm dry wt and 36.6 ppm (organic carbon basis) were determined from polychaete life cycle tests (larvae-emergent juveniles). The author stated that since the probable effect concentrations were designed to provide sediment values above which toxic effects might occur, the close comparison with the NOEC values indicated that the sediment effect concentrations were appropriate.

Sea urchin fertilization efficiency was significantly reduced at 5 and 10 ppm Aroclor 1254 in filtered seawater and 500 mg/L acetone (Adams and Slaughter-Williams, 1988). However, solvent co-toxicity effects of acetone on PCB uptake have been noted elsewhere at concentrations as low as 10 µl/L (Mac and Seelye 1981). In a recent study the sea urchin, *Lytechinus pictus*, was exposed to marine sediments spiked with PCB congener 47 at dry weight concentrations ranging from 0.56 to 22.5 ppb for 35 days. Embryos developed normally at greater than 90% in all test groups, and no significant differences were seen (Schweitzer et al., 2000).

In general, Dexter and Field (1989, cited in NOAA, 1991) found for sediments and PCB toxicity that lethal and sublethal effects and apparent thresholds were reported in the range of 0.1 to about 4.0 ppm dry wt. Median concentrations for numerous tests fall within the range of 0.1 to 1.0 ppm dry wt. Additionally, PCB toxicity decreases dramatically with increasing concentrations of TOC. The concentration range of 0.1-1.0 ppm dry wt apparently has not been approached or exceeded at the offshore Southern California Islands (NOAA, 1991). Long et al. (1995) have determined that the range of sediment PCB concentrations associated with adverse effects is 0.0227-0.18 ppm dry wt. The effects range low (ER-L, the 10th percentile) is 0.0227 ppm dry wt, and the effects range median (ER-M, the 50th percentile) was 0.18 ppm dry wt. This study also pointed out, however, that there was a weak relationship between the total PCB concentration and the incidence of effects. It was further stated that the guidelines should be used as informal screening tools, and should not preclude the use of toxicity tests to determine biological effects. The 1995 effects range values are lower than those published in 1990 due to the removal of freshwater data. Collectively, the references mentioned above indicate that toxic PCB effects in sediment might occur within a range of 0.0227-4.0 ppm dry wt. Most recently a consensus based approach for developing sediment effect concentrations for total PCBs has been

reported (MacDonald et al., 2000). In terms of dry weight the threshold effect concentration was determined to be 0.048 ppm, the midrange effect concentration was determined to be 0.47 ppm, and the extreme effect concentration was determined to be 1.7 ppm. This range of values is also similar to those reported in earlier studies cited above. The authors point out that the consensus based sediment effect concentrations are comparable to estimated chronic toxicity thresholds and equilibrium partitioning models.

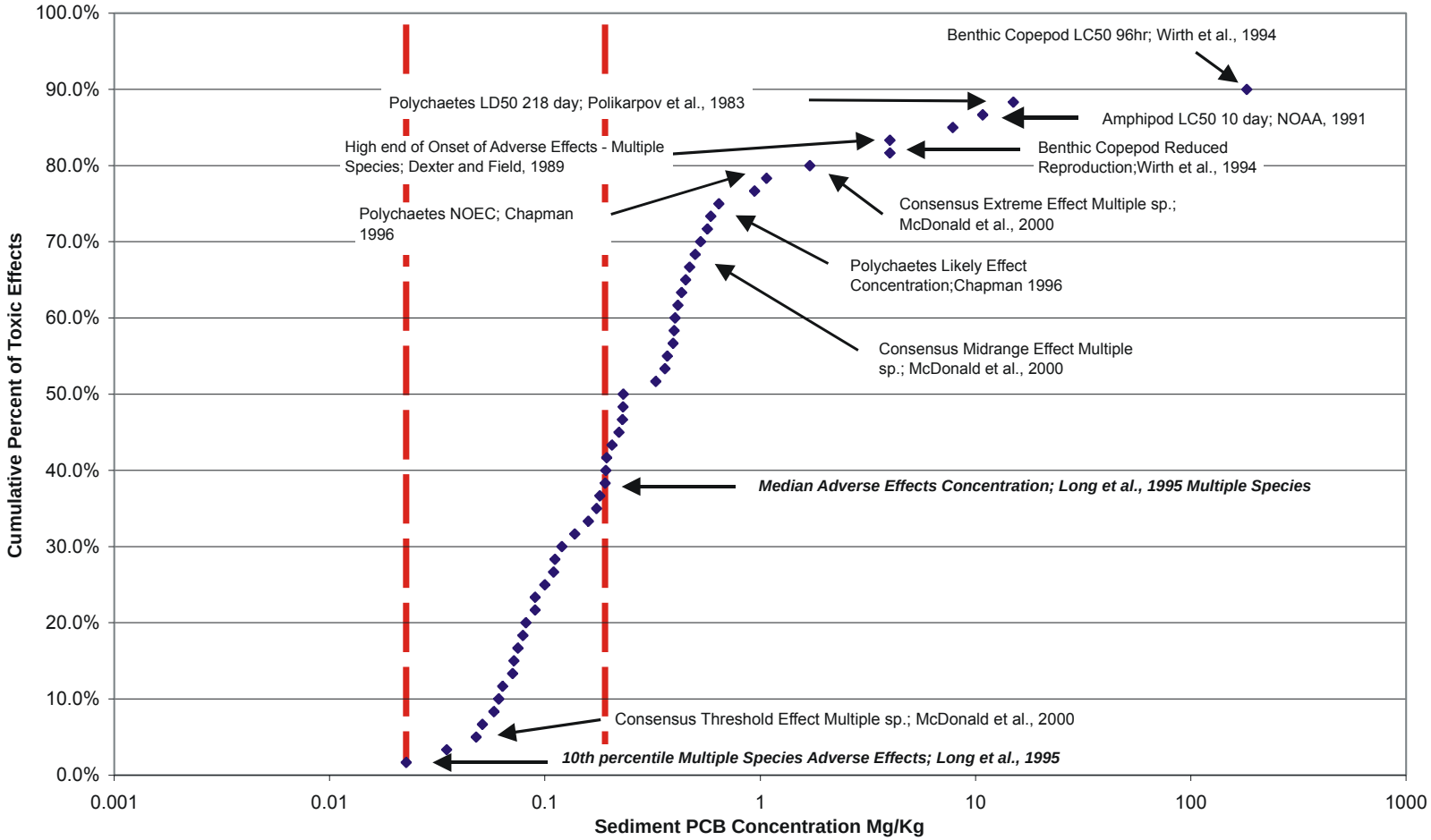


Figure 2-14. Sediment PCB toxicity (the PCB concentration reported may be as an Aroclor equivalent, or sum of several congeners). The majority of effects levels reported is from large data sets or from multiple studies and thus represents a derived value such as a mean or median (e.g., PCB toxicity data of Long et. al., 1995; note their individual data below the 10th percentile value is not available). Other literature data are also included as individual observations. Hence the cumulative distribution presented is a collective summary of studies, some of which may represent a better estimate of toxic effects than others.

2.3.4.4. Bioaccumulation

The term "bioaccumulation" is used to define PCB concentration per unit tissue taken up from the water or sediment by a species. Data may be in terms of unit tissue wet weight, dry weight, or specific tissue (e.g., muscle, liver) weight normalized to lipid content (wet weight divided by percent lipid content). Bioaccumulation is often species specific and can depend on the species, lipid content, age, growth rate, sex, and reproductive condition (Dexter and Field, 1989). Bioaccumulation has been described by such approaches as the equilibrium partitioning theory, which predicts that, when at a steady-state, a contaminant is at equilibrium with the organic carbon content of the environment and the lipid content of an organism, independent of the organism's uptake route. However, it has been recently reported that the equilibrium partitioning theory may underestimate bioaccumulation of hydrophobic organic compounds such as PCBs by selective benthic suspension and deposit feeders (Gunnarson and Skold, 1999).

2.3.4.4.1. Pelagic uptake

PCBs are found in virtually all species. In the water column, bioaccumulation of PCBs may occur via respiration, dermal sorption or indirectly through consumption of contaminated food (Dexter and Field, 1989). Small organisms such as phytoplankton concentrate PCBs directly from water. PCB uptake rates in laboratory cultures of grazing and non-grazing protozoa were compared, and indicated that the dominant pathway of chlorobiphenyl uptake was diffusion for these organisms, and not ingestion (Kujawinski et al., 2000). Zooplankton may also concentrate PCBs directly from water, but uptake is primarily from ingestion as in higher trophic levels. No accumulation of PCBs (sum of 3 congeners; 101, 180, and 209) was observed in two different zooplankton size classes relative to ambient seawater exposures from 10-20 pg/L (Hargrave et al., 2000). In another recent study with a herbivorous copepod (*Calanus hyperboreus*), it was concluded that persistent organic pollutants such as PCBs do not biomagnify (Fisk et al., 2001). PCB concentrations (sum of 89 different congeners) ranging from 15.66 to 33.8 ppb dry weight were reported in the specimens collected. The lower chlorinated congeners were most frequently found in tissue. The authors commented that higher chlorinated congeners are seldom seen in marine zooplankton or seawater.

Some evidence has recently appeared that PCB concentrations in zooplankton have decreased over an approximate 20-year prior period in marine zooplankton samples collected from 1993 to 1994 at 25 stations along the East Coast of Newfoundland and Labrador (Ray et al., 1999). Zooplankton were principally composed of calanoid copepods, with some cyclopoid copepods, free-swimming tunicates, and arrow worms. Total mean PCBs in ng/g wet weight were 0.667, and 85.7 ng/g lipid weight. The concentrations of total PCBs in zooplankton samples were low in comparison with other data in the literature for marine zooplankton, reported much earlier and well below the 4-450 ng/g wet weight range reported by others in zooplankton from the North Atlantic zooplankton (Ray et al., 1999).

Algal uptake of Carbon-14 PCB 77 was studied in a closed continuous flow system for an experimental period of 14 days (Moy and Walday, 1996). Algae (*Fucus* sp.) were exposed to the toxicant and uptake was rapid and significant. PCB 77 concentrations varied between 0.1 and 0.7 µg/L, which is in order of magnitude lower than previously reported lethal levels for Aroclor 1242. Algae exposed to PCB 77 reached the steady state level within 24 hours. Mussels (*Mytilus edulis*) were also tested and showed an approximately linear uptake rate with no observed steady state over the course of the experiments. Organisms moved to a clean environment showed no

reduction in PCB 77 content. The results indicated a steady state accumulation averaging 32 µg PCB 77 per kg wet weight algae when exposed to 0.5 µg PCB 77 per liter. The bioconcentration factor was estimated to be 64. The content of PCB 77 in seaweed was measured on whole plant tissue. Therefore absorption in the plant and adsorption onto the mucous surface of the seaweed was not quantified. More recent data for PCBs in Antarctic algal tissues has been reported (Montone et al., 2001). The total PCB concentration (sum of 12 congeners) measured in samples of *Desmarestia* sp., a macroalgae, ranged from 0.46 to 3.86 ppb (dry weight). The most abundant of the measured congeners were 52, 101, 110, 138, and 153.

A summary of literature values documenting PCBs in pelagic and benthic species tissue samples, shown in Figure 2-15 gives some indication of field concentrations in whole body and muscle tissues, and may be compared to the 2 ppm wet weight U.S. Food and Drug Administration (FDA) action levels for PCBs in fish and shellfish (Boyer et al., 1991). Only two values exceed the FDA limit (10,000 ppb dry weight using a conversion factor of five from wet weight to dry weight). Concerning the lower values found, only a single data point was found for a deep-sea fish (rattail). The muscle tissue PCB concentration reported was2 ppb dry weight.Much of the literature data is reported as lipid-normalized, or from other tissues such as liver. The data presented in Figure 2-15 are not intended to be a complete summary of tissue PCB concentrations in marine species, but rather a comparative summary of tissue PCB concentrations from open-ocean and deep-sea specimens. Where some reports presented both shallow and deep-water collections, we selected the deeper samples for inclusion in Figure 2-15. It is clear from the data presented in Figure 2-15 that tissue PCB concentrations within the included phyla vary over a wide concentration range. Echinoderms, mollusks and fish exhibited the full range of PCB tissue concentrations, while crustacean tissue PCB data were found only in the upper 50th percentile of concentrations.

A broader summary of PCB concentrations from several tissues from several phyla reported as either wet weight, dry weight or lipid normalized is presented in Appendix C. Some broad ranging data such as NOAA's national mussel watch study (tissue PCB grand median) and National Status and Trends median for fish liver tissue are also included in Appendix C, for comparative purposes. Concerning coastal species, it was noted that in flatfish PCBs in tissues decreased dramatically with distance from shore (NOAA 1988). Recent results from the National Benthic Surveillance Project, a complement of NOAA’s National Status and Trends Program, have shown that the concentrations of PCBs in white croaker livers were about 10 times higher than in sediments from the area were the fish were caught. Highly significant correlations were found between concentrations of PCBs in sediment and levels of these compounds in fish livers of all target species combined (Brown et al., 1998). A Mussel Watch survey was conducted in 1988-1989 along the Mediterranean coast of France and Italy. Concentrations of PCBs, expressed by reference to Aroclor 1254, in mussels collected in 1988 to 1989 averaged 527 ng/g dry weight with a range of 50-3,500 ng/g dry weight. PCBs measured in mussels collected in 1973-1974 at the same stations averaged 2,430 ng/g dry weight and ranged from 271-6,578 ng/g dry weight. This reflects a decrease by a factor of 4.5 in 15 years and is in agreement with the gradual cessation of PCB production in the 1970s and 1980s (Villeneuve et al., 1999). This mean PCB value of 527 ng/g approximates 85% of the literature values summarized in Figure 2-15. Note that echinoderms, mollusks and fish exhibited the full range of PCB tissue concentrations, while PCB levels for crustaceans were documented only in the upper 55th percentile.

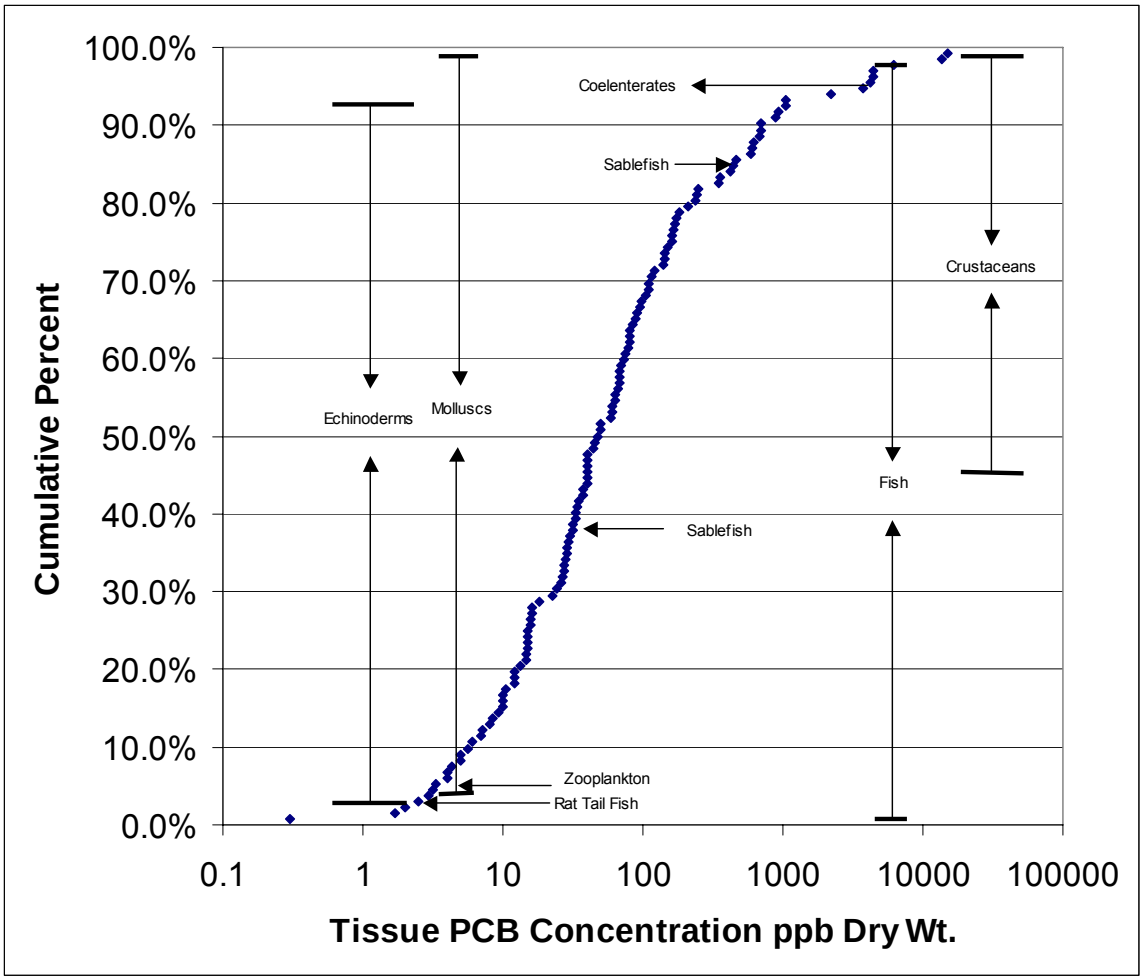


Figure 2-15. PCB concentrations in marine species whole body or muscle tissues (wet weight values were converted to dry weight by a multiplication factor of either 4 or 5 depending on the phylum). The PCB concentration may be as an Aroclor equivalent, or as the sum.

Additional information concerning PCB concentrations in other tissues, or reported as lipid normalized is discussed further below. The data cited in this summary has referred to lipid normalization where the bulk lipid in a given tissue sample has been used as the lipid factor. A recent study (Bergen et al., 2001) has indicated that lipid classes such as total triglycerides or total non-polar lipid fractions can give better correlations with total PCB concentrations. The authors suggest that the standard approach of normalizing tissue PCB concentrations to total lipid may not be appropriate, particularly when the species has a relatively low lipid content (less than 6% in their study). However, if this approach is adopted, then it will be necessary to measure lipid fraction compositions in future studies.

Comparisons of PCB concentrations in fish tissues may not provide clear correlations with sediment PCB concentrations because of fish mobility, particularly for fish living close to or on the bottom. In a National Benthic Surveillance Project, bottom fish (English sole, *Parophrys vetulus*) from two sites, one having 50 times more PCB in the sediment than another (330 ppb compared to 6 ppb) had similar liver PCB concentrations (Malins et al., 1986). However, some association with sediment and tissue PCB concentrations has been reported. Varanasi et al.

(1993) noted that statistically significant correlations between PCB concentrations in sediment and liver tissue of several benthic fish (sole, turbot, flounder, bass, etc.) indicate the bioavailability of sediment-sorbed PCBs. Deep-sea fish, collected by Tucker trawl in the Gulf of Mexico, showed considerable variation in tissue PCB levels among different species taken at the same depth. The same species, of similar size, collected at different stations, were observed to have PCB concentration differing by several orders of magnitude (Baird et al., 1975). Eight species of deep-sea fish caught at various depths off the western coast of Greenland at depths from 200-2,100 m exhibited low to moderate PCB contamination. Hepatic levels of total PCBs (sum of 19 individual PCB congeners) ranged from 110 ng/g lipid weight in Wolf fish to 1,156 ng/g in blue hake (lipid content ranged from 0.3-4.5 percent in muscle tissue). The blue hake is believed to be a benthic feeder. No simple relationship was found between PCB contamination and depth range of the investigated species (Berg et al., 1997). All specimens were captured near the bottom.

Accumulation of highly chlorinated congeners appears to be common in deep-sea fish. This may be explained by particle bound transportation from the surface of the ocean to the deep sea. PCB congeners with the highest chlorination are relatively more adsorbed to suspended solids than PCB congeners with low chlorination. Additional congener accumulation data in deep-sea fish has recently been reported (Porte et al., 2000). Total PCBs as the sum of seven congeners measured in fish caught from 1,500-1,800 m in the Northwestern Mediterranean Sea ranged from 2.5 to 10 ppb wet weight. These concentrations are similar to those seen in relatively clean coastal areas of this region. The highly chlorinated congeners 138, 153, and 180 accounted for 49 to 50 percent of the total PCBs detected in two species (*Coryphaenoides guentheri* and *Bathypterois mediterraneus*) of three species caught, and 38 percent in the third species (*Lepidion lepidion*). The highly chlorinated hexa- to octochloro isomers accounted for 80 to 92 percent of the total detected PCBs. Particle bound transport and selective sorption to particles by higher chlorinated congeners were also given as likely reasons for greater exposure in the deep sea environment in this study. Although it is known that metabolic rates of deep sea fish decrease with depth due to factors such as low temperatures, low food availability, and poor locomotory abilities, enzyme indicator measurements with these deepsea species suggested that enzymes systems studied (Cytochrome P450, Glutathione S-transferase, and antioxidant enzymes) were as catalytically efficient as those of shallow water species. Recent data from Morid cod (*Mora moro*) collected at an approximate depth of 1,000 m in the northwest Mediterranean Sea exhibited total PCB values (sum of 22 congeners) of 24 ppb wet weight in muscle tissue (Sole et al., 2001). The authors stated that compared to surface fish profiles, the PCB patterns indicated a relatively higher contribution of heavier components such as the hepta- and octochlorinated PCBs. No clear bioaccumulation dependence on fish weight or size was seen in gill, digestive tube or liver tissue when normalized to lipid content. The PCB concentration in muscle tissue did decrease with fish size, which may be due to a dilution effect with increasing body weight.

It has been recently reported that the deep sea biota (>800 m) show significantly higher PCB burdens (10x) as compared to surface-living species (0-200 m) of the same region in the North Atlantic. In contrast to the North Atlantic, the differences between PCB burdens in surface and deepwater fish from Monterey Bay Canyon is only about a factor of three to four (Froescheis et al., 2000). In the North Atlantic samples total PCBs reported as four times the sum of seven indicator congeners (28, 52, 101, 118, 138, 153, and 180) were 1,180 ppb lipid weight for brittle star (*Amphiura archystata*) whole body samples, and were 2,080 ppb in Grenadier fish fillets (*Coryphaenoides armatus*) caught at 2,900 m. Brittle star (*Amphiura archystata*) PCB

concentrations were nearly 5,000 µg per kg normalized to lipid content in samples from the Monterey Canyon in 1995 (Looser et al., 2000). Tissue samples from sea stars (*Asterias rubens*) collected at 24 stations in the North Sea have exhibited PCB values (sum of seven congeners as ppb lipid normalized) ranging from 101 to 1,001 ppb (den Besten et al., 2001). Tissue PCB values in the open North Sea ranged from 200-400 ppb.

Muscle tissue of yellowtail flounder, and winter flounder, collected from an offshore location off North Eastern Newfoundland Canada were analyzed for PCBs as the sum of 22 congeners. Offshore total PCB values averaged 2.38 ng/g, as wet weight. As lipid weight, the average was 369 ng/g (Ray et al., 1998). Data from mesopelagic fish (Myctophids) collected in the western north pacific at depths of 50-600 m expressed as lipid weight (sum of 117 congeners) from whole body homogenates were similar to the lipid weight values from Newfoundland offshore data cited above. These values ranged from 20 to 370 ng/g lipid weight. Of the 10 values reported, four ranged from 200-370 ng/gram (Takahashi et al., 2000). Additional recent tissue data from mesopelagic fish collected between 350 and 450 meters has been reported from the western Mediterranean Sea (Garcia et al., 2000). Total PCB values (as the sum of seven congeners) were reported as 2.1, 166, 20.3, and 5.1 ppb wet weight for muscle, liver, gill, and digestive tract tissue, respectively, from *Lepidorhombus boscii*, a benthic species. Total PCB values were 1.0, 327, 2.9, and 8.8 ppb wet weight for muscle, liver, gill, and digestive tract tissue, respectively, from *Phycis blennoides*, a demersal species. Both species exhibited depletion of the lower PCB congeners, particularly those containing meta and para vicinal H-atoms, in liver tissue. These types of congeners are more easily metabolized. Data from midwater fish (myctophid and hatchet fish) collected at 300-1500 m in the western North Atlantic reported low PCB values of 2.3-5.7 ppb (wet wt as Aroclor 1254) in whole body minus liver samples (Stegeman et al., 2001). These concentrations are similar to the muscle tissue values reported by Garcia et al. (2000) above.

Dover sole (*Microstomus pacificus*) and Sablefish (*Anoplopoma fimbria*) collected at 500-1000 m (continental shelf) in the vicinity of the Farallon Islands disposal site had liver wet weight PCB concentrations up to 7 ppm. Sediment concentrations of 190 ppb dry weight were collected at the disposal site (Melzian et al., 1987). The liver concentrations in the Dover sole are comparable to those found in Dover sole collected off Palos Verdes, CA; site of the Hyperion waste water outfall (Sherwood et al., 1980). Additional data concerning PCBs in Sablefish has been reported at two sites off the southern California coast near Orange County at depths of approximately 300-500 m (Cross, 1984, and Thompson et al., 1984). Total PCB concentrations in muscle tissue ranged from 8-110 ppb wet weight, and from 67-250 ppb wet weight in liver tissue. Some of the fishing sites were approximately 50 km from the coast; while not as far as the ex-AGERHOLM site (approximately 190 km), this distance places these specimens in the general area of the Southern California Bight. Concentrations of 24 PCB congeners were recently reported in grouper and shark liver tissue from the northwest African Atlantic. Total PCBs ranged from 39.4-4,723 ppb wet weight (Serrano et al., 2000). Congeners 138 and 153 were the dominant congeners found. The order of abundance of the most toxic congeners was 77>126>169.

Concerning PCB congener bioaccumulation, for a given species increased exposure to PCBs did not lead to increased relative concentrations of non-ortho-substituted (dioxin-like) congeners. Limited data on congeners 77 and 126 in sediment, sea urchins, or for horn sculpins suggested

that, with increasing trophic status, these congeners were diminished rather than enriched relative to the total PCB concentration (Bright et al., 1995).

2.3.4.4.2. Benthic uptake

For the case of PCB uptake by benthic organisms from sediment, PCB sediment concentrations were found to be the most important influence on uptake. Bioavailability tends to be directly related to solubility of the compound and the sediment grain size, and inversely related to sediment organic carbon concentration. Sediments highly contaminated with PCBs are a source of uptake to indigenous organisms, which may accumulate PCBs to high levels, and are probably more important than water concentrations in determining tissue burden. However, information is lacking regarding toxicity to organisms that have accumulated PCBs to higher-than-background levels. A linear relationship between body concentration and log sediment concentration is generally found (Shaw and Connel, 1982). Polychaete worms have been shown to desorb two thirds of available hydrophobic organic contaminants such as tetrachlorobiphenyl from ingested sediment within one minute in their gut via gut-fluid surfactant compounds (Aherns et al., 2001). A study examining the relative importance of sediments and water as a source of PCBs to polychaete worms found that sediments contribute the bulk of PCBs taken up by exposed worms and perhaps other infaunal species (Fowler et al., 1978). Sediments dredged from 150 m were spiked with PCBs to concentrations of 9.3 ppm and 80 ppm. Polychaete bioconcentration factors of three to four were observed for sediment exposure. The worms exposed to 80 ppm PCB did not survive a 90-day test period. When the experimentally derived concentration factors were applied to ambient sediment and water PCB concentrations in waters off the coast of Monaco, the authors calculated that 89-99% of the total uptake would be derived from sediment exposure, depending on the seawater exposure concentration. The same concentration factors predicted that 85% of the accumulated PCB would be due to sediment exposure when applied to open Mediterranean sediment and water concentrations (1 ppb PCB in sediment and 0.8 pptr PCB in water), again indicating the dominant role of sediments as the source of PCBs.

Another study reported accumulation of PCBs in freshwater fish from a sediment-zooplankton exposure route in large experimental ponds. Initial sediment PCB concentrations were 2.7-3.8 ppm PCB dry weight. Maximum PCB concentration in fish adipose extracts was 400 ppm over a 14-month exposure period, yielding a bioconcentration factor of approximately 100. No mention of toxic effects was made, however (Larsson, 1986).

Body burdens in two benthic copepods, *Amphiascus tenuiremis* and *Microarthridon littorale* were determined using Aroclor 1254 (Wirth et al., 1994). Maximum levels in *A. tenuiremis* (0.39 ng/µg dry wt, =390 ppm) and *M. littorale* (0.23 ng/µg, =230 ppm) were reached after 8 days of sediment exposure at a concentration of 83.3 ng/mg dry wt. These values agree with other uptake values in the literature for *A. tonsa* and *A. clausi* exposed to 10 ng/L PCBs for 36 hr (0.248 and 0.224 ng/µg dry wt, respectively).

Fiddler crabs and pink shrimp exposed to 61.0 ppm PCB in natural sediment from Escambia Bay, Florida for 30 days accumulated PCBs ranging from 80-240 ppm wet weight, respectively. At an exposure level of 1.4 ppm PCB in natural sediment, no accumulation was seen in the fiddler crabs, while the pink shrimp accumulated 0.2 ppm PCB in their hepatopancreas tissue. Toxic effects during the testing period were not mentioned (Nimmo et al., 1971a).

Uptake of tetrachlorobiphenyls was studied in the laboratory using benthic microcosms and the brittle starfish *Amphiura filiformis* and *A. chiajei*. Infaunal brittle stars may accumulate

contaminants either by direct contact with contaminated sediment particles, or via respiration in interstitial and overlying water, or by ingestion of contaminated food. Differences in tetrachlorobiphenyl concentrations were still significant after normalization to lipid content, suggesting that selective feeding behavior (i.e., response of increased feeding to enriched food availability) rather than equilibrium partitioning was the cause of the increased tetrachlorobiphenyl burden (Gunnarsson and Skoeld, 1999). In brittle stars collected in the natural environment, total PCBs in sediment and brittle stars were approximately three times higher than samples collected from a coastal station than from an offshore station. Biota sediment accumulation factors determined from the laboratory and field exposures ranged from 1.5-5.9. The study also suggested that eutrophication processes, such as increased phytoplankton production, might contribute to increasing the accumulation of organic pollutants in benthic sediment ingesting fauna. It has been shown that 8-15 percent of the PCB burden in sand dabs from the Bay of Seine could be explained by ophiuroid consumption (Gunnarsson and Skoeld, 1999). This suggests that *Amphiura* communities may play an important role in the accumulation, remobilization and transfer of PCBs and other sediment associated contaminants to higher trophic levels. Such a condition may also exist at the ex-AGERHOLM site due to the ubiquitous distribution of brittle stars in benthic communities.

Sea urchins (*Lytechinus pictus*) fed radio-labeled PCB congener 47 spiked sediment at 0.56 to 22.47 µg/g dry weight for 35 days accumulated this congener by factors of 10.1 to 13.3 in eggs, and by factors of 1.6 to 2.8 in gonadal tissue (Schweitzer et al., 2000). This degree of bioaccumulation was considered significant at the environmentally realistic sediment PCB concentrations tested.

The bioaccumulation upon exposure to mean PCB concentrations of 3.6 ppm in river sediment on bottom-feeding sand worms (*Nereis virens*), a clam (*Macoma nasut*a), and the grass shrimp (*Palaemonetes pugio*) has been reported (Rubinstein et al., 1990). Sandworms accumulated 2.9 ppm PCB (180 days), while controls accumulated 0.7 ppm. Clams accumulated 0.9 ppm (120 days), while shrimp accumulated 0.8 ppm (28 day exposure). Mortality during the testing periods was similar in controls and exposure treatments (no effect). In another study with these species, sandworms (*Nereis virens*), clams (*Macoma nasuta*), and grass shrimp (*Palaemonetes pugio*) were exposed to Passaic River sediment (434 ng/g dry wt. for 10 congeners) for 180 days under flow-through conditions in a laboratory. Although final contaminant concentrations were highest in the sandworms, accumulation factors were generally higher for the clams and shrimp and lower for sandworms. Clams showed preferential accumulation of lower molecular weight PCB congeners, which may be due to the very low lipid content in this species. Specific congener concentrations ranged from 10 ng/g dry wt (PCB 206) to 80.7 ng/g (PCB 153) and mortality did not exceed 22.5% with clams. In general, PCB 153 showed the highest accumulation factor (AF) values (1.4-2.08). The more chlorinated congeners 194, 206, and 209 showed small AFs relative to those of other congeners suggesting that the highly chlorinated compounds are not as effectively accumulated as the less chlorinated congeners. The authors noted that mass and lipid content of the organism are two important factors in the kinetics of PCB uptake. Studies have indicated that arthropods, annelids and mollusks do not possess an aromatic hydrocarbon (Ah) receptor, the presence that might result in the expression of “dioxin like” toxicity (Pruell et al., 1993).

Clam sediment accumulation factors for hexa-, hepta-, and octochlorobiphenyls were estimated at an intertidal marsh, the Turtle River Brunswick estuary located in southern coastal Georgia

where Aroclor 1268 contamination was present. However, accumulation factors for PCB congeners with octanol-water partition coefficients (K_{OW}) >6.5 were only in the range of 0.07-0.88, indicating that their uptake was hindered by such factors as steric inhibition, contamination levels, non equilibrium conditions and stronger affinity to sediment organic carbon. It was also shown that biota sediment accumulation factors (BSAFs) were negatively correlated with log K_{OW} of super hydrophobic PCB congeners also indicating less availability of the higher chlorinated PCBs (Kannan, 1999). In this study, total PCB concentrations in surface sediments and clam tissues were 430 µg/g dry weight and 17.9 µg/g wet weight, respectively. Based on lipid and organic carbon normalized total PCB concentrations in the clam tissue and sediments the BSAF was estimated to be 0.21, much less than a tentative guideline limit of 1.7 suggested for benthic organisms by McFarland and Clarke (1988) based on equilibrium partitioning as the only bioaccumulation factor. Filter feeding clams exhibited lesser BSAFs for PCB congeners than deposit feeders and predators, suggesting that despite the relatively high concentrations of PCBs in sediments, bioavailability of the more chlorinated congeners present in the sediment may be less for filter feeding bivalves. Filter feeding clams ingest PCBs primarily from overlying water, which may not be at equilibrium with the sediments due to the tidal cycle at this intertidal location. Therefore, non-equilibrium conditions could also have contributed for the reduced BSAFs.

More information concerning PCB accumulation in clams has recently been reported from New Bedford Harbor (Burgess and McKinney, 1999). The filter feeder *Mulinia lateralis* and the deposit feeder *Yoldia limatula* were studied using sediment samples collected from a station in upper New Bedford Harbor during September 1995. Uncontaminated sediment from Long Island Sound was used as a reference. Samples were analyzed for 23 PCB congeners. Differences in feeding strategies between species were indicated, especially early in the exposures when *Y. limatula* was accumulating far more PCBs than *M. lateralis*. This differential uptake was probably a result of direct sediment ingestion, and is in agreement with other studies where it has been observed that deposit feeders accumulated more contaminants such as PCBs than filter feeders.

In highly contaminated New Bedford sediments, PCB concentrations of 10,000 ppm resulted in body burdens of 49 ppm dry weight and 86 ppm wet weight in mussels and lobster, respectively (Farrington et al., 1985). Bioaccumulation of PCBs from sediment has not been found to approach rates in water where PCB is more bioavailable, but it should be kept in mind that PCBs in sediments can reach much higher concentrations than in water and lead to high body burdens, (Neff and Breteler, 1983).

2.3.4.5. Biomagnification/Food Chain

The term “biomagnification” is used to define an increase in PCB concentration from one trophic level to another. Little data are available for natural, open water communities and we have found none for deep sea benthic communities. Some evidence is presented from shallow, benthic communities. PCB levels in birds and mammals are not discussed in this report, since these animals are not part of the food chain found at SINKEX depths and are not felt to be significantly exposed to PCBs released at these depths. Similarly, human health issues are not raised here, but are discussed in a separate Human Health Risk Assessment. Few fisheries operate at proposed SINKEX depths, except perhaps for the Sablefish fishery examined in the Human Health Risk Assessment. PCBs released at depth, as discussed previously, probably do

not enter surface waters either through advection or diffusion and remain trapped in the benthic boundary layer. As discussed below, there is no evidence that vertical migration of marine organisms spans these depths, and hence no evidence of a natural biological link exposing SINKEX PCBs to surface communities.

Concerning the relevance of deep sea food chain relationships to SINKEX, the studies discussed below collectively suggest that PCB biomagnification in deep sea food chains may not occur. Several factors such as feeding strategies, biochemical adaptations to depth, and differences in lipid and lipid types may in part be responsible for the lack of biomagnification in the deep sea. These factors and others are discussed in the studies below. Additional information concerning trophic transfer of PCBs in shallow waters, and in some freshwater systems, is also mentioned in order to present a broad view of PCB biomagnification and food chain issues.

A bacterial-microplankton study using three polychlorinated biphenyl PCB congeners (3, 52, and 153) was recently reported (Wallberg et al., 1997). In oceanic waters bacterial biomass may even exceed the phytoplankton biomass. Of the PCB fraction that initially adsorbed to particles, 60-100 percent was associated with the bacterial fraction and 0-5 percent with the microplankton fraction. Approximately 75 percent of the increase in bioconcentration of hexachlorobiphenyl in the microplankton fraction can be explained by bacterial grazing. The results indicate the microbial food web can contribute to a rapid uptake of higher chlorinated PCBs in grazing organisms, particularly in oligotrophic ecosystems were the bacterial biomass dominates. A later study using congener 153 indicated that the transfer rate through a microbial food web was coupled to the carbon flux, and may be less efficient in the transfer of PCBs to higher trophic levels during nutrient-limited periods. When nutrient input increases, a shift toward increased food web diversity with larger planktonic organisms and more efficient carbon transfer is seen (Wallberg and Andersson, 2000). Wallberg et al. (2001) have recently reported that trophic transfer can be a dominant PCB uptake pathway in heterotrophic microplankton such as ciliates, exhibiting higher PCB concentrations than in phytoplankton.

However, lower food chain levels (plankton and invertebrates consumed by fish) do not biomagnify PCBs to the extent seen in upper food chain species such as mammals and birds (Harding, 1986; Shaw and Connell, 1982). The consumption of contaminated food is the major source of xenobiotics for predating birds and mammals. Additionally, the direct uptake of chemicals from the environment (i.e., from water, sediment and air) is only of minor relevance for upper food chain species (Nendza et al., 1997).

In a microplankton-euphausiids-shrimp food chain in the Mediterranean no evidence of trophic magnification of PCBs was found. Most predator-prey biomagnification factors are close to or greater than unity (Harding, 1986). In open Atlantic upper water column fish, no PCB biomagnification was apparent on a wet or lipid weight basis and, in fact, the mixed plankton assemblages captured by net had higher PCB concentrations (average 200 ppb) than any fish. Plankton hauls rich in phytoplankton reached 1 ppm (Harvey et al., 1973). Dry weight concentrations of zooplankton taken in North Atlantic shelf and slope waters off the U.S. had a median PCB concentration of 150 ppb. Zooplankton lipid PCB concentrations (median value of 60 ppm) were comparable to the PCB levels in fish caught in Long Island Sound (Risebrough and Vreeland, 1972). Trophic transfer and biomagnification of PCBs did not appear to occur on contact with contaminated sediments, although direct uptake through equilibrium partitioning was thought to explain tissue burdens in the Hudson River (Pierce et al., 1981). A more recent study has also concluded that uptake of PCBs by fish is mainly direct from water, without

significant influence from diet or age for both pelagic and demersal fish. An observation was made that fish tissue from relatively uncontaminated waters (e.g., capelin from Icelandic waters) may contain as little as 10 ng per gram dry weight of total PCBs (Ali et al., 1997). For a linear food chain including fish, however, biomagnification between trophic levels may occur for the more chlorinated congeners (Bruggeman et al., 1984). The sum of 28 PCB congeners was recently measured in whole body samples of copepods (44-46), euphausiids (28-29), amphipods (38-50), and cod fish liver (108-205) as ppb lipid weight (Borga et al., 2001). The predation step from crustaceans to fish indicated biomagnification factors of 2.2 to 5.9 for the sum of nine congeners (28, 31, 47, 99, 105, 118, 138, 149, and 153). The amphipod predator - copepod prey step indicated bioaccumulation by a factor of 1.1. The relatively low biomagnification factors may reflect the ability of these species to eliminate contaminants such as PCBs through their respiratory surfaces (Borga et al., 2001).

In a recent report concerning food-chain biomagnification in a marine pelagic food chain, Harding et al. (1997) state that evidence for biomagnification is contradictory. Several studies are cited where biomagnification has, and has not been observed. In their study Harding et al. (1997) found a ten-fold increase in PCB concentrations when PCBs were normalized to lipid content. Fish had ten-fold more PCBs than plankton. The point is made that it is not clear whether the increased fish PCB content relative to plankton can be explained by trophic accumulation, or increased exposure as a result of the greater longevity of fish compared to plankton. This clearly has implications for other studies as well. The best predictors of PCB contamination in fish in St. Georges Bay were determined to be lipid content followed by size and age (Harding et al., 1997). Size related differences in uptake rate have been demonstrated in bivalve mollusks. Larger individuals exhibit lower tissue PCB concentrations (Gilek et al., 1996).

Polychlorinated biphenyl congener bioconcentration patterns in tissues of marine macrophytes, urchins, mollusks, sea cucumbers, and fishes at Midway Atoll in the North Pacific Ocean have been recently reported (Hope et al., 1998). This study measured the concentration and distribution of 20 PCB congeners (8,18, 28, 44, 52, 66, 77, 101, 105, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, and 209) in marine sediments, surface waters, and tissues of 12 species of marine biota from near shore waters at Midway Atoll. PCB congeners 28, 52, 101, 118, 138, 153, and 180 are among those recommended by the International Council for Exploration of the Seas for assessing marine pollution by PCBs. The 12 species collected can be nominally grouped into four trophic levels: primary producers, herbivorous primary consumers, omnivorous secondary consumers, and carnivorous tertiary consumers. While it is unlikely that all of these species are simultaneously involved in the same food web, some feeding relationships are suggested by field observations. Calculated mean lipid-normalized logarithmic bioconcentration factors for PCB congeners in various marine species ranged from 3.75-6.97. There was no evidence of a consistent pattern of progressive increase in bioconcentration (e.g., biomagnification) of PCB congeners with increasing trophic level in this near shore ecosystem. It was noted that normalizing the measured BCFs to lipid content of the organism reduces the effects of intra-and interspecies variability. Laboratory-derived BCF values would be expected to deviate from field-derived BCF values, depending on the contributions made through food consumption. Only four congeners (153, 170, 180, and 187) displayed a slight upward trend in bioconcentration for primary producers to tertiary consumers. These findings suggest that a general model for trophic level differences in PCB concentrations may not be attainable.

Deep-sea organisms collected in Suruga Bay, Japan showed no consistent trend between organochlorine concentrations and prey-predator relationships. No consistent trend between PCB concentrations and food chain relationships was observed in the deep-sea organisms. Deep-sea fish ranged from 450 (Argentine fish, *Glossanodon semifasciatus*) to 1900 (Cusk-Eel, *Hoplobrotula armata*) ng/g lipid weight (Takahashi et al., 1998).

Less is known for pelagic species living at bathyal depths. Factors such as exposure to sediment, differing life histories, feeding strategies, and biochemical adaptations to high pressures and low temperatures argue caution in extrapolating shallow water food chain data to deep water organisms (Harvey et al., 1974). In a lengthy literature review, Fowler (1990) noted that deep-sea feeding strategies probably differ from those in shallow water and found no evidence of food chain magnification in the deep sea. Similarly, Grassle et al., (1986) hypothesized that differences in lipid levels and lipid types in deep ocean mesopelagic communities may be partly responsible for lack of food chain biomagnification.

A sediment-copepod-fish trophic study has been reported (Dipinto and Coull, 1997). Aroclor 1254 was used in a benthic-based trophic model where sediments, benthic copepods, and juvenile fish were studied. Field collected benthic copepods (*Microarthridion littorale*) were exposed to sublethal levels of Aroclor 1254 in sediments for 96 hrs. Accumulation of PCB was measured in the copepods, and these contaminated copepods were fed to juvenile *Leiostomus xanthurus* (Sciaenidae) in uncontaminated sediments. Copepods exposed to PCB contaminated sediments at 90 µg per gram accumulated PCBs to 326 µg per gram dry weight. Accumulation of PCB in fish feeding in contaminated sediments was five times higher than that in fish feeding on contaminated prey in uncontaminated sediments. Fish fed meals of copepods exposed to Aroclor 1254 contaminated sediments accumulated PCBs from the copepods at a 33 percent transference of PCBs from prey to predator. It is likely that continued feeding on contaminated copepods during the five to seven months the juvenile fish spend in the estuary would have led to a much greater PCB accumulation in fish over time (Dipinto and Coull, 1997).

Analysis of chlorine homolog groups revealed that the fish preferentially accumulated the tetrachlorinated congeners relative to copepods and sediments. The pentachlorinated congeners exhibited the opposite trend: they were accumulated by fish at levels lower than their source groups. The congener patterns in fish reflected the congener patterns in their respective PCB sources, which were either the PCB contaminated sediments or the PCB contaminated copepods. The lowest K_{OW} group (tetrachlorobiphenyl) accumulated more in the fish relative to their PCB sources. These congeners are less hydrophobic, do not associate strongly with sediments, and as a result are more bioavailable than the higher K_{OW} congeners. However, because these congeners are typically more readily metabolized and eliminated, they do not typically accumulate as highly in higher trophic levels as in environmental samples (Dipinto and Coull, 1997).

Some information has recently appeared concerning fish to fish transfer, and fish to mammal trophic level magnification of PCBs. A total 17 PCB congeners were measured in a local marine food chain near Jarfjord northern Norway. The species represented included the lesser sand eel (fish), cod, harbor seal, and gray seal. The data suggested that the bioaccumulation mechanisms at lower trophic levels (fish) depend primarily on physio-chemical factors, such as the water solubility and lipophilicity of the pollutants. At higher trophic levels (seals) the bioaccumulation mechanisms are primarily affected by biochemical factors, such as the metabolic capacity of the organisms. The biomagnification factors determined showed that total PCBs demonstrated a sharp increase in concentrations with higher trophic level. In cod, higher chlorinated biphenyls

constituted higher proportions of total PCBs than in the sand eel. For cod/sand eel PCB concentrations in tissue, the biomagnification factor was 3.2 for total PCBs. For Harbor seal/sand eel tissues, the biomagnification factor was 28.0. The gray seal/sand eel biomagnification factor was 32.2, the harbor seal/cod biomagnification factor was 8.7, and the gray seal/cod biomagnification factor was 10.0 (Ruus et al., 1999).

Although the data are from freshwater species, it has been shown that Lake Michigan coho salmon bioaccumulated various PCB congeners from their food. The retention efficiency for the pentachloro congeners averaged 38 percent, while retention efficiencies for higher chlorinated congeners ranged from 43 to 56 percent. An overall average net trophic transfer efficiency of 49.3 percent was calculated for all 21 congeners studied (Madenjian, et al., 1999).

In another freshwater study, trophic transfer of chlorinated organic contaminants was investigated in an aquatic community composed of zooplankton, benthic invertebrates, and fish from the Detroit River (Russell et al., 1999). Samples were analyzed for PCB congeners 52, 87, 101, 138, 153, and 180. Biomagnification measured as the increase in lipid-based chemical concentrations in predator over that in prey was observed for high K_{OW} chemicals (log K_{OW}>6.3). PCBs 87, 101, 138, 153, and 180 have log K_{OW}s ranging from 6.4-7.4. Accumulation of PCBs in the food web showed a relationship to the trophic level of the organisms, and biomagnification was evident indicating that the process of biomagnification is K_{OW} dependent. Low K_{OW} chemicals (log K_{OW}<5.5) did not biomagnify in the food web, and chemicals with log K_{OW} between 5.5 and 6.3 showed some evidence of biomagnification. Trophic level differences in chemical accumulation in the food web could not be attributed to bioconcentration into increasing trophic levels with increasing lipid levels, as no relationship was observed between trophic position and lipid content of organisms. It was concluded that trophic interactions play a crucial role in the distribution of High K_{OW} chemicals, but not for low K_{OW} chemicals.

A laboratory study using polychaete worms exposed to Passaic River (NJ) sediment for 70 days, and subsequently fed to the American lobster for up to 112 days showed accumulation of 31 PCB congeners by the worms. Lobsters also accumulated some PCB congeners. The non-ortho-substituted congeners 77 and 126 were particularly enriched (6×) in the lobsters relative to the sediment, likely because they were not metabolized (Pruell et al., 2000).

2.3.4.6. Tissue Burden Toxicity

In addition to water and sediment toxicity data, some studies have reported tissue PCB concentrations where toxic effects were also noted. Additionally, some reports have appeared where critical body concentrations and the mammalian dioxin toxic equivalents model are discussed with respect to marine species. These studies are discussed in the following sections.

2.3.4.6.1. Tissue Concentrations

The processes of bioaccumulation (from pelagic or benthic uptake) and biomagnification may both contribute to tissue PCB concentrations, which may be toxic to a given species. Exposure to PCBs in sea stars fed mussel tissue led to reduced steroid metabolism which could lead to reproductive impairment. Sea stars were exposed to PCB levels (26 ppm lipid) comparable to those found in specimens from polluted field sites (8.92-15.87 ppm lipid) (den Besten et al., 1990). In a food chain experiment consisting of algae-mussels-sea stars using Clophen A50, an almost linear PCB concentration increase was seen in sea star pyloric caeca and gonads. PCB exposed female sea stars had significantly lower gonad indices than in unexposed specimens.

The total PCB concentration in unexposed gonads was 2.96 ppm lipid wt, while exposed gonads had 17.3 ppm (den Besten et al., 1990). In this study total PCBs were the sum of eight congeners (52, 70, 87, 101, 118, 138, 153, 180). It was stated that comparable PCB concentrations have been found in other polluted areas, and that this study may be considered relevant for sea star populations in polluted areas. A recent study of regeneration responses with a crinoid echinoderm (*Antedon mediterranea*) showed abnormal cellular and tissue resposes involving limb regeneration following exposure to total PCBs as Aroclor 1260 at 14 ng/L for 14 days. The exposed specimens accumulated total PCBs at 2,257 ppb lipid wt, which the authors reported as not very different from those measured in other filter-feeding invertebrates along the Mediterranean coasts (Candia Carnevali et al., 2001; Candia Carnavali et al., 2001a).

Cultured clam (*Mercenaria mercenaria*) gametes from sites in Long Island Sound having elevated concentrations of PCBs showed significantly lower fertilization and meiotic success than gametes from other Long Island Sound sites. The PCB concentration in Long Island clam gonadal tissue was generally low, ranging from 0.027-0.048 ppm PCB wet weight. Copper and cadmium were also present in samples (Stiles et al., 1991). Hemocytic neoplasia develops in soft-shelled clams and coastal locations throughout the world. The prevalence of neoplasm was higher than background levels when clams were collected from a PCB contaminated site, New Bedford Harbor. PCBs preferentially localized in normal and neoplastic sites, in the ovary and in several other tissues (Strandberg et al., 1998). This study commented that PCBs are not mutagenic, and that it is not known whether PCBs act as tumor inducers or promoters, although recent evidence supports the latter interpretation. Numerous examples of hemocytic neoplasia in clams from New Bedford Harbor were found. These clams were found where there were persistent high levels of PCBs, although concentrations were not reported. The findings show that clam tissues and blood cells accumulate PCBs.

Incubated North Sea whiting (*Merlangus merlangus*) eggs having PCB concentrations of 3-370 ppb wet weight showed low viable hatching success. Considerable scatter in hatching success was noted at low levels, while higher levels consistently showed low hatching success. A threshold value of 200 ppb wet weight was calculated for ovary contamination, above which impairment of reproductive success (viable hatch below 10%) was likely to occur. Artificially inseminated eggs were incubated and developmental effects correlated with PCB residues in the ovaries. PCB concentrations higher than 200 ppb were found in 16% of all specimens. The parameter measured was a “contamination factor” which was a composite measurement of all contaminants found in the tissues, however. The effects of DDT on viable hatch were 10 times as great as those found with PCBs in Baltic herring. The authors commented that the large number of low hatch events seen could not be explained by the simultaneously occurring contaminants measured and suggested that additional contaminants not measured may have been highly influential (Westernhagen et al., 1989). Viable hatching success (50%) was significantly affected at ovary PCB threshold concentrations of more than 120 ppb wet weight in Baltic herring (*Clupea harengus*). However, 94.5% of the specimens examined had less than this concentration in their ovarian tissue. The authors state that other causative agents not measured may also be responsible for the low PCB threshold since other reports cite toxic levels on the order of 15-17 ppm (Hansen et al., 1985).

Viable hatching success was not affected by PCB concentrations of 5.9-44.9 ppb wet weight in Atlantic herring (Rosenthal et al., 1986). In another study, Atlantic croaker (*Micropogonias undulatus*) fed Aroclor 1254 at a rate of 5 ppm body weight per day for 17 days showed impaired

reproductive function (impaired ovarian growth) (Thomas, 1989). Percent viable hatch success was generally 60% or better in winter flounder (*Pseudopleuronectes americanus*) from Long Island Sound and Boston harbor at egg levels of 0.25-1.0 ppm PCB wet weight as Aroclor 1254 (Nelson et al., 1991). A previous reference suggested that 0.12 ppm wet weight might be a viable hatch threshold limit (Hansen et al., 1985).

Eggs from winter flounder from New Bedford Harbor had significantly higher PCB content (39.6 ppm dry weight as Aroclor 1254) compared to a reference area (1.08 ppm dry weight). Larvae hatched from these eggs were significantly smaller in length and weight than from the reference site. A significant inverse relationship between PCB content of eggs and length or weight at hatch was noted (Black et al., 1988). Exposure to Aroclor 1254 in solution at 1.0 ppb increased cytochrome P450 activity in liver tissue from a goby (*Zosterisessor ophiocephalus*) after a two week exposure. No statement concerning biological significance was made, however (Nasci et al., 1991).

2.3.4.6.2. Critical Body Burdens

Some discussion of critical body burdens has recently appeared concerning marine hazard and risk assessments (Nendza et al., 1997). Contaminant residues in prey organisms (critical body burdens-CBB) may be used for marine hazard and risk assessments. Evaluations solely from aquatic exposure concentrations are not adequate to account for potential secondary effects in marine ecosystems in the view of the authors. Combining the BCF values with toxicity data, the no observed effect concentration (NOEC), yields the critical body burden, which is the contaminant level inside the organism, above which the fish will be impaired.

$$\mathbf{CBB = NOEC \times BCF}$$

However, some variability has been noted. A comparison of measured and calculated critical body burdens has shown that the calculated CBBs tend to be systematically higher than the measured burdens. Measured CBB data may vary by one order of magnitude within a population, with approximately 50 percent of this variation due to the lipid content of the individuals. The actual hazard potential of the individual chemical(s) for direct fish toxicity can be characterized by comparing the critical body burdens with measured concentrations in marine fish (measured concentration [e.g., ng/g tissue] divided by critical body burden=quotient value). The smaller the critical body burden and the actual contaminant concentration in the fish, the less likely is the direct toxic impact of the given chemical. Quotient values <0.01 indicate compounds for which the actual contaminant concentration is at least 100 times lower than the lowest observed toxicity value. Quotient values between 0.01 and 0.1 indicate compounds for which the actual contaminant concentration is at least 10 times lower than the toxicity threshold. Quotient values between 0.1 and 1 indicate compounds for which the actual contaminant concentration is in the same order of magnitude as the toxicity threshold. A direct impact by these compounds may be assumed. If the quotient values exceed one, toxicity is likely to occur (Nendza et al., 1997).

A recent study has reported critical body residue (CBR; e.g., burden) levels for an amphipod *Ampelisca abdita* (Fay et al., 2000) exposed to 2,2′, 4,4′ tetrachlorobiphenyl for 10 days in spiked sediment from Long Island Sound. Nominal sediment tetrachlorobiphenyl concentrations above 710 µg/g dry wt caused 100% mortality. A 50% lethal residue (LR_{50}) concentration of 0.57 µmol/g wet wt (18 µmol/g lipid wt) was determined. These values are 166 µg/g wet wt and 5.3 mg/g lipid wt, respectively. This lethal body residue burden falls below the range of CBRs

predicted for acute narcosis from hydrophobic neutral organic chemicals (2-8 µmol/g wet wt [584-2336 µg/g lipid? wt] - 63-255 µmol/g lipid [18.4-74.5 mg/g lipid] for an organism with 3.14% lipid), indicating that *A. abdita* is a sensitive species for acute toxicity testing. Acute narcotic CBRs can display ranges of values that appear to be species specific, however. The tendency toward over prediction of CBR levels is consistent with the higher calculated vs. measured critical body burdens discussed by Nendza et al., 1997 in the previous paragraph. The principal advantage of the CBR approach in determining toxicity, relative to standard comparisons of concentrations in the exposure medium, is the more direct dose-response relationship rather than an assumption of external exposure being a surrogate for internal dose.

2.3.4.6.3. TEF/TEQ

The TEF/TEQ (toxic equivalency factor/toxic equivalents quotient) approach in mammals has never been validated for invertebrates. The TEFs for dioxin-like congeners are based only on the vertebrate model of the chemical binding to the AhR (aryl hydroxylase) enzyme, which mediates the induction of P450 enzymes (Ahlborg et al., 1994; Safe, 1994; WHO, 1997). The toxic equivalency factor (TEF) of a congener is defined as the EC50 of TCDD divided by the EC_{50} of the congener. In one study, body burdens of radio-labeled PCB congeners 47, 77, and 153 accumulated from seawater were used to determine median effective concentrations (EC_{50}s) for developmental and cytogenic effects in sea urchin embryos following a 72 hour exposure (Schweitzer et al., 1997). Of these three, only congener 77 is considered to be a dioxin-like congener. Congener 47 was found to be at least four times more toxic than congener 77, a non-ortho substituted coplanar tetra chlorobiphenyl, with EC_{50}s of 47 mmol/kg (13.7 ppm) and >218 mmol/kg (63.7 ppm), respectively, using an embryo development assay. This result contradicts the structure activity prediction of the mammalian-based toxic equivalents (TEQ) approach, since congener 77 is a dioxin-like congener should exhibit more toxicity, based on the mammalian model. Congener 153, a di-ortho substituted hexachlorobiphenyl, was virtually nontoxic in terms of developmental effects at the highest dose of 102 mmol (36.8 ppm) per kilogram achievable at its limit of water solubility. Dose-response relationships were established with mitotic activity being the most sensitive endpoint because the PCBs appear to inhibit mitosis. Congener 77 was found to be at least two times more toxic (EC_{50} equals 30 mmol per kilogram, or 8.8 ppm) than congener 153 (EC_{50} equals 67 mmol per kilogram, or 24.2 ppm), but not as toxic as congener 47 (EC_{50}<16 mmol per kilogram, or 4.7 ppm) using mitotic activity as the endpoint for toxicity (Schweitzer et al., 1997).

For congener 47 at doses above 80 µg/L, most embryos had started to disintegrate by 48 hours. Significant toxicity was not observed with congener 77 in the uptake study. The average BCFs for congeners 47, 77, and 153 in the sea urchin embryos in this study were 186,000, 51,000, and 87,000, respectively. The measured water concentrations in this study included both the particulate and dissolved fractions. This demonstrates the limitations of using water measurements to predict tissue concentrations. It is unclear what congeners were bioavailable in the water. The question of bioavailability along with variability within and between the regressions for the three congeners demonstrates the importance of reporting tissue concentration instead of water concentration as the dose. Congener 47 was not only accumulated more readily than congeners 77 and 153, but was also the most toxic among the three congeners at comparable tissue doses. Congener 47 was at least four times more toxic than congener 77 using the development endpoint. The application of the current TEQ approach to invertebrates may yield inaccurate risk estimates for PCB congeners. Because differential toxicity was found among

these three PCB congeners, total body burden of PCBs in invertebrate tissues may be inadequate for predicting effects (Schweitzer, et al., 1997). In a recent conference on the use and development of TEFs and TEQs, it was felt that the development of TEFs for invertebrates is not recommended because there is limited evidence for ligand activation of Ah Receptors or for TCDD like toxicity in invertebrates (Van den Berg et al., 1998).

2.4. The Conceptual Model, Assessment Endpoints, and the Analysis Plan

At this point in the Problem Formulation of the SINKEX Ecological Risk Assessment, the "integration of available information" was completed. What was known (prior to the development of the sampling and analysis plans) about the site's stressor and exposure characteristics, the ecosystems potentially at risk, and observed ecological effects associated with the site and sources under investigation had been reviewed. The next step in Problem Formulation was to develop a conceptual model and select assessment endpoints.

2.4.1. Development of Preliminary Site Conceptual Model

A site conceptual model is a tool, which permits the ecological risk assessor to progress from the Problem Formulation phase to the Analysis Phase. It describes the best technical "understanding" of the study site and relationships among stressors (i.e., contaminants of concern) and exposure pathways through which the stressors may act to affect the environments potentially at risk. Many of the assumptions stated in the site conceptual model are the same ones used for the numerical model previously described in Section 2.3.3.6. The previous sections (Problem Formulation: Integration of Available Information, Exposure Characteristics, and PCB Levels and Effects in the Deep Ocean) have outlined what was known, a priori, about the ex-AGERHOLM site and the presence and behavior of PCBs in the deep ocean environment (focusing on Southern California waters). Figure 2-16 depicts the Site Conceptual Model for the ex-AGERHOLM risk assessment study.

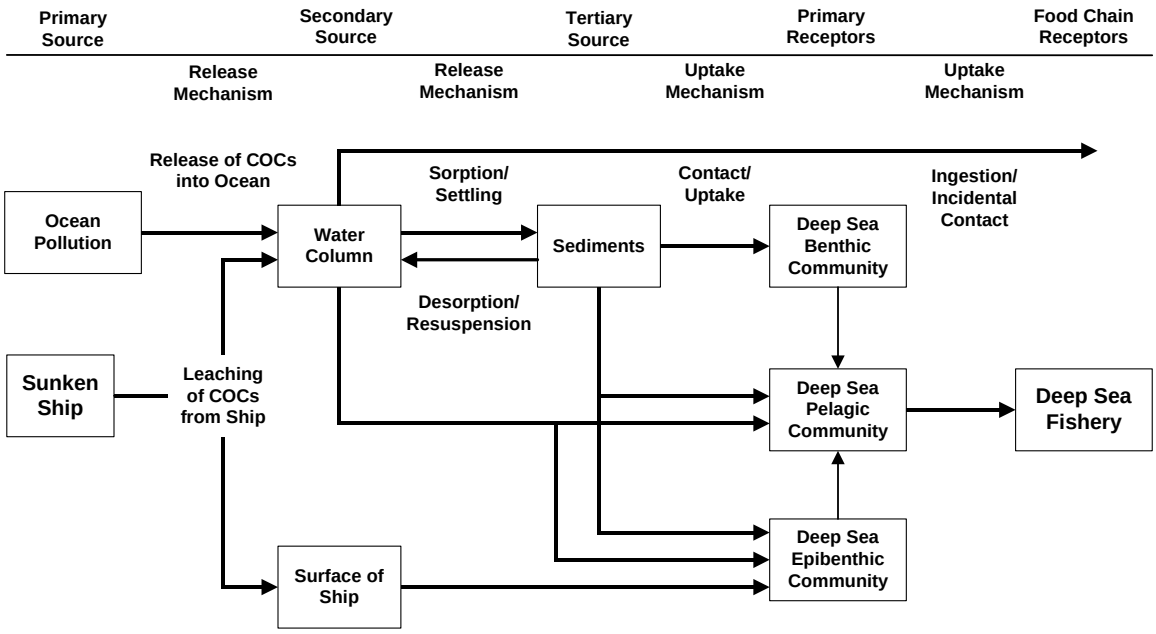


Figure 2-16. Preliminary site conceptual model for SINKEX ecological risk assessment.

2.4.2. Exposure Pathway Analysis for Ecological Receptors of Concern

In the model, primary sources (e.g. PCBs onboard ex-AGERHOLM vs. PCBs entering the system from other polluting sources) are acted upon by release mechanisms (e.g., PCBs leaching from the hulk vs. PCBs released into the ocean via industrial discharges, coastal runoff, food chain transfer from other PCB-contaminated ocean sites, and atmospheric deposition) to collect in secondary sources (e.g., water column and surface of ship). A second input pathway (e.g., sorption/settling and desorption/resuspension) represents the deposition of PCBs into a tertiary source (e.g., sediments) and the double arrows represent subsequent cycling between sediments and the water column. The next process is represented by the biological uptake mechanism that transfers contaminants (e.g., PCBs) from environmental media or matrices (e.g., sediment and water) into primary ecological receptors (e.g., benthic, epibenthic, or pelagic communities). The final step of the site conceptual model is represented by a secondary uptake mechanism (ingestion of contaminated organisms by higher level organisms, including humans) that initiates transfer through the food chain.

In the previous section, which discussed exposure scenarios for the release of PCBs, the following hypothesis was proposed:

> Leaking and leaching are the primary release mechanisms for PCBs into the water column. Solid-bound PCBs and metals in shipboard materials at the bottom of the deep ocean would tend to leach out into the water column at very slow rates, with decades or centuries passing by before an “average” PCB load would completely dissolve into the water column. Minimal contaminant release from the hulk was expected because: (1) the relatively small amount of contaminants onboard SINKEX vessels, and (2) the contaminants of concern (e.g., PCBs and metals) would be incorporated into solid matrices, are only slightly soluble in water, and even less so at expected ambient temperature and pressure.

Currents would affect the dispersion and "footprint" for the dissolved PCBs, but it was thought that a long-term "average" current on the bottom would be slow at this location, resulting in the PCBs quickly adsorbing to hull paints, organic material and sediments close to the hulk. The primary route of exposure to deep ocean organisms would probably be ingestion of sediments and particulates onto which PCBs have adsorbed. Biota receiving direct internal exposure through incidental ingestion of PCBs in settling particulates or bedded sediments would include benthic infauna and epibenthic fauna, as well as bottom-scavenging fish.

Through the mechanical actions of burrowing worms, feeding on benthic macroinvertebrates by predators and movements of underwater currents, secondary release mechanisms such as desorption and resuspension of sediments may occur, causing the PCBs to re-enter the water column.

Ultimately, contaminants from the hulk may enter the water column and sediment, initiating trophic transfer within a food chain, first from benthic and epibenthic organisms and bottom scavengers, and then to deep-sea fish predators such as sablefish. It is not believed that there is a high probability of vertical transfer through the water column, since deep-sea predators tend to stay deep and do not typically migrate long vertical distances. However, there is a potential for transfer of PCBs from these predators to humans via direct consumption of market fish caught by commercial fisheries. The sablefish, or black cod (*Anoplopoma fimbria*), represents the one such example found to still be an active fishery in the waters off of Southern California.

2.4.3. Selection of Assessment Endpoints

An assessment endpoint is defined by the following characteristics:

- a component of the ecosystem that may be impacted by the stressors of concern,
- has high ecological or societal value, and
- represents a component of the ecosystem that can be protected.

Generally considered to symbolize valued environmental conditions or processes, assessment endpoints often cannot be directly quantified. Instead, data on exposure levels and information that relates the exposure to the ability to cause effects to the assessment endpoint are needed to perform the risk assessment (USEPA, 1998a). In order to relate exposure levels to the assessment endpoint, "receptors of concern" must be identified to develop measures of exposure and measures of effects (previously, but still occasionally referred to as "measurement endpoints") that will define the Analysis Plan. By being representative components of the assessment endpoint, receptors of concern are species (or communities of species) that can be directly assessed at the site, measured under laboratory conditions, or used to develop toxicological thresholds (e.g., benchmark effects-concentrations from the literature). Because risks to the assessment endpoints are deduced from the results obtained from measures of exposure and effects, it is necessary to evaluate the uncertainties associated with the measures and appraise their ability to infer harm to the assessment endpoints.

Based on the exposure characteristics and pathways discussed previously, the following biotic communities were selected as "assessment endpoints," each represented by a receptor box (three primary plus one food chain receptor) in the Site Conceptual Model (Figure 2-16).

- Benthic infaunal invertebrates of the benthic community
- Epibenthic invertebrates of the epibenthic community

- Demersal scavenging and predatory fish of the deep pelagic community (Supplement I)
- Human populations consuming predatory fish (Supplement II)

2.4.4. Analysis Plan

Following the development of the Conceptual Model and Selection of Assessment Endpoints, the last major step remaining in the Problem Formulation Phase of Ecological Risk Assessment was development of the Analysis Plan. This plan represented the basis for the sampling and analysis design, including the temporal and spatial distribution of projected field sampling, the methods that would be used to collect, preserve, and analyze samples for physical, chemical, and biological characteristics, and finally the protocols for laboratory experiments that would measure responses of standard test organisms not obtained from the field.

2.4.4.1. From Assessment Endpoints to Measures of Exposure and Effects

The first critical step in developing an Analysis Plan was to provide the rationale for the sampling and analysis design by selecting Measures of Exposure and Measures of Effects (formerly referred to collectively as "measurement endpoints) that could be directly linked to the Assessment Endpoints.

Table 2-7 presents the thought process adopted in this ERA to proceed from Assessment Endpoints to Receptors of Concern and finally to Measures of Exposure and Measures of Effects. It is important to note that the sampling design evolved from conception (i.e., development of the analysis plan) through the numerous field collection efforts made during a two-year period, forcing a modified sampling design with reduced spatial coverage and replacement of target species with surrogates. The most important factors that dictated this evolution were:

- The bottom sediment in some areas proved to be difficult to sample due to the presence of hard, phosphated calcium nodules, requiring engineering re-design of sampling equipment; and
- Inclement weather and equipment problems, which adversely impacted sampling opportunities.

For the Benthic Community, Receptors of Concern were represented by surrogate species typically used in environmental bioassays to measure exposure and effects from contaminated sediments to organisms. These included one amphipod (sediment-burrowing crustacean), two species of polychaetes (sediment-dwelling worms), and one clam (sediment-dwelling bivalve).

Because initial scoping efforts resulted in the discovery of a commercial sablefish fishery in the vicinity and depth of the ex-AGERHOLM, it became necessary to investigate the potential for food chain transfer of PCBs from the ex-AGERHOM to humans via this species. Sablefish or *Anoplopoma fimbria* (Pallas, 1814), also known as black cod or butterfish (Fishbase 2002a, Patterson et al. 2001), are important members of the deep sea community. They are found on soft ocean bottoms in deep water from the Bering Sea to Baja California (OTP 2002), and are generally prized as a food fish because of their high oil content and exceptional flavor (NOAA 2002). Occupying a niche relatively high on the food chain with a trophic level of 3.4 – 4.3 (Fishbase 2002b), sablefish feed on fishes, worms, and crustaceans. Characteristics of Sablefish that make them a suitable species for evaluating higher food chain effects include the following:

- Sablefish prefer to inhabit depths near the ocean bottom (demersal), and can be found at the depth of the ex-AGERHOLM;
- Sablefish feed on the bottom, consuming detritus and/or benthic epi/infauna;
- Sablefish have a high lipid content relative to other deep-sea fish, and when exposed to persistent organics like PCBs, the chemicals are expected to bioaccumulate by partitioning to lipids;
- For their size, Sablefish have been observed to be generally non-migratory and have moderate (on the order of 50 km to 50 nm) habitat ranges. Some observations have noted a tendency for territorial or residential behavior;
- Sablefish have been commercially harvested in offshore waters of California since 1915 or earlier; and
- PCBs have been detected in sablefish from the Pacific Ocean at concentrations higher than in many other sampled organisms.

Table 2-7. Development of measures of exposure and effects from assessment endpoints and receptors of concern.

Assessment Endpoint	Receptors of Concern	Measures of Exposure	Measures of Effects
invertebrates of the benthic/epibenthic community and entry point to larger pelagic food chain	(1)amphipod (*Rhepoxynius abronius*) (2) polychaetes (*Nephtys caecoides* for exposure; *Neanthes arenaceodentata* for Effects) (3) clam (*Macoma nasuta)*	(1) Tissue concentrations of PCBs, metals, and PAHs in surrogate clams (*Macoma nasuta)* and polychaetes (*Nephtys caecoides)*, after 28-day laboratory exposures to SINKEX sediments. (2) Sediment concentrations of PCBs, metals, and PAHs.	(1) 10-day acute toxicity test (survival & reburial) on surrogate amphipods (*Rhepoxynius abronius)*, exposed to SINKEX site and reference site sediments. (2) 28-day chronic toxicity test (growth and survival) on surrogate polychaetes (*Neanthes arenaceodentata*), exposed to SINKEX site and reference sediments.
Scavenging, demersal, and predatory fish in pelagic community	Sablefish/Black Cod (as primary consumer of contaminated sediment/water).	(1) Tissue concentrations of PCBs (2) Water column concentrations	Estimation from exposure
humans	Sablefish/Black Cod (representing step in food chain)	Edible tissue (i.e. muscle) concentrations of PCBs	Human Health Risk Assessment (separate report)

Notes:

(1) Efforts were made during several cruises to capture epibenthic infauna, but too few specimens were collected to address quantitatively.

(2) Because PCB exposure to ecological receptors of concern through water absorption/consumption was considered to be a far less important pathway than sediment ingestion, and because sampling the water column in a robust manner to be statistically valid was problematic, water sampling was a very small component of the Analysis Plan (a few samples were taken to see if PCBs could even be detected) and was not part of the decision matrix. However, a short discussion on water sampling is found in the first part of Section 3.2 (Characterization of Exposure to COCs).

For the reasons that made Sablefish a suitable choice as a Receptor of Concern for the Human (food chain) Assessment Endpoint, this organism proved to be suitable as a surrogate species for the Epibenthic Community and Pelagic community. For the Epibenthic Community, the original plan included epifaunal specimens collected on the hulk and on the sediment nearby. However, because numerous attempts to collect specimens resulted in too few organisms to warrant statistical comparisons, the decision was made to use the sablefish as a surrogate indicator of epifaunal contamination as the first step in the epifaunal food chain (a predator of these organisms, and likely to forage in the same territory over long periods of time).

2.4.4.2. Overall Technical Approach

From the above Measures of Exposure and Effects, a Decision Matrix and resulting Lines of Evidence were formulated. As explained in the Introduction, the overall technical approach meets USEPA and Army Corps of Engineers requirements (referred to as the "Green Book" for simplicity) for the investigation of potentially contaminated sediments, prior to ocean disposal (USEPA/USACE, 1991). However, the approach also fits into the framework of the EPA's Ecological Risk Assessment, exemplified by the prior discussion and logical progression and selection of ecological values, receptors, and assessment/measurement endpoints (USEPA, 1992). Consequently, the Decision Matrix and corresponding Lines of Evidence, from which the sampling and analysis designs were constructed, reflected the integration of these two guiding programs.

Four guiding principles were used in the overall technical approach:

5. A standard two-site hypothesis design, employing an experimental site ("ship") vs. a reference site ("site distant from the ship but in similar deep ocean environment"), as typically used in environmental studies;
6. Statistical power of replication within each data set was achieved with multiple sediment stations along the perimeters of an "inner" ship ring (Inner Ring) and an "outer" reference ring (Outer Ring). All stations on a ring were to be considered as a single station for statistical purposes.
7. A weight of Evidence Approach in considering multiple Lines of Evidence, as typically done in ERAs and in sediment studies under the Green Book (e.g., Sediment Quality Triad); and
8. Emphasis on a Decision Matrix using the Green Book ocean disposal approach (e.g., toxicity testing which follows those protocols), in accordance with USEPA Office of Water guidance provided prior to development of the sampling plan.

The null hypothesis addressed by this investigation is: There is no significant difference between the Inner Ring (Ring 1) and the Outer Ring (Ring 4) in the primary lines of evidence for chemical concentrations in sediment or tissue, or for adverse effects levels in acute and chronic toxicity tests.

2.4.4.3. Sampling Design

Given the above null hypothesis, the sampling design was intended to assist in answering the following questions:

- Are the sediments near the ex-AGERHOLM statistically elevated for PCBs, metals, or PAHs, when compared to the reference site?

- Are the sediments near the ex-AGERHOLM toxic to marine life?
- Do marine life bioaccumulate contaminants from the sediments collected near the ex-AGERHOLM in amounts greater than the reference site?
- Are there elevated levels of contaminants in demersal fishes collected near the ex-AGERHOLM compared to the reference site?

This study design was to be based on sediment and tissue chemistry, toxicity, bioaccumulation, and limited benthic community analysis. It was initially thought that enough benthic infauna data could be collected, within reasonable cost constraints, to provide adequate statistical comparisons in community structure or function for evaluating benthic community impacts. This aspect of the sampling plan was revised prior to the at-sea sampling effort, because the "natural" benthic community of the area was not well understood. Lacking site specific information on community variability it was determined that the number of samples necessary to develop appropriate statistical treatment was not possible, and what limited sampling that could be accomplished could not be effectively used to test differences between rings. However, data on the benthic community was assessed to provide a general physical and ecological description of the site environment, evaluate potential pathways for food chain accumulation and to obtain basic knowledge of representative infauna organisms in the sample area. For these reasons, benthic community analyses are not entered in Table 2-7.

The initial sediment sampling strategy consisted of four concentric sampling rings radiating outward from the ship (Figure 2-17). Sampling difficulties during the first several cruises (mentioned earlier in this section, see Methods Section 2.5 for more details) precluded sampling all stations at all rings. The sampling design was simplified to a single Inner Ring close to the ship (Figure 2-18) representing conditions at the ship site, and an Outer Ring 1000 meters from the ship representing "reference" conditions (Figure 2-19). Table 2-8 lists all sampling locations including sediment stations, oceanographic monitoring stations, sediment traps and fish traps.

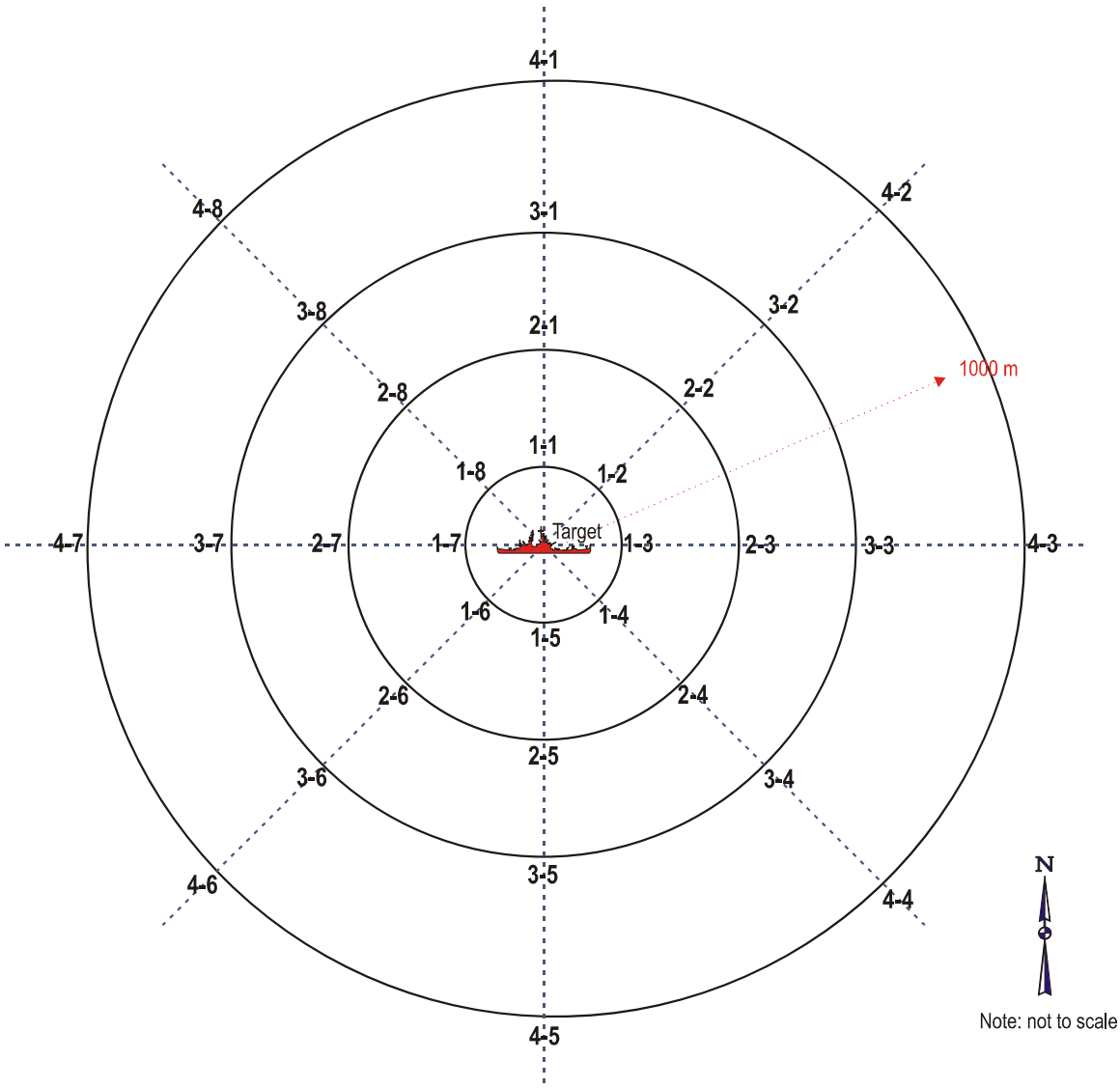


Figure 2-17. Initial sampling plan.

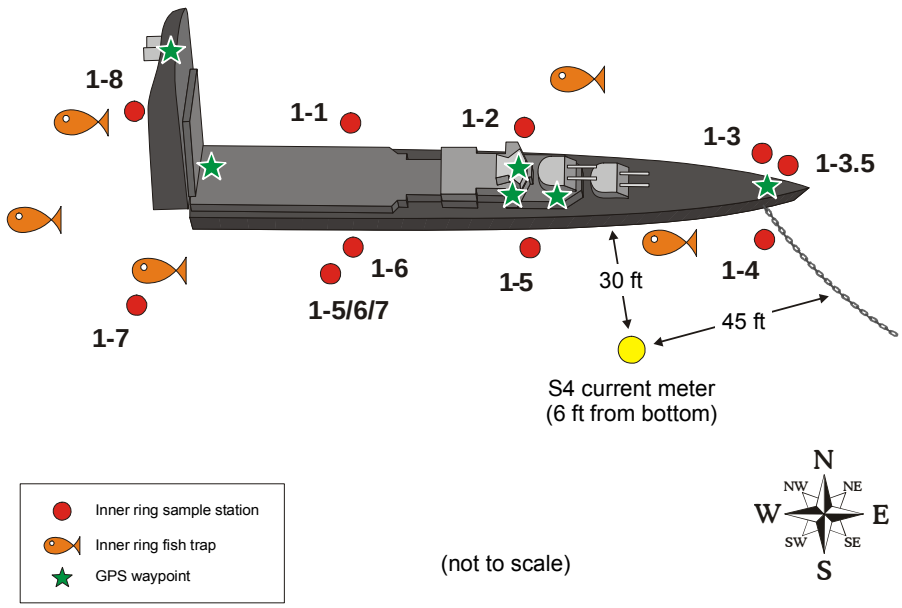
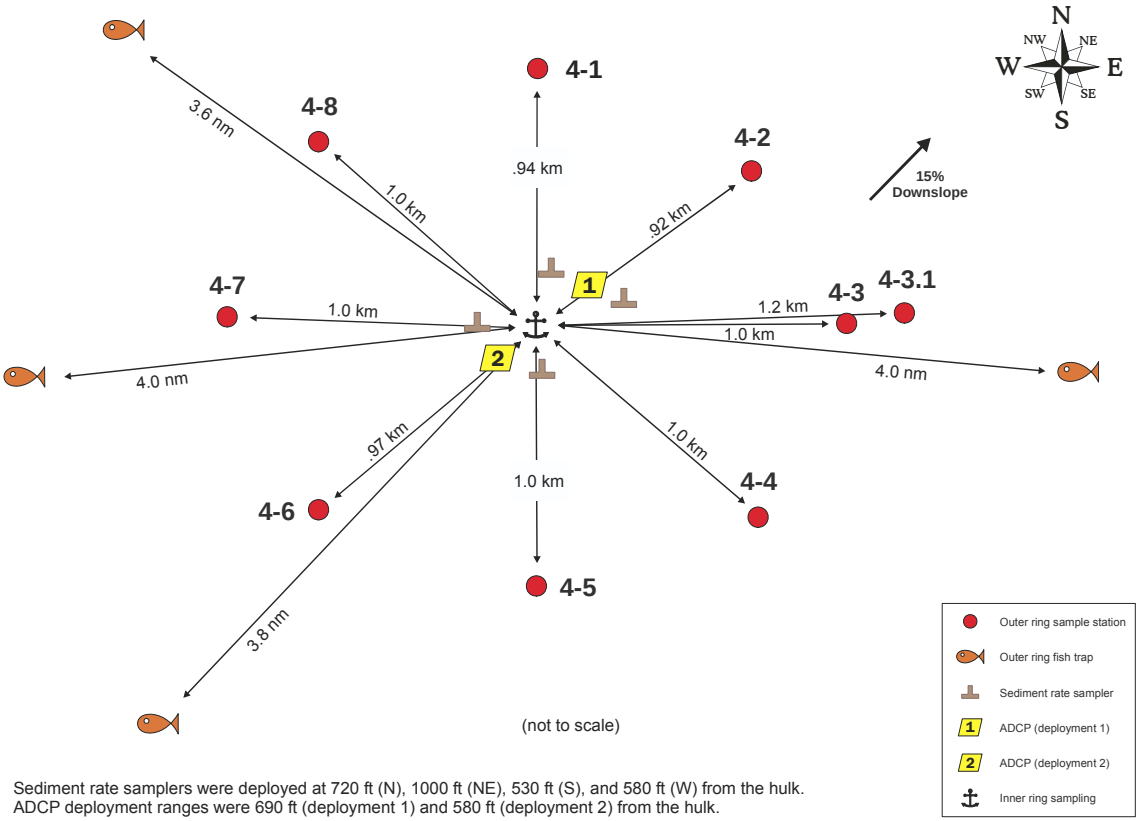


Figure 2-18. Final sampling plan, Inner Ring.



Sediment rate samplers were deployed at 720 ft (N), 1000 ft (NE), 530 ft (S), and 580 ft (W) from the hulk.
ADCP deployment ranges were 690 ft (deployment 1) and 580 ft (deployment 2) from the hulk.

Figure 2-19. Final sampling plan, Outer Ring.

Table 2-8. Sample locations.

SITE NAME	X COORDINATE(°)	Y COORDINATE(°)	Z COORDINATE (m)	SITE AS OF	Distance from hulk (ft)*
Station 1-1	-119.593274	32.756996	-846	18.Sep.98	6
Station 1-2	-119.593058	32.756988	-838	18.Sep.98	9
Station 1-3	-119.59276	32.756951	-834	18.Sep.98	12
Station 1-3.5	-119.592729	32.756937	-824	17.Nov.99	8
Station 1-4	-119.592758	32.756839	-842	18.Sep.98	6
Station 1-5	-119.593051	32.756833	-838	18.Sep.98	8
Station 1-5/6/7**	-119.5933	32.756791	-834	17.Nov.99	15
Station 1-6	-119.593272	32.756827	-838	18.Sep.98	28
Station 1-7	-119.593564	32.756813	-830	18.Sep.98	45
Station 1-8	-119.593544	32.757009	-835	18.Sep.98	32
Station 4-1	-119.592994	32.765967	-1143	01.Nov.99	3290
Station 4-2	-119.585617	32.762117	-1006	03.Aug.97	3010
Station 4-3	-119.582293	32.757043	-975	03.Aug.97	3330
Station 4-3.1	-119.580312	32.757495	-985	01.Nov.99	3960
Station 4-4	-119.585417	32.750433	-808	01.Nov.99	3330
Station 4-5	-119.593007	32.747821	-739	04.Aug.97	3330
Station 4-6	-119.600484	32.750609	-580	18.Sep.98	3170
Station 4-7	-119.603678	32.757183	-727	01.Nov.99	3270
Station 4-8	-119.600542	32.76321	-892	01.Nov.99	3280
S4 Current Meter	-119.592668	32.75673	-780	15.Nov.99	50
ADCP First Deployment	-119.591412	32.758092	-855	04.Aug.97	690
ADCP Second Deployment	-119.594292	32.755788	-800	15.Nov.99	580
Starboard Bow Fish Trap	-119.592755	32.756848	-843	15.Sep.98	5
East Fish Trap	-119.51395	32.756816	-1363	15.Nov.99	24300
Port Stern Fish Trap	-119.593525	32.757057	-821	15.Nov.99	20
Starboard Stern Fish Trap	-119.593488	32.756797	-821	15.Nov.99	35
Northwest Fish Trap	-119.6264	32.80828333	-1297	15.Nov.99	21870
Southwest Fish Trap	-119.646	32.71183	-707	15.Nov.99	23090
Portside Amidships Fish Trap	-119.592944	32.75701	-814	15.Sep.98	20
Rear Fish Trap	-119.59354	32.756879	-900	17.Nov.99	10
West Fish Trap	-119.671845	32.75718833	-771	17.Nov.99	24300
East Sediment Trap	-119.589982	32.757793	-860	04.Aug.97	1000
North Sediment Trap	-119.592448	32.758813	-823	04.Aug.97	740
South Sediment Trap	-119.592778	32.755485	-790	04.Aug.97	530
West Sediment Trap	-119.595008	32.75709	-810	04.Aug.97	630
ex-AGERHOLM Center	-119.593028	32.756901	-832	04.Aug.97	N/A
ex-AGERHOLM Over Starboard Bridge Wing	-119.5930217	32.75689	-828	15.Nov.99	N/A
ex-AGERHOLM Over Bow	-119.5926583	32.75687333	-834	15.Nov.99	N/A
ex-AGERHOLM Over Pilot House	-119.5930117	32.7569	-825	15.Nov.99	N/A
ex-AGERHOLM Near Stern	-119.59347	32.75689	-807	15.Nov.99	N/A
ex-AGERHOLM Next to Turret	-119.5929683	32.75687333	-834	15.Nov.99	N/A

* if <1km, distance from hulk; if ≥1km, distance from centroid of hulk. **Station 1-5/6/7 was selected to revisit suspect samples at Station 1-6 which had been collected on a previous sampling effort. Due to extreme heterogeneity, it is considered distinct.

2.4.4.4. Decision Matrix

Decision matrices were developed a priori to represent possible outcomes and conclusions that could be derived from the data, with the knowledge that the magnitude of toxicity and bioaccumulation, as well as other factors, would greatly influence the decision process. The decision matrices were to be used to evaluate the weight of evidence and characterize ecological risk for the SINKEX Ecological Risk Assessment (ERA). The decision matrix developed (Table 2-9) outlines risk criteria for various outcomes from inference tests. For example, a "+" would occur in a test when there is a statistically significant and unfavorable (i.e., indicating risk) difference between the Inner Ring and Outer Ring Reference stations. Conversely, a "-" would indicate that the average Inner Ring measurements were not different than the average value measured for the Outer Ring. The matrices only show those outcomes that are reasonably expected. For example, it is assumed that significant infaunal tissue bioaccumulation will not occur in the absence of significantly elevated sediment chemistry. Significant effects related to either toxicity or bioaccumulation in samples from the Inner Ring would be assumed to be from the hulk, in the absence of any other causal agent at the study site. The final matrix shows six possible risk outcomes. Since ecological risk is better determined through direct measurement of ecological and biological effects, more weight would be given to toxicity, and laboratory bioaccumulation than to sediment chemistry. Furthermore, results from the benthic community study were not to be used to affect the risk result displayed in the matrix. These data were to be used to support additional studies such as food chain contaminant transfer and human health risks (i.e., biomagnification potential). Note that Sablefish was not included in the Decision Matrix, because its consideration in the overall study - first for human health and later for supplementary ecological risk data – occurred after development of this Decision Matrix.

Table 2-9. Decision matrix for sediment data.

Outcome Number	Sediment Chemistry	Acute/Chronic Toxicity	Laboratory Bioaccumulation	Benthic Community	Risk
1	–	–	–	±	None
2	+	–	–	±	None
3	–	+	–	±	Potential Local
4	+	+	–	±	Probable Local
5	+	–	+	±	Potential Food Chain
6	+	+	+	±	Probable Local and Potential Food Chain

+ Significant unfavorable difference for Inner Ring for any endpoints measured.
– No significant unfavorable difference for Inner Ring for endpoints measured.
± No effect on decision.

The simplest risk determinations to explain are the first and last ones with respect to chemistry, acute toxicity and laboratory bioaccumulation from sediment data. Negative findings for all categories (Outcome 1) would indicate that there is no likely risk, and no reason to assume that the ship has had an adverse impact on the deep-water environment. Positive findings in all categories (Outcome 6) would indicate that there is a probable risk to the deep-water environment and a potential food chain risk associated with the sunken ship.

The middle four risk determinations (Outcomes 2-5) are increasingly subject to interpretation because they each involve disparate outcomes among the measures of exposure and effects. For Outcome 2, elevated chemistry data alone would not indicate risk from sinking ships because sediment chemistry receives less weight than the combined measures of effects (as explained previously); and in the absence of toxicity or bioaccumulation, there is no reason to assume risk.

A positive toxicity result combined with negative results from sediment chemistry and bioaccumulation (Outcome 3) would indicate a potential localized risk. However, the source characterization efforts resulted in selection of the most likely toxic constituents for measurement; thus, it is unlikely that toxicity would be caused by the presence of the hulk without a concurrent elevation of corresponding sediment chemistry values. Nevertheless, other parameters which can affect toxicity testing, such as grain size and TOC, were also measured to provide additional data for consideration.

If the data were to result in conflicting results (Outcomes 4 and 5), the conclusion about risk is more difficult to make. With respect to Outcome 4, there is positive chemistry and toxicity, but the evidence does not indicate a bioaccumulation problem. Thus, the only apparent effect would be toxicity, localized in the vicinity of the hulk and the weight of the two positive results indicates a risk determination of a probable local effect. For Outcome 5, positive bioaccumulation results coupled with positive chemistry indicates a probable transfer of contaminants to higher trophic organisms. In both cases, there is enough potential for effects to ecological receptors to warrant further discussions of risk management options between the Navy and EPA. It may be determined that additional data is required to resolve the conflicting outcomes in this case.

Best professional judgment will always be taken into consideration. The relative strengths of each of the statistical outcomes (i.e., + or –) will vary and should be considered. Additionally, the magnitude of toxicity or bioaccumulation observed could greatly affect the final risk determination.

Finally, best professional judgment also is significant when discussing study findings that are difficult to address in statistical testing. Examples are:

- Possible confounding variables not considered in designing the study, but are only discovered during the course of data collection and/or analysis,
- Mitigating factors related to bioavailability as determined from normalizing measurements (grain size, TOC, etc.), taking into account currents and mixing zones, and
- Uncertainties created by errors in sampling, analysis, or other data collection activities.

2.5. Materials and Methods

2.5.1. Search Operations

Dedicated search operations were conducted in two separate cruises [*insert cruises & years]* to find a suitable sunken vessel at approximately 6000 ft depth, long before the ex-AGERHOLM was opportunistically discovered by the Navy research submarine. This was a more difficult task than anticipated. In any situation involving locating an object in the deep ocean, it is imperative that good positional data be available (i.e. latitude/longitude/depth) that can be related to the time of sinking. Unfortunately, that information was not adequately determined or archived in the

case of SINKEX vessels sunk prior to 1990. Therefore, this effort required extensive investigation into the historical details of a number of particular target exercises and subsequent sinkings. Additional evaluation was necessary to determine the confidence one would have in recorded positional information, as well as the characteristics of a selected site making it amenable for deep-ocean survey and sampling. After considerable effort, three primary targets were selected based on the following criteria: time on the bottom, confidence in positional data, depth in the vicinity of sinking, bottom topography, representativeness of vessel type to typical sinking operations, and information on probable PCB content. Two separate cruises were mounted using the latest side scan sonar technology to search deep ocean areas in the vicinities where the sinkings occurred off the Southern California coast. The searches centered on the best-estimated positions of the ex-HIGBEE and the ex-BAUSELL, both WWII era destroyers selected as the best candidate vessels for study. Neither vessel was located, however, a third candidate hulk (ex-AGERHOLM), which had been sunk since 1982, was located in July of 1996 by the USS DOLPHIN. The crew of this research submarine subsequently provided photography and videography to positively identify the hulk.

2.5.2. Overview of Sampling Efforts

Sampling operations were executed during five oceanographic cruises to the ex-AGERHOLM site. The study design was constrained due to difficulties in sample collection. The study site is, deep, the wreckage was hazardous, and the native sediments were phosphatized into a heterogeneous matrix consisting of soft sediment and hard nodules that was not amenable for typical sediment sampling devices (e.g., a large oceanographic boxcore). After encountering the unusual substrate during Cruise I, it was determined that an ROV would be more successful collecting Ring 1 sediment samples directly adjacent to the ship. The use of ROV technology coupled with development of an innovative sampler resulted in a successful Cruise II. However, due to decreased efficiency in using the ROV, new methods and equipment were developed in an attempt to collect samples more rapidly on Rings 2, 3 and 4. This follow-on effort held promise as an improvement, however it was determined that an unacceptable amount of further development was required to be successful. As a result, sampling methods returned to the ROV operations and focused only on Ring 4 during Cruises IV and V.

Sampling focused on six areas of interest:

1. Sediment (chemical, physical, toxicological, bioaccumulation, and benthic community)
2. Sedimentation rate (Deployment and retrieval of long term samplers)
3. Oceanographic Currents (RD Instruments ADCP and InterOcean S-4 current meters)
4. Bioaccumulation (demersal fishes and crabs caught with fish traps)
5. Water (discrete samples with Niskin bottles and profiles with Seabird SBE-19)
6. Video reconnaissance of the sunken hulk and the surrounding bottom environment

For the successful sampling efforts, the following groups supported SSC-SD in the field:

- Submarine Development Squadron Five (SDS-5)/Deep Submergence Unit (DSU)
- MEC Analytical Systems, Inc. (MEC)
- Arthur D. Little, Inc. (ADL)

Submarine Development Squadron Five (SDS-5)/Deep Submergence Unit (DSU) provided field sampling support, specifically for the operation of the Remotely Operated Vehicles (ROVs). Support included boxcores for ±3 meter sampling accuracy and the actual recovery of sediment from the ocean floor immediately surrounding the hulk. Sub-sampling of sediment for physical measurements, bioassays, and benthic community analysis was performed by MEC. Arthur D. Little, Inc. performed sub-sampling of sediment and tissue for trace-level chemical analyses.

Cruises I and III utilized vessel support from Scripps Institution of Oceanography. No useful data was obtained from these cruises, and thus are not included in this report.

2.5.2.1. Field Sampling Overview

Cruises II, IV and V utilized vessel support from the Chouest group (M/V Laney Chouest Cruise II) and M/V Kellie Chouest (Cruise IV and V). The Deep Submergence Unit provided underwater operations with the ROV SCORPIO 1 for Cruise II and ROV SCORPIO 2 for Cruises IV and V. Twenty-four hour operations were maintained throughout the cruises. The satellite-based Global Positioning System (GPS) provided navigational reference. The dynamic positioning capability of M/V Laney Chouest and the M/V Kellie Chouest allowed the ships to maintain position within a ten-meter box. The SCORPIO ROVs consist of tethered, sled-mounted ROVs with two manipulating arms, hydraulic controls, lighting systems, camera systems, and sonar systems (Figure 2-20). A launching crane, winch and cabling system, and take-up spool with Kevlar®-coated tether was used to launch and recover the SCORPIO ROVs. The SCORPIO ROVs were also used to deploy and retrieve fish traps and collect biological specimens. The water depth at the ex-AGERHOLM site is 2,750 ft (838 m), round trip wire time was approximately one hour (wire speed of about 92 ft/min) for each deployment.

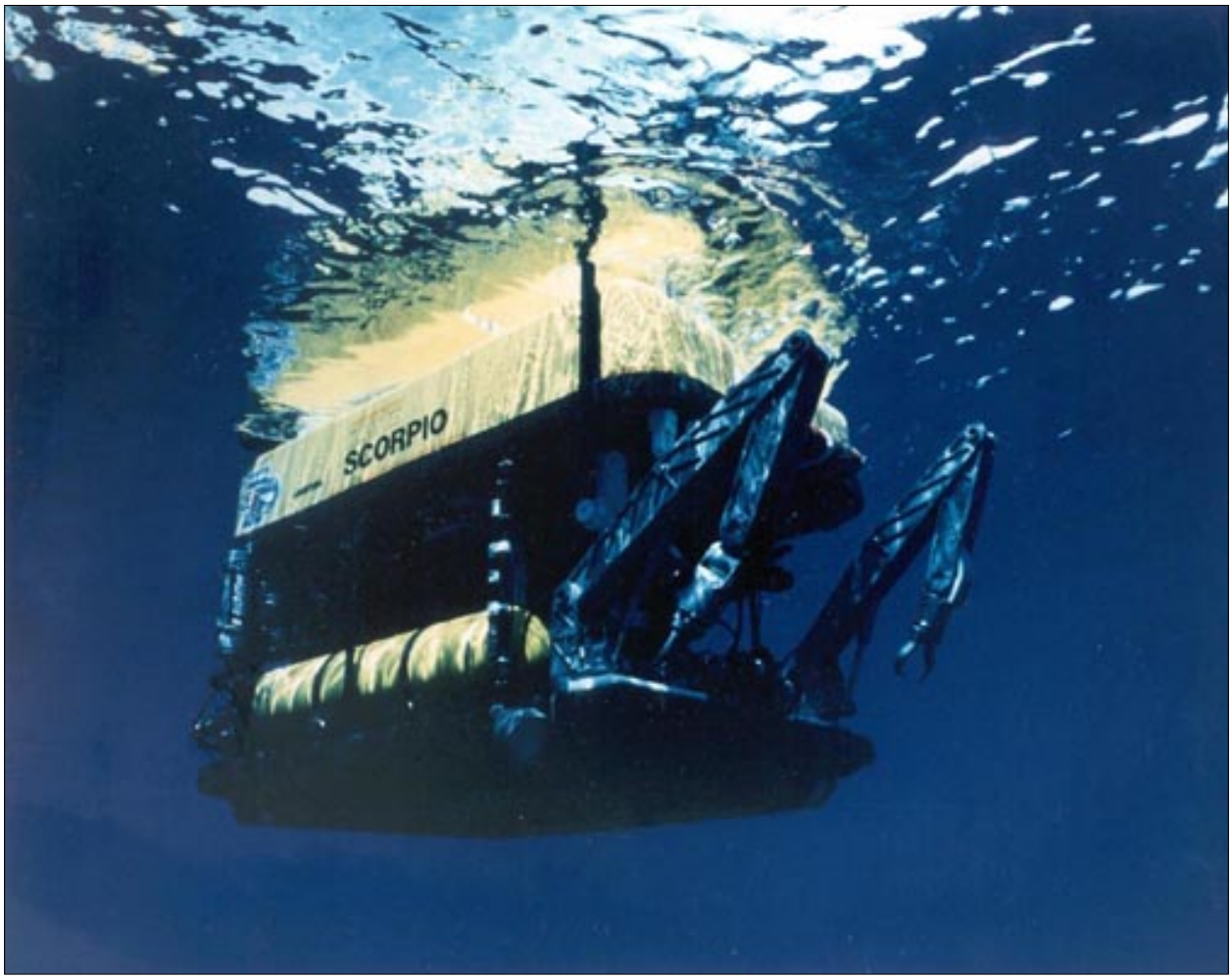


Figure 2-20. The SCORPIO ROV.

When a traditional 0.25 m^2 boxcore was unable to sample the unique site sediments on Cruise I, multiple small (0.05 m^2) boxcorers were designed and fabricated by the team for use with the SCORPIO ROVs. The small boxcores for chemical analysis were fabricated from stainless steel, and ones for bioassay analysis and benthic infauna were made from steel. These small units allowed a high degree of real-time control over placement of the coring devices and the ability to avoid large rocks that impacted the standard large boxcore. On the version used for Cruise II, a stainless steel cable was pulled by the manipulator arm to close the sampler door after a sample was collected. For Cruise IV and V, the coring devices were modified by installing gear-driven shafts to close the door (Figure 2-21). This design took advantage of the manipulator arms ability to rotate, saving time and providing a higher percentage of acceptable samples. All of the new boxcores were constructed with stainless steel. For locations where the substrate was too hard to sample by just pressuring the sampler in with the manipulator arm of the ROV, a hammering device was designed (Figure 2-22). The manipulator arms placed the hammering device on top of the boxcorer and utilized the ability to rotate to repeatedly lift and drop the hammer and drive the core into the substrate. This device was attempted at only one site (unsuccessfully) and all other grab samples were obtained without the assistance of the hammering device. Sediment samples collected approximately 1-2 m from the hulk were designated as Inner Ring samples

(Ring 1) and samples collected approximately 1,000 m from the ship were designated as Outer Ring samples (Ring 4). Sediment samples were collected using the ROV that utilized these small boxcore samplers to collect multiple samples at each sampling location. As explained previously in Section 2.4.4.3, samples were collected along points on these two rings, shown geographically in Figure 2-18 and Figure 2-19, with additional details provided in Table 2-8).

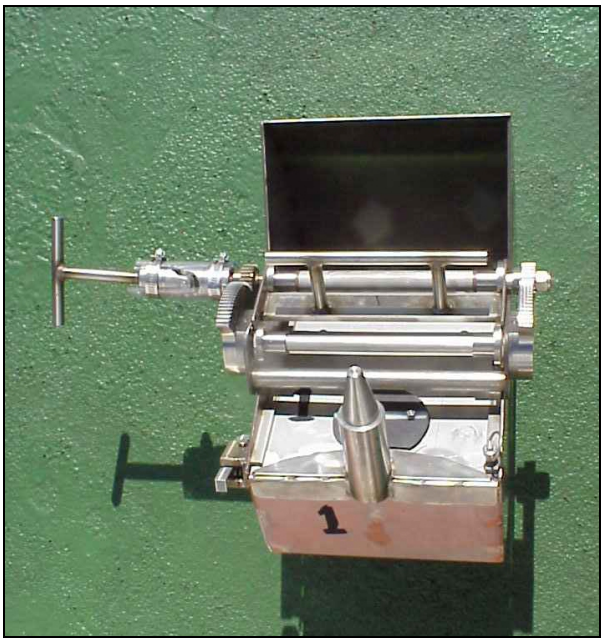


Figure 2-21. Stainless steel boxcore with gear-driven shaft.

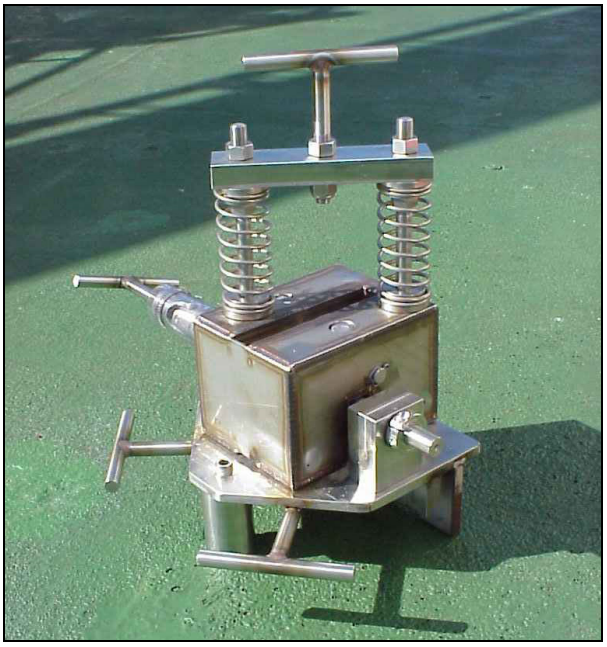


Figure 2-22. Stainless steel boxcore with added "jackhammer" device.

2.5.2.2. Sediment Sampling Overview

Between September 1998 and November 1999 (SINKEX Cruises II, IV, and V), samples were collected along points on two rings at separate distances from the ex-AGERHOLM, as described above. Sediment samples collected approximately 1-2 m from the hull were designated as Inner Ring samples (Ring 1) and samples collected approximately 1,000 m from the ship were designated as Outer Ring samples (Ring 4). Sediment samples were collected using a Remotely Operated Vehicle (ROV) that utilized small boxcore samplers to collect multiple samples at each sampling location.

Temporal differences between surveys were not considered significant and all sediment samples from SINKEX surveys II (September 1998), IV (September 1999) and V (November 1999) were considered as one data set. The null hypothesis was that there was no difference between the Inner and Outer Ring sediment samples. Significance criteria was $p<0.05$, which would suggest rejection of the null hypothesis. Before statistical testing was conducted the data was tested to determine if it fit the assumptions of the ANOVA statistical test (Table 2-11).

When on station, the ROV was lowered over the side and monitored as it descended to the bottom. A pilot and copilot sat at an operations console on the ship and controlled the ROV. Once on the bottom the pilot guided the ROV to the sampling location. The ROV was positively buoyant, thus requiring upward thrusting to stay on the bottom. This reduced sediment disturbance compared to normal ROV operations when the ROV is negatively buoyant, and

thrusters aimed toward the bottom are required to levitate the ROV and significantly disturb the benthic sediments.

On the initial dive at a station, a station marker was brought down by the ROV and placed on the bottom. Subsequent dives located samples within 20 ft of the station marker. Care was taken to not disturb the bottom before the samples are taken.

After the pilot positioned the ROV at the desired sampling location, the copilot activated the manipulator arms of the robot. Using both cameras to guide the manipulator arm, the copilot picked up the coring device and placed it as gently as possible into the sediment at the desired sampling location. The coring device was then pushed into the sediment, with gentle rocking often utilized. Holding the sampler steady, the copilot reached over with the other arm, pulled out the safety cotter pin that kept the shovel open, and pulled the cable that drew the door closed (Cruise II) or turned the gear drive to close the door (Cruise IV and V). The copilot returned the sampler to the ROV, and repeated the process for the second and third samples. This operation was very tedious and time consuming due to the difficulties in penetrating the coring devices through the gravel and rock. Closing the sample device door was sometimes difficult due to the abundance of gravel and rock. When sampling was completed, the diving supervisor gave the command to wind in the cable and the pilot started the accent to the surface. After reaching the surface and the ROV was recovered back on deck, the sediment samples were inspected by the Principal Investigator and Duty Scientist on watch to assure that the samples were intact, undisturbed, and representative of the top 3 cm of the sediment. Any samples which were compromised, from washing out of the sampler during the transit to the surface, excessive mixing of the surface layer, or appear to be unrepresentative of the sediment surface were rejected and not used for quantitative analysis (analytical chemistry, bioassay, or infaunal analysis). Rejected samples were screened for qualitative analysis of biota only.

All equipment used to collect and handle sediments was cleaned with Alconox® and rinsed with freshwater then site seawater prior to re-deployment.

2.5.2.3. Boxcore Sampling Procedures

Sediments were collected using a 0.05 m^2 surface area stainless steel box-core (23 cm × 23 cm × 23 cm), drawn and shown in Figure 2-23 and Figure 2-24, respectively. Cellulose-acetate-butyrate (CAB) cylinders (6.4 cm diameter) were used to obtain subsamples for profiling PCBs and age dating. Detailed box-core sample collection procedures are described below. Procedures were modified in the field based on sea conditions and personnel safety concerns.

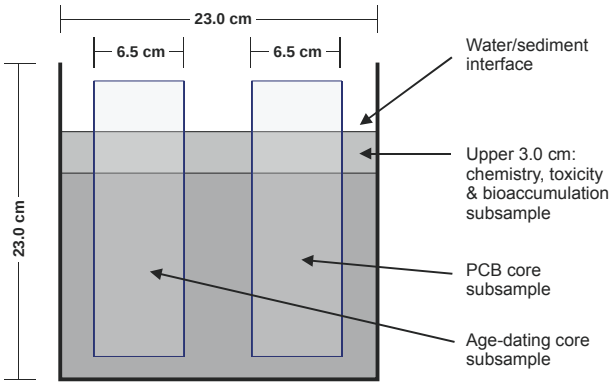


Figure 2-23. ROV boxcore sample schematic.

Prior to sediment sampling activities, the boxcore will be decontaminated as follows:

1. Wash with Alconox® and water;
2. Rinse three times - once with tap water, twice with de-ionized water;

3. Rinse twice with an organic solvent (typically ethanol and hexane);
4. Rinse three times with de-ionized water; then
5. Cover the decontaminated boxcore with a polyethylene plastic sheet and secure until next deployment.

Following retrieval of the sediment sample by the Remotely Operated Vehicle, the box-core sampler was returned to the deck of the vessel and placed in a protected location to limit exposure to potential sources of contamination (e.g., vessel exhaust gases). The top of the boxcore was carefully removed and overlying water was siphoned off from the lowest corner. The physical condition of the sample was observed and reviewed by the Principal Investigator. If the sample was determined to be undisturbed, then the sample was processed. Prior to the removal of sediment, a photograph was taken of the surface to document levels of disturbance.

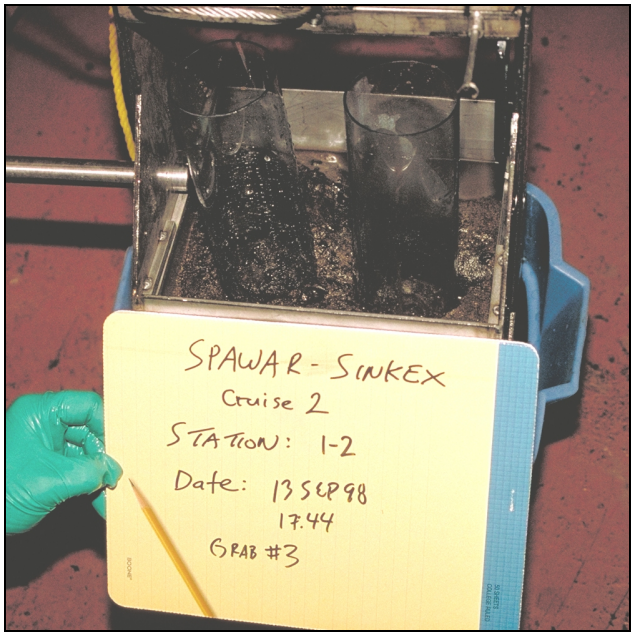


Figure 2-24. Boxcore sediment sample with core tubes.

Sediment sub-samples for chemical analysis were collected, as composites from the top three cm of the sample, with Kynar®-coated aluminum scoops. The scoops were built with three 3-cm deep sides to provide consistent sampling. Care was taken to avoid the sides of the boxcore during sub-sampling. Sediment for analysis was transferred into pre-cleaned clean borosilicate glass for organic chemistry and pre-cleaned HDPE plastic for metals analysis.

Samples for PCB depth profiles and age dating analysis were collected into 6.4 cm diameter by 23 cm long polycarbonate tubes (this was done only on selected samples). The pre-cleaned tubes were inserted slowly into the sediments using gentle oscillating motion to minimize disturbance and ensure that the core liner completely penetrated to the bottom of the boxcore. Once the liner was in place, the polyethylene top cap was placed immediately to minimize potential contamination by stack gases from the survey vessel. The sub-sample was then lifted from the sediment, and the bottom cap was set in place. Air space was left in the sub-sample to allow for expansion during freezing. The outside of the core tube was cleaned with deionized water, and the caps lined with Teflon® film, and then secured and sealed with electrical tape. The sub-sample was labeled, measured, and photographed. The top of the sub-sample was marked to ensure that it was stored in the sample freezer in the vertical position prior to being shipped to ADL.

After collection of the chemistry sub-samples, sediment for toxicity and bioaccumulation testing were removed from the same compartment as the sediment chemistry subsamples with the pre-cleaned Kynar®-coated aluminum scoop and placed into clean, double-layered polyethylene

sample bags. These sediment sample bags were labeled, placed in polyethylene tubs and moved into the vessel's refrigerated storage room at a temperature of 4 °C. A minimum sediment volume of 11 L was collected for toxicity and bioaccumulation testing at 14 stations. Each complete boxcore provided approximately 1.5 L of total subsample given a target sampling depth of 3.0 cm. Approximately seven small boxcores from a single station were required to provide sediment for toxicity and bioaccumulation tests.

Sediments for each infaunal analysis sample were collected from one small boxcore. Sediment for infaunal analysis was screened through two nested sieves. The first sieve was 1 mm mesh, and the second was 0.5 mm mesh. A gentle spray of seawater was used to separate the sediments from the organisms. Infauna were carefully collected from the screens, and placed into labeled jars. Organisms were relaxed with magnesium sulfate in seawater for approximately 10 minutes. After 20 minutes, the organisms were fixed in a solution of 10 percent-buffered Formalin. Within 14 days after collection, organisms were transferred from formalin into 70 percent ethyl alcohol for long-term storage and sample analysis (see Section 2.5.4.3).

2.5.2.4. Sedimentation Rate Sampling

The SINKEX project had a need to determine the natural sediment accumulation rates for the study site to determine the sediment strata most likely to contain contaminants that may have leached from the ex-AGERHOLM. Sedimentation rates from other areas of the Southern California Bight, such as the San Pedro and Santa Monica Basins, were low and ranged from 0.49-0.82 g/m^2/day for water depths between 700 and 850 m. (California Basin Studies summarized by Eganhouse and Venkatesan 1993). Sedimentation rates relate to the amount of material that potentially accumulates upon the bottom, which could bury the strata that might contain contaminants that leached from the target vessel (ignoring the effects of bioturbation or current dispersal). This section describes the results of deploying sediment flux traps at the study site.

Sediment flux traps were constructed from polycarbonate clear tubing with a diameter of 5.7 cm and 0.85 m in length. The tubes were baffled on the open end with a 1-cm grate to minimize resuspension within the tubes and to prevent large organisms from entering the tubes. A concrete base was constructed to hold the sediment tubes and to provide stability during the deployment. These concrete bases were about 1 m in diameter and weighed about 85 kg. Each concrete base held four of the plastic tubes in the upright position placing the open end of the tube about 1.3 m from the sediment surface. Each settling tube was attached to the concrete base with a nylon clip attached to a large stainless steel spring allowing the settling tube to be bent over without breaking with the spring capable of returning and maintaining the settling tube in the upright position. Four sediment trap arrays, each containing four sediment tubes for a total of 16 tubes were built and deployed. Sediment traps were deployed during SINKEX Cruise I on August 4, 1997, and retrieved during Cruise II (September 8-17, 1998). The location and depth of each deployment site is noted in Table 2-8, and can be seen in Figure 2-25. Upon retrieval the tubes and their contents were preserved with 5% buffered formalin for later laboratory analysis. In the laboratory the total suspended solids (TSS) and total volatile solids (TVS) were determined using standard methods. TSS was determined for each settling tube by filtering through a weighed 0.5-micron glass filter and then dried at 103 °C. The filters were then cooled and reweighed, the difference between the filter weights is the amount of solids larger than 0.5 microns retained on the filter. The next step was to take the filters with the retained solids and place them in a muffle

furnace at 550 degrees centigrade for 24 hours. This heating step incinerates the organic material, which is driven off as CO_2 and water vapor. The filters were then again cooled and reweighed allowing determination of the proportion of solids burned off as total volatile solids (TVS). The remaining material is an indication of total suspended solid accumulated as particulate "rainfall" over a 13-month period for a specified area in the vicinity of the ex-AGERHOLM. The laboratory results were converted from the diameter of the settling tubes to values/m^2 using a scaling factor of 392.

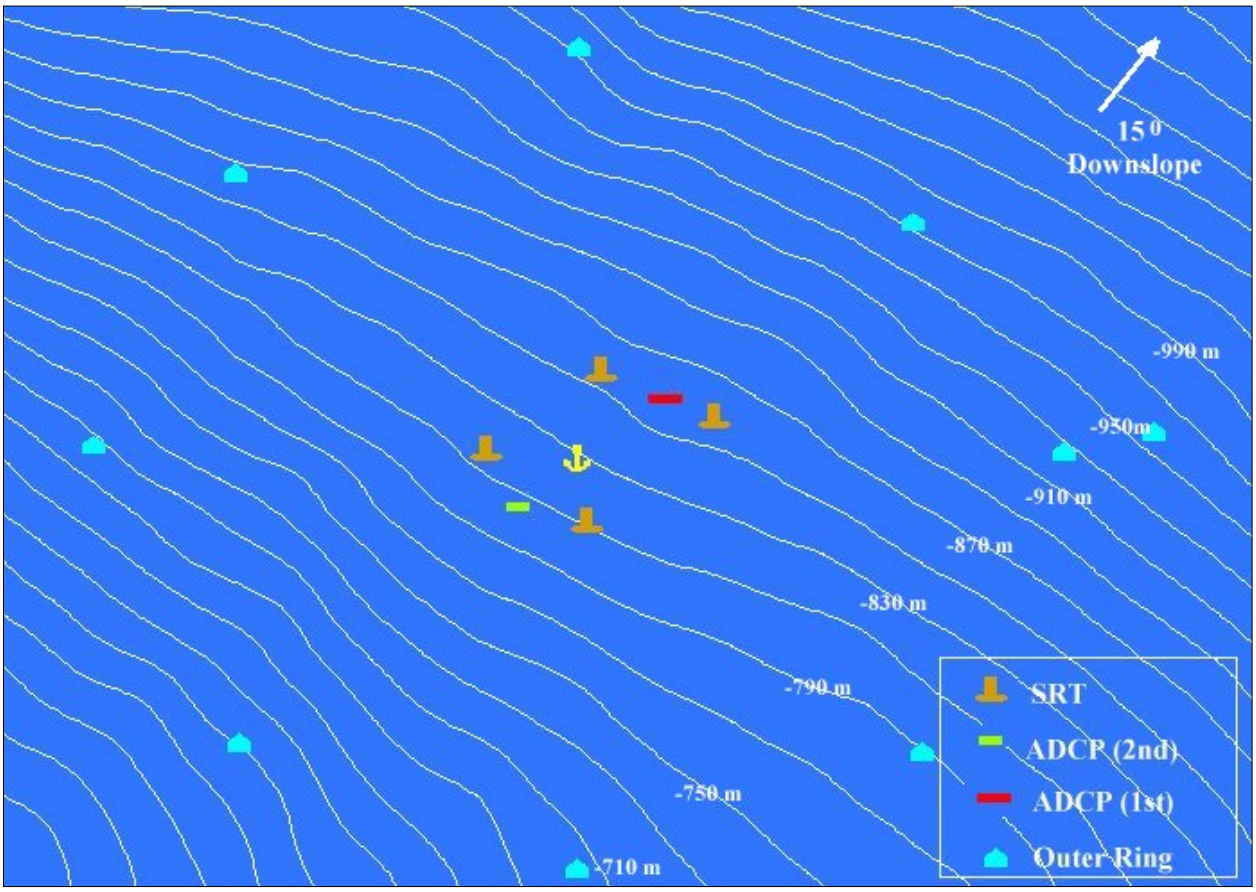


Figure 2-25. Location of Sedimentation Rate Traps and Acoustic Doppler Current Profilers, relative to the Outer Ring of Sediment stations 1 km away from the ex-AGERHOLM (marked by anchor).

Additionally, samples from eight 6 cm x 30 cm sediment cores were analyzed by gamma spectroscopy to obtain sediment geochronologies. Sedimentation rates were calculated using vertical profiles for "excess Lead-210" from each core. Lead-210 (^{210}Pb) is a naturally occurring decay product of Uranium-238 (^{238}U) with a half-life of 22.3 years. "Excess ^{210}Pb" is transferred to the oceans when the daughter radionuclide Radon-222 (^{222}Rn), a gas with a half-life of only 3.8 days, escapes into atmosphere from the land. The ^{222}Rn decays quickly to ^{210}Pb in the atmosphere and rains out into the oceans to be scavenged by particles and carried to the seafloor. The "excess ^{210}Pb" that is carried to the seafloor via the atmosphere decays to non-detectable levels in the sediment column over 4-5 half-lives, about 100 years.

In addition to the atmospheric source of ^{210}Pb, this isotope also is generated in the sediment column from the decay of Radium-226 (^{226}Ra). The ^{210}Pb formed in the sediment from the in situ decay of ^{226}Ra is referred to as "supported ^{210}Pb" (versus the "excess ^{210}Pb" from atmospheric

deposition). The amount of "excess ^{210}Pb" in a sediment sample is determined by measuring the total ^{210}Pb activity of the sample as well as the activity of ^{226}Ra, and then subtracting the activity of the "supported ^{210}Pb" (i.e., the ^{226}Ra activity) from the total ^{210}Pb. Once "excess ^{210}Pb" is buried in the sediment, its activity decreases over time by radioactive decay. Each halving of the activity of "excess ^{210}Pb represents a period of 22.3 years. Sediment accumulation rates can be calculated from the "excess ^{210}Pb" profile. In general, this technique can be used over a range of 4-5 half-lives (i.e., 90-110 years).

Secondary validation of sediment ages determined by the "excess ^{210}Pb" technique is preferred when possible. The fission-produced radionuclide Cesium-137 (^{137}Cs) has been used for this purpose in lacustrine, estuarine and shallow marine systems. However, ^{137}Cs is not commonly detected in deep sea environments.

Age dating of sediments using ^{137}Cs is based on known patterns of atmospheric fallout of this particle-reactive element. Cesium-137 is a radioactive isotope of Cs that is produced from fission of Uranium-235 (^{235}U). The half-life of the isotope is 30.1 years. Cesium-137 was first injected into the atmosphere during the testing of nuclear weapons in the early 1950s, and was not present on Earth before that time. When ^{137}Cs reaches the surface of the Earth as fallout, it is readily adsorbed onto sediment particles. Cesium-137 was first detected as fallout in 1954 and the peak fallout concentrations occurred in 1963. Thus, when viewing a vertical profile for ^{137}Cs in a sediment core, the depth just below where no ^{137}Cs can be detected is assumed to represent sediments deposited circa 1950 and the sediment depth of maximum ^{137}Cs activity is assumed to represent 1963. The thickness of the sediment layer above the 1963 and 1950 horizons can be used to determine sediment accumulation rates and assign approximate ages to different layers in the sediment core.

Because physical and biological processes can mix sediment layers, especially in the top 5-10 cm, use of two isotopes can help deconvolute mixing processes that would be more difficult to resolve with data for only one isotope. A variety of mixing models using data for both ^{137}Cs and "excess ^{210}Pb" are used to carry out this process.

Determination of the activities of ^{210}Pb, ^{226}Ra and ^{137}Cs was made by gamma spectroscopy. About 6 g of sediment from each 0.5-cm thick layer of the sediment core was placed in a 2-cm diameter, 5-cm long polystyrene vial. A rubber stopper was placed in the vial and sealed in place with two-part epoxy to prevent leakage of ^{222}Rn and disruption of secular equilibrium between ^{226}Ra and ^{210}Pb. The sample was set aside for three to four weeks to establish secular equilibrium and then the activities of the various radionuclides were determined by counting.

The sealed plastic vial was placed in an intrinsic Germanium (Ge) well-detector constructed by Princeton Gamma Tech and counted for two to three days or until sufficient counts of the pertinent radionuclides are obtained. The following peaks were monitored: Lead-210 (^{210}Pb) at 46.5 kilo-electron volts (KeV), Lead-214 (^{214}Pb) at 295.2 KeV and 351.9 KeV, Bismuth-214 (^{214}Bi) at 609.3 KeV, and Cesium-137 (^{137}Cs) at 661.6 KeV. The following daughter isotopes were used to determine the activity of ^{226}Ra: ^{214}Pb at 295.2 KeV and 351.9 KeV, and ^{214}Bi at 609.3 KeV. Detector efficiency and counting accuracy were standardized using standard reference river sediment 4350B from the U.S. National Institute of Technology and Standards and other sediment samples made available by the International Atomic Energy Agency (IAEA).

2.5.3. Details of Sampling Operations

Summaries of the individual SINKEX Cruises I through V are presented in the following sub-sections.

2.5.3.1. Initial Sampling Attempt (Cruise I)

During 1997, preparations were made to sample near the ex-AGERHOLM. This included preparation of a detailed sampling plan and the establishment of a peer review committee. The committee consisted of experts in the areas of ocean sediment/infaunal sampling, bioassay protocols, PCBs in the marine environment, sediment quality guidelines, and sampling design. Recommendations from this committee were reviewed, with the majority being incorporated into the workplan. The initial sampling approach was to employ a 0.25 square meter "large boxcore" coupled with a positioning "control vehicle" (launched and recovered from an oceanographic research vessel) to obtain intact sediment samples in a series of concentric rings expanding outward from the hulk (Figure 2-17). After recovery on deck, these boxcore sediments were to be subsampled for chemistry, age dating, bioassay material and benthic infaunal information. Thirty-two stations were to be sampled with two boxcore drops per station, in a period of twelve days supported by 24-hour operations. Sampling commenced on July 31, 1997 and continued until the 6th of August with eight sediment boxcores out of 30 attempts being successfully recovered on deck. However, only one (the first) could be certified as having maintained its integrity from the time it was sampled to the time that it was ready to be subsampled. Three sablefish were also collected from the vicinity of the hulk via long-line rod and reel (opportunistic sampling). Four sediment deposition rate collection devices and one acoustic Doppler current profiler (ADCP) were deployed in the vicinity of the hulk.

The primary issues leading to the failure of the majority of the large sediment boxcore/control vehicle sampling efforts were (in order of importance):

1. The sampling sites contained consolidated material (rock) that prevented the boxcore shovel from sealing against the box of the sampler, thus causing a partial or full washout of most of the sediment samples from the boxcore before it was recovered on deck.
2. Inclement weather and sea state reduced the number of operational days. Weather conditions varied between a Sea State 3, (a moderate sea, waves 3-5 ft, crests beginning to break, wind gentle to moderate 7-15 kts) and a Sea State 4, (a rough sea, moderate waves 5-8 ft, many white caps, some spray, wind moderate to strong breeze 14-27 kts). At no time during the sampling interval were the seas calm. Additionally, an intermittent 10-15 ft swell was experienced during the time on station. Sea State 4 represented both the planned and actual edge of the operating envelope for the safe launch and recovery of the sampling equipment. Sea state also affected station-keeping since the vessel being used had no dynamic positioning (automatic station keeping) capability, and was thus unable to hold station long enough to eliminate the catenary in the cable attached to the boxcore.
3. Finally, the effort was terminated due to the failure of the ship's primary over-the-side equipment-handling crane that had been operating close to its maximum load and extension capability for the entire week.

Following the initial and aborted sampling opportunity, it was deemed unlikely that using the available state-of-the-art large boxcore sampling methodology would result in usable samples on

the Inner Ring (within 3 meters of the hulk), primarily because of sampler placement problems. It was also considered unlikely that undisturbed samples, whether near or far away from the ship, would be recoverable using traditional large boxcore sampling methodology, primarily because of the heterogeneous bottom sediment and the attendant boxcore washout issue.

After a thorough technology assessment, it was concluded that adequate sampling technology was not available to allow the collection of undisturbed samples at this specific site. Specifically, it appeared that due to the nature of the bottom sediments near the ex-AGERHOLM, traditional techniques to bring intact sediment samples to the surface without compromising further subsampling did not exist for this site. It was decided to abandon the conventional boxcore technology and to handle the sampling issue with a two-pronged sampler approach.

The two-pronged effort was selected as the best means to obtain samples next to the ship (near field) and far from the ship (far field). For the near field samples, methodology was developed to use Remotely Operated Vehicle (ROV) technology and a set of small 9-liter boxcore samplers, with the assistance of a video data link, cameras and lights. For the far field samples, an innovative piston sampler was designed to be deployed via a standard surface vessel/over-the-stern (A-frame) method.

During the first effort, the assets of the Navy's Deep Submergence Unit (DSU) were scheduled to support the ROV work. A prototype design for the near field sampling effort was selected in June 1998. Fifteen 9-liter boxcore samplers were constructed and included in the ROV SCORPIO suite of tools. The next available sampling opportunity, 5-18 September 1998, utilized the M/V LANEY CHOUEST as the sampling platform. This near field sampling effort was planned to obtain sediment samples within 3 meters of the ex-AGERHOLM hull at eight different locations around its periphery (the Inner Ring, Figure 2-18). Additional cruise objectives included recovery of the ADCP and the sedimentation rate samplers. Current and sedimentation rate data would be used to further define the site characteristics and would be an instrumental part of the post-cruise analysis.

For the second effort, a prototype piston sampler was designed and constructed by the Marine Physical Laboratory (MPL), the primary field support facility at Scripps Institution of Oceanography (SIO), for the far field sampling effort. This sampler completed prototype testing at the SIO pier; and a group of three of these piston samplers were constructed as a unit to support sediment field testing/sampling in the vicinity of the ex-AGERHOLM.

2.5.3.2. Field Sampling Cruise II

The second sampling attempt was conducted 5-18 September 1998 onboard the M/V LANEY CHOUEST in company with the Navy's Deep Submergence Unit (DSU) and the Remotely Operated Vehicle (ROV) SCORPIO. The weather was generally much more accommodating than during Cruise I and effected operations for only two out of the 11 days on station. The initial launch was intended to videograph the site and to place a marker at each nominal sampling station. The ROV was deployed with eight station markers and was successful in placing the markers around the hulk as close as possible to the hull, spaced about 68-95 feet apart from one another. The two aft station markers (7 & 8) near the stern were about 25 ft apart. Physical extensions of the hulk, other protruding metal and equipment, and miscellaneous debris were necessarily avoided to prevent potential entanglement of the ROV. Station Marker 7 was placed furthest from the hulk, (starboard side, aft) approximately 41 ft from the hulk to the southwest, due to such interferences. Average distance to the hull for the sampling over all dives was as

follows: (1) port side, amidship - 9 ft (2) port side, forward break - 10 ft, (3) port side, bow - 10 ft, (4) starboard side, bow - 8 ft, (5) starboard side, forward break - 8 ft, (6) starboard side, amidship - 11 ft, (7) starboard side, truncated stern - 41 ft, (8) portside, truncated stern - 13 ft. Station 9 (single reference station) was located 225 degrees (true) at 1 km from the position of the hulk. The initial video survey also located the northern Sedimentation Rate Sampler and the ADCP. The ADCP was subsequently recovered, data was downloaded once onboard ship, and the ADCP was prepared for redeployment.

Two major adjustments were made to the mini-boxcores while at sea during the field testing period. Initial difficulties arose while the ROV was being used to handle the transfer cassette (a specially made fiberglass/metal container for six small boxcore samplers), and again while the ROV manipulator arms were employed to pull one boxcore from the cassette without interfering with the cassette's four-point bridle. The first modification was to add Tygon® tubing around the bridle cable to prevent entanglement and provide easier access for the ROV arms. Continued operations indicated that this fix was still not satisfactory, because the ROV was still unable to easily extract the small boxcores from the cassette, and the method of recovering a full cassette turned out to be too time-consuming. A second modification was performed by cutting the cassette in half to allow it to hold only three boxcore samplers at one time, with an objective of attaching this smaller version of a cassette directly to the front of the ROV. When this alteration was judged to be impractical because of the anticipated problems associated with securing the "half" cassette to the ROV, three plastic "milk carton" containers were configured and fastened to allow the ROV to carry, extract and replace three small boxcore samplers during each dive. The cumulative effect of this alteration was that twice as many dives were needed to achieve the same sampling opportunity. While the cassette issues were identified as needing attention prior to sampling, the only option by that time was to proceed with the sampling event and attempt to fix the deficiencies at sea.

In general, the ROV operations consisted of the launch of the ROV, followed by the dive and payout of the tether. The dive to 2,750 feet typically took 30 minutes, with the target being located by sonar until it was picked up visually. The ROV was then maneuvered to the sampling location and landed on the bottom. The ROV was ballasted so that it had a slight positive buoyancy during the sampling evolutions. The three samplers were then extracted sequentially from the improvised milk carton containers and used to obtain a sediment sample. The full samplers were then placed back in their respective containers for the recovery. Average time on the bottom was quite variable due to difficulties experienced in obtaining the samples; the minimum time period was approximately 15 minutes for three "easy" samples while a maximum period was as much as 2 hours for one to three "difficult" samples. Recovery time from the bottom to the surface was approximately 30 minutes. The ROV was remotely controlled by a pilot (responsible for "flying" the vehicle) and a co-pilot (responsible for the sonar and actual manipulation of the "arms"of the craft). The whole event was controlled by a watch supervisor who also managed the amount of tether paid out, the direction in which it was tending and the dynamic positioning (station keeping) of the launch vessel. Upon recovery of the ROV, the samplers were removed from their containers, transported to the wet laboratory, examined to determine if they were "intact", and then processed for subsampling.

The DSU operated the ROV as a "one-dot" transponder field, specifically a single transponder that was strapped to the back of the ROV. The transponder could then be interrogated by the mother ship resulting in a slant range and bearing that was converted to positional data with an

error of ±30 yards. However, in order to obtain an accurate navigation fix, the ROV was required to shut down all hydraulic systems for a period of up to several minutes, requiring that the vehicle have enough weight externally to hold it on the bottom. This was rarely possible for operational reasons, and due to the significant error associated with this approach, it was decided to rely on other navigational methods to obtain positional data for the sampling locations. Specifically, placement of the sample markers was visually correlated with a feature on the existing superstructure and manually plotted on a Damage Control Chart of DD 743 (ex-SOUTHERLAND) supplied by NAVSEASYSCOM (PMS 335D). Scale dimensions from the chart were then utilized to determine sampling positions along the hulk. Sonar readings were obtained for each sample to provide distance from the hulk.

During cruise III, it was possible to obtain an accurate position for recognizable locations on the horizontal deck surface of the hulk (see Cruise III, Section 2.5.3.3) via slow-scan video coupled with the interrogation of a 4-dot transponder field followed by integration with the mother ship's GPS fix. Accordingly, with the accurate positioning of the hulk, it was then possible to convert all Cruise II measurements to LAT/LONG using appropriate software in order to accurately determinate sampling locations for the Inner Ring.

2.5.3.2.1. Initial Dive and Sediment Round-One Sampling

The objective of the initial dive was to retrieve sediment from a reference site from the Outer Ring (1 km from the hulk). This approach was intended to provide the ROV operators with the opportunity to manipulate the boxcores at a site remote from the hulk. The bottom at this site was observed to be very hard and the sampling difficult in this area. Subsequent dives to the bow of the ex-AGERHOLM revealed that the sediment near the hulk was more penetrable. Around-the-clock operations began with round-one sampling (on the Inner Ring) on the second day at sea. A sample judged to be suitable for chemical analyses was required to exhibit an undisturbed water/sediment interface when examined in the wet laboratory after retrieval. Samples that were disturbed during the sampling process with compromise of integrity at the water/sediment interface were used in toxicity tests only. Round one resulted in 27 suitable samples. Sixteen were designated for bioassay testing, five were designated for chemical analyses, and six were "non-samples," having failed due to excessive disturbance or rock interference. The principal investigator made the determination on all samples selected for chemistry analysis.

2.5.3.2.2. Fish Trap Deployment

A total of seven fish traps were used for collecting fish tissue samples. Two fish traps were set near the ex-AGERHOLM (within 10 feet) on the bottom near the stern of the ship on opposing sides (north and south of the hulk). Another set of twin traps were initially set on top of the bow and the last three fish traps were placed at a fish reference site bearing 330 degrees (true) from the hulk at 3.8 nm distance and 4,200 ft. depth. Each trap was baited with mackerel and squid with the intent to leave them in place for 2-3 days before retrieving them. During this period, the first two sediment rate samplers were retrieved (north and south).

2.5.3.2.3. Sedimentation Rate Traps (SRTs), Acoustic Doppler Current Profiler (ADCP), Fish Trap Collection and Sediment Round-Two Sampling

Once chemistry sampling at all nine stations was completed, round-two sampling commenced. Intermingled with the sampling of round two, three other objectives were accomplished: (1) retrieval of the fish traps, (2) retrieval of the remaining SRTs, and (3) the ADCP redeployment.

The two fish traps were recovered from the north side of the hulk, containing two black cod and two large crabs, then were re-baited and replaced on opposite side of the hulk. Upon recovery of the three reference fish traps, it was discovered that they also contained two black cod. With respect to the SRTs, the recovery involved a flying catch of the recovery hook on the SRTs in order that the bottom sediment not be disturbed. This was followed by the recovery of the ROV while paying out 3,000 ft of Kevlar® line attached to the SRT. The line was then detached from the ROV upon recovery and walked up the side of the ship to the amidships D-frame and passed through the block on the D-frame and then to the capstan on the Clyde winch. The Clyde winch was then used to heave in the attached sediment rate sampler. The last of the four sediment rate samplers was recovered, and the ADCP was then redeployed at a bearing of 225 degrees (true), 186 yds from the hulk, at a depth of 2,680 ft. Round two resulted in 30 samples, with twenty-five samples designated for bioassay tests, one sample designated for benthic infaunal assessment, and the last four samples designated for chemical analyses.

2.5.3.2.4. Fish Traps, InterOcean S4™ Current Meter and Sediment Round-Three Sampling

Round three sampling began by retrieving the four fish traps near the hulk. The traps had caught six additional black cod and one crab. The fish traps placed on the actual structure of the hulk did not catch any fish. Although hagfish were observed feeding on the hulk, the general feeding habits of other fish and crabs were observed to occur on the ocean floor rather than on the hulk's structure. Additionally, the InterOcean S4™ current sampler was deployed and placed near the hulk's bow, approximately 1.5ft from the bottom. The S4™ was placed 45 ft from the anchor chain and 30 ft from the hulk in order to avoid magnetic interference. Finally, two fish traps were redeployed in an effort to obtain more reference black cod. This site was at a bearing of 225 degrees (true), and 3.8 nm from the hulk in 2,120 feet of water to provide a similar depth comparison to that of the hulk. The outcome of round three was 27 successful samples. Twenty-one samples were designated as bioassay material, and six samples were to be used for infaunal material.

Cruise II was terminated one day early in response to an emergency requirement for DSU's salvage support capability elsewhere. Due to this unanticipated departure from station, sampling at two sediment stations (5 and 6) amounted to less than the target amount of 11.0 liters, and the two fish traps had to be abandoned at the fish reference site. The cruise was terminated on Thursday, 17 September 1998. Sediment and tissue chemistry samples were transferred to AD Little and Florida Institute of Technology for organics and metals assays, respectively. Sediment, infauna, and epifauna samples were provided to MEC, Inc. Sediment was also transferred to MEC, Inc. for use in acute, chronic and bioaccumulation bioassays.

2.5.3.3. Field Sampling Cruise III

The third sampling attempt was made during the period 11-18 November 1998 onboard the R/V ROGER REVELLE in company with the SIO MPL personnel, their Deep Tow Side Scan Sonar, their Payload Control Vehicle and a to-be-field-tested Piston Corer. On the way out to the sampling site, the new coring device was tested SE of San Clemente Island, resulting in three successful sample grabs of sandy silt. Upon arrival at the site of the ex-AGERHOLM, the weather was found to be calm and did not hinder mission objectives.

2.5.3.3.1. Transponder Field/Navigation

Before sampling attempts, MPL personnel laid down an underwater acoustic transponder field to increase the accuracy of the navigational data that would be recorded during each sampling attempt. Between 0100 and 0300 on 12 November 1998, four transponders were placed at cardinal points, each moored about 1 km from the hulk and about 100 m above the ocean floor. The calibration between the ship's GPS and the field transponders required approximately 10 hours and was carried out between 1200 and 2200 on 12 November 1998.

Between 1830 and 2200 on 14 November 1998, the Deep Sea Instrument Interface (DSII) was used to obtain accurate geographical coordinates for recognizable locations on the deck of the hulk to assist in calculating past sampling locations (Cruise II, see Section 2.5.3.2) that had been previously recorded by qualitative and relative visual descriptions.

2.5.3.3.2. Side-Scan Surveys

The DeepTow Fish 6 instrument package (including side-scan sonars) was launched at approximately 0500 on 12 November1998, and was towed at a lateral distance of 500 m from the hulk in a clockwise manner. The intent was to decrease the distance from the hulk on successive tracks, ultimately closing to within 75 m. This close pass was necessary for an atttempt to create a detailed and accurate side-scan mosaic of both sides of the hulk with a view also from the stern. Side scan efforts yielded an excellent pass on the port side at 137 m. However, shortly thereafter, a power failure developed inside the tow fish caused by a leak from a casing penetration seal and resulting in a power supply failure. Side scan operations were subsequently terminated at 1000 on 12 November 1998 and the decision was made to commence coring operations.

2.5.3.3.3. Sediment Sampling Effort

The new coring device, designed by scientists at SIO/MPL, was modified throughout this trip to overcome sampling difficulties associated with the hard bottom type found around the ex-AGERHOLM. The original design consisted of a single hydraulic piston cylinder that was constructed to penetrate the substrate by force utilizing the combined weight of the sampler frame and the cylinders. The production model incorporated this field-tested design into a tri-sampler capable of taking three samples in one location simultaneously. Once buried in the sediment, the piston coring devices were to be extracted by a hydraulic piston lift mechanism, and sealed closed by spring-triggered doors. The coring cylinders were 18 inches in diameter and 10 inches in depth. It was learned during successive equipment deployments that increasing the device's descending momentum would facilitate penetration. The rate of descent near the bottom prior to penetration of the sediment was controlled and modified throughout the experiment, starting with about a 10 m/min, and culminating with about 40 m/min. Since momentum is the product of both velocity and mass, additional weight was added to the sampler frame when successive cores failed to adequately penetrate the sediment. Weights were added until the operating limit of the crane was reached. The final sampling design modification was made when the mass per unit area of penetrating device was maximized through use of the original prototype single corer mechanism as a backup employing the same mode of operation. The final outcome of the sampling effort was an inability to obtain any satisfactory core with either device at the ex-AGERHOLM site.

2.5.3.3.4. Video Monitoring of Sampling Events

Cameras and lights mounted on the coring device were originally oriented to view the sediment as the device approached the ocean floor. The purpose of this design was to detect and avoid large rocks before they caused sampling problems. During the cruise, the cameras were re-mounted to view the coring device and its interface with the sediment. This alteration was made to reveal where sediment was escaping, since previous attempts indicated this was a problem. Review of the videotapes upon sampler retrieval aided in assessment of: (1) the sediment conditions, (2) the potential reasons behind the sampling difficulties, and (3) modifications or manipulations necessary to ensure successful retrieval of a satisfactory sediment core. A satisfactory core for this project would be defined as "a sediment sample that had been taken and transferred undisturbed from the bottom to be placed on deck (with the sediment/water interface undisturbed) in such a fashion as to allow subsampling." Post-cruise analysis determined that while penetration was in fact occurring in the hard bottom sites, that retraction of the piston(s) tended to disrupt the core(s) and the sliding trap door did not operate fast enough to contain the core resulting in loss of sample.

Much discussion was spent in proposing solutions to the penetration and retrieval problems demonstrated by the piston sampler(s). However, modifications to the sampling device resulted in only limited improvements. A gravity core was manufactured onboard and deployed on 17 November 1998 in an attempt to demonstrate how far a conventional method for deep ocean sediment assessment (albeit for minimum sediment volume retrieval) could penetrate the sediment at this particular site. The gravity core resulted in a sample about 8 cm in diameter and about 18-20 cm in depth. A second gravity core yielded no sample, presumably due to rock obstacles.

On 15 November 1998, with several proposed solutions for sampling with the tri-sampler being ineffective, the decision was made to attempt single piston core sampling several times. When no samples were collected in this fashion, the principal investigator terminated the scientific cruise.

2.5.3.4. Field Sampling Cruise IV

Cruise IV was conducted 20-29 September 1999 using the M/V KELLIE CHOUEST with the ROV SCORPIO 2. Elements of Submarine Development Squadron 5, Deep Submergence Unit, were deployed to man and operate the ROV. The Officer in Charge (OIC) and Safety Officers (SO) provided oversight for all ship navigation, launch, recovery, and maintenance on the SCOPRIO 2. A 24-hour, three-section watch rotation was established with a Dive Supervisor, Pilot, Copilot, Winch Operator, and two Line Handlers on duty at all times. The satellite-based Global Positioning System (GPS) provided the navigational reference. The dynamic positioning capability of M/V KELLIE CHOUEST allowed the ship to maintain position within a ten-meter box. The ship, in low sea states, maintained an accuracy of a few meters, and was able to maintain position within the 10-meter box up to Sea State 4. Beyond Sea State 4, all diving operations were curtailed for safety considerations. The SCORPIO 2 consisted of tethered, sledge-mounted ROV with two manipulating arms, hydraulic controls, lighting system, camera system, and sonar system (Figure 2-20). A launching crane, winch and cabling system, and take-up spool with 5,400 ft of Kevlar-coated tether was installed on the ship. This system was used to launch SCORPIO 2 and retrieve sediment samples from the desired locations near the ex-AGERHOLM and reference locations. The SCORPIO 2 was also used to deploy and retrieve fish traps and collect biological specimens, when possible. Upon completion of sampling, a sampling

grid containing all sampled positions accurate to within two meters near the hulk and 10 meters for the reference stations was generated. Since the water depth at the ex-AGERHOLM site is about 2,750 ft (838 m), the round-trip wire time of about one hour (wire speed of about 30 m/min) was expected.

New sampling equipment was developed specifically for Cruise IV. Based on difficulties experienced with sampler penetration and closure on the bottom during previous cruises, design changes were made to the small boxcore samplers used by the SCORPIO ROV during Cruise II. Prior to this cruise, prototypes of the sampler were built and tested by the DSU crew. The same basic design, with a swinging shovel/door to seal the box from underneath, and a sliding lid for easy sediment recovery without disturbance of the surface layer, was used for this cruise, but with the following modifications: the three stainless steel samplers that were ultimately constructed each had a gear and crank mechanism to provide additional force in closing the shovel, and a new pi-shaped handle (replacing the former T-shaped handle) for easier manipulation by the ROV's arms and claws (Figure 2-21). Because the new ROV arms were less capable than the former ones, only two samplers (vice three) could be carried down to the bottom by the ROV in the baskets (milk crates), which were fastened to the front of the ROV. Nevertheless, the sampling procedure was essentially the same as before. Once settled on the bottom, one of the ROV arms would grab a handle on the top of the sampler, lift it from the basket and position it into the sediment in a way so as not to disturb the sediment surface. Once the sampler was embedded in the sea floor, the second arm would grab a handle connected to the gear mechanism located on the sampler and rotate it one full turn. This action would close the shovel of the sampler and seal the sample within the box. After the sampler was successfully closed, the first arm would lift it out of the sea floor and gently place it back into the basket.

Additionally, a special sampler attachment was designed and fabricated for this cruise to facilitate the sampler penetration into the hard ocean sediment that was known to exist in certain areas on the reference ring. It was manual jackhammer device, made of stainless steel, configured to be coupled with the sampler once the two devices were transported to the ocean bottom in separate ROV baskets (Figure 2-22). It was composed of a weight, springs, a cranking device and handles for the ROV to manipulate. Once the sampler and jackhammer reached the bottom floor, the idea was for the ROV operator to manipulate one arm to lift the jackhammer by a T-handle on top, while the second arm kept the sampler from moving. The two devices were to be coupled together after the jackhammer was placed on top of the sampler, with the ROV operator ensuring that the interlocking components matched up for the junction. The second arm then attempted to grab a side handle attached to the cranking mechanism. As that handle was rotated, the weight was cocked against the springs, and then released to create a strong downward force on the sampler. However, it was not necessary to utilize this device to obtain any samples in that sufficient soft material was located at four sites on the outer ring. At the single site where sampling a hard surface was attempted, the device was determined to be very unwieldy.

It was also determined that water samples (for PCB analysis) would be collected with a five-liter Niskin bottle attached to the ROV. It was rigged so that extending one of the ROV arms, when at the desired depth, could trigger it.

Obtaining sediment samples from the far field (reference) stations was the primary objective of the SINKEX IV cruise. Eight far-field stations on cardinal headings had previously been defined on a ring 1,000 meters from the hulk of the ex-AGERHOM. The exact geographic locations of

the stations were ultimately determined by the ability to collect an intact and undisturbed sediment sample from the sea floor. A full compliment of chemistry, bioassay and infaunal sediments was collected at Stations 4-5, 4-2, 4-3.1, and 4-1, in that order. Station 4-3.1 was established and sampled 200 meters east of the predefined Station 4-3, because of the fouling risk posed to the ROV by an abandoned polypropylene line in the vicinity of Station 4-3. Water samples were collected at Stations 4-2 and 4-5.

Fish traps used on this cruise were the 2x3x4-foot wire mesh cages used previously on Cruise II. At each deployment, two traps were used and marked by a benthos sphere floating about six feet above the cages and marked with reflective tape. The whole apparatus was weighted with a 40 lb. link of chain. Two fish traps each were deployed at two fish reference stations established and sampled in previous cruises, located 3.8 nm NW and 3.8 nm SW of the ex-AGERHOLM, respectively. Two other fish traps were deployed at a third station located 3.8 nm E of the ex-AGERHOLM. Fish tissue samples were collected from both of the SW and E stations. Sediment sampling was attempted at Stations 4-3, 4-4 and 4-8, but no samples were collected at these sites. Station 4-3 was abandoned because of the polypropylene line discussed earlier. Station 4-4 was abandoned because the bottom was too rocky to sample. Station 4-8 was sampled to test out the "jackhammer" modification to the sediment samplers. Station 4-8 was chosen for the test because of prior knowledge that there was a relatively hard rock and sand bottom. The testing of this attachment resulted in some suggested modifications for improved performance - proof of concept was achieved but no actual samples. The original cruise sampling plan called for sampling one near-field station for sediment and water, but problems with the ROV's sonar during the latter part of the cruise prevented any dives from being conducted near the hulk. The sonar problems also prevented the retrieval of the ADCP and InterOcean S4™ current meters, both of which had been deployed on a previous SINKEX cruise.

As with the M/V LANEY CHOUEST, M/V KELLIE CHOUEST maintained position using a dynamic positioning system (DPS). Position was determined with a differential global positioning system (DGPS) with an estimated accuracy of ±10 meters. The ship's position was maintained primarily with a Z-drive bow thruster extending below the hull and able to rotate 360 degrees. When the weather got heavier, the two main engines were also required to maintain position.

Once on station the ROV was lowered over the side and monitored as it descended to the bottom. A pilot and copilot who sat at a control console on the ship controlled the ROV. The pilot controlled the movement of the robot. By manipulating a joystick, he was able to make the ROV fly forward, come up in the water column, move from side to side and then land softly on the bottom at the sampling location. During decent the ROV's thrusters thrust down to minimize tether drift and assured that the ROV came down as straight as possible to the sample location. On the bottom the pilot guided the ROV to the sampling location. On the initial dive at a station, a station marker was brought down by the ROV and placed on the bottom. Subsequent dives attempted to locate samples within 20 ft of the station marker. Care was taken to not disturb the bottom before the samples were taken.

After the ROV was positioned at the desired sampling location, the copilot, using specially made controls activated the manipulator arms of the robot. The copilot used the camera to guide the hand to reach down and pick up the sampling shovel out of the milk crate and place it gently in the mud at the desired sampling location. The copilot then applied pressure to the shovel until it sank all the way down in the mud. Holding the sampler steady, the copilot reached over with the

other arm, and pulled out the cotter pin that kept the shovel open, turned the gear drive to close the shovel, and collected an intact sample of mud that included the upper 3 cm of undisturbed mud. Then the copilot replaced the sampler and was ready to repeat the evolution for the second sample. When sampling was finished, the diving supervisor gave the command to wind in the cable and the pilot started the accent to the surface. After the ROV reached the surface and was recovered back on deck, the sediment samples were inspected by the Principal Investigator and Duty Scientist on watch to assure that the samples were intact, undisturbed, and representative of the top 3 cm of the bottom. Any samples which were compromised, from washing out of the sampler during the transit to the surface, excessive mixing of the surface layer, or appearing to be unrepresentative of the surface of the bottom, were rejected and not used for quantitative analysis (analytical chemistry, bioassay, or infaunal analysis). Rejected samples were screened for qualitative analysis of biota only.

Position of the ROV in relation to the ship was estimated to be within about 100 meters of the ship position at any one time, based on the GPS accuracy and the maximum horizontal swing circle created by the length of tether underwater. This accuracy was determined to be sufficient for sampling of the 1,000-meter reference ring. There was no onboard navigation capability for the ROV on this cruise.

Initially, sediment stations on the outer ring were marked with a station marker consisting of a benthos sphere tied to about six feet of polypropylene line and weighted with a 40 lb. link of chain. The ROV sonar was to detect this marker on some of the subsequent dives. Following the location of the marker, samples were then taken at an undisturbed site within a 20-foot radius of the marker's location. However, because of ROV sonar problems, finding the station marker proved to be difficult. One dive, which lasted eight hours, took this much time due to the search operations required to find the marker. After this, the markers were evaluated as being inefficient, and they were no longer used because of the ROV sonar difficulties.

During the penultimate dive, the ROV, with its ineffective sonar, was having a difficult time locating the far east fish traps for retrieval. The last dive terminated at 0815 on 29 September 1999. Equipment was secured and the ship returned to San Diego. Sediment, tissue and water chemistry samples were transferred to A.D. Little and Florida Institute of Technology for organics and metals assays, respectively. Sediment infauna, and epifauna samples were provided to MEC, Inc. Sediment was also transferred to MEC, Inc., for use in acute, chronic and bioaccumulation bioassays.

2.5.3.5. Field Sampling Cruise V

The fifth and final sampling attempt was made during the period 13-18 November 1999 aboard the M/V KELLIE CHOUEST, again in company with the Navy's Deep Submergence Unit (DSU) and their Remotely Operated Vehicle (ROV) SCORPIO. The weather was again calm except for a two-hour suspension of operations due to high winds.

The primary objectives of the SINKEX V cruise were as follows:

- Obtain a single field duplicate, representing a full compliment of chemistry, bioassay and infaunal sediments, all to be collected at an undisturbed location near Station 1-6. This was to assess anomalous data obtained from the Cruise II analyses performed on Station 1-6.

- Obtain two inter-cruise comparison samples for comparison to samples taken on previous cruises: Chemistry and infaunal samples were to be taken near both Stations 1-3 and 4-6 to confirm lack of chemistry or toxicity hits from relatively "clean" near-field (1-3) and reference (4-6) stations analyzed from Cruise II.
- Retrieve both the S4™ and ADCP current meters from their near-field stations adjacent to the hulk and download long-term (several months) current data.
- Collect additional fish specimens at the reference and hulk sites, emphasizing the need to collect more fish at the reference locations: Two fish traps were to be deployed at a reference station located 4 nm W of the ex-ALGERHOLM. Two other fish traps were to be deployed at the rear of the hulk, in between Stations 1-7 and 1-8. Fish tissue samples were to be collected from both of the stations.
- Sample the water column at both reference and near-field stations. Standard 5-L Niskin bottles were to be used to collect discrete water samples for laboratory chemical analysis during four separate dives near the hulk.
- Standard CTD data (salinity, temperature, depth, oxygen) were to be obtained throughout the ROV deployments in order to assess basic water characteristics of the water column. The CTD was to be deployed during 10 dives close to the hulk, and one dive at the 4 nm fish trap site.

The necessity for Cruise V, was related to the ROV sonar casualty on Cruise IV. It was a direct continuation of the far-field sampling effort with some additional objectives. The stainless steel boxcores that were developed for SINKEX IV and successfully used during that cruise were also used during SINKEX V. The sampling method was exactly the same for obtaining samples. Once settled on the bottom, one of the ROV arms would grab a handle on the top of the sampler, lift it from the basket and position it into the sediment in a way so as not to disturb the sediment surface. Once embedded in the sea floor, the second arm would grab a handle connected to the gear mechanism and rotate it one full turn. This action would close the shovel of the sampler and seal the sample within the box. After the sampler was successfully closed, the first arm would lift it out of the sea floor and gently place it back into the basket.

Fish traps were 2x3x4-foot wire mesh cages, used previously on Cruises II and IV. Each cage was weighted with a 40-lb. link of chain and baited with two cans of fresh and frozen whole mackerel. A pinger was attached to one of the cages to assist in recovery. However, the traps were generally seen before a signal was ever received. As before, water samples were collected with a five-liter Niskin bottle attached to the ROV. It was rigged so that extending one of the ROV arms when at the desired depth could trigger it.

A new piece of sampling gear, a SEA-Bird™ CTD, was attached to the stern of the ROV in order to obtain an accurate water column profile. The CTD collected information constantly from deployment until retrieval.

ROV operations were conducted in similar fashion as previous SINKEX Cruises, which used DSU and SCORPIO to collect samples on the Inner Ring. Each numbered dive consisted of a launch of the ROV from the deck of the M/V KELLIE CHOUEST, paying out of tether to lower the ROV to the bottom, bottom operations to collect sediment or biological specimens, and finally paying in of the tether and recovery of the ROV on deck.

There were two major equipment failures while on station, on November 14, The ROV suffered a sonar failure at approximately 500 feet. The ROV was brought on deck and the problem was determined to be a ground in the tether. It took approximately eight hours to remove 15 feet of tether and re-terminate all of the connections. On November 17, the ROV had a hydraulic pressure malfunction - a leak in the port astern thruster hydraulic line - the dive was aborted, and the problem took approximately 6 hours to rectify.

As mentioned previously for cruise IV, M/V KELLIE CHOUEST maintained position using a dynamic positioning system (DPS). Position was determined with a differential global positioning system (DGPS) with an estimated accuracy of ±10 meters. The ship's position was maintained primarily with a Z-drive bow thruster extending below the hull and able to rotate 360 degrees. When the weather got heavier, the two main engines were also required to maintain position. Position of the ROV in relation to the ship was estimated to be within about 100 meters of the ship position at any one time, based on the GPS accuracy and the maximum horizontal swing circle created by the length of tether underwater. This accuracy was determined to be sufficient for sampling of the 1000-meter reference ring and the fish trap reference station. This accuracy, however, was not sufficient to record near-field stations, so visual reference (relative to ship markings and station markers deployed around the ship for the near-field sampling of the Inner Ring during Cruise II) was used for those dives.

The last dive terminated at 1805 on 17 November 1999. Equipment was secured and the ship steamed back to San Diego. Sediment, tissue and water chemistry samples were transferred to A.D. Little/Florida Institute of Technology for organics and metals assays, respectively. Sediment infauna, and epifauna samples were provided to MEC, Inc. Sediment was also transferred to MEC, Inc., for use in acute, chronic and bioaccumulation bioassays.

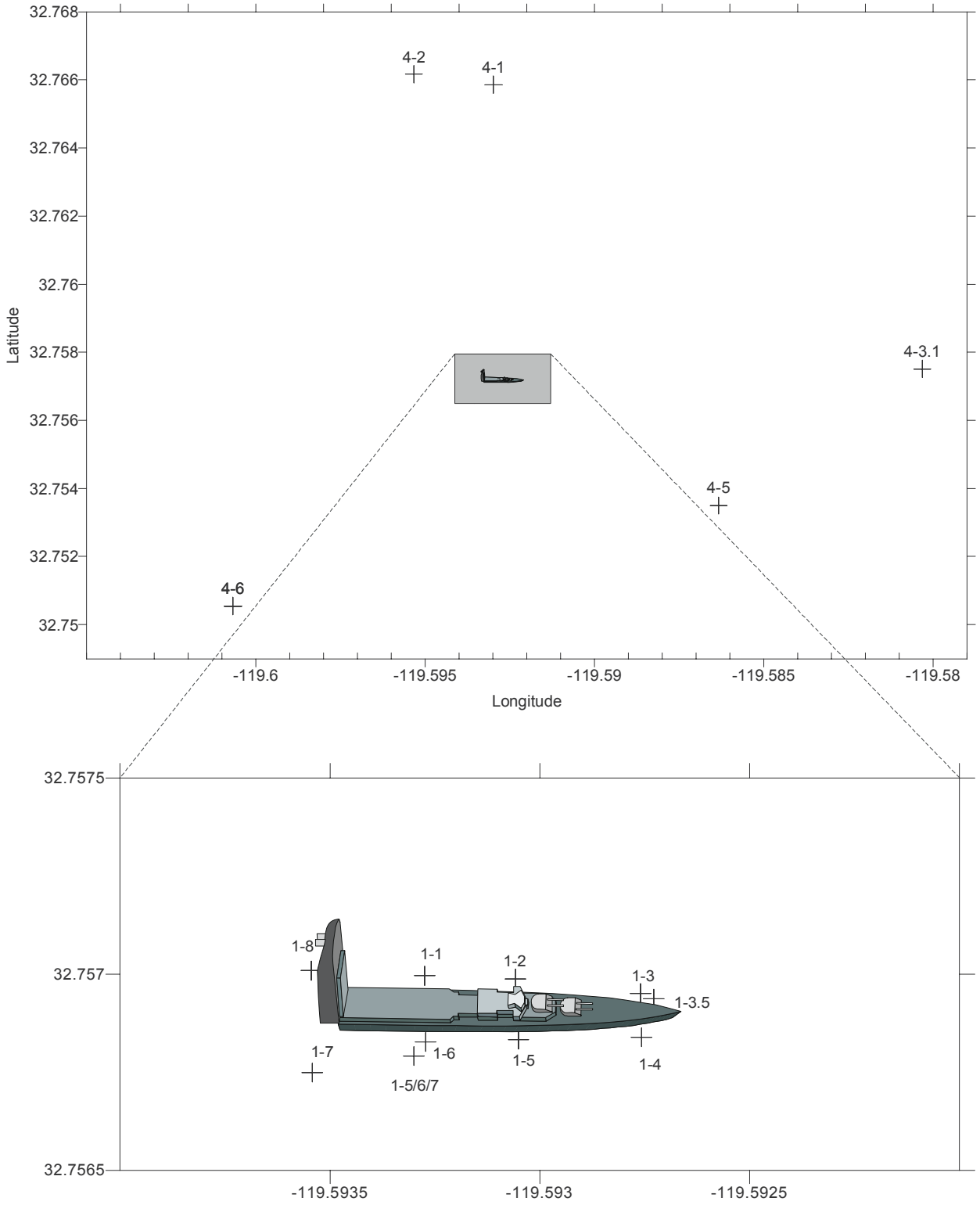


Figure 2-26. Station locations.

Table 2-10. Toxicity, bioaccumulation, grain size, and infauna community analysis summary.

Station	ANALYSES					
	10-Day Solid Phase *Rhepoxynius abronius*	28-Day Solid Phase *Neanthes arenaceodentata*	Bioaccumulation *Nephtys caecoides*	Bioaccumulation *Macoma nasuta*	Grain Size	Infauna Community
Survey - September 1998						
1-1	X	X	X	X	X	X
1-2	X	X	X	X	X	X
1-3	X	X	X	X	X	X
1-4	X	X	X	X	X	X
1-5	X	X	X	X	X	X
1-6	X	X	X	X	X	X
1-7	X	X	X	X	X	X
1-8	X	X	X	X	X	X
4-6	X	X	X	X	X	X
Survey - September 1999						
4-1	X	X	X	X	X	X
4-2	X	X	X	X	X	X
4-3.1	X	X	X	X	X	X
4-5	X	X	X	X	X	X
Survey - November 1999						
1-5/6/7	X	X	X	X	X	X

Table 2-11. Results of testing the toxicity/bioassay data to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best data transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Macoma nasuta survival	Arcsin	0.874	0.047	Non-normal	0.768
Macoma nasuta survival	Non transformed	0.855	0.026	Non-normal	0.272
***Macoma nasuta* survival**	**Ranks**	**NA**	**NA**	**NA**	**NA**
Neanthes arenaceodentata growth	Log 10	0.865	0.035	Non-normal	0.300
***Neanthes arenaceodentata* growth**	**Non transformed**	**0.879**	**0.055**	**Normal**	**0.132**
Neanthes arenaceodentata growth relative to controls	Arcsin	NA	NA	NA	NA
***Neanthes arenaceodentata* growth relative to controls**	**Non transformed**	**0.937**	**0.370**	**Normal**	**0.644**
Neanthes arenaceodentata survival	Arcsin	0.682	<0.001	Non-normal	0.301
Neanthes arenaceodentata survival	Non transformed	0.682	<0.001	Non-normal	0.301
***Neanthes arenaceodentata* survival**	**Ranks**	**NA**	**NA**	**NA**	**NA**
***Nephtys caecoides* survival**	**Arcsin**	**0.931**	**0.304**	**Normal**	**0.444**
Nephtys caecoides survival	Non transformed	0.907	0.139	Normal	0.672
Rhepoxynius abronius reburial	Arcsin	0.858	0.028	Non-normal	0.574
Rhepoxynius abronius reburial	Non transformed	0.800	0.004	Non-normal	0.372
***Rhepoxynius abronius* reburial**	**Ranks**	**NA**	**NA**	**NA**	**NA**
***Rhepoxynius abronius* survival**	**Arcsin**	**0.917**	**0.195**	**Normal**	**0.945**
Rhepoxynius abronius survival	Non transformed	0.785	0.003	Non-normal	0.530

2.5.4. Biota Sampling

2.5.4.1. Fish Sampling

Fish were collected in traps set directly adjacent to the hulk (within 100m) and at two different reference sites approximately 4nm from the hulk. *Anoplopoma fimbria* (Sablefish, Black Cod) was selected for the study based on the rationale provided earlier (see Section 2.4.4.1). An assortment of traps, approximately 3 feet in diameter and 1.5 feet tall were set with the aid of the ROV. Each fish trap was moored to the bottom using an anchor, with the fish trap suspended about 1-3 m above the bottom by a submersible float. Traps were baited with squid, sardines, mackerel or fish heads, and set for one to two days, depending on operations. Following an appropriate period of 72-120 hours (3-5 days), the traps were retrieved. All of the collected fish were photographed, weighed, measured, and eviscerated. Fish selected to provide tissue samples for chemical analysis were wrapped in clean aluminum foil, placed into clean polyethylene bags, assigned a unique identification number, and recorded in the project field notebook. Samples were stored frozen on the vessel and shipped on dry ice to the ADL analytical laboratory. Muscle tissue was dissected at the laboratory in a clean room under controlled conditions prior to digestion and analysis.

2.5.4.2. Invertebrate Epifauna Sampling

An attempt was made to collect epifaunal macro-invertebrates from the hulk. Organisms were to be collected by either (1) pinching and grabbing them directly with the manipulator arm of the ROV, or by (2) scraping them off with a stainless steel scraper held by the manipulator arm of the ROV. Samples were to be placed into a clean cage or bucket by the manipulator arm, secured to the ROV, and returned to the surface. The collected organisms were to be identified, photographed, weighed, and measured. Samples were to be transferred upon receipt into pre-cleaned glass containers, assigned unique identification numbers, and logged into the project field notebook. Samples were to be stored frozen on the ship and shipped on dry ice to the ADL laboratory for tissue residue analysis. However, there was no little direct observation of epifauna available for sampling and only a few specimens of opportunity were collected.

2.5.4.3. Benthic Infauna Sampling

The infaunal community was characterized by 23 boxcore samples collected during Cruises II, IV, and V (Table 2-10). The location and depth for each station are provided in Table 2-8. Infaunal samples were collected using the small boxcores (0.053 m^2 surface area) and the sediments were screened through a 0.5 mm and 1.0 mm sieve. Additional specimens were collected from unused portions (non-quantitative) of sediment samples collected for other objectives and from organisms selectively collected by the ROV. Infauna data for the two sieve sizes was combined so that the infaunal community would be representative of infaunal organisms larger than 0.5 mm and miscellaneous larger organisms. The initial plan for field processing of the infauna samples was to utilize two different sieve sizes in order to provide greater comparability of the data and to partition the sample to facilitate field and laboratory processing. The 1.0 mm sieve for characterizing infaunal communities has been used most often for the few studies completed for the slopes and basins of southern California (e.g., Hartman and Barnard 1960, Thompson and Jones 1987). More recent studies, especially for deeper communities, have been completed using smaller sieves (e.g., 0.5 and 0.3 mm sieves) for processing and characterizing infaunal communities. The 0.5 mm sieve was selected as a default

size when the use of a 0.3 mm proved impractical for the coarse and gravel sediments in the study area.

A gentle spray of seawater was used to separate the sediments from the organisms. Infauna was carefully collected from the screens, and placed into labeled jars. The collected infaunal organisms were relaxed with magnesium sulfate in seawater for approximately 10 minutes. After 20 minutes, the organisms were fixed in a solution of 10 percent buffered formalin. Within 14 days after collection, the organisms were transferred from formalin into 70-percent ethyl alcohol for long-term storage and sample analysis.

Infaunal samples were sorted with the aid of stereoscopic microscopes into five major taxonomic groupings, as follows: molluscs, echinoderms, polychaetes, crustaceans, and other minor phyla. Sorted organisms were distributed to taxonomists who counted and identified individual organisms to the lowest practicable taxonomic level. Wet weight biomass, measured to the nearest 0.01 g, was determined for each of the major taxonomic groups. Number of species, number of individuals, top 20 percent of species, and diversity indices (e.g., Shannon-Wiener) also were reported for each sample (Tetra Tech, 1985; Bergen et. al. 1998; Gray et. al., 1990).

An analyst not involved in performing the analyses reviewed all laboratory data generated to ensure accuracy of all transcribed data. Furthermore, the MEC Project Manager reviewed data for internal consistency and comparability.

Temporal differences between surveys were not considered significant and all infaunal samples from Surveys II, III, and V were considered as one data set. Mean values were used for stations having replicate samples. Infaunal community measures were tested to determine the appropriate data transformations, if any, to fulfill the assumptions of the ANOVA statistical tests (Table 2-12 through Table 2-17 for the 1.0 mm community and Table 2-18 through Table 2-23 for the combined 0.5 and 1.0 mm communities). The null hypothesis was that there was no difference between the Inner Ring (Ring 1), samples collected within 2-3 meters of the target vessel, and the Outer Ring (Ring 4), which represented samples collected approximately 1 km from the target vessel. Significance criteria was $p<0.05$, which would suggest rejection of the null hypothesis. Furthermore, the range of values was evaluated to determine whether the heterogeneity of the environment for the Inner Ring was different from the Outer Ring.

Statistical testing of biological infaunal community measures found some parameters having significant differences between the Inner and Outer Rings. Regression analysis was then used to identify if there was a significant correlation with these infaunal community measures and the physical and chemical measures. To help visualize these relationships the community measures were plotted as a function of grain size measures, TOC, and Principal Component Analysis (PCA) factors to determine if there were significant regressions of infauna community measures with these environmental variables. A similar approach was used on the bioassay data to determine if the variability observed in the bioassay testing could be correlated with physical and chemical measures.

Table 2-12. Results of testing infaunal community measures data (all organisms for 1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best data transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	Non transformed	0.905	0.114	Normal	0.618
Number of Species	**Log 10**	**0.917**	**0.175**	**Normal**	**0.930**
Abundance	Non transformed	0.937	0.334	Normal	0.050
Abundance	**Log 10**	**0.973**	**0.868**	**Normal**	**0.114**
Shannon-Wiener Diversity	Non transformed	0.835	0.010	Non-normal	0.325
Shannon-Wiener Diversity	Log 10	0.751	0.000	Non-normal	0.294
Shannon-Wiener Diversity	**Ranks**	**NA**	**NA**	**NA**	**NA**
Margalef Species Richness	Non transformed	0.962	0.689	Normal	0.990
Margalef Species Richness	**Log 10**	**0.972**	**0.844**	**Normal**	**0.511**
Dominance Index	**Non transformed**	**0.939**	**0.362**	**Normal**	**0.405**
Dominance Index	Log 10	0.799	0.003	Non-normal	0.238
Evenness Index	Non transformed	0.726	0.000	Non-normal	0.182
Evenness Index	Log 10	0.672	0.000	Non-normal	0.190
Evenness Index	**Ranks**	**NA**	**NA**	**NA**	**NA**

P≤0.05=non-normal distribution
NA=Not Applicable

Table 2-13. Results of testing infaunal community measures for Polychaeta (1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best data transformation is highlighted on bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	**Non transformed**	**0.889**	**0.066**	**Normal**	**0.611**
Number of Species	Log 10	0.857	0.021	Non-normal	0.937
Abundance	**Non transformed**	**0.922**	**0.203**	**Normal**	**0.193**
Abundance	Log 10	0.884	0.056	Normal	0.662
Shannon-Wiener Diversity	**Non transformed**	**0.907**	**0.124**	**Normal**	**0.583**
Shannon-Wiener Diversity	Log 10	0.887	0.060	Normal	0.786
Margalef Species Richness	**Non transformed**	**0.900**	**0.098**	**Normal**	**0.561**
Margalef Species Richness	Log 10	0.885	0.056	Normal	0.623
Dominance Index	Non transformed	0.944	0.419	Normal	0.039
Dominance Index	**Log 10**	**0.919**	**0.189**	**Normal**	**0.078**
Evenness Index	Non transformed	0.818	0.006	Non-normal	0.113
Evenness Index	Log 10	0.749	0.008	Non-normal	0.135
Evenness Index	**Ranks**	**NA**	**NA**	**NA**	**NA**

P≤0.05=non-normal distribution
NA=Not Applicable

Table 2-14. Results of testing infaunal community measures for Crustacea (1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best data transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	Non transformed	0.837	0.014	Non-normal	0.235
Number of Species	**Log 10**	**0.956**	**0.678**	**Normal**	**0.757**
Abundance	Non transformed	0.920	0.189	Normal	0.006
Abundance	**Log 10**	**0.905**	**0.174**	**Normal**	**0.706**
Shannon-Wiener Diversity	Non transformed	0.839	0.012	Non-normal	0.304
Shannon-Wiener Diversity	**Log 10**	**0.903**	**0.228**	**Normal**	**0.418**
Margalef Species Richness	Non transformed	0.918	0.326	Normal	0.038
Margalef Species Richness	**Log 10**	**0.918**	**0.326**	**Normal**	**0.368**
Dominance Index	Non transformed	0.848	0.016	Non-normal	0.101
Dominance Index	Log 10	0.841	0.012	Non-normal	0.501
Dominance Index	**Ranks**	**NA**	**NA**	**NA**	**NA**
Evenness Index	Non transformed	0.804	0.016	Non-normal	0.200
Evenness Index	Log 10	0.819	0.024	Non-normal	0.223
Evenness Index	**Ranks**	**NA**	**NA**	**NA**	**NA**

P≤0.05=non-normal distribution
NA=Not Applicable

Table 2-15. Results of testing infaunal community measures for Mollusca (1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best data transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	**Non transformed**	**0.925**	**0.227**	**Normal**	**0.281**
Number of Species	Log 10	0.907	0.160	Normal	0.666
Abundance	**Non transformed**	**0.914**	**0.156**	**Normal**	**0.657**
Abundance	Log 10	0.846	0.024	Non-normal	0.477
Shannon-Wiener Diversity	**Non transformed**	**0.896**	**0.083**	**Normal**	**0.615**
Shannon-Wiener Diversity	Log 10	0.874	0.135	Normal	0.873
Margalef Species Richness	Non transformed	0.962	0.796	Normal	0.006
Margalef Species Richness	Log 10	0.821	0.035	Non-normal	0.035
Margalef Species Richness	**Ranks**	**NA**	**NA**	**NA**	**NA**
Dominance Index	**Non transformed**	**0.916**	**0.170**	**Normal**	**0.367**
Dominance Index	Log 10	0.881	0.050	Normal	0.815
Evenness Index	Non transformed	0.700	0.002	Non-normal	0.013
Evenness Index	Log 10	0.703	0.002	Non-normal	0.014
Evenness Index	**Ranks**	**NA**	**NA**	**NA**	**NA**

P≤0.05=non-normal distribution
NA=Not Applicable

Table 2-16. Results of testing infaunal community measures for minor phyla (1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best data transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	**Non transformed**	**0.925**	**0.228**	**Normal**	**0.248**
Number of Species	Log 10	0.899	0.126	Normal	0.644
Abundance	**Non transformed**	**0.925**	**0.228**	**Normal**	**0.248**
Abundance	Log 10	0.899	0.126	Normal	0.644
Shannon-Wiener Diversity	Non transformed	0.800	0.003	Non-normal	0.032
Shannon-Wiener Diversity	Log 10	0.795	0.036	Non-normal	0.054
Shannon-Wiener Diversity	**Ranks**	**NA**	**NA**	**NA**	**NA**
Margalef Species Richness	Non transformed	0.783	0.028	Non-normal	0.062
Margalef Species Richness	Log 10	0.797	0.038	Non-normal	0.052
Margalef Species Richness	**Ranks**	**NA**	**NA**	**NA**	**NA**
Dominance Index	Non transformed	0.778	0.002	Non-normal	0.022
Dominance Index	Log 10	0.809	0.004	Non-normal	0.032
Dominance Index	**Ranks**	**NA**	**NA**	**NA**	**NA**
Evenness Index	Non transformed	NC	NC	NC	NC
Evenness Index	Log 10	NC	NC	NC	NC
Evenness Index	**Ranks**	**NAF**	**NA**	**NA**	**NA**

P≤0.05=non-normal distribution
NC=Not capable of being calculated, means too similar
NA=Not Applicable

Table 2-17. Results of testing infaunal community measures for Echinodermata (1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best data transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	Non transformed	0.845	0.014	Non-normal	0.344
Number of Species	Log 10	0.617	0.000	Non-normal	0.104
Number of Species	**Ranks**	**NA**	**NA**	**NA**	**NA**
Abundance	Non transformed	0.773	0.001	Non-normal	0.022
Abundance	**Log 10**	**0.950**	**0.627**	**Normal**	**0.377**
Shannon-Wiener Diversity	Non transformed	0.898	0.089	Normal	0.989
Shannon-Wiener Diversity	**Log 10**	**0.945**	**0.669**	**Normal**	**0.826**
Margalef Species Richness	Non transformed	0.849	0.073	Normal	0.030
Margalef Species Richness	Log 10	1.0	0.992	Normal	0.000
Margalef Species Richness	**Ranks**	**NA**	**NA**	**NA**	**NA**
Dominance Index	Non transformed	0.859	0.023	Non-normal	0.514
Dominance Index	**Log 10**	**0.878**	**0.449**	**Normal**	**0.475**
Evenness Index	Non transformed	NC	NC	NC	NC
Evenness Index	Log 10	NC	NC	NC	NC
Evenness Index	**Ranks**	**NA**	**NA**	**NA**	**NA**

P≤0.05=non-normal distribution
NC=Not capable of being calculated, means too similar
NA=Not Applicable

Table 2-18. Results of testing infaunal community measures data (all organisms for 0.5 and 1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	**Non transformed**	**0.929**	**0.262**	**Normal**	**0.733**
Number of Species	Log 10	0.884	0.055	Normal	0.419
Abundance	**Non transformed**	**0.922**	**0.202**	**Normal**	**0.072**
Abundance	Log 10	0.920	0.191	Normal	0.961
Shannon-Wiener Diversity	Non transformed	0.836	0.011	Non-normal	0.127
Shannon-Wiener Diversity	Log 10	0.764	0.001	Non-normal	0.148
Shannon-Wiener Diversity	**Ranks**	**NA**	**NA**	**NA**	**NA**
Margalef Species Richness	**Non transformed**	**0.926**	**0.237**	**Normal**	**0.197**
Margalef Species Richness	Log 10	0.872	0.036	Non-normal	0.218
Dominance Index	**Non transformed**	**0.956**	**0.591**	**Normal**	**0.060**
Dominance Index	Log 10	0.851	0.018	Non-normal	0.111
Evenness Index	Non transformed	0.814	0.005	Non-normal	0.522
Evenness Index	Log 10	0.786	0.002	Non-normal	0.502
Evenness Index	**Ranks**	**NA**	**NA**	**NA**	**NA**

$P \leq 0.05$=non-normal distribution
NA=Not Applicable

Table 2-19. Results of testing infaunal community measures for Polychaeta (0.5 and 1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	**Non transformed**	**0.920**	**0.190**	**Normal**	**0.109**
Number of Species	Log 10	0.890	0.068	Normal	0.174
Abundance	Non transformed	0.952	0.539	Normal	0.646
Abundance	**Log 10**	**0.987**	**0.991**	**Normal**	**0.679**
Shannon-Wiener Diversity	Non transformed	0.830	0.009	Non-normal	0.219
Shannon-Wiener Diversity	Log 10	0.754	0.001	Non-normal	0.242
Shannon-Wiener Diversity	**Ranks**	**NA**	**NA**	**NA**	**NA**
Margalef Species Richness	**Non transformed**	**0.954**	**0.563**	**Normal**	**0.152**
Margalef Species Richness	Log 10	0.938	0.347	Normal	0.186
Dominance Index	**Non transformed**	**0.943**	**0.409**	**Normal**	**0.235**
Dominance Index	Log 10	0.876	0.041	Non-normal	0.276
Evenness Index	Non transformed	0.799	0.003	Non-normal	0.219
Evenness Index	Log 10	0.764	0.001	Non-normal	0.212
Evenness Index	**Ranks**	**NA**	**NA**	**NA**	**NA**

P≤0.05=non-normal distribution
NA=Not Applicable

Table 2-20. Results of testing infaunal community measures for Crustacea (0.5 and 1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	Non transformed	0.936	0.326	Normal	0.748
Number of Species	**Log 10**	**0.967**	**0.797**	**Normal**	**0.336**
Abundance	Non transformed	0.914	0.159	Normal	0.480
Abundance	**Log 10**	**0.971**	**0.842**	**Normal**	**0.333**
Shannon-Wiener Diversity	**Non transformed**	**0.981**	**0.953**	**Normal**	**0.181**
Shannon-Wiener Diversity	Log 10	0.937	0.369	Normal	0.116
Margalef Species Richness	Non transformed	0.919	0.206	Normal	0.781
Margalef Species Richness	**Log 10**	**0.969**	**0.826**	**Normal**	**0.392**
Dominance Index	Non transformed	0.868	0.032	Non-normal	1.00
Dominance Index	**Log 10**	**0.964**	**0.732**	**Normal**	**0.148**
Evenness Index	Non transformed	0.677	0.000	Non-normal	0.220
Evenness Index	Log 10	0.669	0.000	Non-normal	0.199
Evenness Index	**Ranks**	**NA**	**NA**	**NA**	**NA**

P≤0.05=non-normal distribution
NA=Not Applicable

Table 2-21. Results of testing infaunal community measures for Mollusca (0.5 and 1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	Non transformed	0.938	0.350	Normal	0.920
Number of Species	**Log 10**	**0.950**	**0.499**	**Normal**	**0.247**
Abundance	Non transformed	0.860	0.024	Non-normal	0.290
Abundance	**Log 10**	**0.954**	**0.568**	**Normal**	**0.187**
Shannon-Wiener Diversity	**Non transformed**	**0.977**	**0.910**	**Normal**	**0.089**
Shannon-Wiener Diversity	Log 10	0.927	0.269	Normal	0.053
Margalef Species Richness	**Non transformed**	**0.961**	**0.705**	**Normal**	**0.209**
Margalef Species Richness	Log 10	0.921	0.224	Normal	0.084
Dominance Index	**Non transformed**	**0.898**	**0.089**	**Normal**	**0.175**
Dominance Index	Log 10	0.894	0.078	Normal	0.191
Evenness Index	Non transformed	0.953	0.572	Normal	0.613
Evenness Index	**Log 10**	**0.972**	**0.864**	**Normal**	**0.456**

P≤0.05=non-normal distribution

Table 2-22. Results of testing infaunal community measures for minor phyla (0.5 and 1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	**Non transformed**	**0.947**	**0.462**	**Normal**	**0.791**
Number of Species	Log 10	0.920	0.217	Normal	0.741
Abundance	**Non transformed**	**0.967**	**0.770**	**Normal**	**0.271**
Abundance	Log 10	0.935	0.349	Normal	0.761
Shannon-Wiener Diversity	Non transformed	0.877	0.043	Non-normal	0.082
Shannon-Wiener Diversity	Log 10	0.833	0.035	Non-normal	0.042
Shannon-Wiener Diversity	**Ranks**	**NA**	**NA**	**NA**	**NA**
Margalef Species Richness	**Non transformed**	**0.905**	**0.235**	**Normal**	**0.360**
Margalef Species Richness	Log 10	0.889	0.157	Normal	0.065
Dominance Index	Non transformed	0.831	0.009	Non-normal	0.088
Dominance Index	Log 10	0.788	0.002	Non-normal	0.054
Dominance Index	**Ranks**	**NA**	**NA**	**NA**	**NA**
Evenness Index	Non transformed	0.852	0.060	Normal	0.001
Evenness Index	Log 10	0.862	0.077	Normal	0.001
Evenness Index	**Ranks**	**NA**	**NA**	**NA**	**NA**

P≤0.05=non-normal distribution

Table 2-23. Results of testing infaunal community measures for Echinodermata (0.5 and 1.0 mm) to determine compliance with statistical test assumptions and for determining the best data transformation for statistical testing. Best transformation is highlighted in bold.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Number of Species	Non transformed	0.767	0.001	Non-normal	0.285
Number of Species	Log 10	0.544	0.000	Non-normal	0.043
Number of Species	**Ranks**	**NA**	**NA**	**NA**	**NA**
Abundance	Non transformed	0.868	0.031	Non-normal	0.007
Abundance	**Log 10**	**0.956**	**0.626**	**Normal**	**0.214**
Shannon-Wiener Diversity	Non transformed	0.739	0.001	Non-normal	0.242
Shannon-Wiener Diversity	**Log 10**	**0.951**	**0.699**	**Normal**	**0.285**
Margalef Species Richness	Non transformed	0.703	0.001	Non-normal	0.066
Margalef Species Richness	**Log 10**	**1.0**	**0.992**	**Normal**	**0.000**
Dominance Index	Non transformed	0.707	0.000	Non-normal	0.227
Dominance Index	Log 10	0.730	0.000	Non-normal	0.076
Dominance Index	**Ranks**	**NA**	**NA**	**NA**	**NA**
Evenness Index	Non transformed	NC	NC	NC	NC
Evenness Index	Log 10	NC	NC	NC	NC
Evenness Index	**Ranks**	**NA**	**NA**	**NA**	**NA**

P≤0.05=non-normal distribution
NC=Not capable of being calculated, means too similar
NA=Not Applicable

2.5.5. Current Measurement and Water Sampling

2.5.5.1. Measurement of Deep Ocean Currents in Vicinity of ex-AGERHOLM

Two types of current meters were required for this study: (1) an acoustic Doppler current profiler (ADCP) for measuring currents in 2-meter layers or horizontal slices throughout the water column, and (2) a S4 meter used to measure the bottom currents not measurable with the ADCP. Because the ADCP was situated on the bottom, and sent and received its acoustic signals to and from the water column immediately above its location, the very bottom "layer" of water currents represented a "blind spot."

2.5.5.1.1. Acoustic Doppler Current Profiler (ADCP)

A deep water RD Instruments ADCP was used to obtain 27 months of data on currents from two deployments in the vicinity of the hulk. The ADCP was initially deployed during Cruise I and then recovered during Cruise II with the SCORPIO 1, collecting 13 months of data. The current profiler was re-deployed during Cruise II on the opposite side of the hulk on a reciprocal bearing and at a similar distance from the hulk. The meter was recovered during Cruise V, after obtaining 14 additional months of data. The ADCP was mounted in a gimbaled cage and positioned perpendicular to the horizontal memory. The unit was programmed to collect data as detailed in Table 2-24. Water movement data was collected from 30 depth "bins". Each bin was programmed at two meters thick. With the null zone in the data set from the bottom up to

approximately 4.25 meter above the bottom (due to limitations of the equipment) and 4.25 to 64.25 meters above the bottom representing the area of data collection. Upon retrieval, this information was used to estimate advection of water borne contaminants from the hulk and/or sediment transport on the sea floor. The ADCP model used was a 300 kHz system with 20 MB of on-board memory.

Table 2-24. RDI Acoustic Doppler Current Profiler (300 kHz Workhorse) - Deployments 1 and 2.

Parameter	Values	Recorded Measurements
Number of depth bins	30	Average current speed, and direction, for 30 2-m depth intervals. Recorded every 30 minutes. Temperature data also recorded.
Bin Length	2 Meters	
First Bin (blank zone)	4.25 Meters	
Last Bin (range)	64.25 Meters	
Sampling frequency	Every 30 Minutes	
Number of data points averaged per sample	45	
Deployment Duration	13 months; 14 months	

2.5.5.1.2. S4™ Current Meter Measurements

An InterOcean S4™ electro-magnetic current meter was deployed during Cruise II and recovered during Cruise V. The meter was placed approximately 50 ft off of the startboard bow and away from the anchor chain that heads toward the same direction. The S4™ was utilized to supplement the current data being collected by the ADCPs, specifically current direction and speed near the sediment/water interface (1.5 ft above the bottom). Data was collected for 6.5 months.

2.5.5.2. Water Sampling

Water samples were collected with a five-liter Teflon®-lined Niskin samplers during Cruises IV and V. Samplers were attached to the ROV, and triggered by the manipulator arm. Water samples were collected from two locations, one from Ring 1 (Station 1-5/6/7) and one from Ring 4 (Station 4-2).

Samples were transferred into precleaned borosilicate glass jars (for organic chemical analysis) or precleaned polyethylene jars (for inorganic chemical analysis) with appropriate preservatives. Samples for organic chemistry analysis were preserved with hydrochloric acid, samples being analyzed for total organic carbon were preserved at 4 ºC. All water samples were given unique identification numbers and logged. Samples were stored in the dark at a temperature of 4 °C after collection and during shipment to the laboratory for analysis.

2.5.6. Chemical Analyses

2.5.6.1. Sample Preparation

Arthur D Little's (ADL) Environmental Monitoring and Analysis Laboratory, based in Cambridge, MA, conducted the sample preparations.

2.5.6.1.1. Water Sample Preparation for PAH and PCB Analysis

Water samples were extracted for semivolatile organic compounds per Arthur D Little's (ADL) Standard Operating Procedure (SOP) ADL-2824, "Extraction of Semivolatile Hydrocarbons and

PCBs/Pesticides from Water Samples." This method is similar to EPA SW-846 Method 3510B, "Separatory Funnel Liquid-Liquid Extraction." With every sample preparation batch the following quality control samples were prepared: procedural blank (PB), blank spike (BS), and blank spike duplicate (BSD). Each environmental and quality control sample was transferred to a separatory funnel and spiked with polyaromatic hydrocarbon (PAH) and polychlorinated biphenyl (PCB) surrogate solutions prior to the first addition of the extraction solvent. The concentration of the surrogate compounds spiked into the samples was determined based on the expected contamination level in the samples. For this project, all surrogates were spiked at low levels in the water samples. In addition to the surrogate solution, the BS and BSD quality control samples were spiked with a subset of the target PAH and PCB compounds.

Organic compounds were extracted from 2-L water samples using the organic solvent dichloromethane. For each sample, a 120-mL aliquot of solvent was added to the separatory funnel and the separatory funnel was sealed and shaken vigorously for 1-2 minutes. The organic layer was allowed to separate from the water phase and then was drained into a flask. This extraction procedure was repeated two more times with fresh aliquots of solvent. The 3 solvent extracts per sample were combined and water was removed from the combined extract by adding approximately 75 g of sodium sulfate. All extracts were concentrated to approximately 1 mL, using Kuderna-Danish (KD) concentrators and nitrogen evaporation. Extracts were split into archive and working volumes. The working extract volume was further split - one-half was designated for PAH analysis and one-half was exchanged into hexane for PCB analyses.

2.5.6.1.2. Sediment Sample Preparation for PAH and PCB Analysis

Sediment samples were extracted for semivolatile organic compounds per ADL's Standard Operating Procedure (SOP) ADL-2819, "Extraction of Polychlorinated Biphenyls and Chlorinated Pesticides from Sediment or Shoreline Soil Samples." With every sample preparation batch the following quality control samples were prepared: PB, BS, BSD, matrix spike (MS), matrix spike duplicate (MSD), laboratory duplicate (DUP), and sediment standard reference material (NIST SRM 1944). After each sample was thoroughly homogenized, a 30 to 50 gram aliquot of the sediment sample was transferred into a Teflon® jar for sample preparation. Approximately 60 grams of sodium sulfate were mixed into each sample, followed by the addition of 100 mL of 50:50 dichloromethane/acetone. Each environmental and quality control sample was spiked with PAH and PCB surrogate solutions prior to the first addition of the extraction solvent. The concentration of the surrogate compounds spiked into the samples was determined based on the expected contamination level in the samples. For this project, all surrogates were spiked at low levels in the sediment samples. In addition to the surrogate solution, the BS, BSD, MS, and MSD quality control samples were spiked with a subset of the target PAH and PCB compounds.

Organic compounds were extracted from the sediment samples using a 50:50 mixture of the organic solvents dichloromethane and acetone. For each sample, a 100-mL aliquot of solvent was added to the Teflon® jar and the sample was sonicated for three minutes. The samples were centrifuged and the organic solvent layer was decanted into a flask. This extraction procedure was repeated two more times with fresh aliquots of solvent. After the third sonication, the sample jar was placed on an orbital shaker for 1 hour prior to the final centrifuge. The three solvent extracts per sample were combined and water was removed from the combined extract by adding approximately 75 g of sodium sulfate. Copper, alumina column, and high-pressure liquid chromatography (HPLC) cleanups were performed on the sample extracts to remove potential

contamination that would interfere with sample analysis. All extracts were concentrated to approximately 1 mL, using KD concentrators and nitrogen evaporation. Extracts were split into archive and working volumes. The working extract volume was further split - one-half was designated for PAH analysis and one-half was exchanged into hexane for PCB analyses.

2.5.6.1.3. Sample Preparation Procedure of Sediment Core Samples for PCB Analysis

Sediment core samples were sectioned at 0.5-cm intervals for the top 3 cm (six samples) and then at 1-cm intervals for the remaining 4 to 10 cm depth (seven samples). The sediment core samples were extracted for semivolatile organic compounds per ADL-2819, "Extraction of Polychlorinated Biphenyls and Chlorinated Pesticides from Sediment or Shoreline Soil Samples." With every sample preparation batch the following quality control samples were prepared: PB, BS, BSD, and sediment standard reference material (NIST SRM 1944). After each sample was thoroughly homogenized, a 15 to 20 gram aliquot of the sediment sample was transferred into a Teflon® jar for sample preparation. Approximately 40 grams of sodium sulfate were mixed into each sample, followed by the addition of 100 mL of 50:50 dichloromethane/acetone. Each environmental and quality control samples was spiked with PCB surrogate solution prior to the first addition of the extraction solvent. The concentration of the surrogate compounds spiked into the samples was determined based on the expected contamination level in the samples. For this project, all surrogates were spiked at low levels in the sediment samples. In addition to the surrogate solution, the BS, BSD, MS, and MSD quality control samples were spiked with a subset of the target PCB compounds.

Organic compounds were extracted from the sediment samples using a 50:50 mixture of the organic solvents dichloromethane and acetone. For each sample, a 100-mL aliquot of solvent was added to the Teflon® jar and the sample was sonicated for three minutes. The samples were centrifuged and the organic solvent layer was decanted into a flask. This extraction procedure was repeated two more times with fresh aliquots of solvent. After the third sonication, the sample jar was placed on an orbital shaker for 1 hour prior to the final centrifuge. The three solvent extracts per sample were combined and water was removed from the combined extract by adding approximately 75 g of sodium sulfate. Copper, alumina column, and HPLC cleanups were performed on the sample extracts to remove potential contamination that would interfere with sample analysis. All extracts were concentrated to approximately 1 mL, using KD concentrators and nitrogen evaporation. Extracts were split into archive and working volumes. The working extract volume was exchanged into hexane for PCB analyses.

2.5.6.1.4. Sample Preparation Procedure of Tissue for PCB Analysis

The tissue samples were extracted for semivolatile organic compounds per ADL's SOP ADL-2831, "Extraction of Semivolatile Hydrocarbons, PCBs, and Chlorinated Pesticides from Biological Tissue Samples." With every sample preparation batch the following quality control samples were prepared: PB, BS, BSD, MS, MSD, DUP, and tissue standard reference material (NIST SRM 1974a).

Approximately 20 g of homogenized tissue was weighed into a Teflon® jar. Sixty grams of sodium sulfate (or more where necessary) was added to dry the sample, followed by 100 mL methylene chloride. The sample was then spiked with the appropriate amount of PCB and PAH surrogates. Each sample was macerated/extracted using the Tissumizer®, centrifuged, and decanted into an Erlenmeyer flask containing approximately 70 g of sodium sulfate. These

extraction steps were repeated two more times, with all extracts combined following each centrifugation. Sample extracts were concentrated to 10 to 15 mL in round-bottom flasks, and further concentrated under nitrogen after transfer to 25-mL vials. The lipid content was determined gravimetrically for each sample extract, which was then be loaded onto an alumina cleanup column. The cleaned-up sample was be concentrated to 1 mL for HPLC fractionation.

After each sample was homogenized, a 5 to 15 gram aliquot of the tissue sample was transferred into a Teflon® jar for sample preparation. Approximately 60 grams of sodium sulfate were mixed into each sample, followed by the addition of 100 mL of dichloromethane. Each environmental and quality control sample was spiked with PAH and PCB surrogate solutions prior to the first addition of the extraction solvent. The concentration of the surrogate compounds spiked into the samples was determined based on the expected contamination level in the samples. For this project, all surrogates were spiked at low levels in the tissue samples. In addition to the surrogate solution, the BS, BSD, MS, and MSD quality control samples were spiked with a subset of the target PAH and PCB compounds.

Organic compounds were extracted from the tissue samples using the organic solvent dichloromethane. For each sample, a 100-mL aliquot of solvent was added to the Teflon® jar and the sample was macerated at high speed for two minutes using a tissue extraction probe. The samples were centrifuged and the organic solvent layer was decanted into a flask. This extraction procedure was repeated two more times with fresh aliquots of solvent. The three solvent extracts per sample were combined and water was removed from the combined extract by adding approximately 75 g of sodium sulfate. All extracts were concentrated to approximately 1 mL, using KD concentrators and nitrogen evaporation. After sample extraction and concentration, the total extractable lipid weight was determined for each sample. Alumina column and HPLC cleanups were performed on the sample extracts to remove potential contamination that would interfere with sample analysis. Extracts were split into archive and working volumes. The working extract volume was further split - one-half was designated for PAH analysis and one-half was cleaned with sulfuric acid and exchanged into hexane for PCB analyses.

2.5.6.1.5. Tissue Sample Total Extractable Lipid Weight Determination

After extraction with organic solvent and prior to extract cleanup procedures, the volume of the total sample extract was measured by withdrawing the entire extract into an appropriately sized glass syringe, recording the volume to the nearest 0.05 mL, and returning the sample to the vial.

Using a glass-barreled syringe or a glass disposable micropipette, a 20- to 50-µL aliquot of the sample was withdrawn and transferred to a tared aluminum weighing pan. The pan was placed on a hot plate (approximately 60 °C) and sufficient time was allowed for solvent evaporation. The pan was placed on the hanger of a calibrated electrobalance and weighed to the nearest 0.01 mg. The total extract sample volume, aliquot volume, and aliquot weight was then recorded in the project Laboratory Notebook.

The total extractable {lipid} weight was calculated as follows:

> {[weight of extracted residue × (extract volume of sample/volume of aliquot)]/sample weight extracted} × 100%

If dry weight results were to be reported, the sample dry weight extracted in the above calculation was used; if wet weight results were to be reported, the sample wet weight extracted in the above calculation was used.

Sample dry weight extracted=grams of wet sample extracted × decimal percent solids

Total extractable weight only approximates total lipid weight. ADL's method for approximating percent lipid is consistent with the procedures of other laboratories performing tissue analyses (EPA, Battelle). There are more complex and expensive methods, which more accurately determine percent lipids. ADL's method for estimating percent lipids was consistent over the course of the SINKEX project. It should be noted that total extractable weight may provide a high bias estimate of total lipid weight. The magnitude of this bias, however, is variable, dependent upon several factors, and thought to be trivial.

For percent solid determination, a 5-g aliquot of the macerated tissue for each sample was placed in a tared weighing pan, and weighed. The tissue was dried overnight at 103-105 ºC (the temperature of the oven was recorded daily). The sample and pan was allowed to cool and then re-weighed. Percent solids were calculated using the formula:

% solids=(dried sample weight/wet sample weight) × 100%

2.5.6.2. Fractionation by High Pressure Liquid Chromatography (HPLC)

All extracts were fractionated to reduce matrix interferences utilizing an HPLC with a size-exclusion column. The column was calibrated such with the expectation that fractions collected would contain PCBs and PAHs. Half of the 1-mL extract was injected onto a 320 mm x 21.2 mm size exclusion column. The column was then eluted with methylene chloride and the PCB/PAH fraction collected. This fraction was concentrated to 1 mL and exchanged into hexane. The fraction was then spiked with both PAH and PCB recovery standards and submitted for GC/MS and GC/ECD analysis, respectively. The recovery standard compounds were fluorene-d_{10} and chrysene-d_{12} for PAH analysis, and Tetrachloro-m-xylene (TCMX) for PCB analysis.

2.5.6.3. Sample Analysis

The final sample extracts were analyzed by GC/MS and GC/ECD instrumental techniques. Internal recovery standards were spiked into each sample for quantification of all target analytes. The extracts were concentrated to an appropriate pre-injection volume (PIV) and separate volumetric aliquots were delivered into the respective GC ports for each analysis. The amount of recovery standards added to the extracts approximated their concentration in the calibration standards.

2.5.6.3.1. PCB Analysis

The organics analyses were also conducted by ADL. Water, sediment, and tissue sample extracts were analyzed for PCB congeners, homologs, and Aroclors per ADL's SOP ADL-2845, "Determination of PCBs by Gas Chromatography/Mass Spectrometry in the Selected Ion Monitoring Mode." ADL's PAH analysis method is a modified version of EPAs Method 680. The target PCB congeners, homologs, and Aroclors are listed in Table 2-25. The GC/MS was operated in SIM mode to obtain the desired sensitivity that is comparable to that of a GC equipped with an ECD. The GC/MS was first tuned with PFTBA to verify accurate mass assignment and to maximize the sensitivity of the instrument in the mass range of interest (100 to 300 atomic mass units). After tuning, an initial calibration was performed which consisted of five calibration standards, at different concentration levels, spanning the concentration range of interest. Average response factors for each target compound and surrogate are calculated from

the initial calibration standards relative to the internal standard compounds added to the sample extracts just prior to instrumental analysis. Continuing calibration standards, at a mid-range concentration level, were analyzed every 18 hours or after every 12 sample analyses to monitor sensitivity and linearity of the GC/MS. Sample analyses were performed after acceptable calibration analyses were obtained. The average response factors generated from the initial calibration were used to calculate the concentrations of target compounds and surrogates in the environmental and quality control samples. The recoveries of the surrogate compounds spiked into the sample prior to extraction were used to assess sample-specific extraction efficiency. The target compound concentrations were adjusted based on sample-specific surrogate recoveries to correct for differences in extraction efficiency.

Selected water, sediment, and tissue sample extracts were analyzed for PCB congeners per ADL's SOP ADL-2818, "Determination of Chlorinated Pesticides and PCB Congeners by Gas Chromatography/ Electron Capture Detection." ADL's PCB congener analysis method is a modified version of EPA SW-846 Method 8081, using dual, dissimilar columns and dual detectors. A Restek RTX-5 column (or equivalent) was used as the primary column and a DB-17 column (or equivalent) was used as the confirmation column. The target PCB congeners are listed in Table 2-25. Prior to sample analysis, an initial calibration was performed which consisted of five calibration standards, at different concentration levels ranging from 1 to 200 ng/mL. Average calibration factors for each target compound and surrogate are calculated from the initial calibration standards (external standardization). Continuing calibration standards, at a mid-range concentration level, were analyzed every 18 hours or after every 10 sample analyses to monitor sensitivity, retention time stability, and linearity of the GC/ECD. Sample analyses were performed after acceptable calibration analyses were obtained. The average calibration factors generated from the initial calibration were used to calculate the concentrations of target compounds and surrogates in the environmental and quality control samples. When coelution occurred between one or more target compounds or when interference occurred on the primary column, the results were reported from the confirmation column for the affected compounds. Compound identification was based on 1) detecting a peak within the established retention time window for a specific compound on both the primary and confirmation columns and 2) the analyst's judgment. The recoveries of the surrogate compounds spiked into the sample prior to extraction were used to assess sample-specific extraction efficiency. The target compound concentrations were adjusted based on sample-specific surrogate recoveries to correct for differences in extraction efficiency.

Table 2-25. PCB Congener, Homolog, and Aroclor Target, Surrogate, and Internal Standard Compound List.

Compound	Surrogate Reference	Compound	Surrogate Reference
PCB Congeners		**PCB Aroclors**	
8 - 2,4′-Dichlorobiphenyl	2	Aroclor 1221	2
18 - 2,2′,5-Trichlorobiphenyl	2	Aroclor 1232	2
28 - 2,4,4′-Trichlorobiphenyl	2	Aroclor 1242	2
44 - 2,2′,3,5′-Tetrachlorobiphenyl	2	Aroclor 1248	2
49 - 2,2′,4,5′-Tetrachlorobiphenyl	2	Aroclor 1254	2
52 - 2,2′,5,5′-Tetrachlorobiphenyl	2	Aroclor 1260	2
66 - 2,3′,4,4′-Tetrachlorobiphenyl	2	Aroclor 1262	2
77 - 3,3′,4,4′-Tetrachlorobiphenyl	2	Aroclor 1268	2

Compound	Surrogate Reference	Compound	Surrogate Reference
87 - 2,2′,3,4,5′-Pentachlorobiphenyl	2		
101 - 2,2′,4,5,5'-Pentachlorobiphenyl	2	**PCB Homologs**	
105 - 2,3,3′,4,4′-Pentachlorobiphenyl	2	Monochlorobiphenyl	2
118 - 2,3′,4,4′,5-Pentachlorobiphenyl	2	Dichlorobiphenyl	2
126 - 3,3′,4,4′,5-Pentachlorobiphenyl	2	Trichlorobiphenyl	2
128 - 2,2′,3,3′,4,4′-Hexachlorobiphenyl	2	Tetrachlorobiphenyl	2
138 - 2,2′,3,4,4′,5′-Hexachlorobiphenyl	2	Pentachlorobiphenyl	2
153 - 2,2′,4,4′,5,5′-Hexachlorobiphenyl	2	Hexachlorobiphenyl	2
156 - 2,3,3′,4,4′,5-Hexachlorobiphenyl	2	Heptachlorobiphenyl	2
169 - 3,3′,4,4′,5,5′-Hexachlorobiphenyl	2	Octachlorobiphenyl	2
170 - 2,2′,3,3′,4,4′,5-Heptachlorobiphenyl	2	Nonachlorobiphenyl	2
180 - 2,2′,3,4,4′,5,5′-Heptachlorobiphenyl	2	Decachlorobiphenyl	2
183 - 2,2′,3,4,4′, 5′,6-Heptachlorobiphenyl	2		
184 - 2,2′,3,4,4′,6,6′-Heptachlorobiphenyl	2	**Internal Standard**	
187 - 2,2′,3,4′,5,5′,6-Heptachlorobiphenyl	2	Tetrachloro-m-xylene (TCMX)	A
195 - 2,2′,3,3′,4,4′,5,6-Octachlorobiphenyl	2		
206 - 2,2′,3,3′,4,4′,5,5′,6-Nonachlorobiphenyl	2	**Surrogates**	
209 - Decachlorobiphenyl	2	Dibromo-octafluoro-biphenyl (DBOFB)	1,A
		PCB 103	2,A
		PCB 198	3,A

The list of congeners analyzed in SINKEX samples compares reasonably well to congeners reported in environmental samples. Table 2-26 displays 36 congeners found in the environment is reproduced from McFarland and Clarke, 1989:

Table 2-26. PCB congeners reported in the environment (McFarland & Clark, 1989).

IUPAC No.				
Group 1A	Group 1B	Group 2	Group 3	Group 4
77	105	87	18	37
126	118	99	44	81
169	128	101	49	114
	138	153	52	119
	156	180	70	123
	170	183	74	157
		194	151	158
			177	167
			187	168
			201	189

Congeners listed in McFarland and Clarke (1989), but not analyzed in SINKEX samples are highlighted/shaded.

McFarland and Clarke (1989) described their list of 36 congeners by enzyme induction type. Induction of some enzyme types may be linked to metabolic carcinogenic processes. Group 1A-B congeners are most likely to contribute to adverse biological effects in environmental samples. Group 1A congeners are aryl hydroxylase enzyme inducers. Group 1B congeners are mixed type inducers (mixed function oxidase enzyme -type) frequently reported in environmental samples. The toxicity potential of congeners is inferred by mixed function oxidase enzyme induction.

Group 2 congeners are Phenobarbital type mixed function oxidase enzyme inducers prevalent in the environment, and most are relatively abundant in tissues. Group 3 congeners are weak or non-mixed function oxidase inducers, but are frequently found in environmental tissue samples (fish and invertebrates). Group 4 congeners are mixed type inducers that are relatively scarce in environmental samples. They are relatively scarce in tissue samples as well.

2.5.6.3.2. PAH Analysis

Water, sediment, and tissue sample extracts were analyzed for PAHs per ADL's SOP ADL-2827, "Determination of Polynuclear Aromatic Hydrocarbons and Selected Heterocyclic Compounds by Gas Chromatography/Mass Spectrometry in the Selected Ion Monitoring Mode." ADL's PAH analysis method is a modified version of EPA SW-846 Method 8270. Target analytes are shown in Table 2-27. The gas chromatograph/mass spectrometer (GC/MS) was operated in selected ion monitoring (SIM) mode to obtain the desired sensitivity that is comparable to that of a GC equipped with an electron capture detector (ECD). The GC/MS was first tuned with perfluorotributylamine (PFTBA) to verify accurate mass assignment and to maximize the sensitivity of the instrument in the mass range of interest (100 to 300 atomic mass units). After tuning, an initial calibration was performed which consisted of five calibration standards, at different concentration levels, spanning the concentration range of interest. Average response factors for each target compound and surrogate are calculated from the initial calibration standards relative to the internal standard compounds added to the sample extracts just prior to instrumental analysis (internal standardization). Continuing calibration standards, at a mid-range concentration level, were analyzed every 18 hours or after every 12 sample analyses to monitor sensitivity and linearity of the GC/MS. The average response factors generated from the initial calibration were used to calculate the concentrations of target compounds and surrogates in the environmental and quality control samples. The recoveries of the surrogate compounds spiked into the sample prior to extraction were used to assess sample-specific extraction efficiency. The target compound concentrations were adjusted based on sample-specific surrogate recoveries to correct for differences in extraction efficiency (surrogate-corrected).

Table 2-27. PAH target, surrogate, and internal standard compound list.

Compound (Abbreviation)	Surrogate Reference	Compound (Abbreviation)	Surrogate Reference
Naphthalene (C0N)	1	Benzo[a]anthracene (BAA)	3
C1-Naphthalenes (C1N)	2		
C2-Naphthalenes (C2N)	2	Chrysene (C0C)	3
C3-Naphthalenes (C3N)	2	C1-Chrysenes (C1C)	3
C4-Naphthalenes (C4N)	2	C2-Chrysenes (C2C)	3
		C3-Chrysenes (C3C)	3
Acenaphthene (ACE)	2	C4-Chrysenes (C4C)	3
Acenaphthylene (ACEY)	2		
Biphenyl (BIP)	2	Benzo[b]fluoranthene (BBF)	4
		Benzo[k]fluoranthene (BKF)	4
Fluorene (C0F)	2	Benzo[a]pyrene (BAP)	4
C1-Fluorenes (C1F)	2	Benzo[e]pyrene (BEP)	4
C2-Fluorenes (C2F)	2	Perylene (PER)	4
C3-Fluorenes (C3F)	2	Indeno[1,2,3-c,d]pyrene (IND)	4

Compound (Abbreviation)	Surrogate Reference	Compound (Abbreviation)	Surrogate Reference
		Dibenzo[a,h]anthracene (DAH)	4
Dibenzothiophene (C0D)	3	Benzo[g,h,i]perylene (BGP)	4
C1-Dibenzothiophenes (C1D)	3		
C2-Dibenzothiophenes (C2D	3		
C3-Dibenzothiophenes (C3D	3		
Phenanthrene (C0P)	3	**Surrogate Compounds**	
Anthracene (C0A)	3	Naphthalene-d8 (D8N)	1,A
C1-Phenanthrenes/Anthracenes (C1P/A)	3	Acenaphthene-d10 (D10AC)	2,A
C2-Phenanthrenes/Anthracenes (C2P/A)	3	Phenanthrene-d10 (D10PH)	3,A
C3-Phenanthrenes/Anthracenes (C3P/A)	3	Benzo[a]pyrene-d12 (D12BAP)	4,B
C4-Phenanthrenes/Anthracenes (C4P/A)	3		
Fluoranthene (FLANT)	3		
Pyrene (PYR)	3	**Internal Standards**	
C1-Fluoranthenes/Pyrenes (C1F/P)	3	Fluorene-d10 (D10FL)	A
C2-Fluoranthenes/Pyrenes (C2F/P)	3	Chrysene-d12 (D12C)	B
C3-Fluoranthenes/Pyrenes (C3F/P)	3		

2.5.6.3.3. Inorganics/Metals Analysis

The inorganics/metals analyses were conducted by the Environmental Sciences Department of the Engineering Division at the Florida Institute of Technology. Initially, each wet sediment sample was homogenized using a Teflon® mixing rod. Then, a portion (~2 g) of each sample was transferred into pre-weighed plastic vials to determine water content. Once transferred, the wet sediment and the vial were re-weighed. Then, the sediment samples were frozen, freeze-dried and re-weighed to determine the water content. The dried sediment samples were homogenized again using a Teflon® mixing rod.

About ~0.45-g of freeze-dried, homogenized sediment and standard reference sediment (BCSS-1) were totally digested in Teflon® beakers using concentrated, high-purity HF-HNO_3-$HClO_4$. Total digestion of the sediments is preferred because then no doubt remains about the absolute amount of metal associated with a sample. In the digestion process, 1 mL $HClO_4$, 1 mL HNO_3 and 3 mL HF were added to the sediment in a Teflon beaker and heated at 50 °C with a Teflon® watch cover in place until a moist paste formed. The mixture was heated for another 3 hours at 80 ºC and then an additional 2 mL HNO_3 and 3 mL HF were added before heating the sample to dryness. Finally, 1 mL HNO_3 and about 30 mL distilled, deionized water (DDW) were added to the sample and heated strongly to dissolve perchlorate salts and reduce the volume. The completely dissolved and clear samples were then diluted to 20 mL with DDW. This technique is 100% efficient with no loss of the elements studied and has been used successfully in the FIT laboratory for many years with a variety of sediment types.

Sediment for Hg analysis were prepared by heating 2-4 g of wet sediment in acid-washed, polyallomer centrifuge tubes with 4 mL HNO_3 and 2 mL H_2SO_4. Sample tubes were heated for 1 hour in a 90 °C water bath and allowed to cool. Each tube was centrifuged at 2,000 rpm and the supernate decanted into a 25 mL graduated cylinder. The sediment pellet was rinsed twice with 5

mL of DDW, centrifuged and decanted into the graduated cylinder before diluting to a final volume of 20 mL with DDW.

Labware used in the digestion process was acid washed with hot, 8N HNO_3 and rinsed three times with DDW. One procedural blank, one duplicate sample and one or two SRMs were prepared with this set of samples. Standard Reference Material BCSS-1 and MESS-2 issued by the National Research Council of Canada were used for sediment trace metal and Hg analysis, respectively, because BCSS-1 is not certified for Hg.

Samples, standard reference sediments, and procedural and reagent blanks were analyzed by either flame atomic absorption spectrometry (FAAS-Al, Cr, Cu, Fe and Zn) using a Perkin-Elmer 4000 instrument, Zeeman graphite furnace atomic absorption spectrometry (ZGFAAS-Ag) using a Perkin-Elmer 5100 instrument, cold vapor atomic absorption spectrometry (CVAAS-Hg) using a Laboratory Data Control Model 1235 Mercury Monitor or inductively coupled plasma-mass spectrometry (ICP-MS-Cd, Ni and Pb) using an ELAN 5000 instrument. All analytical techniques followed manufacturer's specifications, SOPs on file at FIT and the details provided in the QA/QC section below. These methods are closely akin to the EPA methods described for Series 200 (AAS, GFAAS, CVAAS and ICP-MS as described in EPA (1991). Matrix interferences were carefully monitored for all elements using the method of standard additions.

Table 2-28. Metals target analyte list and methods.

Element	Method
Silver (Ag)*	ICP-MS
Aluminum (Al)	FAAS
Cadmium (Cd)*	GFAAS, ICP-MS
Chromium (Cr)	FAAS, GFAAS
Copper (Cu)*	FAAS, ICP-MS
Iron (Fe)*	FAAS
Mercury (Hg)*	CVAAS
Nickel (Ni)*	FAAS, ICP-MS
Lead (Pb)*	ICP-MS
Zinc (Zn)	FAAS

*Also analyzed as Simultaneously Extracted Metals in sediments
GFAAS - Graphite Furnace Atomic Absorption Spectrophotometry
FAAS - Flame Atomic Absorption Spectrophotometry
CVAAS - Cold Vapor Atomic Absorption Spectrophotometry
ICP-MS - Inductively Coupled Plasma Mass Spectrometry

2.5.6.3.4. Sediment – Grain Size Analyses

Between September 1998 and November 1999 (SINKEX Cruises II, IV, and V), 14 samples were collected and analyzed for grain size at the MEC laboratory in Carlsbad, California (Table 2-10). Eighty grams of sediment were utilized for sediment grain size analysis. Prior to sample analysis sediments were stored on ice but not frozen. Grain size analysis was conducted in two parts: grain sizes larger than 64 microns were measured gravimetrically using graded series of sieves to provide grain sizes in half phi intervals. Sediment particles smaller than 64 microns

were analyzed using a pipette withdrawal procedure following the methods of Plumb (1981) providing grains size in whole phi units.

Sediment samples were thoroughly homogenized with a stainless steel spatula, and approximately 40 g of sediment was transferred into deflocculent bottles. Sodium hexametaphosphate (deflocculent) was added to the sediment in each bottle, and the contents were mixed and allowed to stand for 12 hours. Samples were washed over a 63-µm screen placed over an evaporating dish in order to catch the silt/clay solution. The sand retained on the screen was transferred to a porcelain dish and dried for 12 hours at 50-85 °C. The sand was then placed on the top sieve of an 11-sieve stack (2.00 mm to 63 µm) and shaken for 10 minutes. Starting with top sieve, the sand retained on each sieve was weighed to obtain the contribution of each phi size. The silt/clay solution was transferred from the evaporating dish to a 1,000-mL graduated cylinder, which was then filled to 1,000 mL with deflocculent. The cylinder was placed in a 24 °C water bath, and the sample mixed for one minute with a plunger. Six, 25-mL aliquots of sample were pipetted from the cylinder at different depths and times and transferred to 50-mL beakers. The beakers were dried for 12 hours at 95-105 °C and weighed to determine the percent silt and clay in the sample.

2.5.6.3.5. Sediment - Total Organic Carbon

Between the 1998 and 1999 sampling period, instrumentation used to process sediment TOC was changed at FIT and the new equipment dictated the extraction of a larger quantity of sediment. Re-analysis of the TOC data in spring of 2002 suggested a potential high bias for the 1998 samples, which were dominantly Inner Ring stations. Archived sediment samples from Inner Ring stations 1-1, 1-2, 1-3, 1-4, 1-6, 1-7, and 1-8 were subsequently reanalyzed using the 1999 instrumentation and methods and this data is used throughout this report. Station 1-5 was not reanalyzed, due to low archive quantities, and the original TOC data for this 1998 analysis is used in the report.

Initially, an aliquot of sediment was treated with concentrated HCl and dried to remove any inorganic carbon present. Once dry, the samples were re-weighed to determine the increase in weight due to the addition of acid and resultant formation of $CaCl_2$. Then, approximately 10-20 mg of pre-treated sediment was weighed into tin cups and combusted at 1,020 °C. The total organic carbon (TOC) content of sediment samples was determined using a Carlo-Erba NA1500 nitrogen-carbon-sulfur analyzer following manufacturer's instructions. The TOC concentrations were corrected to account for the increase in sediment weight. Precision was determined by analyzing selected sediment samples in duplicate and averaged about 3%. The accuracy of the OC analyses was obtained by analyzing standard sediment BCSS-1 issued by the National Research Council of Canada (NRC). Results obtained for the standard sediment BCSS-1 agree within the mean ± standard deviation of the values reported by the NRC.

2.5.6.3.6. Sediment - Calcium Carbonate

The calcium carbonate content of these sediments was determined using a carbonate bomb method (Schink et al., 1978). The carbonate bomb is a gasometric device that measures the amount of CO_2 produced when a sample is treated with 6N HCl. Variable amounts of pure $CaCO_3$ (0.10 to 1.00 g) were reacted with 18 mL of 6N HCl to obtain a calibration curve. The calibration curve consisted of plotting the pressure change (amount of evolved CO_2 and measure nanometrically) versus weight of $CaCO_3$. Next, the $CaCO_3$ content of each sample was

determined by reacting a 1.00g sample with 6N HCl and taking the resulting pressure change and finding the corresponding $CaCO_3$ weight on the calibration curve. The percent $CaCO_3$ was obtained by multiplying the corresponding weight by 100.

2.5.6.3.7. Sediment - Acid Volatile Sulfide/Simultaneously Extracted Metals (AVS/SEM)

Approximately 4-9 g of sample were homogenized, weighed and analyzed for AVS using the cold acid purge and trap method (Di Toro *et al,* 1990). The homogenized sample was placed in a flask containing 45 mL of deionized and deoxygenated water as the system had been purged using 99.999% nitrogen. The sulfide in the sediment was then volatilized by injecting 45 mL of deoxygenated 2N HCl through a septum. The flask was continuously stirred while being purged with 99.999% nitrogen. The nitrogen was passed through an impinger containing 45 mL of a sulfide anti-oxidant buffer (SAOB), which acted to trap and prevent oxidation of the sulfide. The SAOB consisted of 2M NaOH, 0.1M ascorbic acid and 0.1M EDTA. After a 1-hour reaction time, the SAOB solution was placed into a 100-mL volumetric flask. The SAOB solution was brought to a final volume of 100 mL by adding the solution obtained from rinsing the impinger flask with a 1:1 solution of SAOB and deionized, deoxygenated water. SAOB sulfide concentrations were determined with a sulfide specific ion probe (Orion Model #9616BN). The probe was calibrated for each analysis using known concentrations of sodium sulfide/SAOB solution with a minimum 5-point curve. The sediment/acid slurry remaining at the end of the reaction was filtered into 100-mL volumetric flasks through a Whatman #40 ashless paper filter. The reaction flasks were rinsed with DDW and the rinse was used to bring the filtrate to volume. This filtrate was then stored in acid washed polyethylene bottles until analysis for simultaneously extracted metals.

Concentrations of Cu, Fe, Ni and Zn were determined by flame AAS using a Perkin-Elmer model 4000 instrument. Values of Cd and Pb were determined by GFAAS using the Perkin-Elmer model 4000 instrument equipped with an HGA-400 heated graphite atomizer AS-40 autosampler.

2.5.6.3.8. Sediment - Geochronology

Approximately 8-10 grams of freeze dried sediment from each layer (0.5-1.0 cm thick) of the sediment cores were ground to a fine powder using a Spex 8000 mixer mill. The samples were then tightly packed into a 2 cm diameter, 5 cm long polycarbonate vial to a depth of 30±1 mm. A rubber stopper was used to seal the vial and was cemented into place with two part epoxy to prevent leakage of ^{222}Rn and disruption of secular[6] equilibrium between ^{226}Ra and ^{210}Pb. The samples were then set aside for at least 20 days to establish secular equilibrium and the activities of the various radionuclides were then determined by counting.

The sealed vial was placed in a well-type intrinsic germanium detector, "WiGe" (Princeton Gamma Tech Model IGW11023). The samples were then counted for a period of 1-2 days or until sufficient counts of the pertinent radionuclides were obtained (>1000 net counts for ^{210}Pb).

The peaks monitored for the purposes of this study were: ^{210}Pb at 46.5 KeV, ^{214}Pb at 295.2 KeV and 351.9 KeV, ^{214}Bi at 609.3 KeV, and ^{137}Cs at 661.6 KeV. The ^{226}Ra daughter isotopes ^{214}Pb

[6] Secular equilibrium is a steady-state condition of equal activities between a long-lived parent radionuclide (e.g., 238U) and it is short- lived daughter (e.g., 234U).

(2 peaks) and ^{214}Bi were used to determine the activity of ^{226}Ra. Detector efficiency and counting accuracy were standardized using standard reference river sediment 4350B (^{137}Cs) from the U.S. National Institute of Technology and Standards and using RGU-1 (^{210}Pb) from the International Atomic Energy Agency.

Sedimentation rates (S) in cm/yr were calculated using the following equations with the assumptions being made that there is no sediment mixing:

For ^{137}Cs:

$$S = \frac{\text{Depth in cm at which Activity}_{Cs\text{-}137}\text{=maximum}}{\text{(Year – 1963) in years}}$$

For ^{210}Pb:

$$S = \frac{\text{(-) decay constant for } ^{210}\text{Pb (0.0311 y}^{-1}\text{)}}{\text{Slope for plot of natural logarithm (ln) excess } ^{210}\text{ Pb vs. sediment depth}}$$

The activity of excess ^{210}Pb was calculated by subtracting the mean of $A_{(Pb\text{-}214,\ Bi\text{-}214)}$ from $A_{Pb\text{-}210}$.

2.5.6.3.9. Tissue - Trace Metals

In preparation for analysis, each tissue sample was homogenized with a Teflon® mixing rod. Then a 0.3 to 1.5 g portion of tissue was transferred to a tared plastic vial and reweighed for determination of percent water content (required for Hg analyses). At this time, an additional weighed portion of each homogenized wet tissue was transferred to a 50 mL polypropylene centrifuge tube to be digested for total Hg content. The plastic vials containing the wet tissue for percent water content were then freeze-dried, reweighed and the water content was calculated.

Tissue samples for determining concentrations of all metals except Hg were prepared using ~4 g of wet sample. The tissue was transferred to pre-weighed, 100 mL glass digestion flasks, reweighed, freeze-dried and the percent water content was calculated. These freeze-dried tissues and ~0.5 g portions of tissue SRM (#1566a, oyster tissue issued by the National Institute of Standards and Technology and DORM-2, a dogfish muscle issued by the NRC) were totally dissolved by refluxing with concentrated, high-purity HNO_3, H_2O_2 and HCl. Once the tissue samples were completely dissolved, the clear solutions were transferred to graduated cylinders, diluted to 20 mL with DDW rinses of the flasks, and then stored for analysis in 30 mL polyethylene bottles.

The wet tissue samples (1-3 g) for Hg analysis, along with 0.2-0.4 g portions of tissue SRM (#1566a, oyster tissue issued by the National Institute of Standards and Technology and DORM-2, a dogfish muscle issued by the NRC), were each digested by refluxing with high-purity HNO_3 and H_2SO_4 in the original polypropylene centrifuge tubes. Once the Hg digestion was complete, the solution was decanted into graduated cylinders, diluted to a final volume of 20 mL with reagent water rinses of the centrifuge tubes and stored for analysis in 30 mL polyethylene bottles.

Metal concentrations in the digested tissue samples, tissue SRMs and procedural blanks were determined by FAAS, GFAAS, ICP-MS or CVAAS in a manner compatible with the EPA Series 200 techniques (EPA, 1991). Concentrations of Al, Cd, Cu, Fe and Zn were measured by FAAS using a Perkin-Elmer Model 4000 system. Chromium and Ni concentrations were determined by

GFAAS using a Perkin-Elmer Model 4000 AAS with an HGA-400 graphite furnace and AS-40 autosampler. Concentrations of Ag were determined using a Perkin-Elmer Model 5100PC AAS with a HGA-600 and AS-60 autosampler. Tissue concentrations of Pb were measured by ICP-MS using a Perkin-Elmer ELAN 5000 spectrometer. Mercury concentrations were determined by CVAAS using a Laboratory Data Control Model 1235 Mercury Monitor. In all cases, the manufacturers specifications were followed and adherence to QA/QC requirements was maintained.

2.5.6.4. Definition of Method Detection Limits

The U.S. Environmental Protection Agency defines the method detection limit (MDL) to be "the minimum concentration of a substance that can be measured and reported with 99 percent confidence that the analyte concentration is greater than zero." The EPA procedure used for establishing MDLs is described in Appendix C to Part 136 "Definition and Procedure for the Determination of the Method Detection Limit - Revision 1.11," 40 CFR 136, 1986. This procedure consists of preparing and analyzing seven aliquots of a standard spiked at three to five times the expected MDL. The MDL is defined as approximately three times the standard deviation of the mean value for the seven analyses. The Arthur D. Little Environmental Monitoring and Analysis laboratory (ADL) calculated sample-specific method detection limits (sometimes referred to as minimum detection limits) by adjusting the MDL for the sample-specific preparation factors such as sample mass extracted, percent solid content, and dilution factors. It was the sample-specific MDL that was reported in the electronic data deliverables submitted to the Navy for the organic compound results. In general, few organic results were detected and reported at concentrations below the MDL. The Florida Institute of Technology, Marine and Environmental Chemistry Laboratories (FIT), calculated MDLs by the same method; however, FIT did not adjust the MDLs for sample-specific preparation factors. It was the unadjusted MDL that was reported in the electronic data deliverable submitted to the Navy for the inorganics/metals analyte results. In general, it was also the unadjusted MDL that was reported in the electronic data deliverable as the minimum reporting limits (MRL) for the inorganics/metals analyte results.

Sample-specific MRLs were calculated for each sample and were based on the concentration of the lowest level calibration standard adjusted for all sample preparation factors. ADL's MRL was consistent with a practical quantitation limit (PQL). The MRL is considered to be the value at which the results can be accurately quantitated. Results reported at concentrations less than the MRL (and thus less than the lowest calibration standard) were qualified with a "J" by the laboratory to indicate that the result was an estimated value. In general, on a compound-by-compound basis, the MRLs were normally 3 to 10 times greater than the MDL.

The fundamental difference between detection limits and quantitation/reporting limits is that detection limits are considered the lowest concentration that can be accurately <u>detected</u> by the method, whereas, quantitation limit is considered the lowest concentration that can be accurately <u>quantitated</u> by the method. Thus, results reported at concentrations below the quantitation/ reporting limit were considered to be estimated values and qualified with a "J". Often, the quantitation limit is set at the concentration equal to the concentration of the lowest level calibration standard. For example, in a CLP Statement of Work for Organics, the contract required quantitation limit (CRQL) is equal to the lowest level calibration standard adjusted for sample preparation factors. However, in the SINKEX Project, quantitation limits were generally

reported for organic nondetect data, and, MDLs or instrument detection limits (IDLs) were generally reported for inorganic nondetect data.

For this project, any compound confirmed to be present in the sample (e.g., mass spectrum meets identification criteria) was reported as a positive result regardless of the result concentration. All results detected at concentrations below the sample-specific MRL were qualified with a "J" by the laboratory. These results were considered to be estimated values due to uncertainty in quantitation below the calibrated range of the instrument. There were also positive results reported at concentrations below the MDL when the compound identification criteria were met. If a compound was not detected or did not meet the compound identification criteria, the compound was reported as a nondetect. In the electronic data deliverables, the nondetect values were reported with a value of zero and a "ND" or "U" qualifier. It was left to the judgment of the data user to determine the best value to be used for nondetects based on specific data analysis needs.

For samples collected during cruises IV and V, the concentration of the lowest level initial calibration standard for Method 8081 (PCB Congeners by GC/ECD) was reduced from 5 parts-per-billion (ppb) to 1 ppb. Acceptable linearity of the instrument was consistently achieved when the 1 ppb standard was included with the initial calibration, indicating that the instrument's sensitivity was adequate to reliably detect and quantitate the compounds to this lower concentration level. However, the Method 8081 MDL study, which was originally performed using 5 ppb standards, was not immediately redone to coincide with initial usage of the lower initial calibration standard. Thus in several cases, the sample-specific MRLs are less than the associated MDLs.

2.5.6.5. Quality Assurance/Quality Control for Chemistry Samples

2.5.6.5.1. Quality Assurance for Organics

Sample processing was conducted in accordance with the laboratory Quality Assurance Plan (QAP). The plan describes the laboratory quality assurance (QA) structure and organization. The following is a general description of some of the plan elements as they apply to this project. Exceptions to the quality control elements are documented and filed with the appropriate laboratory report.

Water, sediment, sediment cores, and tissue samples were collected by SSC-SD with contractor support. The samples were transported to the laboratory at appropriate temperatures and under strict chain-of-custody procedures. Arthur D. Little received the study samples intact and in good condition. A listing of the project samples, copies of the chains-of-custody, sample results, and quality assurance/quality control (QA/QC) results are included in the ADL data reports for Delivery Orders 17, 21, and 24.

Detailed laboratory records were maintained throughout the processing of samples. All raw instrumental data are archived electronically. Completed records or copies of forms were collated into a binder as a final data package with sufficient detail for audit. The final laboratory data package includes:

- Lot numbers, vendor, and preparation records for reagents and standards
- Sample preparation records
- Analytical procedures used that are not documented in laboratory SOPs

- Instrument analysis records
- Instrument raw data hardcopy
- Documentation of observations or deviations encountered

2.5.6.5.2. Quality Control for Organics

A number of measures were added to the processing of samples to monitor quality control (QC) and to aid in the assessment of the usability of the data with respect to the project objectives. An important part of this was the evaluation of specific QC samples for accuracy, precision, and potential contamination. The method summaries included in this report and the project-specific work plan contain details of the quality control samples required for each analytical method. The data quality analyses and Data Quality Objectives (DQOs) for the PAH and PCB analysis methods are summarized in Table 2-29 and Table 2-30.

Each lot of solvent received at the laboratory was tested by the applicable analytical method to determine potential solvent contamination prior to use.

Preparation of analytical standards is described in the relevant laboratory SOP. Prior to spiking the samples with surrogate, matrix spike, and/or internal standard solutions, all standard solutions were analyzed to determine accuracy of preparation and potential contamination.

Instruments were calibrated prior to sample analysis by analyzing standard solutions of containing the target and surrogate compounds at different concentration levels spanning the concentration range of interest. The linearity of the instrument over the selected concentration range was checked. A continuing calibration standard was analyzed regularly to check the stability of the instrument response and the compound retention times. If the variability of either the initial calibration or the daily calibration did not meet the criteria set in the project-specific work plan, a new calibration was run and the affected samples reanalyzed.

To assess the accuracy of the calibration standards, an independent reference material (IRM) was analyzed. Instrument calibration was considered acceptable if the reported concentrations of the compounds in the IRMs were within 15 percent of the target concentrations (for PAH analysis only).

Standard Reference Materials (SRMs) were obtained from National Institute of Standards and Technology (NIST) and are contaminated environmental samples that have been repetitively analyzed to determine certified values. For this project tissue and sediment SRMs were prepared and analyzed with the tissue and sediment samples. SRMs are used to assess the effect of the sample processing procedures and matrix on method accuracy.

A solution of an assayed crude oil was analyzed with each analytical sequence. The results were compared to previously established laboratory means to assess method accuracy. The solution was also used to provide pattern information and aid in sample fingerprinting (for PAH analysis only).

A procedural blank was processed and analyzed with each sample preparation batch in order to monitor potential contamination resulting from laboratory solvents, reagents, glassware, and processing procedures.

Blank spikes and blank spike duplicates were prepared by spiking representative target compounds into a blank matrix to assess the effect of the sample processing procedure independent of sample matrix effects on method accuracy and precision.

Duplicate samples were prepared by extracting and analyzing a second representative aliquot of a sample. Comparisons of the original and duplicate sample results were used to assess the effect of the sample processing procedures and sample matrix effects on method precision.

A surrogate is a known compound, which is not present in environmental samples, that is added to a sample prior to processing. The chemical properties of the surrogate compounds must be close to the target compounds. The surrogate was measured to assess the sample preparation efficiency and impacts of sample handling. Surrogates may also be used to adjust the target compound concentrations to correct for loss during sample preparation (surrogate correction). Surrogates were added to all samples prior to preparation.

Target compound concentrations, surrogate and matrix spike recoveries, and additional QC sample results were determined in the respective laboratories. After careful checking and review, analysts transferred data electronically from the instrument data systems to ADLs data management software for further data review, qualification, and edits. All data summary forms were generated from the data management software and were compared with the instrument quantitation reports for accuracy. The data summaries were arranged in spreadsheet format.

The chemistry data for each analysis were reduced and reviewed by the laboratory staff and then assembled into the final laboratory data package. The assembled package was reviewed and validated by the facility supervisor or staff responsible for each analysis. The data were checked to ensure that data quality objectives were met, that the analyses met the project objectives, and that the data were traceable and defensible. The Project Manager also reviewed the data for compliance with the documented procedures and quality objectives. Data were also reviewed for internal consistency and against expected or known values. All final laboratory data packages and the associated electronic data deliverables were audited by the Quality Assurance Manager or data review specialists according to the procedures outlined in ADLs data auditing SOP.

All results detected at concentrations below the sample-specific minimum reporting limit (MRL), but above the method detection limit (MDL), are estimated values due to uncertainty in quantitation below the calibrated range of the instrument and due to increased variability at concentrations near the method detection limit (MDL). However for the SINKEX study, additional measurements that fell below MDL would have been qualified as estimated (J) due to minor quality control exceedances, if these data were validated according to EPA guidelines.

Table 2-29. Data quality objectives and criteria for PAHs and PCB congeners, homologs, and Aroclors by GC/MS SIM.

Element or Sample Type	Minimum Frequency	Data Quality Objective/Acceptance Criteria
Initial Calibration	Prior to every batch sequence.	5-point curve. %RSD <25% for 90% of analytes and <35% for all analytes.
Continuing Calibration	Must end analytical sequence and every 12 samples or 18 hours, whichever is more frequent.	%D <25% for 90% of analytes and <35% for all analytes.
Procedural Blank	Every batch/every 20 field samples.	No more than 2 analytes to exceed 5x PQL unless analyte was not detected in associated sample(s) or associated sample compound concentrations are >10x blank value.
Blank Spike/Blank Spike Duplicate Sample	Every batch/every 20 field samples.	50-150% recovery, RPD <35%.
SRMs (SRM 1941a for sediment, 1974a for tissue)	Every batch/every 20 field samples.	Values ±35% difference of true value for all certified analytes, two may exceed.
Duplicate Analysis	One per 40 field samples.	RPD <35% for all analytes that are detected at concentrations >10 times the MDL; mean RPD <35%.
Matrix Spike, Matrix Spike Duplicate Sample	Every batch/every 20 field samples.	45-150% recovery, RPD <35%.
Surrogate Standards	Every sample.	40-125%: d8-napththalene, d10-acenaphthene, d10-phenenthrene 40-135%: d12-benzo[a]pyrene 45%-125%: all surrogates, one is allowed out
IRMs	One set per batch of samples after every ICAL.	Values ≤20% difference of true value for all certified analytes.
Oil Reference Standard (North Slope Crude)	One set per batch of samples after every ICAL (PAH analysis only).	North Slope Crude <35% D from laboratory mean for target compounds (use surrogate-corrected values) detected at concentrations > the reporting limit.

Table 2-30. Data quality objectives and criteria for PCB congeners by GC/ECD.

Element or Sample Type	Minimum Frequency	Data Quality Objective/ Acceptance Criteria
Initial Calibration	Prior to every batch sequence.	5-point curve. %RSD <25% for 90% of analytes and <35% for all analytes.
Continuing Calibration	Must end analytical sequence and every 10 samples or 18 hours, whichever is more frequent.	%D <25% for 90% of analytes and <35% for all analytes.
Procedural Blank	Every batch/every 20 field samples.	No more than 2 analytes to exceed 5x PQL unless analyte was not detected in associated sample(s) or associated sample compound concentrations are >10x blank value.
Blank Spike/Blank Spike Duplicate Sample	Every batch/every 20 field samples.	50-150% recovery, RPD <35%.
SRMs (SRM 1941a for sediment, 1974a for tissue)	Every batch/every 20 field samples.	Values ±35% difference of true value for all certified analytes, two may exceed.
Duplicate Analysis	One per 40 field samples.	RPD <35% for all analytes that are detected at concentrations >10 times the MDL; mean RPD <35%.
Matrix Spike, Matrix Spike Duplicate Sample	Every batch/every 20 field samples.	45-150% recovery, RPD <35%.
Surrogate Standards	Every sample.	45%-125% all surrogates, one is allowed out
IRMs	One set per batch of samples after every ICAL.	Values <15% difference of true value for all certified analytes GC-ECD.

2.5.6.5.3. Quality Assurance/Quality Control for Inorganics and Metals

Sample Tracking Procedure. Upon receipt, each sediment and tissue sample received by the Marine & Environmental Chemistry Laboratories at Florida Institute of Technology was carefully inspected to insure that it was intact and that the identification on the sample container matched that found on the custody sheet. All samples were kept refrigerated (~1 ºC) until processed for analysis.

Quality Control Measurements for Analysis. For this project, QC measures included balance calibration, instrument calibration (FAAS, ZGFAAS, CVAAS, Gamma spectrometer, NCS analyzer and ICP-MS), matrix spike analysis for each metal, duplicate sample analysis, SRM analysis, procedural blank analysis, and standard checks. With this batch of samples, one procedural blank, one or two SRM, one duplicate sample and one matrix spiked sample also were analyzed. Data quality objectives (DQOs) for these QC measurements are provided in the table entitled "Data Quality Objectives and Criteria".

Instrument Calibration. Electronic balances used for weighing samples and reagents were calibrated prior to each used with certified (NBS traceable) standard weights. All pipets (electronic or manual) were calibrated prior to use. Each of the spectrometers used for metal analysis, as well as for the calcium carbonate determinations, was initially standardized with a three- or five-point calibration and a linear correlation coefficient of $r \geq 0.999$ required before experimental samples could be analyzed. Analysis of complete three- to five-point calibrations

and single standard checks alternated every 5 to 10 samples until all analyses were complete. In all instances, the linear regression of the standards met the DQOs. The relative standard deviation (RSD) for the initial calibration and subsequent calibration were <15% in all instances.

Matrix Spike Analysis. Matrix spikes were prepared for a minimum of 5% of the total number of samples analyzed and included each metal to be determined. Results from matrix spike analysis, using the method of standard additions, provides information on the extent of any signal suppression or enhancement. Results for matrix spikes were within the 80-120% limit specified in the DQO, for all metals except Hg in the tissue samples (~70%). This low recovery is commonly observed when organic-rich samples are analyzed by CVAAS; thus, all tissue Hg concentrations reported in the tables were corrected for their spike recoveries.

Duplicate Sample Analysis. A duplicate subsample from homogenized field samples (as distinct from field replicates) was prepared in the laboratory. This laboratory duplicate was included as part of the set of sample digestions and analyses and provides a measure of analytical precision. Results for the duplicate sample was below 25% as required by the DQO.

Procedural Blank Analysis. One procedural blank was prepared with this set of samples to monitor any potential for metal contamination. The blank utilized the same reagents, handling techniques and analytical scheme as the experimental samples. No contamination from either of these sources was noted and the analytes did not exceed 5 times the MDL.

Field Blank Analysis. Four field blanks (2 control DI water and 2 tissumizer DI water samples) were analyzed for trace metals to monitor potential contamination from field operations. No contamination from any of these sources was noted and the concentrations of the analytes in the blanks were <5 times the MDL, except for Ag (2nd set of tissue samples in both the control DI water and the tissumizer DI water) and Ni (1st set of tissue sample in the tissumizer DI water).

SRM analysis. A common method used to evaluate the accuracy of environmental data is to analyze SRMs, samples for which consensus or "accepted" analyte concentrations exist. The following SRMs were used: Marine Sediment, BCSS-1, issued by the NRC; Oyster Tissue (NIST SRM #1566a) and Dogfish Muscle (NRC DORM-2). Metal concentrations obtained for the SRMs were within ±20% of accepted values for >85% of other certified analyses as required by the DQO

2.5.7. Toxicity Testing

The toxicity and bioaccumulation testing was conducted by the MEC Inc., Carlsbad, CA. Bioassay/toxicity testing has become a standardized method to determine if sediment samples have the potential to impact sensitive test organisms. It was decided that that the sediments near the study vessel would be sampled and evaluated for toxicity. Additionally, the protocols for conducting these tests and the criteria for evaluating the results would be based on ocean disposal criteria (USEPA/USACE 1991). While toxicity testing for ocean disposal of dredged sediments has become routine, interpretation of laboratory results requires evaluation of the test conditions as well as other environmental factors that can affect test results. Since factors other than the Contaminant(s) of Concern (COC) can affect laboratory test results, toxicity results are often considered just one component of the sediment "triad" approach for evaluating sediment toxicity. In the broader context of evaluating sediment toxicity, the sediment "triad" approach has been widely used and it includes not only toxicity tests but also measures of sediment contaminant concentrations and infaunal communities. Thus, to support an evaluation of

sediment toxicity one should not only consider the laboratory toxicity test results, but also whether there were any chemicals of concern at concentrations that might cause effects as well as the "health" of the infaunal community. For example, sediment toxicity is most likely indicated if there is poor survival in an acute toxicity test coupled with COC at concentrations known to cause effects, and a depauperate infaunal community. There should be no failure of an amphipod acute toxicity test with no sediment concentrations of COC above effects threshold and a diverse and healthy infaunal community. However, if there is failure under these conditions, this would suggest that confounding factors other than sediment toxicity (e.g., grain size effects, quality and availability of organic carbon, ammonia) may be affecting the laboratory toxicity test results and consequently, the tested sediments may not be toxic.

Sampling methods for sediments have been presented earlier. Sediments used for toxicity testing utilized only the upper 3-cm of sediments collected in each boxcore sample as these sediments represent the strata most likely to contain contaminants from the ex-AGERHOLM.

Toxicity tests conducted for the 14 samples included a 10-day solid phase test using the amphipod *Rhepoxynius abronius* and a 28-day solid phase test using the polychaete *Neanthes arenaceodentata*. (Because the concentrations of PCBs in the sediments were of concern with regard to chronic toxicity, the standard methods were tailored to measure longer-term effects. The 28-day solid phase test evaluating growth and survival in *Neanthes arenaceodentata* was chosen over the USEPA/USACE (1991) recommended 10-day test so that the potential chronic sublethal effect of sediment-associated contaminants could be determined.) Twenty-eight-day exposure, bioaccumulation tests with the polychaete *Nephtys caecoides* and the bivalve *Macoma nasuta* were performed for 14 of the samples; tissues from the bioaccumulation test organisms were analyzed for chemistry by ADL. Testing procedures strictly adhered to MEC's bioassay and bioaccumulation protocols, which were developed to comply with ocean disposal criteria (USEPA/USACE 1991).

A reference toxicant (positive control) test using the amphipod test species was performed prior to test initiation. Test results were compared to MEC's reference toxicant response (LC_{50}) database to determine the relative sensitivity of the population and the validity of the bioassay. Five laboratory replicates were tested for each sample and the control sediment. Homogenized test sediment (approximately 200 mL) was placed in a 1-L container and 800 mL of filtered sea water was added. Aeration was provided through plastic tubing, with care taken to avoid disturbing the sediment. Water quality measurements were taken in one replicate chamber daily, and included pH, salinity, temperature and dissolved oxygen. Porewater ammonia and sulfide were measured at the start and finish of the test for each sample. The number of emerged amphipods was recorded daily. After 10 days of exposure, live amphipods were carefully removed by sieving from the sediment in which they were tested and placed into clean seawater. Data obtained from these studies was the number surviving at the end of 10 days. The test was re-run if mean control survival was below 90 percent or individual replicate survival was below 80 percent and there was sufficient test material available.

The polychaete worm bioassay was run with *Neanthes arenaceodentata* using standard 10-day acute test methods (ASTM 1992). Polychaetes were obtained from a culture maintained at California State University, Long Beach, California. Initial stock densities in each replicate were 20 organisms per test chamber for each species. Five laboratory replicates were tested for each sample and the control sediment. Homogenized test sediment (800 mL) was placed in a 3.5 L container and 3 L of filtered sea water was added. Seawater was allowed to flow through the test

chamber at a rate of approximately 100 mL per minute. The number remaining alive at the end of 10 days was recorded. Water quality measurements were taken in one replicate chamber daily and included pH, salinity, temperature and dissolved oxygen. Porewater ammonia and sulfide were measured at the start and finish of the test for each sample. All instruments used were calibrated and logged daily. The number of dead worms were recorded daily. After 10 days of exposure, live worms were carefully removed by sieving from the sediment in which they were tested, placed into clean seawater, and examined under a microscope. Each animal was inspected for survival using gentle prodding. Survival/mortality was recorded for each of the five laboratory replicates and mean survival was calculated and recorded (USEPA/USACE, 1991). The test was re-run if mean control survival is below 90 percent, or individual replicate survival was below 80 percent.

2.5.8. Bioaccumulation Studies

Assessment of bioaccumulation potential was carried out using *Nereis virens* and *Macoma nasuta* over a 28-day exposure period. The tests were initiated using test and control sediments and run with both organisms in single containers. One replicate per station was tested for the 14 composite sediment samples (Table 2-10). A minimum of 10 *Nereis* and 20 *Macoma* were placed in each test chamber. The test chambers were maintained under flow-through conditions (San Diego Bay water), and daily water quality measurements were taken on replicate chambers as specified in the 10-day acute test. On Day 28, the sediments were sieved to remove the worms and clams. The surviving animals were placed in clean flow-through aquaria to depurate for 48 hours. After the 48-hour depuration period, the organisms were chilled to 4 °C and shipped via next-day air to ADL for tissue residue analysis. There was no survival criterion for test acceptance..

Zero Time organisms are those organisms that are tested just prior to the beginning of the test in order to establish a baseline of tissue concentrations and determine if they have been contaminated from a prior exposure. Control organisms are those that are run separately without being exposed to contaminated sediment but at the same time as the exposed organisms, for quality control purposes (primarily cross-contamination issues). The Zero Time organisms are sacrificed and tested prior to the beginning of a bioaccumulation test and the Control organisms are sacrificed and tested at the end of the bioaccumulation test

2.5.9. Sample Custody Procedures

Sample custody control was maintained at each laboratory through the use of several tracking systems designed to protect sample integrity. The sample custodian initiated laboratory chain of custody documentation when the courier relinquished the sample. Samples were inspected for the following to ensure that:

- minimum sample volumes were received,
- appropriate containers and preservatives were used,
- acceptable sample conditions were maintained (e.g., temperature, no breakage), and
- samples were received within allowed shipping time (e.g., next-day air).

Immediately upon receipt by the laboratory, the sample custodian assessed and documented the conditions of the samples and initiated sample log-in. The contents of each shipping container were checked against the information on the chain of custody forms. Temperature blank samples

were checked to verify that samples were maintained within specified temperature ranges. If anomalies were noted in the chain of custody form, the Project Manager was to be informed. Any samples that were not properly preserved were to be noted on the chain of custody form and the Field Manager was be notified immediately. The Field and Project Manager were to determine the necessary corrective action. The laboratory assigned an internal unique identifier to each sample, or used the sample identification number assigned in the field with the container number and project name to track individual sample containers so that the sample would not be confused with samples from another project. There were no anomalies, nor improperly preserved samples noted/reported during the conduct of this study.

The field chain of custody document was completed and maintained in the project file. While within the laboratory, the sample was stored in appropriate areas to maintain sample integrity. Upon completion of the analysis, any remaining sample was placed into long-term storage. When sample analysis and all quality control checks are completed and a final data report is issued, the unused sample portion will be stored up to six months or longer if requested by the SSC SD Principal Investigator. Samples will not be disposed of without the written permission of the SSC SD Principal Investigator. Sample disposal will be documented in the project file.

2.5.10. Methods for Supplementary Sablefish Assessment

The full report for the supplementary Sablefish Assessment is provided in its entirety as Supplement I. Excerpts from the text and some tables and figures have been brought forward to this main document. Methods for individual fish trap deployment and recovery, and fish sampling and analysis were described previously in their respective sections. This additional section summarizes and explains modifications to these methods for the supplementary Sablefish assessment reported in the Exposure (3.2.4) and Effects (3.3.3) sections. Sablefish were collected with fish traps between September 1998 and November 1999 during Cruises II, IV, and V. Tissue samples were analyzed for PCBs by Arthur D. Little, Inc., Cambridge, MA (ADL) using gas chromatography (GC) electron capture detection (ECD, ADL 1999a) and/or GC/mass spectroscopy (MS) with selective ion monitoring (SIM, ADL 199b) (Supplement I). For each method the samples were processed in the same manner for sample preparation, extraction, and fractionation, only the injection and quantification procedures differed. For the ECD method, samples were analyzed for 26 congeners and 8 Aroclors. For the SIM method data on 26 (or 29) congeners, 10 homologs, and 8 Aroclors were obtained (Supplement I).

Fish from ex-AGERHOLM (Ship) and reference sites were collected with fish traps as described in 2.5.3.2.2. Because only limited numbers of fish were collected during each cruise, the fish from all the cruises were pooled to evaluate potential ecological risks (Supplement I).

Following Cruise II, two fish from the reference site and seven fish from the ship were analyzed using the ECD method. The ECD method was not capable of quantifying the total amount of PCBs present. While empirical relationships between the sum of certain congeners measured by ECD and Total PCB are available for coastal and estuarine bivalves and fish (NOAA, 1991, Hyland et al. 1998) no such empirical relationships were available that were applicable to deep sea fish like Sablefish.

Because human health and ecorisk assessments require estimates of total PCB to make accurate determinations of risk, analytical methods were needed to obtain data on total PCB concentrations present in the tissues of Sablefish. Therefore, Sablefish samples were analyzed using the GC/MS SIM method (ADL 1999b) to determine the amount of total PCBs present. In

order to obtain the best data possible and assure comparability between all cruises, fish samples collected during Cruise IV and Cruise V were analyzed by both ECD and SIM methods (App.Q, Table 2). The methods were ADL-2818.05 "Determination of Chlorinated Pesticides and PCB congeners by Gas Chromatography Electron Capture Detection" (ECD, ADL 1999a) and ADL-2845.00 "Determination of PCBs by Gas Chromatography/Mass Spectrometry in Selected Ion Monitoring Mode" (SIM, ADL 1999b). Because the SIM method provided data for PCBs present in each homolog group, the total amount of PCBs (Total PCB) present could then be determined from the sum of all homolog groups (i.e., from mono- to decachlorobiphenyl).

The precision of both methods was evaluated by conducting a method detection limit (MDL) study. The MDL study consisted of 8 repeated measurements of a sample of fish tissue (haddock) spiked with known amounts of the individual PCB congeners. The results were tabulated and the arithmetic mean and standard deviation were calculated to determine the theoretical MDL of the analytes of interest (App. Q, Table 3). The MDLs obtained for both methods were very similar (App.Q, Figure 3). Slightly lower detection limits were obtained by the SIM method for congeners 8, 18, 77, 138, 170, and 209; and slightly lower detection limits were obtained by the ECD method for congeners 101, 105, 153, 156, 180, and 206 (App.Q, Table 3, Figure 3). Differences in the absolute concentration of congeners measured in the fish tissue samples maybe due to differences in the samples and spiking levels used in the MDL studies (i.e., both methods were not performed on the same samples at the same times). The MDL study yielded slightly lower detection limits for ECD for most of the congeners (App.Q, Table 3).

The accuracy of the two methods was evaluated by comparing the results obtained from the analysis of the Standard Reference Material (SRM) 1974a Mussel Tissue. Obtained from the National Institute of Standard and Technology (NIST 1999), the mussel tissue SRM comes with certified values for most of the congeners analyzed. Samples of the SRMs were included in each of the analytical batches and were processed in the same manner as the field samples. Three SRM samples were analyzed by ECD and two SRM samples were analyzed by SIM (App.Q, Figure 4). The upper and lower bounds of the certified value (error bars on Figure 4) indicates the "true value" of the SRM sample. Analytical results from each batch of samples were considered acceptable if no more than two analytes exceeded 35% of the certified value of the SRM. Overall, the two methods obtained very similar results (Figure 4). The ECD method tended to over predict the certified value of congeners 118 and 187 and the SIM method tended to over estimate the certified valued of congeners 28, 118, and 153. For most of the congeners there appeared to be a greater differences in the results of repeated measures of the SRM by the same method than from the two methods (i.e. greater variance between samples than between the methods). There did not appear to be any bias in either of the methods and they accurately measure the levels of PCB present in the SRM mussel tissue samples.

2.5.11. Data Quality Objectives

The data quality objectives (DQOs) of this study were that the data produced must be of sufficient quality to satisfy the intended use of the data in a scientifically sound manner (Stanely and Verner 1985, USEPA 1994). Data quality objectives were established for six aspects of data quality. These were that the data were representative of actual conditions at the site, that the data were complete enough to make a valid assessment, that the data from different stations and different cruises were comparable, and that the data were accurate, precise and reproducible

(Reifsteck et al., 1993). To explicitly define the data quality objectives for this study a seven-step process was followed (USEPA, 1994).

Step 1 – State the problem. Data were obtained that could be used to characterize chemical exposure levels in the sediments adjacent to the ex-AGERHOLM, evaluate the toxicological and bioaccumulation potential of the sediment, evaluate the status of the benthic community, measure the field tissue concentrations of demersal fishes, and determine if there are any adverse impacts related to chemical releases from the ship. In order to determine whether the ex-AGERHOLM may be the cause of any elevated exposure or adverse effects, reference data were required to allow comparison to similar deep-water environments not exposed to the presence of a sunken Naval vessel. Additionally, ancillary data on the currents, characteristics of the sediment substrate, and general ecological properties of the study area were required to provide a context for interpreting the presence of harmful effects and estimating the potential for long-term impacts.

The analysis goals of the ecological risk assessment required analytical chemistry methods that:

- were capable of detecting chemicals below levels that can cause ecological effects as well as at levels associated with background or naturally occurring concentrations,
- were capable of differentiating chemical levels from interferences due to sample matrices, and
- could be reliably reproduced and verified (Johnston and Valenti 1999).

The analysis goals of the ecological risk assessment required methods for biological and toxicological analyses that

- were capable of eliciting ecologically relevant responses,
- could differentiate responses from interferences due to test methods and sampling handling, and
- could be reliably reproduced and verified.

All methods were selected to provide data that are scientifically sound and which can meet the representativeness, completeness, comparability, accuracy, precision and reproducibility required for the risk assessment.

Step 2 - Identify the decision. Quality assurance and quality control (QA/QC) procedures were defined to evaluate the data, to the extent possible, for representativeness, completeness, comparability, accuracy, and precision. Performance-based QA/QC requirements were defined which required the laboratory to use of accuracy materials (e.g., certified or standard reference materials and laboratory control materials), calibration standards, method blanks, matrix spike samples, laboratory duplicates, internal standards, injection standards, and interlaboratory calibrations. The results of these QA/QC analyses were provided to verify the quality of the data, all the raw data to be validated, and assure comparability of the data with data generated by different laboratories using different analytical procedures (Johnston and Valente, 2000).

Step 3 - Identify inputs to the decision. Procedures were implemented that allowed the data to be evaluated and verified for representativeness, completeness, comparability, accuracy, and precision. Representativeness is defined as "the degree to which the data accurately and precisely represented a characteristic of a population parameter, variation of a property, a process characteristic, or an operational condition" (Stanley and Verner, 1985). Representativeness can

be applied to field sampling in two tiers: how representative a station is to the study area within which it is located, and how well the actual samples or data collected from station are representative of the station (Reifsteck et al. 1993). Station location and the overall study design were formulated to assure that the samples collected were representative of the conditions near the ship and reference conditions in similar deep-water environments. Therefore it was very important that the ship's crew, ROV operators, and scientific personnel (i.e., the "SINKEX team") could accurately locate and relocate stations. Once a station was properly located, the SINKEX team took all precautions to assure that intact cores were obtained from the station and that the cores were transported undisturbed to the surface.

Completeness is defined as "a measure of the amount of data collected from a measurement process compared to the amount that was expected to be obtained under the conditions of measurement" (Stanley and Verner, 1985). As applied to the sampling plan, this means that for each station a minimum amount of material was required to complete all of the tests. In the case that not enough material is collected, certain tests would have been "incomplete" because they could not be performed. For example, if only enough sediment material was collected to allow chemistry analysis, then it will not be possible to conduct bioassays or infauna analyses. In that case the data for chemistry would be "complete" but the data for toxicity and infauna would be "incomplete". In designing the sampling program a minimum number of data points were required to complete the assessment. As a result, a completeness goal of 90% was established. This means that to conduct the assessment as planned, 90% of the expected data must be collected (Reifsteck et al., 1993).

Comparability is defined as "the confidence with which one data set can be compared to another" (Stanley and Verner, 1985). To obtain data sets that were comparable among different cruises, literature data, and other sources of information, the samples and data were produced in a consistent, reproducible manner. Therefore, the ROV operator and scientific personnel took care to following the sample and analysis methods defined in the workplan.

Accuracy is defined as the difference between a measured value and the true or expected value, and represents an estimate of systematic error or net bias. Where possible, accuracy was evaluated by analyzing certified standards, laboratory control materials, and appropriate blanks to obtain a measure of accuracy. Precision is defined as the degree of mutual agreement among individual measurements and represents an estimate of random error. Precision was evaluated by the analysis of a suitable number of field replicates, laboratory duplicates, and other control samples. Together these two aspects provided an estimate of the total error or uncertainty associated with individual measurements.

Step 4 - Define boundaries. The boundaries of the assessment for impacts to sediment and water are the bottom environment directly adjacent to the ex-AGERHOLM (Inner Ring), and reference areas defined by sample areas that create reference ring(s) around the hulk, including the Outer Ring located at 1,000 m in all directions from the hulk. Demersal fishes caught in the general area of the ex-AGERHOLM (within ~250 m) were compared to specimens collected 3.8 nm from the hulk.

Step 5 - Develop decision rule. The results obtained from the quality control procedures were to be used to judge the quality and usability of the data. If results indicated that the data did not meet the quality criteria required for the assessment, they were not used in the analysis. A decision matrix was developed to represent the possible outcomes and conclusions that can be

derived from the data. The overall decision was based on the weight of evidence of exposure and effects data collected at the ex-AGERHOLM and reference stations.

Step 6 - Specify tolerable limits on decision errors. Direct observation and oversight by SSC-SD personnel of all field sampling and data collection activities was conducted. Any samples that were not collected intact and transported undisturbed to the surface were rejected. Laboratories conducting chemical and biological analyses were required to demonstrate proficiency through routine analysis of accuracy-based materials. Ongoing performance evaluation exercises were conducted to demonstrate initial capability (i.e., prior to the analysis of actual samples) and on a continuous basis throughout the project. The laboratories were required to initiate corrective actions (e.g., reanalyze the sample) if their performance fell below certain predetermined minimal standards defined in the quality assurance plan.

Step 7 - Optimize the design. Based on the results obtained, the sampling design was modified to reflect new knowledge about the site and address key uncertainties in the assessment. The quality assurance plan was updated to reflect realized performance criteria and allow for improvements in analysis methods and instrumentation.

2.5.12. Data Management and Statistical Analysis

A master database was designed, populated and maintained at SSC-SD. Pertinent project data, including field information, laboratory sample and quality control results, was transcribed or imported into the database under the direction of the principal investigator. Chemical, physical, and biological data were received electronically from various investigators in pre-defined formats. Field data were manually entered by into Excel files directly from field notebooks. Only the principal investigator has authority to allow changes to data placed in the database.

Statistical analyses of project data was performed by computer using the statistical software SAS® Version 8.0 in batch programming mode. Data analysis consists primarily of graphics and statistics used to support interpretation of data generated in the project. Graphical displays of data were generated using a variety of commercial software packages (e.g., CorelDraw®, Surfer®).

The idealized sampling design originally intended for the SINKEX program is presented in Figure 2-17. Eight replicate stations (transects at 45º increments) were to be located on each of four concentric rings. Rings were to be located at increasing distances from the target vessel to permit the analysis of chemical gradients (2, 10, 50, 1000 m from the ship). Difficulties encountered during sampling precluded the successful acquisition of sediments from all 32 stations. Three successful sampling efforts occurred during the period from September 1998 to November 1999. Nineteen separate stations were finally occupied in an attempt to obtain sediment samples. Hypothesis testing in this report takes advantage of the actual distribution of stations and considers stations, within a unique ring, to be replicates of that ring. Inner Ring 1 consists of stations approximately 2-3 m from the ex-AGERHOLM and Outer Ring 4 contains stations approximately 1,000 m from the vessel. Ring 4 is considered the program reference area.

The null hypothesis addressed by this investigation is: There is no significant difference in mean values for each parameter tested between Inner Ring 1 and Outer Ring 4. Parameters tested included chemicals of concern (i.e., Total PCBs, Total PAHs [41 analytes], trace metals [cadmium, chromium, copper, lead, mercury, nickel, silver and zinc]), grain size measures (i.e., median size, % gravel, % sand, % fines, % silt, and % clay), organic and inorganic carbon,

bioassay and bioaccumulation end points (e.g., % survival, growth, and tissue contaminants), and infaunal community measures (e.g., number of species, abundance, and diversity indices). Total PCBs have historically been reported by a wide variety of calculations including summation of Aroclors, subsets of congeners, homologs and the sum of all 209 congeners. The EPA and USACE provide guidance for the determination of total PCBs in the publication "Evaluation of Dredge Material Proposed for Ocean Disposal, Testing Manual" commonly referred to as the Green Book (USEPA/USACE, 1991). Total Green Book PCBs are determined through the summation 21 specific PCB congeners, with non-detected congeners given a value of zero. The SINKEX investigation analyzed all 26 congeners (21 Green Book congeners plus an additional five common to other environmental studies). The most conservative measure for total PCBs includes all 26 congeners measured during the SINKEX program with the assignment of ½ the sample specific method detection limit (MDL) for all non-detected congeners. Between these two summation strategies lay totals represented by Green Book 21 congeners with non-detected values represented by ½ the MDL and all 26 SINKEX congeners with non-detected values represented by zero. In general, this report presents summary statistics (e.g., mean, minimum, maximum) for all four types of total PCBs, but statistical comparisons between Inner and Outer Rings and graphical presentation are limited to one, or both, of total Green Book 21 (zeros for non-detected congeners) and all 26 SINKEX congeners (½ MDL for non-detected congeners). Totals for PAHs are handled similarly and the logic is presented in the PAH report sections.

2.5.12.1. Descriptive Statistics

Descriptive statistics were performed on all physical, chemical, and biological data. These statistics included computations for number of samples, means, standard deviations, ranges of values, coefficient of variation, and frequencies of detectable concentrations. Computations were performed on final results data that have passed quality control review. These data included detectable chemical concentration results, percent survivals from bioassays, and various benthic community parameters.

2.5.12.2. Coefficient of Variation

The coefficient of variation (CV) was used to describe the variation in several populations of physical and chemical sediment parameters. The CV can be expressed as a percent and is generally defined by Snedecor and Cochran (1980) as:

$$\text{CV} = \frac{\text{standard deviation}}{\text{population mean}} \times 100$$

2.5.12.3. Comparative Statistics

Comparative statistics (e.g, ANOVA) were used to test the null hypothesis. The Null Hypothesis is stated as: "Samples taken near the ship (i.e, Inner Ring) are the same as samples taken approximately 1 km from the vessel (i.e., Outer Ring) or stated mathematically with the Null Hypothesis as H_0: $\rho=0$ versus the alternate H_1: $\rho\neq0$ with a stated p-value ≤ 0.05. Significance criteria was $p<0.05$, which would suggest rejection of the null hypothesis.

Before statistical testing was conducted, the data were tested to determine if it fit the assumptions of the statistical test. One-way Analysis of Variance (ANOVA) requires that the data be normally distributed with equivalent variances. The Shapiro-Wilks test (Snedecor and Cochran

1980) was used to determine whether the data was normally distributed. The Shapiro-Wilks test produces two values: a W statistic indicating the tendency for the data to be normally distributed and a p-value indicating the probability of normality, the null hypothesis is predicated on the assumption that the data to be tested is normally distributed. For this test, rejection of the null hypothesis (i.e., the data was non-normally distributed) was set at $p<0.05$ (e.g., Table 3-47).

Levene's test was used to determine if the variances were equal and $p<0.05$ indicated that the variances were not equal. When the data were not normally distributed and/or the variances were not equal the data were transformed using standard data transformations (i.e., percentage data was arcsine and rank transformed; numeric data was Log 10 and rank transformed). The use of non-transformed data is preferred for hypothesis testing but when data transformations were needed to normalize the data, the bias was to select transformations (i.e., Log 10 and arcsine) that still utilized parametric statistical tests rather than using non-parametric transformations. Rank transformations are rather radical and a t-Test using rank transformed data is equivalent to Kruskal-Wallis Nonparametric Test, which is less robust and sensitive, compared to an ANOVA test (Tate and Clelland 1957). The transformation selected for hypothesis testing was the one that gave the highest W value, normal distribution, and equivalent variances. When none of the transformations fulfilled all criteria, then the data was rank transformed and the mean of the laboratory replicates was used as the value for each station.

ANOVAs were performed using stations within each ring as spatial replicates. Tukey's "Honestly Significantly Different" (HSD) test was used to determine significant differences among separate groups of stations (e.g different rings). To allow statistical comparisons of data, chemical concentrations less than detection limit are assigned a concentration of the sample specific detection limit divided by two. This is a common and simple method that has been shown to provide a higher level of discriminative power than representing non-detected chemical concentrations as zero. Among the ten methods investigated by the USACE using dredge sediment study data, using the detection limit, one half the detection limit, or zero tend to allow more power in analysis when compared to more complex methods such as uniform replacements, maximum likelihood estimation, and linear regression (US EPA, 1996a; US EPA, 1998b). One half the detection limit is preferred over the entire detection limit in cases such as this study where low levels of contamination exist and high numbers of non-detected analytes may be encountered.

ANOVA testing utilized the General Linear Model (GLM). Under the GLM, a continuous response, or dependent variable (e.g., total PCB) is measured under experimental conditions identified by classification, or independent variable (e.g., Ring). The variation in the response is explained as being due to effects in the classification, with random error accounting for the remaining variation (Searle, 1971). Laboratory duplicate values are commonly processed to address internal analytical quality control and, when present, were not included in the data analysis, since laboratory duplicates are not appropriate in the comparison of field-replicated stations. When laboratory duplicates are encountered in the analytical process, results were chosen based on "first sample processed" and when this is not possible, "first SAS data set occurrence" was used. Laboratory duplicate/QC data were included in hard copy reports provided by individual processing laboratories.

The final experimental design necessitated the implementation of a two-way GLM (see Table 2-31). Independent variables were assigned to both "Ring" (1 and 4) and "Cruise" (1998 and 1999). Two-way GLM processing permits the examination of between site (Ring) comparisons,

between time (Cruise) comparisons, and interactions where site and time are not independent. Unfortunately, the GLM cross-matrix (Table 2-31) is very weak with respect to time and highly biased.

Table 2-31. GLM cross matrix.

Sample Area	Stations Sampled		
	Cruise 1998	Cruise 1999	Total
Ring 1 (sediment)	8	2	10
Ring 4 (sediment)	1	7	8
Total (sediment)	9	9	18
Ring 1 (bioaccumulation)	8	1	9
Ring 4 (bioaccumulation)	1	4	5
Total (bioaccumulation)	9	5	14

The number of observations in matrix cells "Cruise 1998 Ring 4" and "Cruise 1999 Ring 1" are very low, making comparisons specific to time meaningless. Corrections for multiple-testing errors are not necessary, since contaminants are examined as singular dependent variables.

2.5.12.4. Principal Component Analysis

Principal Component Analysis (PCA) was conducted on all the physical and chemical measures (i.e., % gravel, % sand, % silt, % clay, % silt-clay, median phi in mm, % TOC, % $CaCO_3$, 10 individual metals, Total PAH, and Total PCB). PCA was done to reduce the number of measured environmental variables to only those variables that co-vary, to explore possible explanations for any observed adverse effects (e.g., toxicity). That is, while many different environmental parameters were measured for the study (e.g., 10 metals, sum of PAHs and PCBs, and six grain size parameters) many of these constituents show similar concentrations and spatial patterns. Total PAHs and PCBs were used rather than individual analytes because many analytes were non-detectable in some samples causing data gaps in the analysis matrix, which defeats the purpose of the test. Thus, some of the environmental measures are not independent from each other but co-vary among them. For example, certain metals, because of their size and ionic charge, react in a similar pattern and thus are not independent of each other. PCA reduces the number of apparent independent variables to fewer variables that are truly independent from each other and thereby eliminating the potential for multiple testing of non-independent environmental measures.

PCA produces derived environmental measures, called factors, which identify those environmental measures that co-vary in a similar pattern. These PCA factors resolve the data into axes that account for the variability of the data. Thus, the first PCA factor is the most significant axis accounting for the greatest variance in the data set. The second axis is set orthogonal to the first axis, making it independent of axis 1, the analysis then proceeds with axis 2 accounting for as much of the variability in the data set as possible. This process continues until all the variability of the data set has been taken into account. For most data sets the number of axes is about four, plus or minus one, as most of the variability in a data set can be reduced to three or four gradients (i.e., axes). The PCA produces a table of Eigen values that attributes the proportion each environmental measure contributes to each factor (i.e., axis). The Rotated Factor

Pattern table shows that the PCA produced four factors. Under Factor 1 there were four measures (mercury, nickel, silver, and TOC). All had Eigen values greater than 0.75 indicating that Factor 1 was largely determined by the values of these environmental measures. Environmental measures having Eigen values of less than 0.75 do not make a significant contribution for that axis or Factor. Thus, Factor 1 was renamed the "Hg, Ni, Ag, and TOC Factor" because these were the variables that largely determined this factor and which accounted for most of the variability within the data set. The four measures delineating Factor 1 are grouped together because they covary in a similar pattern and significant regressions related to Factor 1 cannot be attributed to any one of the measures comprising this factor.

The PCA utilized an arcsine transformation of all percentage data before the analysis. Factor 1 accounted for 31.1% of the total variance in the data set. Factor 2 accounted for 27.7% of the variance and was characterized by grain size measures: % silt, % clay, % silt-clay, and a negative correlation with median grain size. Factor 2 was renamed the "Fine Grain Size Factor". Factor 3 accounted for 20.7% of the variance and was determined largely by Cd, Cu, and total PCBs. It was renamed the "Cd, Cu, PCB Factor". Factor 4 accounted for 20.5% of the variance and was mainly characterized by $CaCO_3$, Al, Cr, and Fe. It was renamed the "$CaCO_3$, Al, Cr, and Fe Factor".

2.5.12.5. Correlations and Regressions

If significant differences between Inner and Outer Rings for physical and chemical measures were found, the next step was to determine whether these differences could be correlated with results of the bioassay, bioaccumulation, or infaunal biology.

Correlative analyses were performed on the separate data sets using the Pearson product-moment, which measures the strength of the linear relationship between two variables. If one variable (e.g., Total PCB) can be expressed exactly as a linear function of another variable (e.g., TOC or number of species), then this represents a perfect correlation with an r^2 value =1, or -1 if the variables are inversely related. A correlation of $r^2=0$ between two variables suggests that each variable has no linear predictive ability for the other. If the values associated with the variables are normally distributed, a correlation of 0 also means the variables are independent of one another. Additionally, this also gives a probability (p) value that the regression (i.e., linear plot of the correlation) has a non-zero slope. If $p<0.05$, then the null hypothesis is rejected, which means that the regression line has a significant slope and there is a significant relationship between the variables.

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3.0. Analysis Phase/Results: Characterization of Environment at the Site, and Characterization of Exposure and Effects

3.1. Characterization of Environment at the Site

This section provides a broad-based physical and biological background description for the ecosystem being investigated and the various lines of scientific investigation (i.e., sediment chemistry, sediment toxicity, and bioaccumulation). It was first tentatively described in the Problem Formulation section, based on information available to SSC SD prior to commencing the field and laboratory studies. A detailed characterization is now given in three parts:

1. Description of the epibenthic, benthic, and pelagic organisms and communities observed from videography recorded during numerous sampling dives of a remotely-operated vehicle (ROV).
2. Physical description of the sediment based on standard laboratory analyses of grain size, total organic carbon (TOC), and calcium carbonate ($CaCO_3$), factors which influence the behavior, and therefore interpretation, of chemical and biological processes which occur in the sediment bed. Sediment age dating and sediment rate trap analysis are also included here to evaluate the association between the depth of sediment sampled during the study and the depth of sediment expected to have deposited since the AGERHOLM was sunk.
3. Analysis of benthic infaunal ecology, based on standard community measures, which provide a detailed (i.e., categorization, enumeration, and comparison of species and higher level groupings between ship and reference site) overview of the ecosystems being studied.

3.1.1. Characterization of the Benthos Using Video Observations

The U.S. Navy ROV "SCORPIO" (see Figure 2-20) was used to assist in the study of possible environmental impacts from a sunken navy vessel. Normal ROV operations utilize video cameras to supply real-time video to the ROV pilot and co-pilot to facilitate vehicle operations, locate stations and obtain samples. Video streams were fed into industry-standard VHS and Hi-8 recorders and tapes were retained for future assessments of general operation efficiency and sediment sample integrity. Incidental to the video documentation of sampling operations, various large macroinvertebrates and fishes were recorded. While the intent of the video documentation was not to assess resident faunal assemblages, the hundreds of hours of recordings are useful in providing insight into the animal community in the vicinity of the ex-AGERHOLM.

From the literature, macroinvertebrates (those easily seen) and fishes from upper slope depths in the Southern California Bight (SCB) have been described primarily from organisms collected in benthic trawls. Few direct video-observations have been made at upper slope depths. Studies related to petroleum development and undersea cable installations have provided some of the best descriptions of the more common fauna off Southern California in water depths less than 400 m. Macroinvertebrates beyond these depths are not well known by sight, particularly the sponge (Porifera) and cnidarian fauna (e.g., anemones), whereas the echinoderms (e.g., starfish) and the fish fauna are relatively well known.

A list of the macroinvertebrates and fishes observed is presented in Table 3-1. Observations in the vicinity of the ex-AGERHOLM revealed a biotic community typical of the SCB lower shelf and upper slope community. Eighteen fish and 40 macroinvertebrate taxa were observed. The most common fishes were sablefish (*Anoplopoma fimbria*, Figure 3-1), longspine thornyhead (*Sebastolobus altivelis*), and black hagfish (*Eptatretus deani*). A picture of capture from video of a thornyhead sculpin is presented in Figure 3-2. Less common were rattails (i.e., grenadiers Macrouridae, Figure 3-3), shortspine thornyheads *(S. alascanus*), and filetail cat sharks (*Parmaturus xaniurus*). These are well known species of the upper SCB slope.

Figure 3-1. The sablefish, *Anoplopoma fimbria* (NOAA photograph).

Figure 3-2. A longspine thornyhead, *Sebastolobus altivelis*, observed at the ex-AGERHOLM site.

The most common invertebrates were suspension feeding sponges and anemones. Most of these sedentary species cannot be easily visually identified to species or genus level, but several are well known from other areas within the SCB, based upon their video descriptions of color and morphology. Several of these suspension feeders are known to occur in the Santa Barbara Channel.

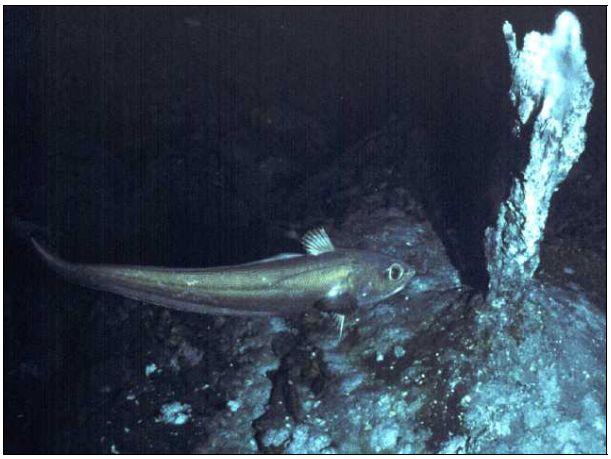

Figure 3-3. The rattail, *Coryphaenoides acrolepis* (NOAA photograph).

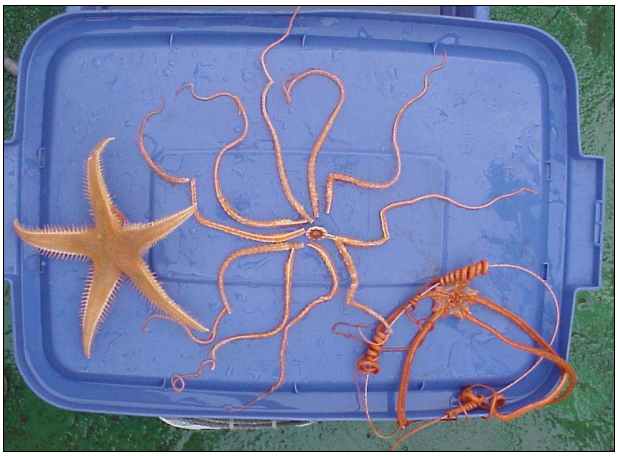

Figure 3-4. Representative echinoderms recovered from the ex-AGERHOLM site.

The echinoderm fauna was well represented and many are well known including motile predatory sea-stars (*Myxoderma platyacanthum, Poraniopsis inflata and Thrissacanthias* sp.),

and the scavenging motile sea cucumber (*Pannychia moseleyi*). The presence of the multi-armed brisingid starfish (*Brissingia* sp.) was noted, which is an organism that extends its arms into the water column to filter-feed. A single pink urchin (*Allocentrotus fragilis*), a well known spiny species from shallower depths of the outer continental shelf, was observed. A few burrowing urchins (*Brissopsis pacifica*) were also observed. An example of representative echinoderms is presented in Figure 3-4. This photo (as seen in the lower right) also shows an interesting commensal relationship of a brittle star (probably *Asteronyx loveni*) clinging to the upper portion of a sea whip (possibly *Distichoptilum* sp.). This relationship has been observed in different locations and found on display at the deepwater Monterey Bay Aquarium Research Institute (MBARI) exhibit in Monterey, CA.

In conclusion, the species observed in the vicinity of the ex-AGERHOLM collectively appear to represent a typical assemblage for the continental slopes and upper basins off the Southern California coast.

Table 3-1. List of species observed on sampling documenting video recordings in vicinity of ex-AGERHOLM depth range 640 to 1,143m offshore of San Diego, west of San Clemente Island, September and November 1998.

	FISHES
Anoplopomatidae	
Anoplopoma fimbriata	Sablefish
Apristuridae	
Apristurus brunneus	Brown Cat Shark
Parmaturus xaniurus	File-tail Cat Shark
Argentinidae	
Argentina sialis	Pacific Argentine
Macrouridae	
Nezumia stelgidolepis	California Rattail
Melanostomiatidae	
Melanostomiatid (unident)	Scaleless Dragonfishes
Merluciidae	
Merluccius productus	Pacific Hake
cf. Myctophidae	Lanternfish (unident)
Myxinidae	
Eptatretus deani	Black hagfish
Eptatretus stoutii	Pacific Hagfish
Pleuronectidae	
Microstomus pacificus	Dover Sole
Rajidae	
Raja rhina	Longnose Skate
Raja sp.	Longnose or Deep Skate
Scorpaenidae	

Sebastolobus alascanus	Shortspine Thornyhead
Sebastolobus altivelis	Longspine Thornyhead
Sebastolobus sp.	Thornyhead

Scyliorhinidae

Parmaturus xaniurus	File-tail Cat Shark

Zoarcidae

cf. *Lycodesdiapterus*	Black Eel-pout

INVERTEBRATES

PORIFERA

Unident. Sponge 1	orange, amorphous
Unident. Sponge 2	large, white, amorphous
Unident. Sponge 3	white vase sponge
Unident. Sponge 4	white urn sponge
Unident. Sponge 5	fuzzy grey urn sponge
Unident. Sponge 6	grey-white ball sponge (cf *Geodia* sp.)
Unident. Sponge 7	"Friar Tuck" ball sponge

CNIDARIA

hydroids (unident.) - many on hard substrates	
Tealianthus sp.	Anemone
Liponema brevicornis	Anemone
Pennatulidae (cf. *Pennatula phosphorea*)	Sea Pen
Umbellula sp.	Sea Pen
Unident. Anemone 1	anemone (thin purple tentacles, flush with sediment)
Unident. Anemone 2	anemone (cream colored base, stout dark tentacles)
Unident. Anemone 3	anemone (extended, long clear tentacles)

MOLLUSCA

Gastropoda

Unident. Gastropod 1	Whelk like

Pelecypoda

cf. Mytilidae - unident.	Mytilid, hard substrate

Cephalopoda

Unident. Squid	Squid (short webbed tentacles)

ANNELIDA

Sedentariata unident. Sp. 1	worm tubes

ECHINODERMATA

Asteroidea

Brissingia sp.	Multi-armed Starfish
Myxoderma platyacanthum	Starfish
Poraniopsis inflata	Starfish
Rathbunaster sp.	Sun Star
Thrissacanthius sp.	Starfish
Unident Starfish 1	Long thin arms (not *Thrissacanthias)*

Echinoidea

Allocentrotus fragilis	Pink Urchin
Brissopsis pacifica	Sea Urchin

Holothuroidea

Pannychia moseleyi	Sea Cucumber
cf. *Pteraster*	Sea Cucumber
Unident holothoroid (nr. *Scotoplanes*)	Sea Cucumber
Ophiuroidea	
Asteronyx longifissus	Brittle Star
Ophiomusium lajollanensis	Brittle Star
CRUSTACEA	
Decapoda	
Chorila longipes	Longhorn Decorator Crab
Lithodes cousei	King Crab
Majidae (cf. *Chionoecetes bairdii*)	Tanner Crab
Munida quadrispina	Galatheid Crab
Parilithoides sp.	Red King Crab
Unident. Spider Crab	Spider Crab
Euphausiacea	
Euphausiid shrimp	Krill

3.1.2. Sediment Characteristics

Grain size, total organic carbon (TOC), $CaCO_3$, sediment rate trap, and age-dating (geochronology) results for the Inner Ring (Ring 1) and reference Outer Ring (Ring 4) are presented in this section. These physical parameters can influence chemical distributions and benthic community structure in sediments and are therefore important in the interpretation of data. Grain size, TOC and $CaCO_3$ results are summarized for stations sampled in September 1998 and September/November. Grain size analyses were performed by MEC Analytical Systems, Inc. (Carlsbad, CA). TOC, $CaCO_3$ and age-dating samples were processed at the Florida Institute of Technology, Marine and Environmental Chemistry Laboratories (Melbourne, FL). Between the 1998 and 1999 sampling period, instrumentation used to process sediment TOC was changed at FIT and the new equipment dictated the extraction of a larger quantity of sediment. Re-analysis of the TOC data suggested a potential high bias for the 1998 samples, which were dominantly Inner Ring stations. Archived sediment samples from Inner Ring stations 1-1, 1-2, 1-3, 1-4, 1-6, 1-7, and 1-8 were subsequently reanalyzed using the 1999 instrumentation and methods and this data is used throughout this report. Station 1-5 was not reanalyzed, due to low archive quantities, and the original TOC data for this 1998 analysis is used in the report. Note that throughout the results section (3.0), data associated with Station 1-5/6/7 (e.g., Table 2-6) refer to a single station sampled in 1999 (Cruise 5). This station was re-sampled to provide confirmation of apparent anomalous results produced by sediments collected from Station 1-6 in 1998 (Cruise 2).

Sediment grain size characteristics are notable for their controlling influence upon sedimentary community dynamics, and because they often correlate with biologically meaningful variables such as sediment porosity, compaction, oxygen tension, water content and retention of organic matter. Grain size characteristics are equally important in controlling sediment chemical concentrations due to the increase in adsorptive capacity with finer-grained particles. Since many contaminants are strongly bound to organic material that is often complexed with fine mineral particles, there is a high potential for contaminant accumulation in environments where settlement of fine-grained sediment occurs.

Most studies of marine and brackish sediments show a high positive correlation between fine-grained particles and organic carbon. Deposition, re-suspension and sorting processes influenced by hydrodynamic factors (e.g., current regime) normally create a gradient of diminishing grain size proceeding offshore. After they are introduced into the coastal system, the smallest particles remain in suspension for the longest period of time and can move the farthest from shore. Following deposition, smaller particles are more readily resuspended from the seabed by waves, currents and turbidity flows. This ex-AGERHOLM site is bathymetrically complex with the hulk located on a steep gradient (15 percent slope). Additionally, the vessel is situated on the seaward side of a deep basin, which could serve as a physical barrier from normal nearshore sedimentary inputs. Ultimately, at this site, it is thought that fine-grained sediments with high organic content would progress downslope into the deeper, stable basins that would serve as the ultimate repositories for contaminants. Site sediments consist of deep ocean minerals mixed with large phosphatized-calcium carbonate precipitates and living biogenic material, mainly large benthic protozoans (Foraminifera). Poorly sorted sediments typify the study area. Cold water (4 ºC), available phosphate, $CaCO_3$ and hydrostatic pressure produce an abundance of phosphatized-calcium carbonate rocks ranging in size from pebbles to cobble.

3.1.2.1. Grain Size, TOC and $CaCO_3$

Sediment grain size characteristics (Figure 3-5 through Figure 3-8), TOC (Figure 3-9 and Figure 3-10), and $CaCO_3$ are summarized in Table 3-2 for the ex-AGERHOLM site (Inner Ring, #1), and reference area (Outer Ring, #4). Most Ring 1 sediment samples were collected from a water depth of approximately 850 m while Ring 4 ranged from 640 to 1015 m. Inner Ring 1 sediments were significantly different from Outer Ring 4 sediments. Inner ring sediments (median size=0.246 mm) exhibited almost twice the average grain size of the outer ring (median size=0.136 mm), with Ring 1 having almost five times more gravel (Ring 1=24.4% versus Ring 4=4.4%), a little less sand (Ring 1=51.7% versus Ring 4=65.5%), and about half of the silt content (Ring 1=10.9% versus Ring 4=18.0%) of the outer ring. Inner ring fine sediments (silts + clays) averaged 23.9% compared to 30.1% for the outer ring. This difference was almost entirely due to the difference in silt content, since the proportion of clay was essentially the same for both locations (Ring 1=13.0% clay versus Ring 4=12.2% clay). Inner ring sand and gravel averaged 51.7% and 24.4%, respectively, while the Outer reference ring contained only minimal quantities of gravel and was dominated by sand.

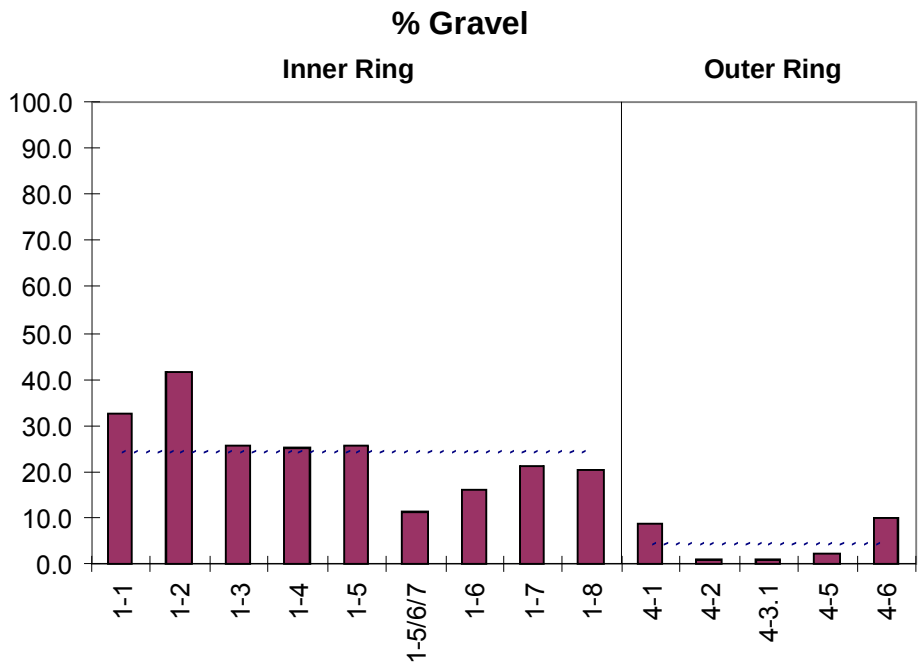

Figure 3-5. Inner and Outer Ring sediment % gravel.

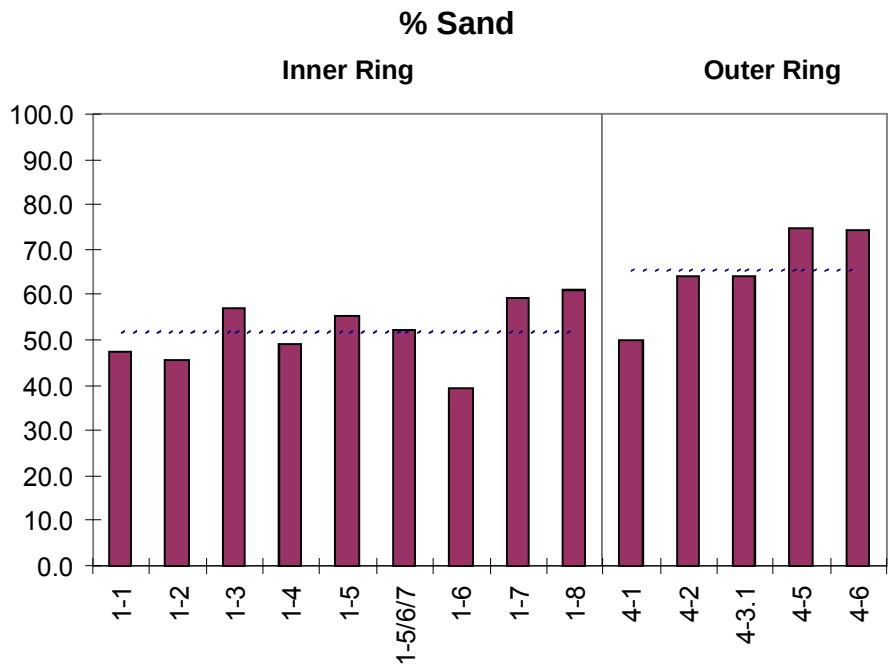

Figure 3-6. Inner and Outer Ring sediment % sand.

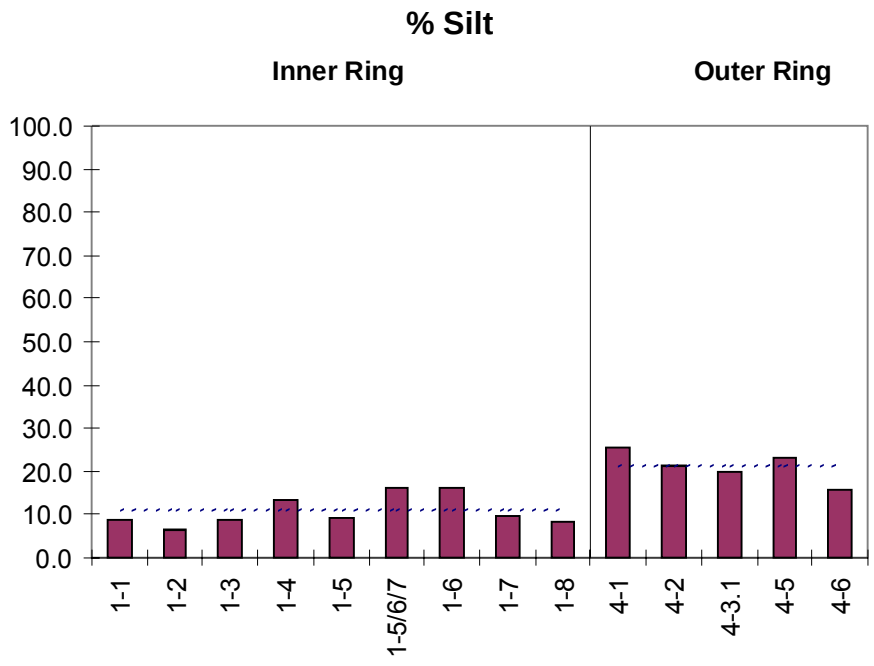

Figure 3-7. Inner and Outer Ring sediment % silt.

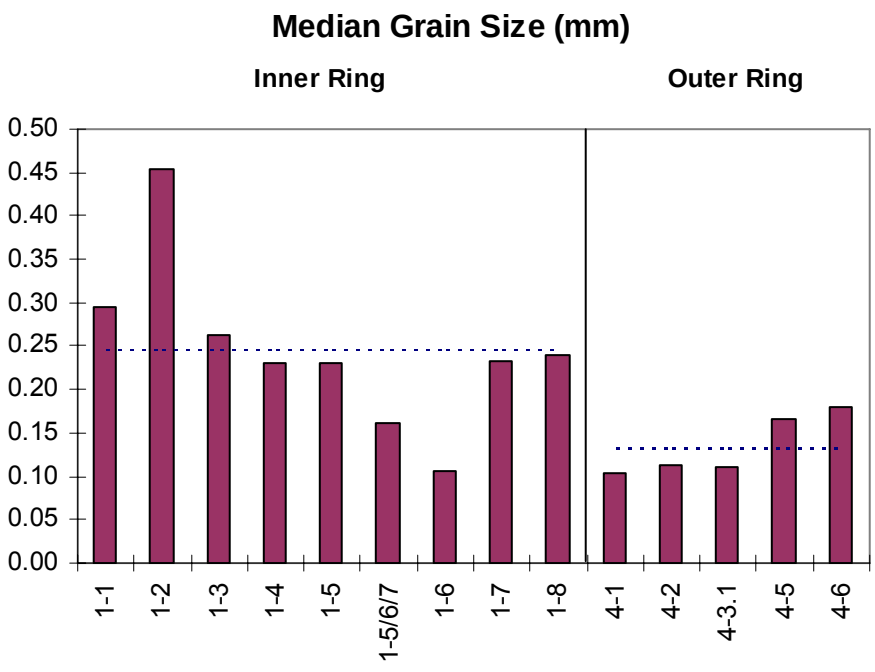

Figure 3-8. Inner and Outer Ring sediment median grain size (mm).

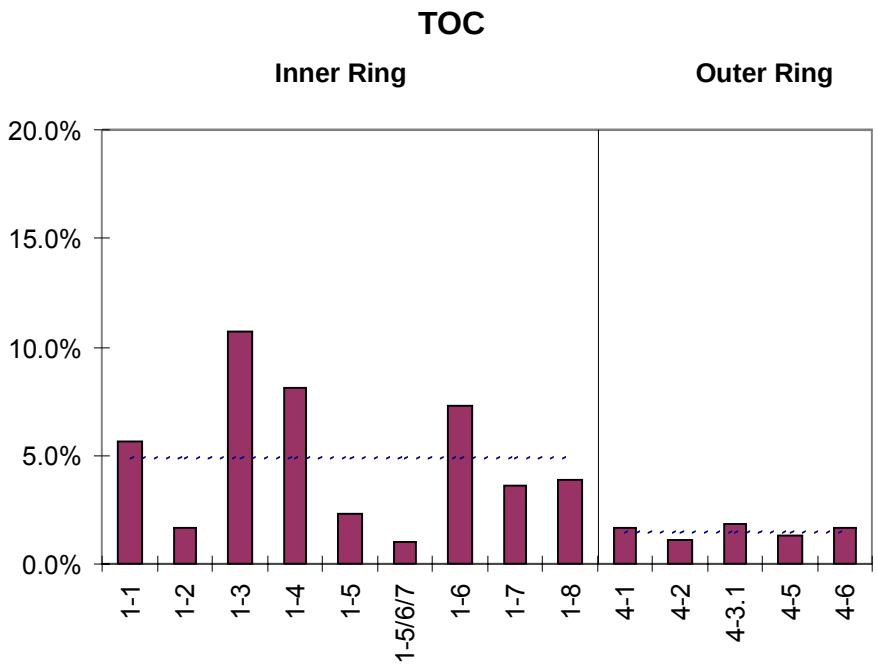

Figure 3-9. Inner and Outer Ring sediment TOC.

Table 3-2. Descriptive statistics - sediment % fines, gravel, sand, TOC and $CaCO_3$.

Analyte	Site	N	Mean (%)	Minimum (%)	Maximum (%)	Standard Error	% CV
Fines (<0.063 mm)	ALL	14	26.13	13.0	44.6	2.77	36
	Ring 1	9	23.91	13.0	44.6	3.42	36
	Ring 4	5	30.13	15.8	41.2	4.63	33
Gravel (≥2 mm)	ALL	14	17.23	0.7	41.6	3.31	71
	Ring 1	9	24.35	11.2	41.6	2.97	36
	Ring 4	5	4.41	0.7	9.8	2.02	101
Sand (>0.063, <2 mm)	ALL	14	56.64	39.4	74.8	2.79	19
	Ring 1	9	51.74	39.4	60.9	2.36	14
	Ring 4	5	65.46	50.0	74.8	4.53	19
Total Organic Carbon	ALL	18	3.16	1.01	9.06	0.61	81
	Ring 1	10	4.38	1.01	9.06	0.94	67
	Ring 4	8	1.64	1.1	2.57	0.16	26
Calcium Carbonate	ALL	17	47.85	22.2	69	3.15	27
	Ring 1	9	45.49	22.2	69	5.12	33
	Ring 4	8	50.51	27.7	57.4	3.55	19

Total organic carbon (TOC) concentrations provide an indication of the amount of organic matter present in bottom sediment. The generally high organic content of the sediment coupled with the observed high variability is atypical of low-energy, fine-grained deep ocean depositional areas. Benthic Foraminifera may be contributing the bulk of measured TOC.

Average TOC values for Ring 1 (4.38%) were over 2.5 times higher than the values for Ring 4 (1.64%); but due to the high variability, this difference was not statistically significant (Kruskal-Wallis analysis, p=0.068). Mean TOC concentrations for Ring 1 were similar to reported values of 3-4% for southern California basins (Finney and Huh 1989), but higher than levels in typical deep-sea sediments, of less than 1%. Reefs, whether natural or artificial, alter the physical characteristic of sediments adjacent to these structures (Diener et al., 1997). The ex-AGERHOLM represents a large reef, and the physical characteristics of the sediments within 100-200 m could have been affected by the trapping of finer sediments and organic matter on the upcurrent side of the hulk and erosion of finer sediment on the downcurrent side. Thus, TOC values near the hulk may be elevated due to this reef effect. For most marine sediments, there is usually a significant correlation between fine sediments and TOC concentrations; however this relationship was not significant for the study (Pearson's correlation p=0.57), suggesting that other factors contribute to TOC values. For example, large numbers of living benthic Foraminifera (protozoa) were found in the sediments samples and could contribute to higher TOC measurements, but these observations were incidental and not quantified. The presence of these Foraminifera was significant, because the high proportion of $CaCO_3$ measured in the sediments was largely due to the tests of both living and dead foraminifera.

As shown in Table 3-2, almost half of the sediments collected were comprised of calcium carbonate (48%) and values ranged from 22.2% (Station 1-2) to 69% (Station 1-5/6/7). No significant differences were found between either rings or cruises (p=0.61, 0.82 respectively). When compared to other physical sediment parameters, $CaCO_3$ was found to weakly correlate with gravel (r^2=0.35, p=0.03). A similar relationship is observed between $CaCO_3$ and Al ($r^{2=}$0.39, p=0.008;) Figure 3-11. See Appendix D for further information.

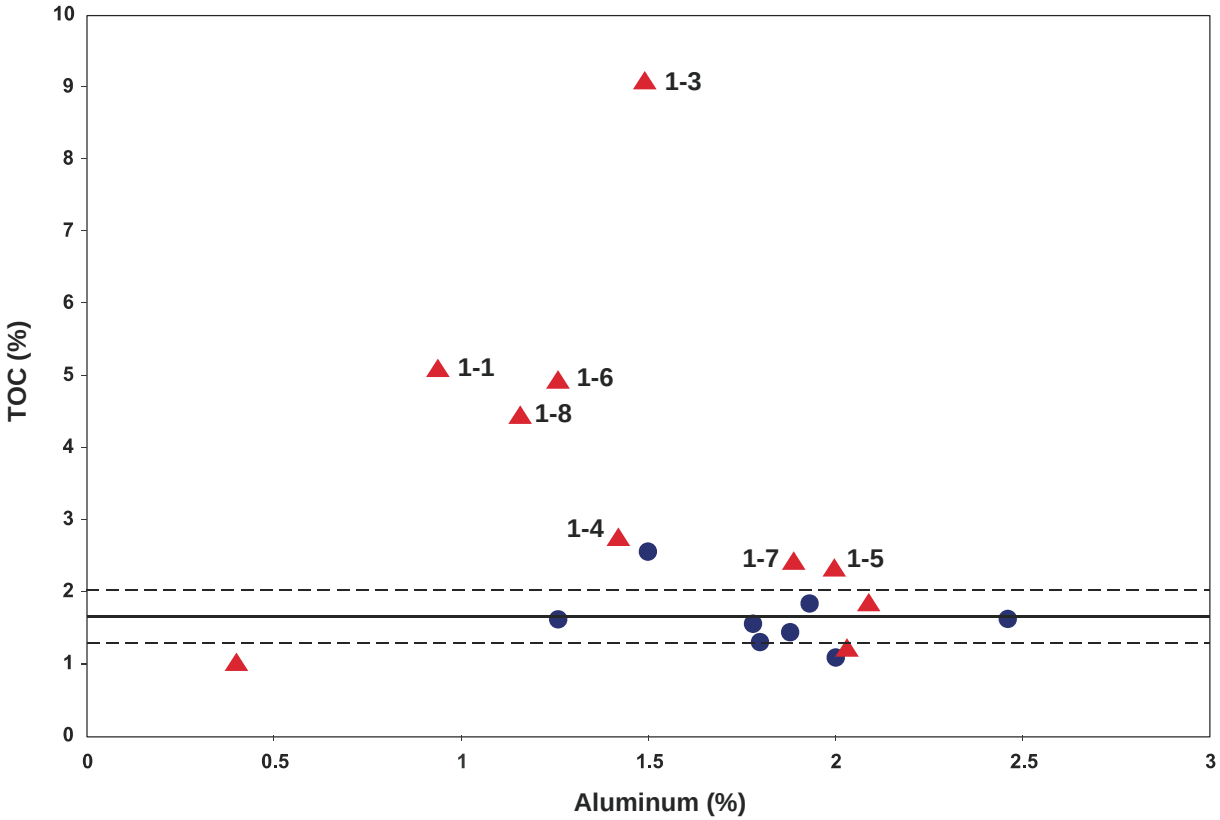

Figure 3-10. Concentrations of Al vs. total organic carbon (TOC) for sediment samples from Ring 1 (red triangles) and Ring 4 (blue cirles). Solid line is linear regression of Ring 4 samples, dotted lines are 95% predictive interval.

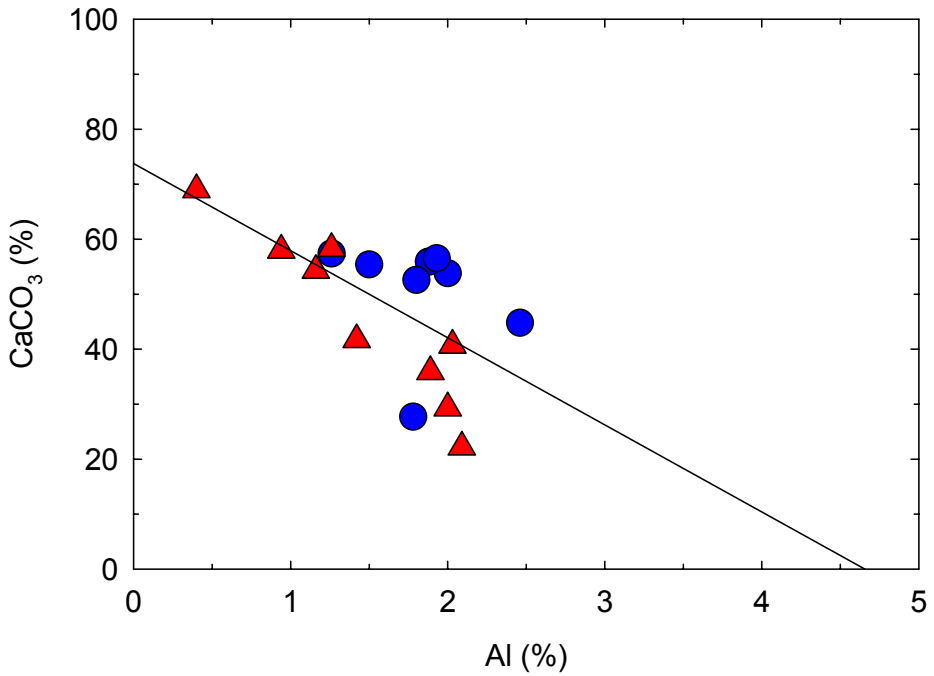

Figure 3-11. Concentrations of Al versus calcium carbonate ($CaCO_3$) for sediment samples from Ring 1 (red triangles) and Ring 4 (blue circles). Solid line shows result from linear regression calculation for entire data set.

3.1.2.2. Sedimentation Rate Evaluation

Preliminary estimates of sedimentation rates were calculated from two sediment cores obtained prior to the sampling of the inner/outer rings. One core (EXAG001) had a sedimentation rate of 0.12 +/- 0.04 cm/y, while the other one's (EXAG002) rate was 0.067 +/- 0.005 cm/y, corresponding to 16-year sediment depositions of 1.2-2.4 cm and about 1 cm respectively, since the sinking of the ex-AGERHOLM in 1982. This slow accumulation appeared to be reasonable for this area off the Southern California coast, based on Huh et. al.'s (1990) estimates of 0.02-0.03 cm/yr for a flat, deep basin and 0.05-0.13 cm/yr for a dynamic slope, as well as the mass-based estimates of Eganhouse and Venkatesan (1993) calculated to be 0.5-0.8 g/m^2/day. To maintain a balance between sampling enough sediment depth to ensure collection of material deposited since 1982, and collecting too much material that could potentially dilute any signal from the sunken vessel, 3 cm surface grabs and cores were selected. As described in Section 2.5.2.4, two different methods were employed during this study to calculate sedimentation rates: (1) estimates based on deposition of sediment in traps deployed over a period of time in the vicinity of the ex-AGERHOLM, and (2) estimates based on radiological age-dating techniques from relatively additional undisturbed sediment cores recovered from the site.

3.1.2.2.1. Rate Calculated by Deposition in Sediment Rate Traps

Four sediment trap arrays each containing four settling tubes were successfully deployed and recovered providing a sampling exposure period of 405 days. Sediment mass collected as particulate fallout (i.e., suspended solids) was about half of the values reported for basins located nearer the mainland and continental shelf. There was some difference in the sedimentation rates based upon location within the study site. Traps located to the north had the greatest

sedimentation rates and those from east the least. The results of the sediment trap data are presented in Table 3-3. Using the total amount of sediment collected in the traps, Total Suspended Solids (TSS) ranged from 103.68-182.45 g/m^2 while Total Volatile Solids (TVS), representing the organic fraction that can be volatilized in the laboratory, ranged from 18.83-38.77 g/m^2. Therefore, based upon an average deployment of 405 days, TSS sedimentation rates ranged from 0.256-0.450 g/m^2/day, corresponding to 93.4-164 g/m^2 /year, or 1.49- 2.6 kg/m^2 for the entire 16 years during which the hulk rested on the ocean bottom. This is about half of the rate reported by Eganhouse and Venkatesan (1993) and most likely is a reflection of lower primary production and terrestrial particulates associated with the distance (120 nm) of the study site in Tanner Basin from the mainland compared to the Santa Monica (12 nm) and the San Pedro Basins (7 nm). The thickness of the solid particulates accumulated at the bottom of each settling tube was between 1 and 2 mm. Thus, the best estimate for the site-specific sedimentation rate/yr is in the range of 1-2 mm/year or 1.6-3.2 cm/16 years. The particulates trapped in the settling tubes contained a significant amount of organic material as indicated by the TVS values, which averaged between 15.01-29.91% of the sample weight. Since the ratio of TVS/TOC for the sediments in the study area is about two, then about 7.5-15% of the material settling into the basins is of organic origin.

Table 3-3. Results of settling tubes, total suspended solids, total volatile solids, and percent volatiles.

Location	TSS Total Dry Wt. (g/m^2)	TVS Total Volatiles (g/m^2)	Percent Volatiles
North 1	138.13	20.73	15.01
North 2	153.30	25.77	16.81
North 3	123.52	18.83	15.24
North 4	171.59	28.09	16.37
North Mean	146.64	23.36	15.86
South 1	113.80	24.62	21.64
South 2	123.70	24.98	20.20
South 3	156.14	25.42	16.28
South 4	154.04	26.55	17.23
South Mean	136.92	25.39	18.84
East 1	103.63	22.84	22.04
East 2	110.55	23.97	21.68
East 3	122.77	21.53	17.54
East 4	118.96	19.86	16.70
East Mean	113.98	22.05	19.49
West 1	129.66	38.77	29.91
West 2	109.95	23.23	21.13
West 3	182.45	36.58	20.05
West 4	125.81	26.03	20.69
West Mean	136.97	31.15	22.95

3.1.2.2.2. Rate Calculated by Sediment Age-Dating (Geochronology)

Eight sediment core samples from various locations surrounding the ex-AGERHOLM were analyzed by gamma spectroscopy (Section 2.5.2.4) to obtain sediment geochronologies.

Sedimentation rates were calculated using vertical profiles for excess ^{210}Pb from each core. ^{210}Pb is a naturally occurring decay product of ^{238}U with a half-life of 22.3 years. Excess ^{210}Pb is transferred to the oceans when the parent radionuclide ^{222}Rn, a gas with a half-life of only 3.8 days, escapes to the atmosphere from the land. The ^{222}Rn decays quickly to ^{210}Pb in the atmosphere and rains out to the oceans to be scavenged by particles and carried to the seafloor. The excess ^{210}Pb that is carried to the seafloor via the atmosphere decays to non-detectable levels in the sediment column over 4-5 half-lives, about 100 years. The profile for the activity of the excess ^{210}Pb versus depth in the sediment column can be used to calculate sedimentation rates. Secondary validation of sediment ages determined by the excess ^{210}Pb technique is preferred when possible. The fission-produced radionuclide ^{137}Cs (first introduced to Earth by bomb testing during the early 1950s) has been used for this purpose in lacustrine, estuarine and shallow marine systems. However, ^{137}Cs is not commonly detected in deep sea environments and was not used to validate aging in this study. Table 3-4 summarizes age-dating results that are discussed in more detail below. The discontinuity depth in Table 3-4 refers to the depth below which no excess ^{210}Pb is observed; "none" is written in Table 3-4 when no excess ^{210}Pb is observed in any layers of the core from a given site.

Table 3-4. Sedimentation rates determined by analysis of radionuclides in sediment cores.

Station	Ring	Cruise	Discontinuity Depth (cm)	Sedimentation Rate (cm/y)
1-2	1	1998	None	not determinable
1-4	1	1998	>13 cm, unreliable	???
1-6	1	1998	~4 cm, less reliable	0.03-0.07
1-8	1	1998	None	not determinable
4-1	4	1999	0.5	no rate calculated, top ~2 cm may be missing
4-3	4	1999	3.0	0.031
4-3.1	4	1999	3.0	0.036
4-5	4	1999	0.5	0.038, less reliable, based on 2 points, top ~2 cm may be missing

Sediment from Ring 4 appears to be less disturbed than sediment collected from Ring 1 and thus results for the outer ring sites (particularly in the deeper areas) provide better results. Vertical profiles for the natural logarithm (ln) of excess ^{210}Pb in sediment cores from Stations 4-3 and 4-3.1 show predicted exponential decay to zero within the top 3 cm of the sediment column (Figure 3-12,Figure 3-13). Sedimentation rates calculated for these two cores are 0.031 and 0.036 cm/yr, respectively. Thus, sediment at a depth of about 3 cm, at this particular spot, may have been deposited almost 100 years before collection. However, it is unclear whether data collected at the bottom of the basin can be extrapolated to predict deposition at the ex-AGERHOLM, which is situated upslope from stations 4-3 and 4-4.1. These rates derived from age dating are 3-6x less than the rates estimated from the sediment traps, which are comparable to the initial estimates derived from the literature. The difference suggests either advective transport away from the area, or a bias in the sediment traps.

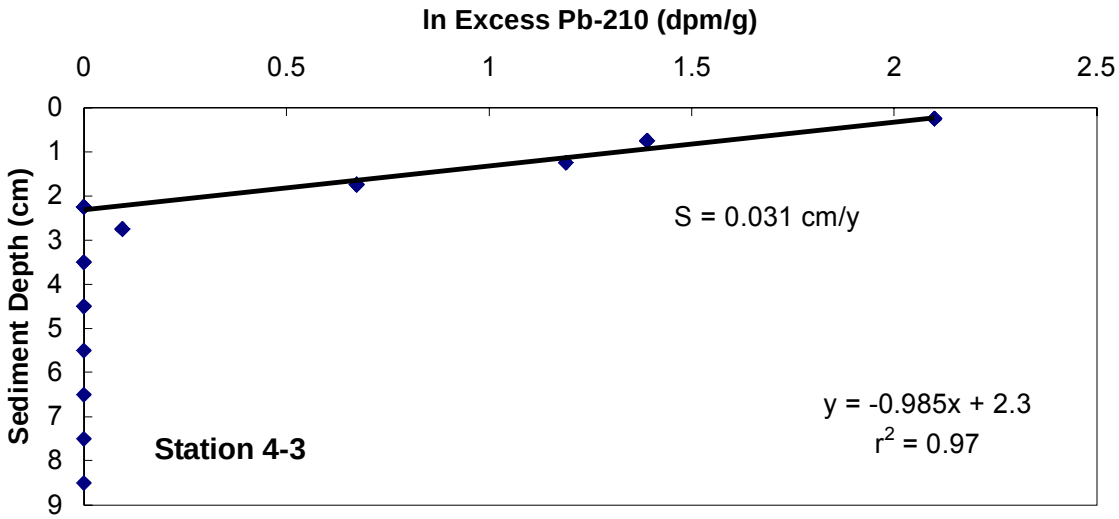

Figure 3-12. Vertical profile for ln excess ^{210}Pb in sediment core from Station 4-3. The equation for the linear regression, the coefficient of determination (r^2) and the sedimentation rate (S) are listed in the figure.

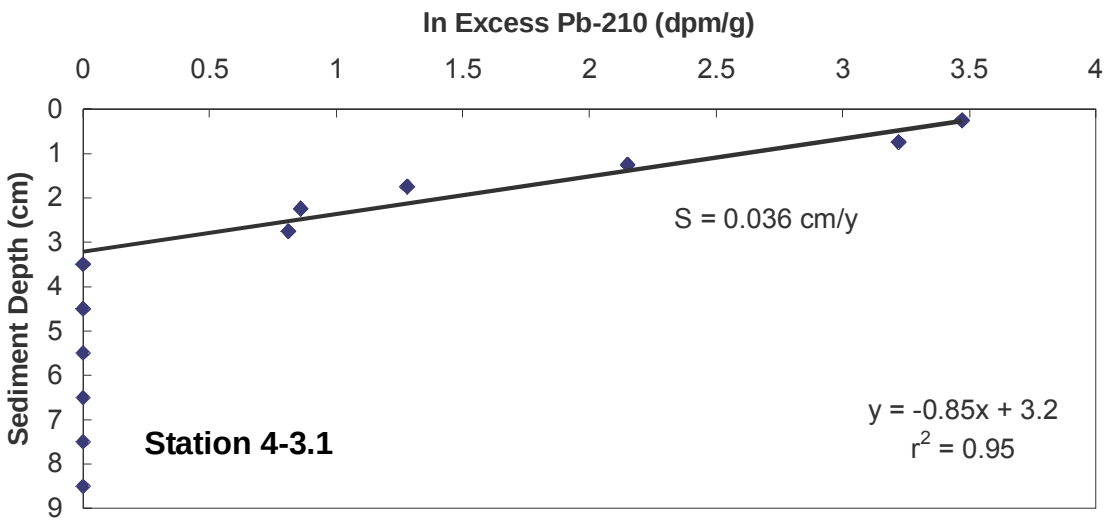

Figure 3-13. Vertical profile for ln excess ^{210}Pb in sediment core from Station 4-3.1. The equation for the linear regression, the coefficient of determination (r^2) and the sedimentation rate (S) are listed in the figure.

The results obtained here for Ring 4 show reasonable agreement with those obtained by Huh et al. (1990) for the southern California basins. In the flat, deeper portions of the Santa Monica and San Pedro Basins, the data from Huh et al. (1990) can be used to calculate sedimentation rates that range from 0.02 to 0.03 cm/yr, assuming a wet (bulk) sediment density of 1.6 g/cm$^{3.}$ Along the more dynamic slope portions of the two basins, sedimentation rates are considerably higher at 0.05-0.130 cm/yr (Huh et al. 1990). The activity of excess ^{210}Pb in the top 0.5 cm of sediment at Station 4-3.1 in the SINKEX study is 32 dpm/g, a value consistent with those obtained by Huh et al. (1990). This good comparison in absolute activities supports confidence in recovery of an intact sediment core or a core that had not been disturbed in situ prior to sampling. The activity

of ^{210}Pb in the core from Station 4-3.1 was lower at 8 dpm/g, possibly due to loss of the very top layer (<1 cm) of sediment due to some natural disturbance and/or sample collection.

Excess ^{210}Pb was found only in the top 0.5 cm section of the other two cores from Ring 4 (Stations 4-1 and 4-5) as shown on Figure 3-14 and Figure 3-15. Therefore, no reliable sedimentation rates could be calculated. The activities of excess ^{210}Pb in these two top layers are low at 1.5 dpm/g (Station 4-5) and 0.3 dpm/g (Station 4-1). Such low levels of excess ^{210}Pb in the surface layer of sediment are consistent with loss of the top ~2 cm of sediment due to natural disturbance or loss during sampling. Despite the apparent loss of sediment, and with only one core slice containing excess ^{210}Pb, a sedimentation rate of 0.038 cm/yr is obtained for Station 4-5. The extremely low, single value for excess ^{210}Pb at Station 4-1 (Figure 3-15a), yields an incompatible sedimentation rate of 0.01 cm/yr. However, as shown on Figure 3-15b, if the single point obtained for excess ^{210}Pb at 0-0.5 cm in this core is viewed as a point at a depth of about 2 cm (due to a probable loss of the top layer of sediment) and excess ^{210}Pb profile data are taken from the upper layer data from intact cores, a sedimentation rate similar to those found at Stations 4-3 and 4-3.1 can be inferred.

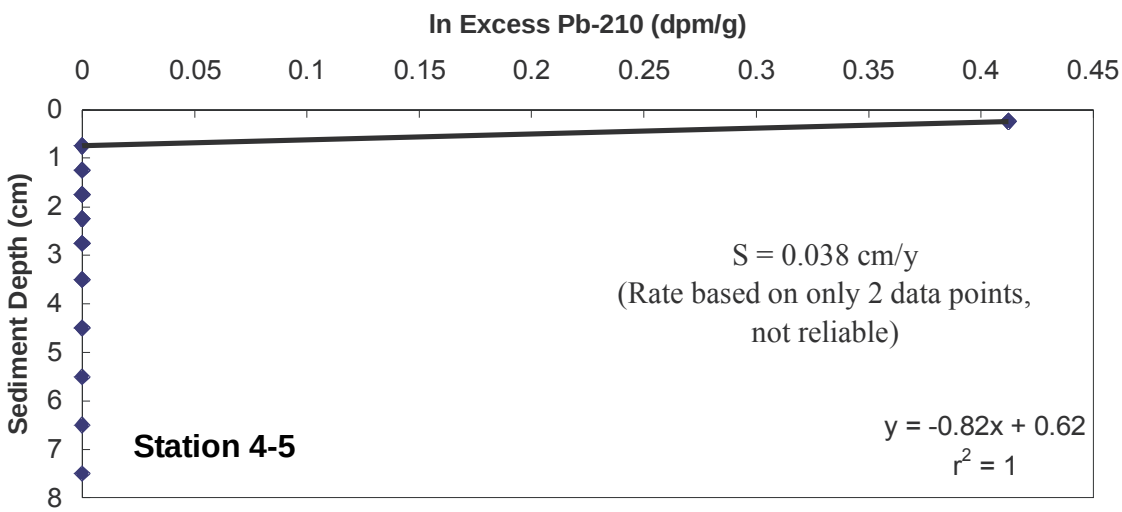

Figure 3-14. Vertical profile for ln excess ^{210}Pb in sediment core from Station 4-5. The equation for the linear regression, the coefficient of determination (r^2) and the sedimentation rate (S) are listed in the figure. The value for S is not considered to be reliable because it is based on just two points.

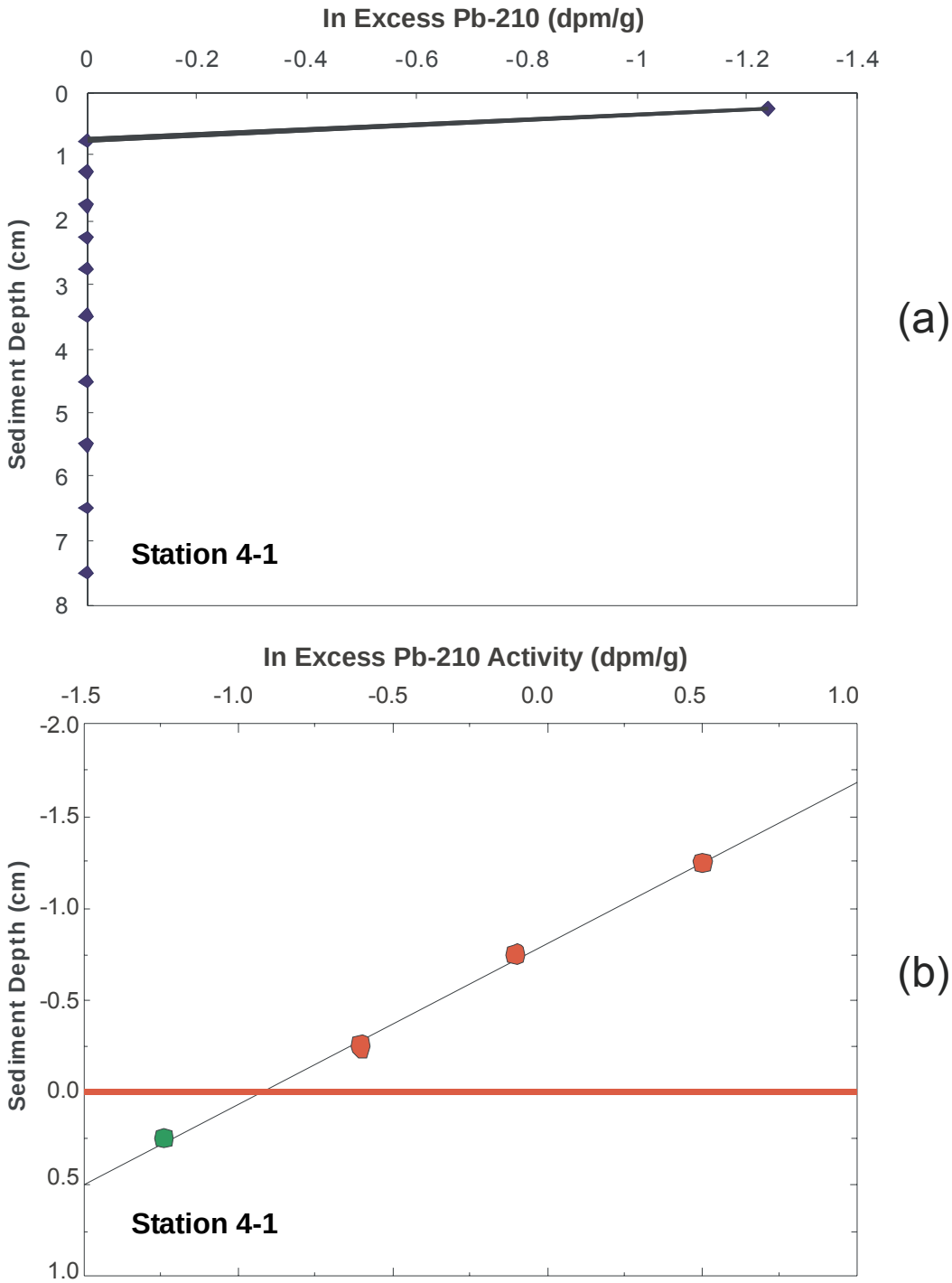

Figure 3-15. Vertical profiles for ln excess ^{210}Pb from Station 4-1. (a) Original profile showing the presence of excess ^{210}Pb only in the top 0.5 cm of the core. (b) Extrapolation of the vertical profile for excess ^{210}Pb at Station 4-1 based on data from sites 4-3 and 4-3.1. If about 1.5-2 cm of sediment were lost, the observed trend at Station 4-1 would be consistent with the trends at Stations 4-3 and 4-3.1 and a sedimentation rate of ~0.03 cm/yr.

The geochronology data for Ring 1 are much less reliable than those from Ring 4 and support considerable disturbance of the sediment at this near-ship location. No excess ^{210}Pb was

measured in any samples from Stations 1-2 and 1-8, both directly adjacent to the port side of the sunken ship. The observed lack of excess ^{210}Pb could result from having the top layer of most recently deposited sediment removed by a natural event (e.g., turbidity flow), an anthropogenic disturbance (e.g., hydraulic wake from the sinking ship), or a sampling artifact.

Excess ^{210}Pb was measured in sediment cores from Ring 1 at Stations 1-4 and 1-6 on the starboard side of the sunken ship. However, the vertical profiles were somewhat disturbed for Station 1-6 and greatly disturbed at Station 1-4 as shown Figure 3-16 and Figure 3-17. If only the top three samples from Station 1-6 are considered, the sedimentation rate is about 0.031 cm/yr and consistent with cores from Ring 4 (Figure 3-16a). If all samples are included in the calculation, the sedimentation rate increases to 0.072 cm/yr (Figure 3-16b). No clear explanation fits the observed profile; however, some disturbance of the sediment most likely occurred. At Station 1-4, high levels of excess ^{210}Pb are observed over 13 cm of the core. No reliable sediment accumulation rate can be obtained from this core, which probably contains several different layers of surface (top 2 cm) sediment that were re-suspended and then settled back on the seafloor.

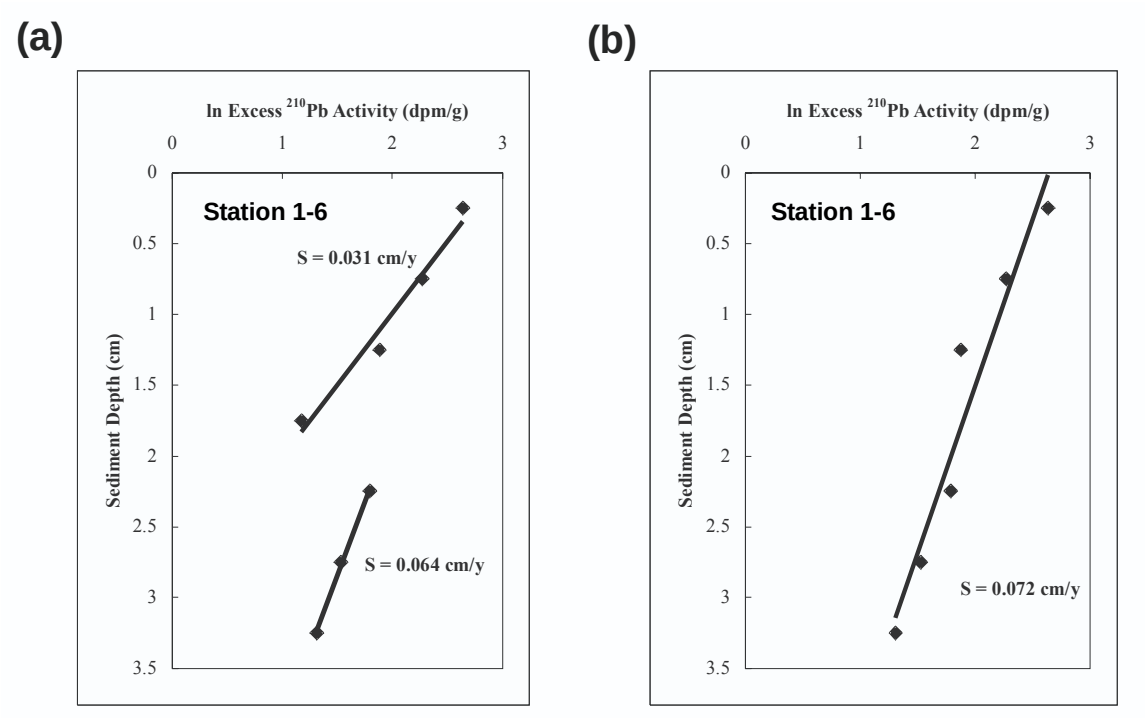

Figure 3-16. Vertical profiles for ln excess ^{210}Pb from Station 1-6. Sedimentation rates (S) are shown on each figure. (a) Possible interpretation of shifts in sedimentation rate of addition of sediment during re-suspension event caused when the ex-AGERHOLM sunk and (b) Interpreted sedimentation rate assuming no sediment disruption, just non-steady-state deposition over time.

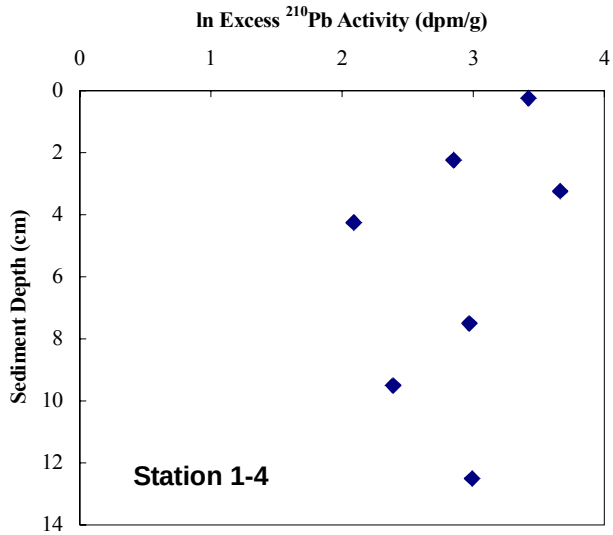

Figure 3-17. Vertical profiles for ln excess ^{210}Pb from Station 1-4.

Overall, the results for Ring 4 yield a sedimentation rate of about 0.03 cm/yr (using one significant figure). This rate seems reasonably reliable for this area and consistent with previous studies at somewhat comparable locations (e.g., Huh et al. 1990). Thus, <1 cm of sediment has most likely been deposited since the sinking of the ex-AGERHOLM in 1982.

Sediments have been more disturbed in Ring 1, and most likely occurred during the sinking of the ex-AGERHOLM. The geochronology data can be used to suggest possible effects. For example, sediment at two stations on the port side of the ship in Ring 1 was most likely moved away from the ship by the hydraulic wake caused during sinking. Some surface sediment most likely resettled at Station 1-4, and possible Station 1-6, on the starboard side of the sunken ship.

3.1.2.2.3. Summary of Sedimentation Rate Evaluation

Attempts to date the sediments through calculation of sedimentation rates by using two different methods produced slightly different results. The analysis of sediment collected in the traps yielded a rate of 0.1-0.2 cm/yr, which corresponds to an accumulation of 1.6-3.2 cm for 16 years, while radiological age-dating techniques yielded a lower rate of about .03 cm/yr, corresponding to an accumulation of about 0.5 cm for the 16 years. If the latter rate is closer to the truth, then the top 3 cm of material integrated deposition of sediment from a longer time period than 16 years. The comparison of ship site vs. reference site, in which any dilution of sediment is constant across the study, renders this issue more of an academic one rather than a critical confounder. Finally, these ranges of values did not differ from those found in the relevant literature.

3.1.3. Infaunal Community Assessment

The infaunal community, small invertebrates living on and in the sediments, was sampled to assist in describing the existing biological conditions and to provide a baseline description of this community at the study site. Quantitative (i.e., boxcore samples) and non-quantitative (e.g., ROV collection, organisms on retrieved equipment, and processing of residual sediments in boxcores where the majority of the sediments were collected for chemistry or bioassay) evaluation of the

epibenthic macroinvertebrates/infaunal community was conducted on samples retrieved from both the Inner Ring (ship site) and Outer Ring (reference site).

Infaunal samples were collected using small boxcores (0.053 m2 surface area) and the sediments were screened through nested sieves with two different size mesh: first, through a 1.0 mm sieve (retains all the organisms larger than 1.0 mm) and then through a 0.5 mm sieve (retains organism smaller than 1.0 mm but larger than 0.5 mm). The size fractionation of infauna samples permits comparisons to historical information. For example, studies characterizing the infaunal communities of southern California slopes and basins utilized a 1.0 mm sieve (e.g., Hartman and Barnard 1960, Thompson and Jones 1987). More recent studies in other areas, especially for deeper communities, have been using smaller sieves (e.g., 0.5 and 0.3 mm sieves) for processing and characterizing infaunal communities. The 0.5 mm sieve was selected as a default size when the use of a 0.3 mm proved impractical for the coarse and gravel sediments in the study area. The 1.0 mm sieve characterizes the community for the larger organisms retained only on the 1.0 mm sieve. The infaunal community characterized by the 0.5 mm sieve includes all of the organisms larger than 0.5 mm, which involves the addition of the organisms retained on the 1.0 mm sieve and the additional organisms retained by the 0.5 mm sieve representing the whole community larger than 0.5 mm.

3.1.3.1. Community Composition

3.1.3.1.1. Species Richness and Abundance

This evaluation found 240 unique taxa (exclusive of nematodes and calanoid copepods) representing 1,508 organisms (Appendix A). The infaunal community, sampled (both 0.5 mm and 1.0 mm fractions) by quantitative boxcores, was represented by 203 unique taxa representing 1,290 organisms (Table 3-5). This represents a species richness of 25.8 taxa/0.053 m^2 boxcore with an infaunal density of 1,058 individuals/m^2 for the community defined by the 0.5 mm sieve. For the infaunal community defined only by the 1.0 mm sieve, the species richness was 13.5 taxa/boxcore with a density of 445 individuals/m^2 (Table 3-6). The non-quantitative miscellaneous collections added another 36 unique taxa representing 225 individuals. Nematodes (218) and calanoid copepods (7) were not included, because nematodes tend to be small and are poorly sampled by the 0.5 mm sieve, whereas the calanoid copepods are typically found in the water column and therefore are not representative of the infaunal community.

3.1.3.1.2. Comparisons with Relevant Historical Data

For comparison, the density of infauna reported by Hartman and Barnard (1960) for the 1.0 mm community for the slopes and basins of southern California was only 71 organisms/m^2. Studies of the 1.0 mm infaunal community off of southern California (Thompson and Jones 1987) found an average of 20.4 taxa/0.06 m^2 boxcore with an average density of 642 individuals/m^2 for the nearshore upper slope infaunal community (161-632 m deep), 13.5 taxa/boxcore with a density of 458.5 individuals/m^2 for the nearshore lower slope communities (480-851 m deep), and for the offshore basins lower slopes (541-1768 m deep), an average of 28.4 taxa/boxcore with a density 1,275 individuals/m^2 was reported. The results of this SINKEX study for the 1.0 mm community (13.5 species/boxcore and 445 individuals/m^2) characterize a community that appears almost identical to the nearshore lower slope community, but more depauperate than the offshore basin lower slope community. This difference may reflect the coarse gravel nature of the sediment from the SINKEX study site that tends to be less favorable habitat for small infaunal organisms.

Table 3-7 and Table 3-8 summarize the infaunal community for the major taxonomic groups for the community represented by the 0.5 mm and 1.0 mm sieve fractions from the ex-AGERHOLM site, respectively. The community represented by just the 1.0 mm fraction had about half the number of taxa/sample and abundance of the 0.5 mm fraction. For communities on the continental shelf, the 0.5 mm fraction will typically have about two to three times the abundance of the 1.0 mm fraction and about 50% greater species richness Thus, the abundance proportions for 1.0 and 0.5 sieve sizes from the ex-AGERHOLM site are similar to that of shallower communities, but the samples from the study area indicate a greater number of smaller species. This is consistent with the general concept that deeper water communities tend to have smaller sized organisms (Theil, 1975; Gage and Tyler, 1991).

3.1.3.1.3. Overall Composition by Major Taxonomic Groups

The total infaunal community (including both 0.5 mm and 1.0 mm fractions, Table 3-5) was dominated by segmented worms (polychaetes), representing 42.8% of the species and 55.7% of the abundance. Crustaceans were the second most important group of organisms representing 33.5% of the species and 21.1% of the individuals. Mollusca were the third most important group representing 12.8% of the species and 11.5% of the abundance. These groups were followed by minor phyla having 4.4% and 2.3%, echinodermata having 3.4% and 8.5%, and cnidaria having 3.0% and 0.9% of the species and individuals, respectively.

For the 1.0 mm community these proportions (of species and abundance, respectively) were as follows:

- polychaetes 43.5% and 57.6%,
- crustaceans 26.6% and 16.2%,
- molluscs 15.6% and 9.45%,
- echinoderms 4.5% and 11.6%,
- cnidaria 3.9% and 1.5%,
- and for minor Phyla 5.8% and 3.7% for proportion of taxa and abundance, respectively.

These proportions for the 1.0 mm fraction are similar or intermediate between those values representing slope and shelf communities (Hartman and Barnard 1960). These results suggest that the community at the study site is somewhat of a transitional community between upper slope/shelf and lower slope communities.

Table 3-5. Summary of number of species and abundance by major taxonomic group for the combined 0.5 and 1.0 mm infauna size fractions and for miscellaneously collected organisms.

	Infauna No. of Species	Misc. Invertebrate No. of Species	Total No. of Species	Infauna Abundance	Misc. Invertebrate Abundance	Total Abundance
Cnidaria (anemones, hydroids, sea pens)	6	4	10	12	17	29
Polychaeta (worms)	87	12	98	718	89	807
Minor Phyla (sponges, nemertea, bryozoan, etc)	9	1	10	30	6	36
Crustacea (amphipods, Cumacea, shrimp, crabs)	68	12	81	272	51	316
Mollusca (snails, clams, chitons, and tooth shells)	26	1	27	148	17	165
Echinodermata (starfish, brittle stars, sea cucumbers)	7	6	13	110	45	155
TOTAL	203	36	240	1290	225	1508

Table 3-6. Summary of number of species and abundance by major taxonomic group for the 1.0 mm infauna size fractions and for miscellaneously collected organisms.

	Infauna No. of Species	Misc. Invertebrate No. of Species	Total No. of Species	Infauna Abundance	Misc. Invertebrate Abundance	Total Abundance
Cnidaria (anemones, hydroids, sea pens)	6	4	10	8	17	25
Polychaeta (worms)	67	12	79	313	89	402
Minor Phyla (sponges, nemertea, bryozoan, etc)	9	1	10	20	6	26
Crustacea (amphipods, Cumacea, shrimp, crabs)	41	12	53	88	51	139
Mollusca (snails, clams, chitons, and tooth shells)	24	1	25	51	17	68
Echinodermata (starfish, brittle stars, sea cucumbers)	7	6	13	63	45	108
TOTAL	203	36	240	543	225	768

Table 3-7. Summary of community measures by station for the combined 0.5 and 1.0 mm infaunal community.

Station	Number of Replicates	Number of Species	Abundance	Shannon-Wiener Diversity	Margalef Species Richness	Dominance Index	Evenness Index
1-1	2	19	33	2.71	5.07	13	0.94
1-2	1	32	53	3.48	7.81	23	1.00
1-3	1	20	29	2.81	5.64	14	0.94
1-3.5	1	31	58	3.33	7.39	20	0.97
1-4	1	10.5	22	1.86	3.18	5.5	0.79
1-5	1	21	42	3.00	5.35	15	0.99
1-5/6/7	2	27	59.5	3.14	6.34	17	0.96
1-6	1	24	70	2.88	5.41	11	0.91
1-7	2	33	65	3.37	7.66	21	0.97
1-8	1	22	51	3.04	5.34	13	0.98
4-1	2	35.5	91.5	3.25	7.63	18	0.91
4-2	2	27	47.5	3.23	6.74	18.5	0.99
4-3.1	2	24.5	45.5	3.10	6.16	15.5	0.97
4-5	2	25	45.5	3.02	6.36	16.5	0.95
4-6	2	35.5	88	3.21	7.67	18.5	0.92
Average	-	25.8	53.4	3.03	6.25	16.0	0.95

Table 3-8. Summary of community measures by station for the 1.0 mm infaunal community.

Station	Number of Replicates	Number of species	Abundance	Shannon-Wiener Diversity	Margalef Species Richness	Dominance Index	Evenness Index
1-1	2	11.5	19.5	2.32	3.73	7.5	0.95
1-2	1	11	13	2.46	3.90	9	1.03
1-3	1	15	24	2.50	4.41	10	0.92
1-3.5	1	16	29	2.59	4.45	10	0.94
1-4	1	10	29	1.48	2.67	3	0.64
1-5	1	11	22	2.41	3.24	9	1.01
1-5/6/7	2	12	19.5	2.48	3.71	9.5	1.00
1-6	1	12	26	2.19	3.38	7	0.88
1-7	2	17	27.5	2.83	4.80	13	1.01
1-8	1	10	22	2.29	2.91	7	0.99
4-1	2	15	28	2.64	4.22	10.5	0.99
4-2	2	15.5	18.5	2.68	4.98	11.5	0.99
4-3.1	2	13	18.5	2.54	4.15	9.5	0.99
4-5	2	11.5	15	2.49	3.91	10	1.02
4-6	2	21.5	42.5	2.87	5.45	13.5	0.94
Average	-	13.5	23.6	2.45	3.99	9.33	0.95

3.1.3.2. Inner versus Outer Ring Comparisons – Total Community (0.5 & 1.0 mm)

The infaunal community was further evaluated using common community measures, which are summarized in Table 3-9 and Table 3-10. For the total infaunal community (both 0.5 and 1.0 mm sieve) there were no significant differences for any of the community measures between the Inner and Outer Rings, and no significant regressions with grain size and TOC measurements, or PCA factors (Table 3-9).

Table 3-9. Mean value of community measures and results of statistical hypothesis testing for the combined 0.5 and 1.0 mm infaunal community.

Community Measure	Data Transformation	Mean Inner Ring (Ring 1)	Mean Outer Ring (Ring 4)	Statistical Significance	Interpretation: Reject Null Hypothesis
No. of Species	Non transformed	24.0	29.5	P=0.148	No
Abundance	Non transformed	48.3	63.6	P=0.161	No
Shannon Wiener Diversity Index	Ranks	2.96	3.16	P=0.411	No
Margalef Species Richness	Non transformed	5.92	6.91	P=0.170	No
Dominance Index	Non transformed	15.3	17.4	P=0.388	No
Evenness Index	Ranks	0.945	0.948	P=0.728	No

Table 3-10. Mean value of community measures and results of statistical hypothesis testing for the 1.0 mm infaunal community.

Community Measure	Data Transformation	Mean Inner Ring (Ring 1)	Mean Outer Ring (Ring 4)	Statistical Significance	Interpretation: Reject Null Hypothesis
No. of Species	Log 10	12.6	15.3	P=0.112	No
Abundance	Log 10	23.2	24.5	P=0.964	No
Shannon Wiener Diversity Index	**Ranks**	**2.36**	**2.65**	**P=0.021**	**Yes**
Margalef Species Richness	Log 10	3.72	4.54	P=0.050	No*
Dominance Index	Non transformed	8.50	11.0	P=0.075	No
Evenness Index	Ranks	0.936	0.987	P=0.728	No

* Criteria for significance p<0.05

3.1.3.2.1. Species Richness and Abundance

The number of taxa/sample, considered by many as the only true measure of community diversity, ranged from 10.5 to 35.5 taxa/sample. On average the Outer Ring (29.5 taxa/sample) had 5.5 more taxa/sample than the Inner Ring (24.0 taxa/sample); however, this difference was not statistically significant (p=0.148). Abundance values ranged from 22 to 91.5 individuals/sample and showed a pattern similar to the number of taxa/sample with the Outer Ring (63.6 individuals/sample) having 15.3 more individuals/sample than the Inner Ring (48.3 individuals/sample); however, this difference also was statistically not significant.

3.1.3.2.2. Diversity Indices

Diversity indices are derived community measures that utilize number of species and their abundance for their calculation. For diversity indices to be useful they should have some biological meaning, be easy to interpret, and they should be sensitive to community changes caused by pollutant impacts. Diversity is considered important because it appears to be an attribute of natural or organized community. Diversity is a widely used term to define observed frequencies of species with a particular number of individuals within a population or community. Diversity has two aspects that relate to the number of species (taxa) in a sample and the relative abundance of each species. Because the number of species and the equitability of the species abundance often vary independently, numerous methods of calculating diversity have been

developed. Four diversity indices have been used to evaluate the infaunal data: Shannon Wiener Index, Margalef's Species Richness, Dominance, and Evenness.

The Shannon Wiener (H′) index has been widely used to evaluate community responses to impacts. This index is based upon information theory, tends to be normally distributed and independent of sample size. This index tends to be more sensitive to the distribution (equitability) of individuals among the species and less sensitive to species richness.

For the total community (0.5 and 1.0 mm communities), Shannon Wiener values Table 3-7) ranged from 1.86 (Station 1-4) to 3.48 (Station 1-2). Low values indicate that a few species dominated the sample, while higher values reflect a more equitable distribution of the individuals among the species. Shannon Wiener values appeared to be lower for the Inner Ring (2.96) than the Outer Ring (3.16), but this difference (as for all the comparisons of diversity indices) was not statistically significant.

Margalef's Species Richness Index (Table 3-9) transforms the community measures by normalizing the number of taxa by the natural log of the abundance. This transformation produces a numeric value that is essentially a dominance measure; thus, this index is more sensitive to the distribution of individuals among the species rather than the number of taxa. Margalef's Species Richness values ranged from 3.18 (Station 1-4) to 7.81 (Station 1-2), and the Inner Ring (5.92) had lower values than the Outer Ring (6.91). This difference was also non-significant (Table 3-9).

The Dominance Index used in this study is the number of taxa that accounts for 75% of the individuals. Dominance (Table 3-7) ranged from 5.5 (Station 1-4) to 23 (Station 1-2). Inner Ring Dominance averaged 15.3 while the Outer Ring averaged 17.4. This means that the Outer Ring required 17.4 taxa to account for 75% of the abundance, while the Inner Ring required 15.3 taxa. However, this difference was not significant (Table 3-9). Higher values of Dominance (i.e., fewer dominant species) correlate with higher values of Shannon Wiener and Margalef's indices. The Evenness Index is another measure of the equitability of the individuals among the taxa. Evenness (Table 3-7) ranged from 0.79 (Station 1-4) to 1.00 (Station 1-2). The Inner Ring (0.945) had slightly lower Evenness than the Outer Ring (0.948), but again this was non-significant (Table 3-9). Note that the lowest values for all these diversity indices were at Station 1-4, while Station 1-2 had the highest values. This indicates that while these diversity measures have very different values, they all tend to show the same pattern and trends.

3.1.3.2.3. Community Measures and Habitat Variability

In addition to the differences in the physical environment between the Inner and Outer Rings and the statistical differences for some of the community measures, there were also differences in the ranges of values observed for the community measures. Notably, the infaunal community (both 0.5 and 1.0 mm) for the Outer Ring was generally more consistent and less variable than comparable measures for the Inner Ring (Table 3-13). The increased range of values for the Inner Ring measures is consistent with increased habitat variability associated with possible "reef effects" of the sunken vessel, and/or possible physical alterations (e.g., more exposed and broken gravel/cobble) of the sediments from the impact of the sunken vessel as it contacted the bottom. For the major taxonomic groups, except for polychaetes, the greatest values for number of species and abundance were in the Outer Ring. All but one station within the Inner Ring (Station 1-7) had at least one measurement for number of species and/or abundance that were less than the range observed at the Outer Ring stations.

3.1.3.3. Inner versus Outer Ring Comparisons – Focus on 1.0mm Community

A similar type of community analysis was conducted just for the 1.0 mm community (Table 3-10). This analysis found no significant difference for four of the six community measures, but Margalef's Species Richness was at the cutoff of significance (p=0.050) and Shannon-Wiener Diversity was significantly (p=0.021) higher for the Outer Ring. This difference reflects the fact that the Outer Ring tended to have a few more species, and they tended to be more equitable in their distribution.

3.1.3.3.1. Correlations Among Composition and Physical/Chemical Properties

Regression analysis was used to determine if observed differences in community measures were correlated with differences in physical (i.e., grain size measures) and chemical properties (i.e., PCA factors) of the sediments. Regression analysis found only marginal correlations for both Margalef's Species Richness and Shannon-Wiener Diversity with % sand (r^2=0.27, p=0.057 and r^2=0.024, p=0.078, respectively). Since there was no significant difference between the rings for the primary community measures (i.e., number of species and their abundance) and for two of the four derived measures there is little indication of effects from the vessel on the infaunal community. The lack of significant correlations with sediment contaminants and only marginal correlations with sediment grain size measures suggests that the statistical results are most likely due to physical differences between the rings and not from sediment contaminants.

3.1.3.3.2. Inner Versus Outer Rings – Major Taxonomic Groups (0.5 & 1.0 mm)

A more detailed examination of the infaunal data for the five major taxonomic groups found more significant differences among the community measures in these groups between the Inner and Outer Ring than in the total community measures (Table 3-11 and Table 3-12). Thus, when considering the total infaunal community (both 0.5 and 1.0 mm) there appears to be no differences between the Inner and Outer Rings and only one minor difference for the 1.0 mm community. However, because of significant differences in physical characteristics (e.g., Table 3-48) between the Inner and Outer Rings, some differences in the infaunal community would be expected.

For the total infaunal community (0.5 and 1.0 mm) there were no significant differences for annelids and minor phyla for any of the community measures (Table 3-11). In contrast, the Outer Ring had more crustacean species, with greater diversity; more molluscan taxa with greater abundance; and greater echinoderm dominance, than the Inner Ring.

Table 3-11. Summary of statistical hypothesis testing for infaunal community measures by major phyla for the 0.5 mm and 1.0 mm size fractions. Bold values indicate statistical significance p<0.05; 4>1 denotes that the values for the Outer Ring (4) were greater than the Inner ring (1).

Group	Number of Species	Abundance	Shannon-Wiener Diversity	Margalef'sSpecies Richness	Dominance index	Evenness
Annelida	P=0.400	P=0.847	P=0.411	P=0.413	P=0.551	P=0.411
Crustacea	**P=0.034** **4 > 1** **9.40 > 4.55**	P=0.138 4 > 1 16.2 > 8.25	P=0.064 4 > 1 1.95 > 1.27	**P=0.011** **4 > 1** **3.11 > 2.05**	**P=0.022** **4 > 1** **6.30 > 3.40**	P=0.950
Mollusca	**P=0.015** **4 > 1** **4.70 > 2.50**	**P=0.047** **4 > 1** **8.90 > 4.20**	**P=0.031** **4 > 1** **1.36 > 0.82**	P=0.088 4 > 1 1.81 > 1.39	P=0.061 4 > 1 3.00 > 2.20	P=0.070 1 > 4 0.988 > 0.922
Minor Phyla	P=0.759	P=0.396	P=0.719	P=0.336	P=0.803	P=0.118
Echinodermata	P=0.678	P=0.213	P=0.916	P=0.404	**P=0.020** **4 > 1** **1.60 > 1.20**	P=1.000

Table 3-12. Summary of statistical hypothesis testing for infaunal community measures by major phyla for the 1.0 mm size fraction. Bold values indicate statistical significance p<0.05, 4 > 1 denotes that the values for the Outer Ring (4) were greater than the Inner Ring (1).

Group	Number of Species	Abundance	Shannon-Wiener Diversity	Margalef's Species Richness	Dominance Index	Evenness
Annelida	P=0.073 4 > 1 8.15 > 6.20	**P=0.016** **1 > 4** **17.3 > 10.0**	P=0.261	P=0.372	P=0.333	P=0.457
Crustacea	P=0.086 4 > 1 4.00 > 1.55	P=0.135 33	P=0.653	P=0.052 4 > 1 2.06 > 1.32	**P=0.005** **4 > 1** **3.30 > 1.60**	P=0.492
Mollusca	**P=0.005** **4 > 1** **1.19 > 0.84**	**P=0.037** **4 > 1** **3.10 > 1.45**	**P=0.014** **4 > 1** **0.92 > 0.27**	**P=0.031** **4 > 1** **1.83 > 1.01**	**P=0.009** **4 > 1** **2.60 > 1.40**	P=0.197
Minor Phyla	P=0.901	P=0.901	P=0.403	P=0.068 1 > 4 1.63 > 1.44	P=0.402	NC
Echinodermata	P=0.170	P=0.580	P=0.693	P=0.504	P=0.117	P=1.000

NC=Not Capable of being calculated; i.e., too similar

3.1.3.3.2.1. Crustaceans

There were significantly (p=0.034) more crustacean taxa/sample for the Outer Ring (9.40) compared with the Inner Ring (4.55). This also caused crustacean Dominance and Margalef's Species Richness to be significantly higher (p=0.022 and p=0.011, respectively). For the number of crustacean taxa, a significant, but weak, positive correlation was found with % sand (r^2=0.26), and similar negative correlations were found with % TOC (r^2=0.22) and % gravel (r^2=0.31) (Figure 3-18). Only the regression with gravel was statistically significant (p=0.037). However, the regressions with % sand (p=0.058) and % TOC (p=0.089) were close to the statistical cutoff and support the observation that the greater number of crustacean taxa away from the target vessel may be related to changes in these measures that are natural, or possibly due to reef

effects. These results suggest that the sediments adjacent to the target vessel, because of their higher % gravel, TOC, and lower % sands, are not as favorable a habitat as the Outer Ring for crustaceans. These results also support the theory (to be discussed later in the Effects section) that the one negative toxicity[7] result (refer to Section 3.3.1) for *Rhepoxynius abronius* (a crustacean) at Station 1-6 may have been due to the physical characteristics of the sediments, and not from sediment toxicity at this location. This station had the lowest proportion of sand, high TOC and % clay (Table 3-45 and Table 3-46).

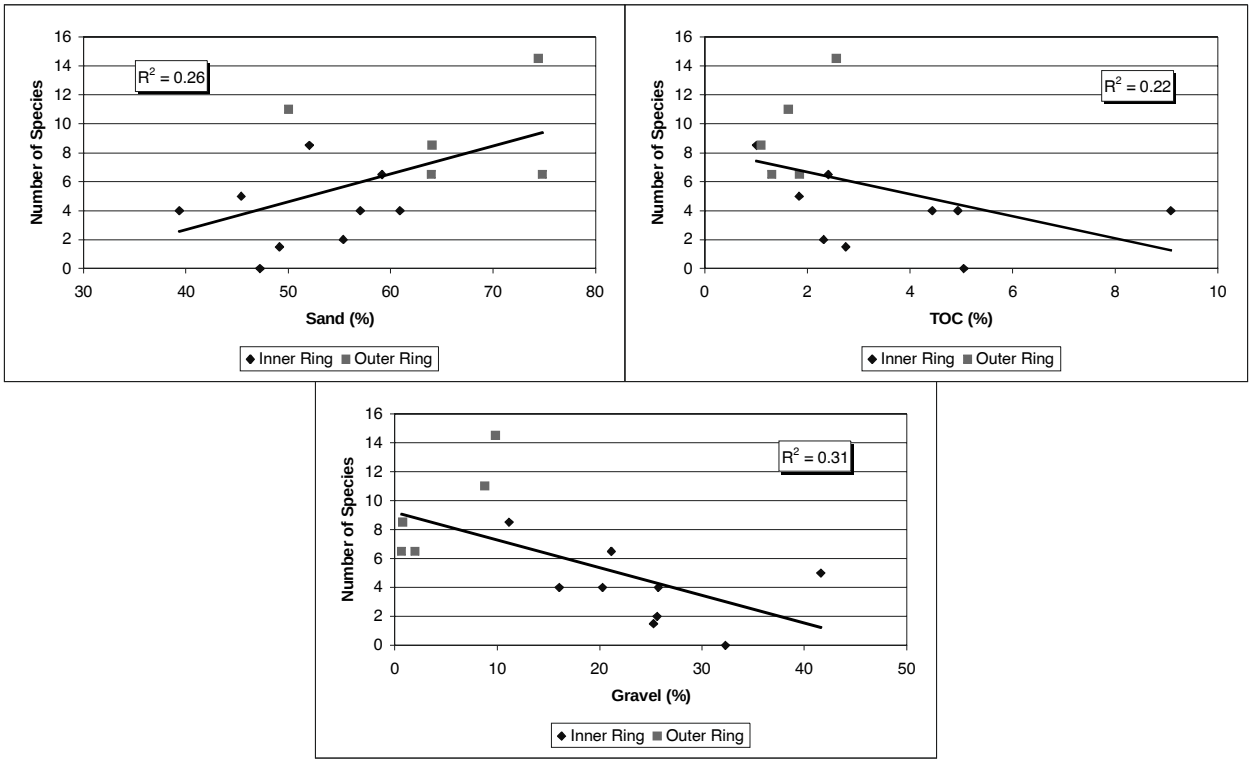

Figure 3-18. Regression of number of crustacean species (0.5 + 1.0 mm) with physical characteristics.

3.1.3.3.2.2. Molluscs

Mollusc patterns were similar to crustacean patterns. There were significantly more mollusc taxa per sample at the Outer Ring (4.70) than at the Inner Ring (2.50). In addition, there was significantly greater mollusc abundance at the Outer Ring (8.9) than the Inner Ring (4.2). These differences between the rings also contributed to a higher Shannon Wiener Diversity for the Outer Ring. Regression analysis found significant, albeit weak, positive correlations for the number of mollusc taxa with % silt (r^2=0.38, p=0.018) and Fine Grain Size Factor (PCA Factor 2, see Section 2.5.12.4) (r^2=0.39, p=0.017), and a similar significant negative correlation with % gravel (r^2=-0.30, p=0.043) (Figure 3-19). Thus, as found for the crustaceans some of the difference between the Inner and Outer Ring for molluscs appears to be due to the differences in the composition of the sediments. The lack of significant correlations with PCA Factors 1, 3 and 4 (mainly characterized by primary and secondary contaminants of concern) supports the

[7] Due to the complexity of this report, it is necessary to refer to toxicity in this section, although the topic is not discussed in detail until a following section.

contention that differences in the molluscan community were due to differences in physical parameters, and not chemical contaminants contributed by the sunken ship.

3.1.3.3.2.3. Echinoderms

For the echinoderms there was only one community measure that indicated some difference between rings. The Outer Ring had a significantly higher echinoderm Dominance Index than the Inner Ring (Table 3-11), but there were no significant correlations with any measured environmental parameters. This observation suggests that the statistical difference may not have ecological significance.

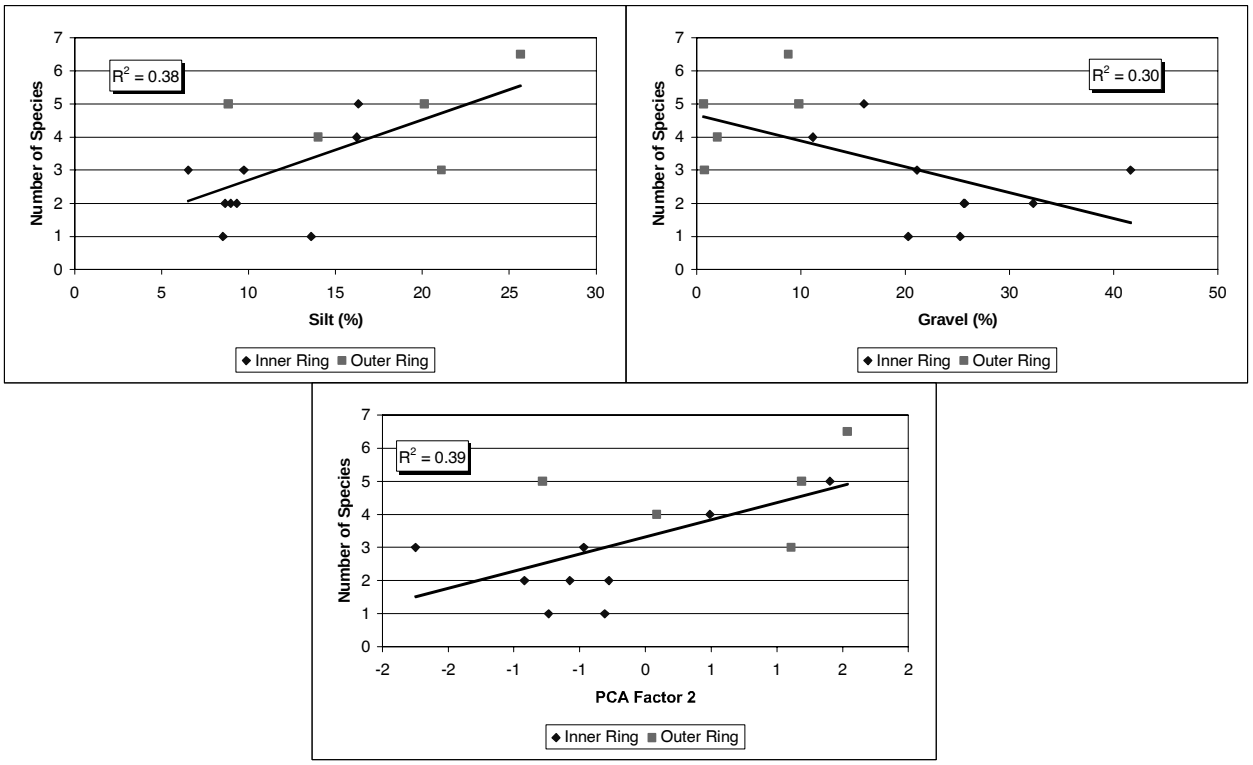

Figure 3-19. Regression of number of molluscan species (0.5 + 1.0 mm) with physical characteristics.

3.1.3.3.3. Inner versus Outer Rings – Major Taxonomic Groups (Focus on 1.0 mm)

A slightly different perspective was provided by considering only the community represented by the 1.0 mm fraction (Table 3-12). There were no significant differences for the echinoderms and minor phyla for any of the community measures. There were more worms (annelids) and fewer crustaceans and molluscan species with lower abundance at the Inner Ring, compared to the Outer Ring.

3.1.3.3.3.1. Annelids

There were significantly more annelids at the Inner Ring (17.3) compared to the Outer Ring (10.0). This difference was significantly, but weakly correlated with % TOC (r^2=0.34, p=0.030), indicating that the increase in the number of worms near the target ship may have been in part related to the higher concentration of TOC at the Inner Ring, a possible indicator of food, and not

due to sediment contaminants (Figure 3-20). It is unknown whether such food abundance would preferentially favor worms over other organisms.

3.1.3.3.3.2. Crustaceans

As found for the combined 0.5 and 1.0 mm fractions, crustaceans tended to have more individuals and species away from the target vessel (although this was not significant), and this influenced the calculations for diversity indices. Of the four crustacean diversity measures, only crustacean Dominance was significantly higher for the Outer Ring. However, number of species ($p=0.086$) and Margalef's Species Richness ($p=0.052$) were close to being significant. Crustacean Dominance was significantly and strongly correlated with % sand ($r^2=0.59$, $p=0.010$) and negatively, though weakly, correlated with % gravel ($r^2=0.25$, $p=0.067$) (Figure 3-21). Thus, physical differences in grain size parameters correlate with crustacean community measures and may contribute to observed community differences between Inner and Outer Rings.

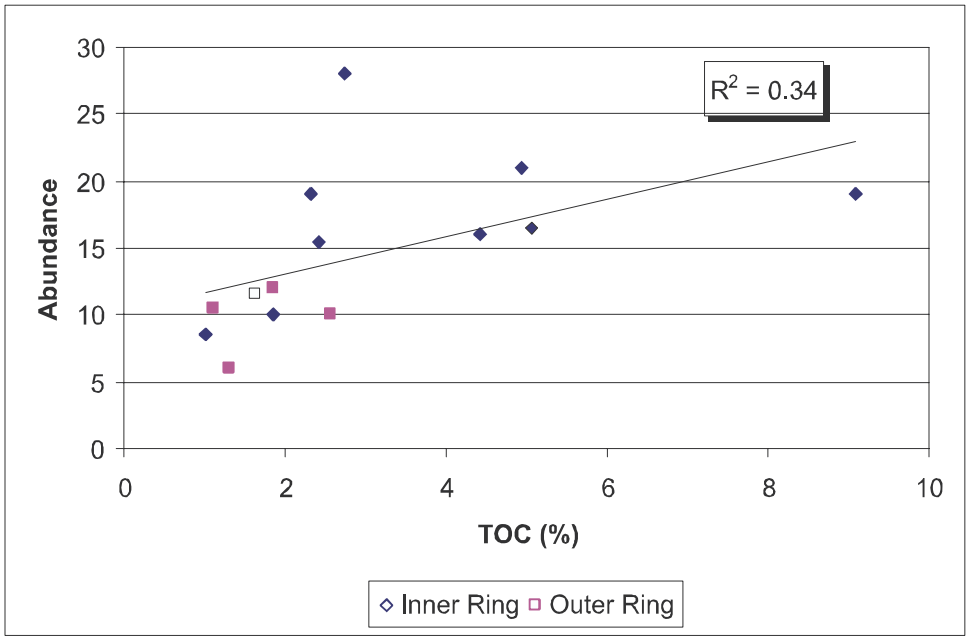

Figure 3-20. Regression of polychaete abundance (1.0 mm) with TOC.

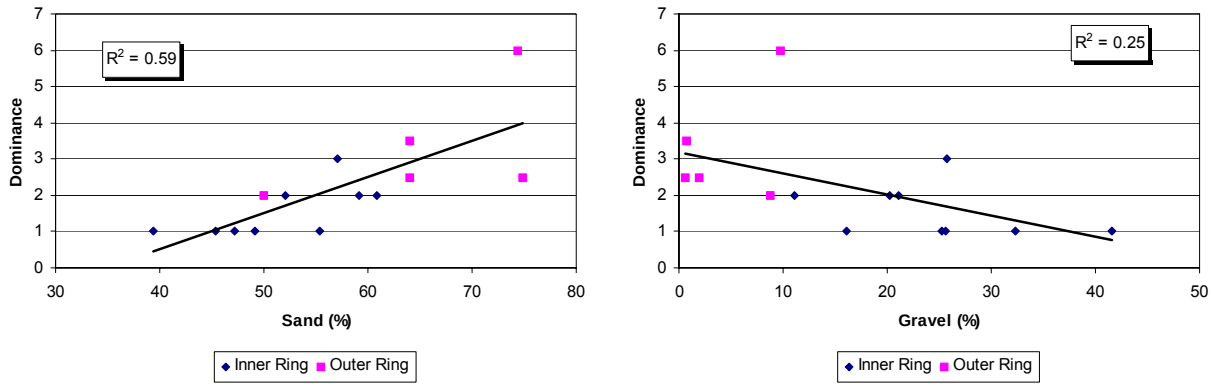

Figure 3-21. Regression of crustacean dominance (1.0 mm) with grain size measures.

3.1.3.3.3.3. Molluscs

Molluscan community measures followed the pattern of the crustaceans (Table 3-12). There were significantly more mollusc taxa with greater abundance at the Outer Ring and this difference contributed to a higher diversity values for Shannon Wiener Diversity, Margalef's Species Richness, and Dominance. The number of molluscan taxa was negatively correlated with % gravel (r^2=0.27, p=0.0593), and molluscan abundance was not significantly correlated with the measured environmental variables. Derivation of PCA factors is discussed in Section 2.5.12.4. Thus, as found for other major phyla groups, the differences in grain size measures correlate with some of the differences observed for the infaunal community measures. In contrast to crustaceans and molluscs, the polychaetes had greater abundance closest to the target vessel. This apparent enhancement effect correlates with polychaete abundance being positively correlated with TOC, which was highest nearest the target vessel (Figure 3-20). The data for all of the above measurements are tabulated in Table 3-13.

Table 3-13. Ranges of infaunal community measures and differences between Inner and Outer Rings.

Measurement Type	Mean ± SD First set of data are Outer Ring summaries/Second set of data are for Inner Ring summaries - Highest values are in bold		Range of Measurement First set of data are Outer Ring summaries /Second set of data are for Inner Ring summaries - Highest values are in bold		Differences Ranges (Maximum/minimum) First set of data are Outer Ring, Second set of data are for Inner Ring bold data indicates least variable set of data		Differences % Difference between Outer Ring and Inner Ring ranges
Species							
Total	29.50±5.56	24.00±6.99	24.5-35.5	10.5-33.0	1.45	3.14	0.46
Mollusca	4.70±1.30	2.50±1.27	3.0-6.5	1.0-5.0	2.17	5.0	0.43
Annelida	12.90±1.08	14.45±3.84	11.5-14.5	7.5-21.0	1.26	2.80	0.45
Arthropoda	9.40±3.40	4.55±3.10	6.5-14.5	0.0-10.0	2.23	∝	∝
Echinodermata	1.10±0.55	0.90±0.32	0.5-2.0	0.0-1.0	4.0	∝	∝
Other	1.40±0.82	1.55±0.90	0.5-2.5	0.0-3.0	5.0	∝	∝
Abundance							
Total	63.6±23.92	48.3±16.11	45.5-91.5	22-70	2.01	3.18	0.63

Mollusca	8.90±5.92	4.20±3.20	4.5-19.0	4.5-11.0	4.22	2.44	173
Annelida	29.60±6.70	31.3±9.97	23.0-37.5	22.0-70.0	1.63	3.18	0.51
Arthropoda	16.2±9.65	8.25±7.45	7.0-29.5	0.0-22.5	4.21	∝	∝
Echinodermata	7.50±7.98	2.55±1.98	1.0-20.5	0.0-6.0	20.5	∝	∝
Other	1.40±0.82	2.00±1.39	0.5-2.5	0.0-4.0	5.0	∝	∝
Shannon-Wiener Diversity							
Total	3.16±0.10	2.96±0.46	3.02-3.25	1.86-3.37	1.08	1.82	0.59
Mollusca	1.36±0.20	0.82±0.47	1.1-1.5	0.0-1.6	1.36	∝	∝
Annelida	2.44±0.12	2.47±0.42	2.3-2.6	1.5-3.1	1.13	2.07	0.55
Arthropoda	1.95±0.37	1.27±0.69	1.5-2.5	0.0-2.5	1.67	∝	∝
Echinodermata	0.43±0.16	0.20±0.33	0.3-0.7	0.0-1.0	2.33	∝	∝
Other	0.42±0.29	0.51±0.47	0.0-0.7	0.0-1.0	∝	∝	∝
Margalef Diversity							
Total	6.91±0.71	5.92±1.42	6.16-7.63	3.18-7.81	1.25	2.46	0.51
Mollusca	1.81±0.24	1.39±0.47	1.4-2.0	0.6-2.2	1.43	3.67	0.39
Annelida	3.57±0.27	3.93±0.92	3.2-3.8	2.3-5.6	1.19	2.43	0.49
Arthropoda	3.11±0.67	2.05±0.57	2.7-4.3	1.2-3.1	1.59	2.58	0.62
Echinodermata	0.06±0.13	0.29±0.55	0-0.3	0.0-1.4	∝	∝	∝
Other	1.62±0.36	1.39±0.50	1.4-2.2	0.7-1.8	1.57	2.57	0.61
Dominance Diversity							
Total	17.4±1.34	15.3±5.20	15.5-18.5	5.5-23	1.19	4.18	0.28
Mollusca	3.0±0.35	2.2±0.82	2.5-3.5	1-4	1.4	4	0.35
Annelida	8.4±1.14	9.3±3.14	7-9.5	3.5-15	1.36	4.29	0.32
Arthropoda	6.3±2.11	3.4±1.85	5-10	1-7	2.0	7.0	0.29
Echinodermata	1.6±0.22	1.2±0.35	1.5-2	1-2	1.33	2.0	0.67
Other	1.6±0.42	1.65±0.58	1-2	1-2.5	2.0	2.5	0.8
Evenness Diversity							
Total	0.95±0.03	0.95±0.06	0.91-0.99	0.79-1.00	1.08	1.27	0.85
Mollusca	0.92±0.06	0.99±0.06	0.8-1.0	0.8-1.1	1.25	1.38	0.91
Annelida	0.96±0.03	0.94±0.07	0.9-1.0	0.8-1.0	1.11	1.25	0.89
Arthropoda	0.94±0.10	0.96±0.06	0.8-1.0	0.8-1.0	1.25	1.25	1.00
Echinodermata	0.25±0.56	0.75±0.66	0.0-1.0	0.0-1.3	∝	∝	∝
Other	1.00±0.00	1.21±0.25	1.0-1.0	1.0-1.5	0.0	1.5	∝

3.1.3.4. Benthic Infaunal Assessment Conclusions

The study site represents a unique slope habitat characterized by a thin veneer of coarse to fine sediments overlying consolidated substrate, which has been characterized as "borderland hard bottom" (Svedrup et al., 1942). Because of the predominately coarse and rocky nature of this habitat and the difficulty of sampling, the infaunal community associated with this habitat is largely unknown. Thus, this study provides a first look at a relatively diverse, but not abundant, community. Taxonomic analysis was difficult as many of the taxa collected are represented by

one to few individuals, tend to be small, and were often damaged by sample processing due to the gravel and rocks in the samples. There appear to be several new species collected during this study, which would require further effort to describe and characterize.

There were essentially no significant differences between the Inner and Outer Ring community measures for the whole infaunal community, and only one of the five community measures was different for the community defined for just the 1.0 mm sieve fraction and none for the 0.5 sieve fraction. Thus, at the whole community level there is no indication of an effect from the target vessel. However, a further evaluation of the infaunal community, focused on each of the major phylogenetic groups, identified significant differences between the Inner and Outer Ring. That community differences might be expected was indicated by significant differences in grain size and TOC measures, which typically affect infaunal communities. There were no significant correlations with PCA factors that incorporated sediment chemistry. The infaunal community appears to have responded to these sediment differences resulting in there being more crustaceans and molluscs at the Outer Ring and fewer at the Inner Ring, but this was offset by there being more polychaetes (annelids) at the Inner Ring. Thus, some differences in the distribution and diversity of the major phylogenetic groups were evident. However, most of these differences were correlated with grain size and TOC gradients. While there were some significant differences for some of the major infaunal taxonomic groups, these differences appear to reflect differences in sediment measures between the Inner and Outer Rings. Whether these differences in sediment measures represent natural gradients of the study area, “reef effects”, impact disturbance, or some combination of these influences cannot be determined.

3.2. Characterization of Exposure to COCs

Characterization of Exposure to COCs is divided into the following three environmental compartments relevant to the SINKEX study:

1. Deep Ocean Water Exposure
2. Deep Ocean Sediment Exposure
3. Deep Ocean Tissue Exposure

Exposure is characterized primarily with the use of chemical concentrations of COCs in these three environmental compartments.

3.2.1. Deep Ocean Water Exposure

It was previously mentioned (Development of Conceptual Model and Analysis Plan) that the water column was thought to be of minor importance, relative to exposure pathways in this study. While the primary emphasis of deep ocean water exposure is on the chemical concentrations of COCs in water, there are two other aspects relating to the water column to consider: (1) the deep ocean currents and their role in transporting contaminants in particular directions, and (2) leaching mechanisms of PCBs in shipboard solids (evaluated in greater detail in a separate PCB leach rate study at SSC-SD).

3.2.1.1. Measured currents and estimated transport

This section summarizes current data collected in vicinity of the ex-AGERHOLM from 6 August 1997 through 15 November 1999. This effort was undertaken to describe general water current speed and direction in the vicinity of the hulk. Information obtained for this study provides

insight into sediment and chemical distributions, and can be used to identify a potential contaminant footprint. Water current speed and direction were recorded at three locations near the ex-AGERHOLM on the outer rim of the Tanner Basin (Figure 2-18 and Figure 2-19). An RD instruments Acoustic Doppler Current Profiler (ADCP) was deployed twice. The first deployment, located approximately 400 m northeast of the ex-AGERHOLM at a depth of 855 m, recorded current speed and direction at two-meter intervals from 5 m above the seafloor to 64 m between August 6, 1997 and September 5, 1998. The second deployment, located approximately 200 m southwest of the ex-AGERHOLM at a depth of 800 m, recorded current speed and direction at two-meter intervals from 5 m above the seafloor to 64 m between September 6, 1998 and November 15, 1999. Finally, an InterOcean S4™ located 10 m from the near the starboard bow collected current speed and direction data from a depth of 780 m between August 17, 1998 and April 5, 1999.

Figure 3-22 depicts current speed data for all instruments and across all depths for the entire program. Typical of deep waters, current speeds are quite low, with the 99.6 percent of the bottom (S4™) measurements less than 15 cm/sec (0.3 knots). Bottom current speeds averaged 3.26 cm/sec (0.06 knots, Table 3-14) with a maximum hourly average of 19.05 cm/sec (0.37 knots). Current speeds tended to increase with distance from the bottom, as measured by the ADCP, averaging 7.89 cm/sec (0.15 knots) through the water column. Between 5 and 10 meters above the seafloor, all average hourly current measurements were below 20 cm/sec, with a maximum measured value of 97 cm/sec (1.88 knots) 41 m above the seafloor. This relatively high velocity is unique to the entire data.

Table 3-14. Summary statistics for current speeds measured by ADCP and S4™ instruments.

Instrument	Depth Strata	Mean Current Speed (cm/sec - knots)	Minimum Hourly Speed (cm/sec)	Maximum Hourly Speed (cm/sec)	Number of Hourly Observations	Standard Deviation
ADCP	5 - 65 meters from seafloor	7.89 – 0.15	1.80	25.35	19693	3.42
S4	1 meter from sea floor	3.26 – 0.06	0.11	19.05	4813	1.99

Figure 3-23 and Figure 3-24 are representative frequency diagrams summarizing current direction measured through the water column (ADCP) and near the bottom (S4™), respectively, for four months in 1998. Currents measured away from the ex-AGERHOLM by the ADCP tend to favor a west-northwest and southeast direction of motion. A review of animated ADCP data supports this observation and suggests that the measured currents may be dominated by daily tidal cycles. Bottom current direction (Figure 3-24) measured next to the ex-AGERHOLM shows a much higher aggregation of westerly currents, when compared to measurements taken during the same time period, but some distance from the hulk. This pattern is distinct; however, it is mostly likely the result of extreme “shadowing” created by the physical presence of the ex-AGERHOLM.

In summary, bottom water currents are minimal and probably do not contribute to the large scale movement of sediments. Current direction appears to be dominated by daily tidal cycles. Due to the presence of very low energy bottom currents relative to the dynamics associated with sorption and settling that would cause deposition into the sediments after any release of dissolved

PCBs into the water column, any contaminants originating from the ex-AGERHOLM are not expected to differentially accumulate with directionality in the near hulk sediments.

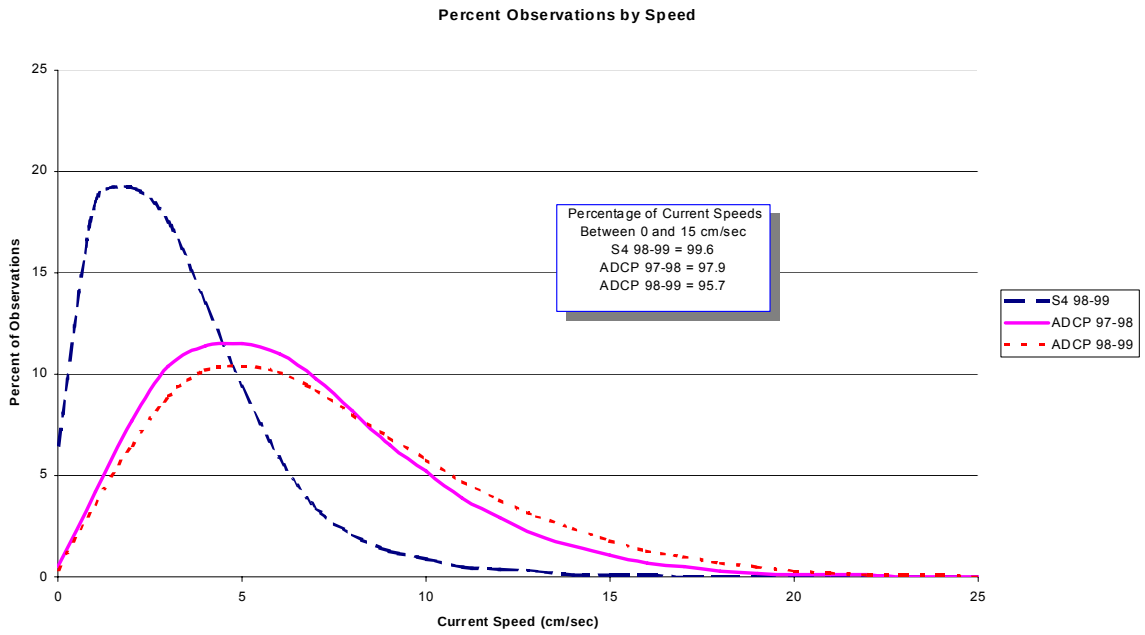

Figure 3-22. Frequency (percent) of measurements by current speed (cm/sec). ADCP data included measurements at all depth intervals.

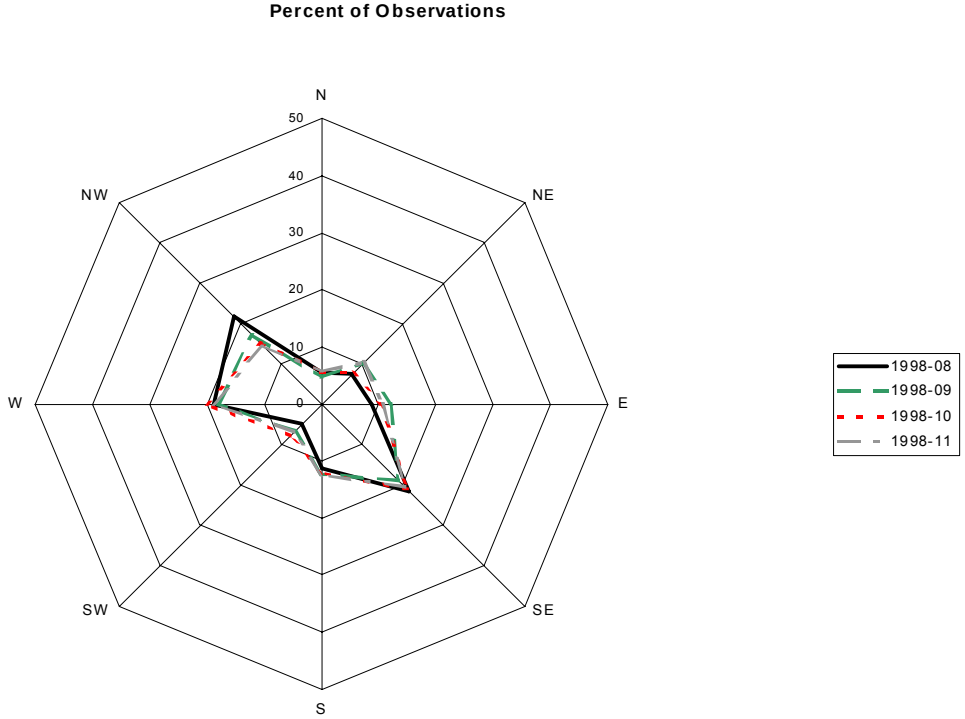

Figure 3-23. Typical frequency (percent) of measurements by current direction (8 cardinal points) by ADCP. Data depicted represent four months in 1998. Data included measurements at all depth intervals.

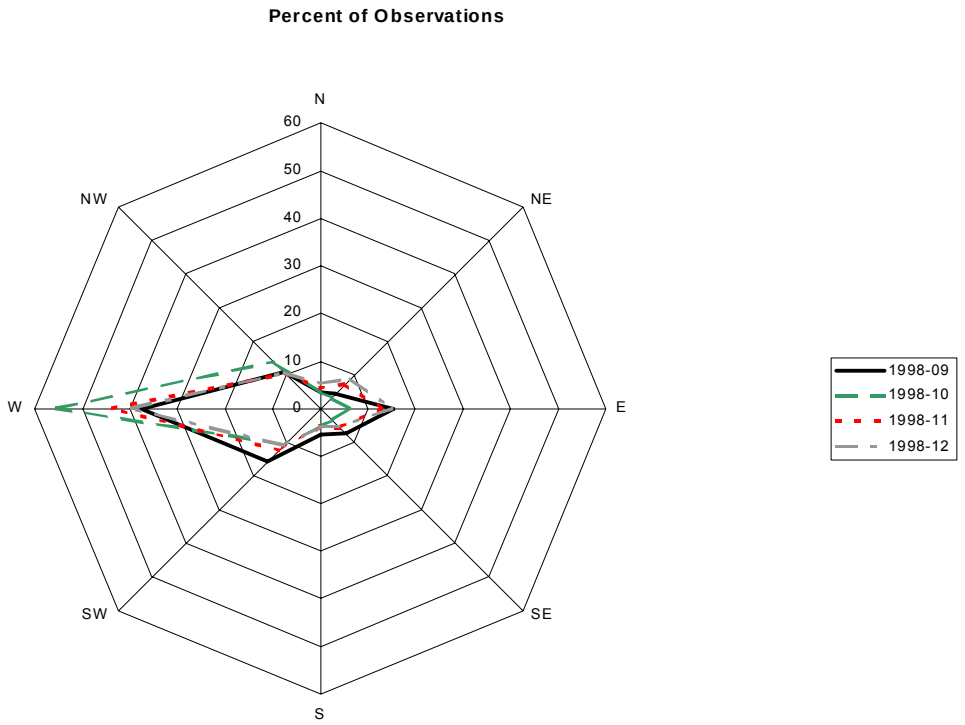

Figure 3-24. Typical frequency (percent) of measurements by current direction (8 cardinal points) by S4. Data depicted represent four months in 1998. Data included measurements at all depth intervals.

3.2.1.2. Measured concentrations of COCs in water

As discussed earlier in Section 2.4.2, release of PCBs from solid materials into the water column is expected to be very slow, but considered the only contaminant pathway through which contaminants are released from the ex-AGERHOLM and end up in the bottom sediments. The released PCBs are expected to be at measurable concentrations in their dissolved phase for a short time before some is transported away and diluted through advection and diffusion, while another fraction sorbs onto settling particulate matter and is deposited into the bottom sediment. Additionally, Section 2.4.2 explained that the water column is a very minor pathway for either ecological or human exposure. Consequently, the sampling plan was not designed to rigorously examine water concentrations of PCBs. Four replicate water column samples were obtained during the 1999 sampling period at Inner Ring Station 1-5/6/7 and four replicates at each of two Outer Ring stations (4-2, 4-5) utilizing a 6L Van Doran bottle activated remotely by the ROV. Samples were captured approximately one meter from the seafloor, recovered, and 2L sub-samples were shipped to ADL/FIT for PCB and TOC analysis. None of the 26 PCB congeners investigated were detected (MDL range 0.1 to 0.43 µg/L; see Appendix E). TOC reported by FIT ranged from 1.9 to 2.8 mg/L. Salinity measured at Station 1-5/6/7 averaged 33.6 parts per thousand.

3.2.1.3. Leaching Mechanisms of PCBs in Shipboard Solids

Laboratory leaching experiments were conducted during the preparation of this report and are to be reported separately (George et. al., in preparation). These leaching experiments were performed with representative shipboard materials, under varying temperature and pressure

conditions simulating different ocean depths. Some speculation about leaching mechanisms pertinent to the deep ocean environment of SINKEX is warranted here for purposes of understanding the possible impact the deep-ocean environment may have on solid materials that contain PCBs.

Possible effects of pressure on PCB leaching, are expected to be minor, but may include a) and/or b):

a) A direct effect on PCB solubility. A slight solubility decrease, if any, is expected. This is being verified empirically in current laboratory studies.

b) An effect during sinking if compressive forces inward change the seawater-accessible surface area of the shipboard solid such as loss of structural integrity during sinking or what might occur normally during leaching due to conditioning of the solid (e.g. degradation, dynamic wettability, etc.). This could also be related to localized inward pressure gradients within the solid matrix.

The dominant effect of depth on leaching was expected to be a reduction in solubility due to temperature. Preliminary leaching measurements have shown that this effect results in approximately an order of magnitude reduction in leach rate upon decreasing the temperature from 25°C to 4°C (George et. al., in preparation). These results were used for updating the source loading input to the recently updated modeling study (See Section 2.3.3.6). Empirical leach rates were used with estimated PCB concentrations in shipboard solids thought to be onboard ex-AGERHOLM (JJMA Report, 1998) to model the vessel PCB release during sinking and in the first 405 days subsequent to sinking.

A smaller, secondary effect due to a hydrostatic pressure increase was considered possible, though unlikely. This effect was expected to be chemical in nature, manifested by a solubility decrease, rather than a physical transport of bulk/liquid undissolved PCBs. Leaching experiments under simulated deep-ocean conditions (cold/high pressure, 4°C /300bar) exhibited behaviors that appear to be largely insensitive to an increase in hydrostatic pressure. The observed leach rates were more dynamic initially, but result in leach rates very similar to those observed under simulated cold shallow-ocean conditions (cold/low pressure, 4°C /1bar) (George et. al., in preparation). These observations support the conclusion that temperature has a much larger effect than hydrostatic pressure on leaching behavior.

Finally, it is useful to note that without a net pressure gradient from within the solid matrix outward, semi-solid or liquid (neat) PCBs (if such a phase is present in the shipboard solid) cannot be physically squeezed out of a shipboard solid matrix, even if it is very porous. Pressure-induced transport of PCBs is only possible if a net pressure gradient outward could exist (similar to squeezing a liquid-filled sponge between two hands, resulting in liquid transport out of the material in a direction normal to the applied opposing forces). In the deep ocean, this type of behavior does not occur because there is a uniform hydrostatic pressure from all directions inward, resulting in no net pressure gradients out of the material. [8]

[8] From classical oceanography and centuries of observation it is well known that hydrostatic pressure in the deep ocean provides a known amount of uniform compression on the outside of an object, meaning, under deep ocean conditions, only zero or inward net pressure gradients exist. Liquids are generally not very compressible, and if a structurally stable solid has pores that are completely PCB-filled (no voids), they would be compressed uniformly, leading to little/no effect on leaching (i.e., incompressible liquid/solid behavior). If a void-containing solid under

3.2.2. Deep Ocean Sediment Exposure

Evaluation of deep ocean sediment exposure proceeds in this report from PCBs, the primary COC, to the secondary COCs, which are PAHs and metals. Within the evaluation of PCBs, the main focus is on total PCBs, with supporting data (grouped with the secondary COCs) from PCB congeners, PCB Aroclors, and PCB core profiles. There is also a review of preliminary modeling estimates made in the 1994 study, which is then compared to the findings of this study.

3.2.2.1. Measured concentrations of PCBs in sediment

Total PCBs have historically been reported by a wide variety of calculations including summation of Aroclors, subsets of congeners, homologs and the sum of all 209 congeners. The EPA and USACE provide guidance for the determination of total PCBs in the publication "Evaluation of Dredge Material Proposed for Ocean Disposal, Testing Manual" commonly referred to as the Ocean Testing Manual or Green Book (USEPA/USACE, 1991). As discussed in Section 2.5.12, total PCBs in this study have been calculated in four ways (Table 3-15) to make the data broadly comparable to other studies. The least conservative measure (producing the lowest total concentration) is generated using Green Book criteria, since only 21 of 26 congeners measured in this study are included and non-detected congeners are assigned zero concentrations. The most conservative measure for total PCBs includes all 26 congeners measured during the SINKEX program and assigns non-detected congeners a concentration of ½ the sample specific method detection limit (MDL). These two methods are employed throughout the remaining discussion.

Figure 3-25 and Figure 3-26 graphically present total Green Book PCBs (21 congeners with non-detected values assigned zero concentrations) and Grand Total PCBs (26 congeners with non-detected values included as ½ the method detection limit). Figure 3-27 is a projection of total Green Book PCBs for the Inner Ring stations only. The highest Green Book total PCB concentration measured in sediment at the ex-AGERHOLM site was 9.8 µg/kg (Station 1-7). This value is 43% of the Effects Range-low (ERL) value of 22.7 µg/kg reported by Long et al., 1995. "Effects range" values for various chemical contaminants have been developed by the National Oceanographic and Atmospheric Administration's National Status and Trends Program (NOAA, NS&T) for use as informal interpretive tools in the absence of national sediment chemical criteria for estuarine environments for estuarine and marine environments. The ERL is the concentration of a chemical below which adverse biological effects are rarely realized. Alternatively, a higher benchmark,the Effects Range-median (ERM) is a chemical concentration, above which adverse biological effects frequently occur. There is some controversy as to the

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compression is structurally unstable over time, it could rearrange or lose its integrity, leading to non-uniform pressure gradients within the material (i.e., localized gradients from one region/pore to another). It is possible that small pressure gradients such as these can cause localized PCB movement or PCB transport inward within the solid material as the solid matrix equilibrates with pressure (e.g. flow from outer PCB-filled pores into inner voids). If a net flow of PCBs to an internal leaching surface occurs, it is conceivable that the observed rate of PCB leaching could be enhanced slightly. However, if there is a solid integrity failure, decrease in the seawater-accessible surface area is also possible, effectively blocking seawater from reaching a previously accessible surface, and resulting in a subsequent leach rate decrease. Depending on the extent of these opposing effects, a small net leach rate increase or decrease could result. Unfortunately, in the absence of solid-specific seawater accessible surface areas and/or knowledge of their dynamic behavior as a function of time and/or leaching scenario (properties), it is not possible to quantify these contributions, though they are likely to be minor. And, as a result, changes in seawater accessible surface area is considered an integral component of any solid-specific leaching behavior (i.e., any variation in surface area is considered part of the unique leaching behavior specific to each type of solid).

relationship between these benchmark values and observed toxicity; however this ER-L/M approach tends to be commonly applied in the California coastal region. Additionally, several other benchmarks have been used and are displayed for comparison purposes: TEL (Threshold Effects Level) and PEL (Probable Effects Level) from FDEP (1994), and the AET (Apparent Effects Threshold) from Barrick et al., 1988.

Table 3-15. Summary of total sediment PCBs.

Analyte	Ring	n	Mean (µg/kg)	Minimum (µg/kg)	Maximum (µg/kg)	Standard Error	CV
Total PCB (Green Book 21, ND=0)	1	10	3.90	0.49	9.80	1.13	91
	4	8	1.52	0.41	4.28	0.45	84
	ALL	18	2.84	0.41	9.80	0.70	104
Total PCB (Green Book 21, ND=0.5*MDL)	1	10	4.46	1.05	10.50	1.12	79
	4	8	1.82	0.69	4.60	0.45	70
	ALL	18	3.29	0.69	10.50	0.72	92
Total Congeners (26, ND=0)	1	10	4.88	0.62	13.55	1.41	91
	4	8	1.75	0.56	4.48	0.46	74
	ALL	18	3.48	0.56	13.55	0.87	106
Total Congeners (26, ND=0.5*MDL)	1	10	5.54	1.36	14.25	1.38	79
	4	8	2.07	0.87	4.85	0.46	63
	ALL	18	4.00	0.87	14.25	0.88	93

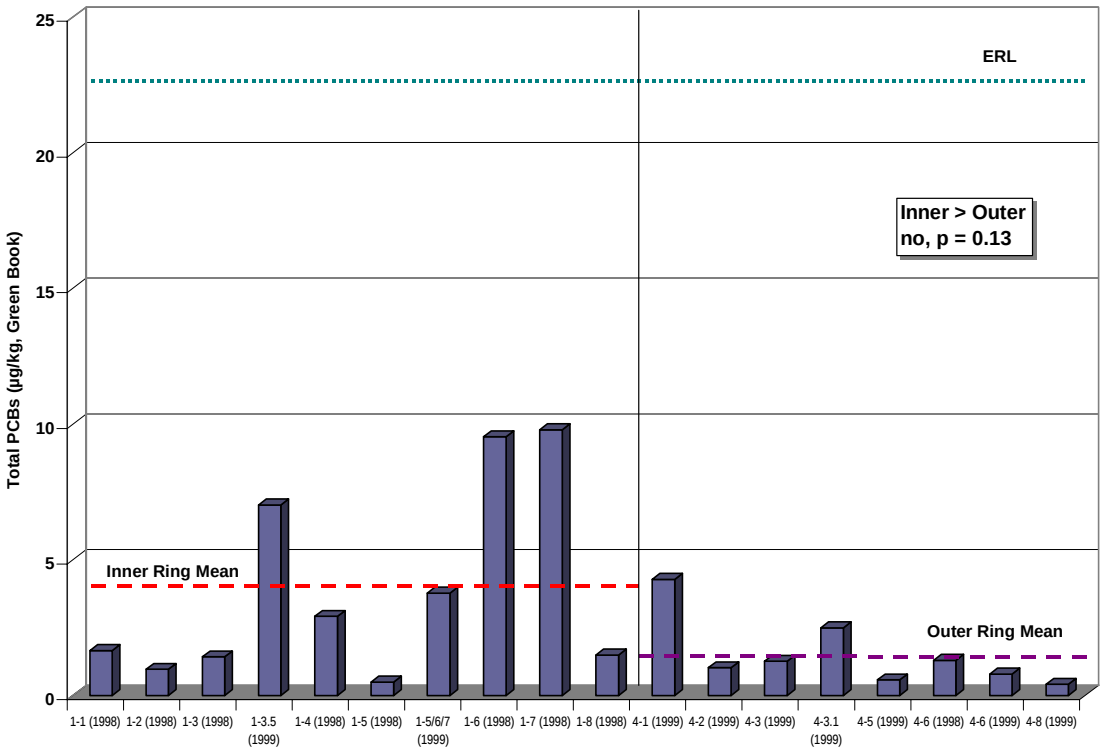


Figure 3-25. Total of 21 Green Book PCB Congeners measured at all SINKEX stations.

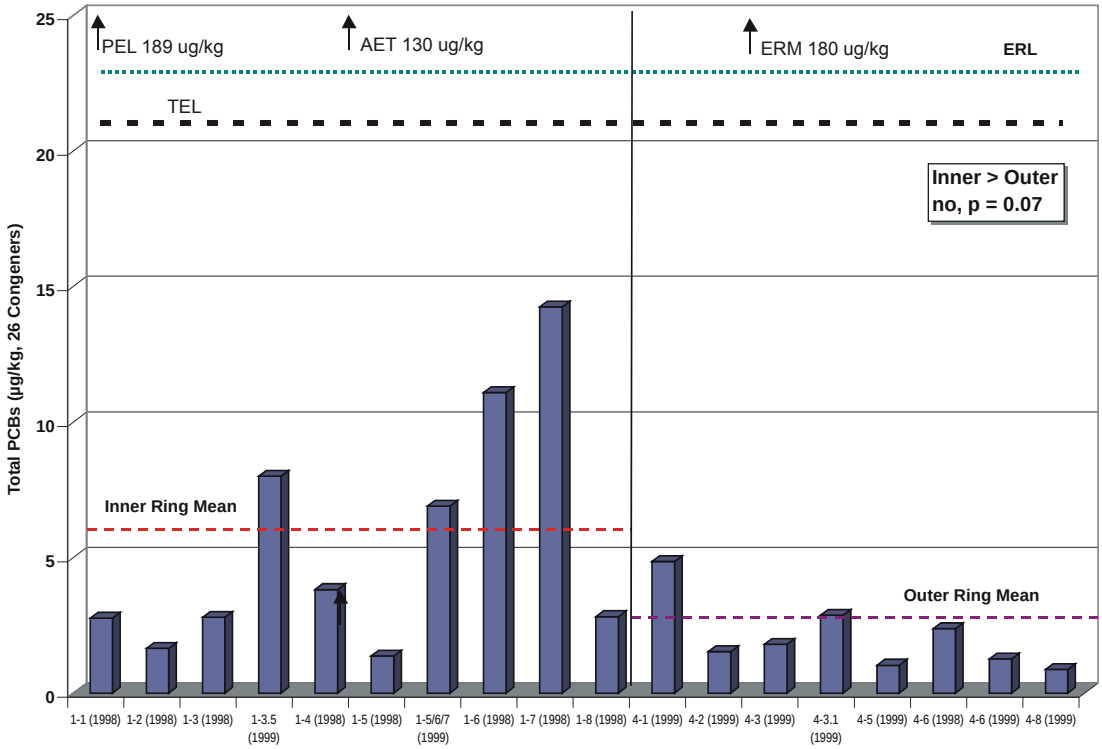


Figure 3-26. Total of 26 PCB Congeners measured at all SINKEX stations. One-half detection limit used when congener was not detected.

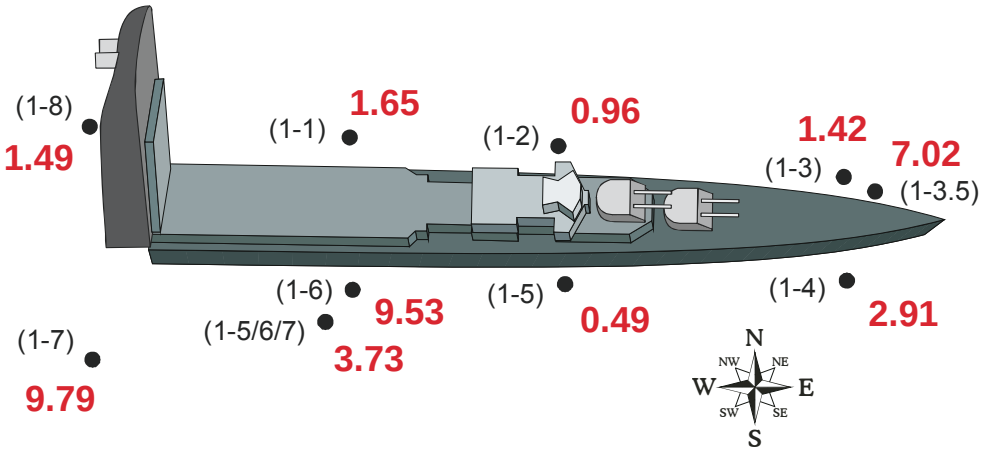


Figure 3-27. Ring 1 Total Green Book PCBs (µg/kg dry weight). Station numbers in parentheses.

The experimental design permitted a statistical comparison between near hulk samples (Inner Ring) and reference area samples (Outer Ring). Green Book and Grand Total (i.e., the 26 SINKEX congeners) PCB distributions were evaluated for normality using the Shapiro-Wilks test procedure prior to hypothesis testing. Non-normal distributions were subjected to three

normalization/standardization procedures (log transformation, ratio of total PCB to percent total organic carbon, and the log of the ratio of total PCB to percent total organic carbon). If the distribution remained "non-normal," a non-parametric hypothesis test was applied to the untransformed data. Table 3-16 presents the results of normality and Generalized Linear Model (GLM) hypothesis testing performed on both Green Book and Grand Total PCB sediment concentrations. A simple log transformation provided the best normal distribution for both types of PCB totals. Inter-ring comparisons of log normalized Green Book PCBs congeners did not disprove the null hypothesis and thus ex-AGERHOLM and reference areas were not significantly different at the 0.05 level (p=0.126). Similarly, inference testing of differences between Rings 1 and 4 of Grand Total 26 PCB congeners proved insignificant at the p=0.05 level. The highest value measured for Total PCBs, based on a summation of 26 congeners, was found at Inner Ring station 1-7 (14.3 µg/kg dry weight) and is within the 0.8 to 25 µg/kg background range reported by Kennish (1992) for ocean sediments untainted by a point source of anthropogenic inputs.

Table 3-16. Two-way comparison of SINKEX rings (1 and 4 and cruises 1998-1999). Values in bold provided best normalization of raw data (differences significant if p<0.05).

		GLM Hypothesis Test Results			Normality Test Results		
Total PCB Measurement	Transformation Standardization Method	Inter-Ring Comparison (different if ≤0.05)	Inter-Cruise Comparison (different if ≤0.05)	Ring-Cruise Interaction (important if ≤0.05)	Shapiro-Wilks Value	Probability of Non-Normal Distribution (non-normal if ≤0.05)	Distribution
Green Book PCBs (21)	**log**	**0.1259**	**0.5531**	**0.4299**	**0.9521**	**0.4588**	**Normal**
	logtoc	0.1428	0.0310	0.1210	0.9512	0.4434	Normal
	none	0.1411	0.5965	0.6846	0.7521	0.0003	Non-Normal
	toc	0.0048	0.0022	0.0109	0.7263	0.0002	Non-Normal
All Sinkex PCBs (26)	**log**	**0.0658**	**0.6857**	**0.2882**	**0.9518**	**0.4533**	**Normal**
	logtoc	0.0922	0.0213	0.0537	0.9267	0.1699	Normal
	none	0.1070	0.6729	0.5724	0.7704	0.0006	Non-Normal
	toc	0.0009	0.0005	0.0017	0.6920	0.0001	Non-Normal

Values in **bold** provide best normalization of raw data.

3.2.2.1.1. PCB Congener Concentrations

Table 3-17 summarizes sediment PCB congener concentrations for Ring 1, Ring 4 and both rings combined (All). Congeners not detected during analysis were assigned a value of one half the sample specific MDL for the statistical presentation (most conservative). Sample specific method detection limits were very low, varying by sample and congener and ranging from 0.024 µg/kg (congener 206) to 0.47 µg/kg (congener 170). A discussion of detection limits can be found in Section 2.5.6.4.

A total of 468 individual congener-station combinations (26 congeners × 18 stations) were examined (see Appendix F). Even considering the low levels of detection employed during this investigation, 142 measurements (30%) fell below sample specific MDLs. Six stations (1-5/6/7, 1-6, 1-7, 4-1, 4-3.1) had individual congener concentrations greater than 1.0 µg/kg, with the highest single congener value detected at station 1-6 (209 decachlorobiphenyl, 2.6 µg/kg). Across all sediment stations, only nine of the 26 congeners (congener number 8, 87, 49,52, 118, 153, 184, 206, 209) had concentrations greater than 1.0 µg/kg. Congeners 87 and 153 are moderate enzyme inducers (toxicity is inferred by mixed function oxidase enzyme induction)

prevalent in the environment. Congeners 49 and 52 are weak or non-mixed function oxidase inducers frequently found in the environment (McFarland and Clarke, 1989). The most toxic congeners, 77, 126, and 169 (those most likely to contribute to toxic effects as discussed in McFarland and Clarke, 1989) had sub-ppb mean values of 0.09 to 0.29 µg/kg in Inner Ring samples, and mean values of 0.06 to 0.13 µg/kg in Outer Ring samples.

Table 3-17. Summary of 26 PCB congener concentrations in mg/kg dry weight for Ring 1, Ring 4 and all stations combined. Non-detected congeners processed as ½ MDL. Congeners used in calculating total Green Book PCBs are highlighted.

Analyte	Ring	n	Mean (µg/kg)	Minimum (µg/kg)	Maximum (µg/kg)	Standard Error	CV
8 - 2,4'-Dichlorobiphenyl	1	10	0.21	0.02	1.6	0.15	227
	4	8	0.03	0.02	0.06	0	34
	ALL	18	0.13	0.02	1.6	0.09	274
18 - 2,2',5-Trichlorobiphenyl	1	10	0.12	0.02	0.17	0.01	33
	4	8	0.05	0.03	0.13	0.01	64
	ALL	18	0.09	0.02	0.17	0.01	56
28 - 2,4,4'-Trichlorobiphenyl	1	10	0.05	0	0.11	0.01	92
	4	8	0.04	0.01	0.04	0	33
	ALL	18	0.04	0	0.11	0.01	75
44 - 2,2',3,5'-Tetrachlorobiphenyl	1	10	0.15	0.02	0.87	0.08	171
	4	8	0.02	0.02	0.03	0	13
	ALL	18	0.09	0.02	0.87	0.05	214
49 - 2,2',4,5'-Tetrachlorobiphenyl	1	10	0.38	0.08	1.7	0.15	130
	4	8	0.08	0.03	0.36	0.04	149
	ALL	18	0.24	0.03	1.7	0.09	162
52 - 2,2',5,5'-Tetrachlorobiphenyl	1	10	0.22	0.03	1.4	0.14	191
	4	8	0.02	0.01	0.05	0	49
	ALL	18	0.13	0.01	1.4	0.08	243
66 - 2,3',4,4'-Tetrachlorobiphenyl	1	10	0.16	0.02	0.6	0.06	126
	4	8	0.07	0.02	0.15	0.02	69
	ALL	18	0.12	0.02	0.6	0.04	130
77 - 3,3',4,4'-Tetrachlorobiphenyl	1	10	0.29	0.03	0.99	0.12	124
	4	8	0.13	0.02	0.78	0.09	202
	ALL	18	0.22	0.02	0.99	0.08	147
87 - 2,2',3,4,5'-Pentachlorobiphenyl	1	10	0.2	0.02	1.2	0.12	184
	4	8	0.05	0.02	0.09	0.01	39
	ALL	18	0.14	0.02	1.2	0.07	207
101 - 2,2',4,5,5'-Pentachlorobiphenyl	1	10	0.17	0.03	0.8	0.07	134
	4	8	0.06	0.01	0.1	0.01	49
	ALL	18	0.12	0.01	0.8	0.04	145
105 - 2,3,3',4,4'-Pentachlorobiphenyl	1	10	0.13	0.03	0.57	0.05	123
	4	8	0.03	0.01	0.05	0	39
	ALL	18	0.09	0.01	0.57	0.03	149
118 - 2,3',4,4',5-Pentachlorobiphenyl	1	10	0.4	0.04	1.9	0.18	141
	4	8	0.09	0.04	0.13	0.01	39
	ALL	18	0.26	0.04	1.9	0.11	169
126 - 3,3',4,4',5-Pentachlorobiphenyl	1	10	0.12	0.04	0.31	0.02	62
	4	8	0.06	0.04	0.11	0.01	33
	ALL	18	0.09	0.04	0.31	0.02	69

Analyte	Ring	n	Mean (μg/kg)	Minimum (μg/kg)	Maximum (μg/kg)	Standard Error	CV
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	1	10	0.06	0.01	0.21	0.02	94
	4	8	0.03	0.01	0.05	0.01	55
	ALL	18	0.05	0.01	0.21	0.01	96
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	1	10	0.37	0.04	1.4	0.15	125
	4	8	0.08	0.03	0.11	0.01	36
	ALL	18	0.24	0.03	1.4	0.09	154
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	1	10	0.48	0.06	1.3	0.13	88
	4	8	0.14	0.07	0.21	0.02	36
	ALL	18	0.33	0.06	1.3	0.08	108
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	1	10	0.11	0.05	0.23	0.01	43
	4	8	0.02	0.01	0.05	0.01	78
	ALL	18	0.07	0.01	0.23	0.01	82
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	1	10	0.09	0.01	0.13	0.01	35
	4	8	0.08	0.04	0.12	0.01	32
	ALL	18	0.08	0.01	0.13	0.01	34
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	1	10	0.09	0	0.2	0.03	92
	4	8	0.07	0.01	0.1	0.01	49
	ALL	18	0.08	0	0.2	0.01	79
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	1	10	0.23	0.04	0.79	0.08	102
	4	8	0.09	0.02	0.25	0.03	93
	ALL	18	0.17	0.02	0.79	0.05	117
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	1	10	0.11	0.01	0.25	0.03	77
	4	8	0.04	0.01	0.07	0.01	43
	ALL	18	0.08	0.01	0.25	0.02	92
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	1	10	0.28	0.05	1.7	0.16	179
	4	8	0.06	0.02	0.22	0.02	105
	ALL	18	0.18	0.02	1.7	0.09	210
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	1	10	0.21	0.02	0.72	0.07	103
	4	8	0.13	0.02	0.49	0.06	133
	ALL	18	0.17	0.02	0.72	0.05	113
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	1	10	0.05	0.01	0.11	0.01	62
	4	8	0.02	0.01	0.04	0	32
	ALL	18	0.03	0.01	0.11	0.01	70
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	1	10	0.32	0.02	1.5	0.15	147
	4	8	0.47	0.01	2.3	0.29	172
	ALL	18	0.38	0.01	2.3	0.15	162
209 - Decachlorobiphenyl	1	10	0.52	0.02	2.6	0.27	160
	4	8	0.12	0.01	0.47	0.05	129
	ALL	18	0.34	0.01	2.6	0.15	189

3.2.2.1.2. Aroclor PCB Concentrations

Aroclors (trade name) were manufactured in the United States until 1978 by bulk chlorination, resulting in differing distributions of individual congeners, creating oils (Aroclor 1016, 1221, 1232, 1248), viscous liquids (Aroclor 1254), sticky resins (Aroclor 1260 and 1262) and white powders (Aroclor 1268, 1270). The type of Aroclor is identified by the percentage of chlorine in a mixture of congeners. For example, Aroclors 1221 and 1268 are manufactured by chlorinating biphenyl to a final chlorine content of 21 and 68 percent, respectively (Kennish, 1992). Much of

the historic PCB literature lacks direct reporting of individual congeners and, as such, eight Aroclors (1221, 1232, 1242, 1248, 1254, 1260, 1262, 1268) were measured in this study to provide some level of historical perspective. Due to the extremely low concentrations of PCBs, only four Aroclors patterns could be identified in sediments at the ex-AGERHOLM site (1254, 1260, 1262 and 1268). Ring summary statistics are provided in Table 3-18 and complete data can be found Appendix F. Figure 3-28 presents Aroclor concentrations (averaged by station). The highest average Aroclor concentration measured was Aroclor 1254 (31.3 µg/kg dry weight) and was only found at Station 1-7. Station 1-7 also had the highest concentrations of Aroclors 1260 and 1262 (20.0 and 8.1 µg/kg dry weight, respectively), and total PCBs. Aroclor 1262 was identified at 1-4 (6.4 µg/kg dry weight). Aroclor 1268 was found at stations 1-6, 1-5/6/7, 1-3.5, 4 1 and 4-3.1 (concentrations 15, 1.7, 3.3, 8.2, 3.8 µg/kg dry weight, respectively).

The minimal number of uniquely identified Aroclors precludes rigorous statistical comparisons, however, individual Aroclors are reported in concentrations greater than total PCBs, based on congener analysis presented in Section 3.2.2.1.1 and are mostly likely the result of differing analytical methods, with Aroclor analysis begin considerably less precise than congener specific identification. Stations 1-6 and 1-7 are the only stations where more than a single Aroclor was identified (Figure 3-28).

It should also be noted that SINKEX sediment Aroclors were estimated using different analytical techniques, gas chromatography electron capture device (GC-ECD) in the 1998 survey, and gas chromatography/mass spectroscopy selective ion monitoring (GC-SIM) methods in the 1999 survey. The reporting of Aroclor data reported in this document includes measurements from both analytical methods and does not discriminate between the methods. Where duplicate measurements made, the resulting values were averaged and reported. Inter-method comparisons will be discussed in a subsequent report.

Table 3-18. Summary of sediment Aroclor concentrations (µg/kg dry weight).

Analyte	Ring	n	Mean (µg/kg dry weight)	Minimum (µg/kg dry weight)	Maximum (µg/kg dry weight)	Standard Error	CV (%)
Aroclor 1221	1,4	none found					
Aroclor 1232	1,4	none found					
Aroclor 1242	1,4	none found					
Aroclor 1248	1,4	none found					
Aroclor 1254	1	4	24.0	1.9	35.0	7.5	62.9
	4	none found					
Aroclor 1260	1	2	20.0	15.0	25.0	5.0	35.4
	4	none found					
Aroclor 1262	1	2	7.3	6.4	8.1	0.9	16.6
	4	none found					
Aroclor 1268	1	3	6.7	1.7	15.0	4.2	108.9
Aroclor 1268	4	2	6.0	3.8	8.2	2.2	51.9

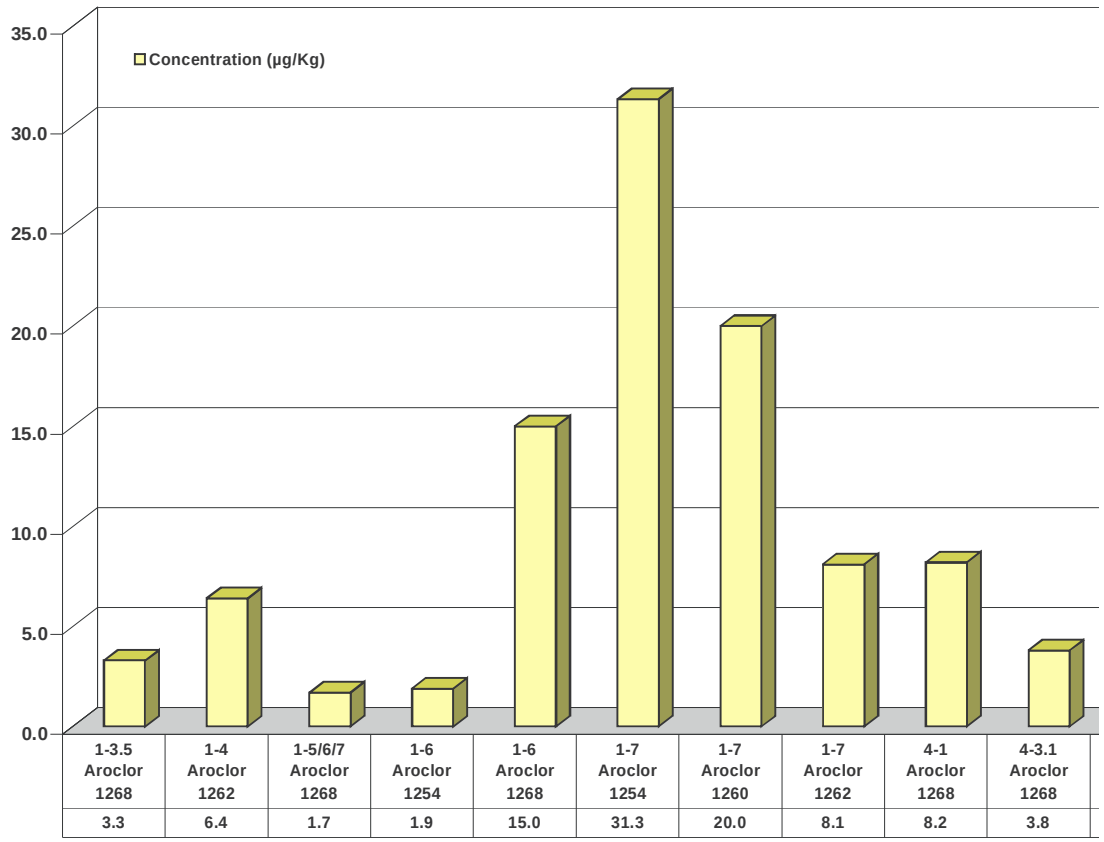


Figure 3-28. Average Aroclor sediment concentrations. Data presented only for detected Aroclors.

3.2.2.1.3. Core Sample PCB Concentrations

Sediment core samples were processed from eight locations to assess vertical distribution of PCB compounds in subsurface sediments (see Appendix G). Five samples were analyzed from Ring 1 and three samples from Ring 4. Cores were collected in 1998 (four from Ring 1) and 1999 (one from Ring 1, three from Ring 4). Core tubes were frozen in a vertical position and maintained vertically during transport to the laboratory. Cores were sectioned to a depth of approximately 10 cm, and varied in slice thickness for samples taken between 1998 and 1999. One-centimeter slices were taken in 1998, whereas thickness ranged from 0.5 cm to 1 cm in 1999. It has already been suggested that the integrity of the geo-chronological record is questionable on the inner ring, most likely due to mixing throughout the sediment depths when the hulk hit the bottom.

Vertical profiles of total PCBs are presented as a sum of 26 SINKEX congeners for the eight cores (Figure 3-29 and Figure 3-30). Total PCB concentrations for the top 3 cm of coincident surface grab samples and the average concentrations from the upper 3 cm of each core are also displayed. Deeper core segments (i.e., further from the sediment surface) had fewer detectable congeners. To avoid overstating total PCB estimates for these very old sediments, congeners not detected during analysis were assigned a value of zero for use in summation and subsequent data analysis and presentation. This was done. Sample specific method detection limits were typically very low during the program, varying by sample and congener and ranging from 0.017 µg/kg (congener 206) to 4.1 µg/kg (congeners 87 and 156). A discussion of detection limits can be found in Section 2.5.6.4.

A total of 2,522 individual congener-station-depth combinations were examined (26 congeners × 8 cores × average of ~12 depth sections per core). Sums of the 26 congeners ranged from 1,860 µg/kg at Station 1-5/6/7 in 1999 to non-detected (three sections from Station 1-2 and five sections from Station 1 6). Cores from only three stations (1-5/6/7, 1-7 and 4-2) had individual core congener concentrations greater than 1.0 µg/kg, with the highest single value detected at station 1-5/6/7 (138-2,2′,3,4,4′,5′-Hexachlorobiphenyl, 350 µg/kg).

Two cores (1-5/6/7 and 1-7) contained total PCBs concentrations one to three orders of magnitude higher than the other six cores and 0-3 cm (depth) surface sediment composites taken from within the same box-cores (i.e., those from the same sampling locations). Only eight of the 96 sections had total PCB concentrations at or above 10 µg/kg. These eight sections originate from Inner Ring Stations 1-5/6/7 and 1-7. Stations 1-5/6/7 and 1-7 are located near the starboard stern area of the ship and are close to a major break in the hull (Figure 2-18). This extremely large separation in the hulk structure provides an increased potential for exposure to PCB materials as well as other contaminants.

It is interesting to note that within both Inner and Outer Ring samples, concentrations below 2.5 cm typically exceed concentrations in the upper 0-2 cm of the core and in surface sediment samples (0-3 cm composite). Levels greater than 5 µg/kg were measured at multiple depths within the two higher concentration cores; eight sections from Station 1-5/6/7 and four from Station 1-7. It is reasonable to speculate that advective transport and subsequent reburial may have been the cause for these unusual results.

There are no concurrent age-dating data available from Stations 1-5/6/7 and 1-7, hence a quantitative assessment of disturbance within the sediment is not known. There is evidence, both quantitative (age dating) and qualitative (observations from videotape), that the sinking of the ex-AGERHOLM perturbed the sediments in the near vicinity of the ship. While this historic surface disturbance is very evident, the precise mechanism for mixing of PCBs within the top five to eight centimeters suggested by the data is not known. Age dating results with the highest degree of confidence describe settling rates ranging from 0.03-0.04 cm/year placing an age of material at one cm and below at greater than 25 years, and therefore not subject to direct influence from the ex-AGERHOLM. It is possible that the top layer of sediment adjacent to the impact site was blown off in some areas by the sinking ship's hydraulic wake, rendering near hulk age dating analyses ineffective. This appears to be the case at Stations 1-2 and 1-8. Alternatively, PCBs-ISM may also have dislodged from the ship and reached the sediment surface faster than the sediment suspended during the sinking. Small pieces of falling PCBs-ISM could have been covered by "older" sediments, thus creating an apparent age-dating discontinuity. Nevertheless, these possibilities do not explain the anomalous spikes of PCBs observed at the 7 and 10 cm depths (i.e., older than first use of PCBs in the world) on the Outer Ring at Station 4-2 (Figure 3-30). One explanation is that there was sample contamination between sample collection and analysis. There are a couple of possible explanations for the horizontal transport from ship to reference site, albeit speculative without data. One would be that a piece of PCBs-ISM that broke off the floating or sinking ship, finally landing out on the reference ring (debris was observed some distance from the hulk). Another explanation is sediment transport caused by occasional, strong, bottom currents.

Bioturbation may explain the presence of the higher concentrations of PCBs in sediment at depths that predate wide scale commercial production of these contaminants. However, benthic infaunal analysis indicates a minimally active deep burrowing community and does not appear to

support bioturbation as a primary mechanism for sediment mixing. The minimal extent of bioturbation in the area of the ex-AGERHOLM is confirmed by consistent age dating transitions at stations 4.3 and 4.3.1 (Section 3.1.2.2.2).

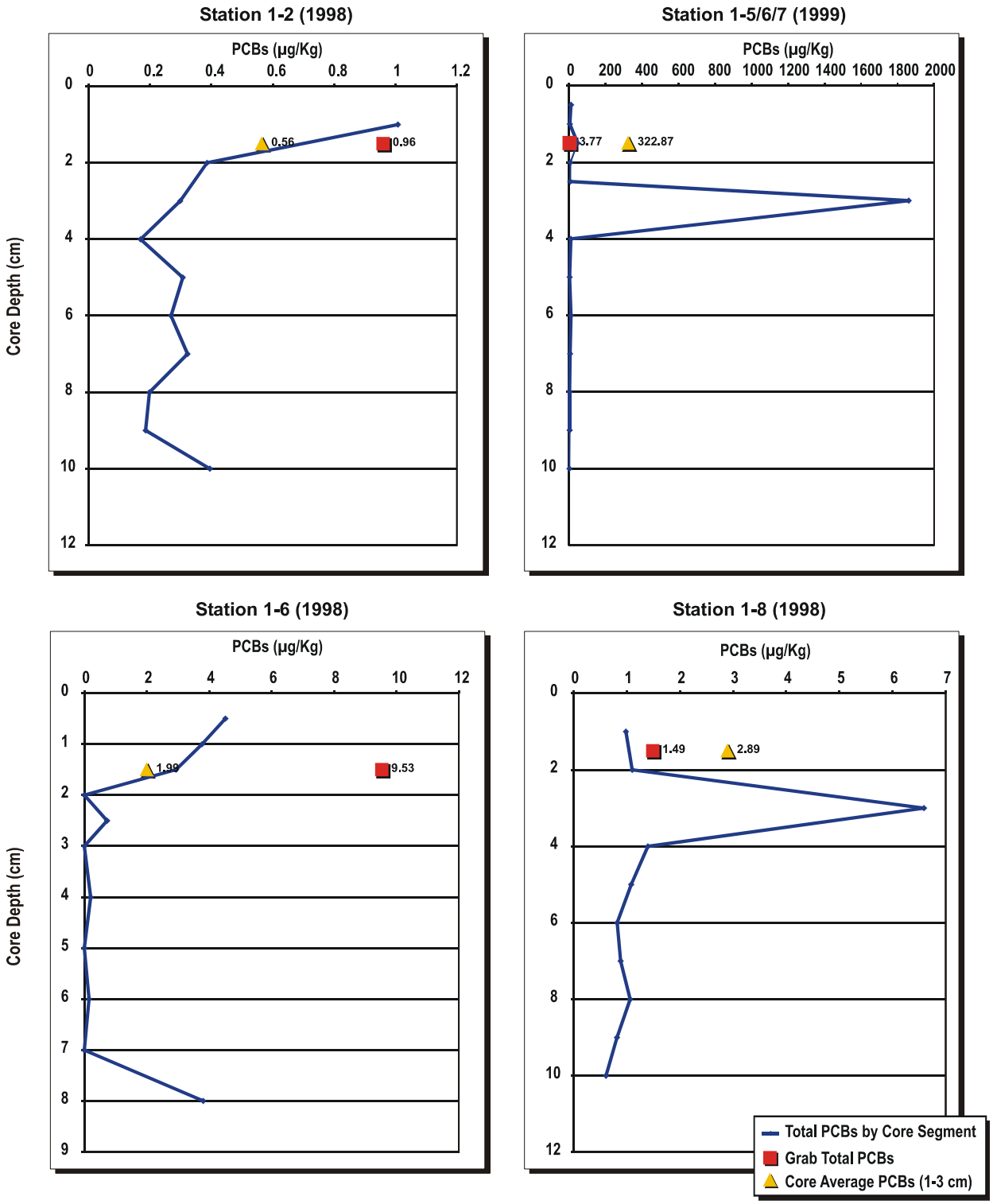


Figure 3-29. Vertical distribution of PCBs at Stations 1-2, 1-5/6/7, 1-6 and 1-8.

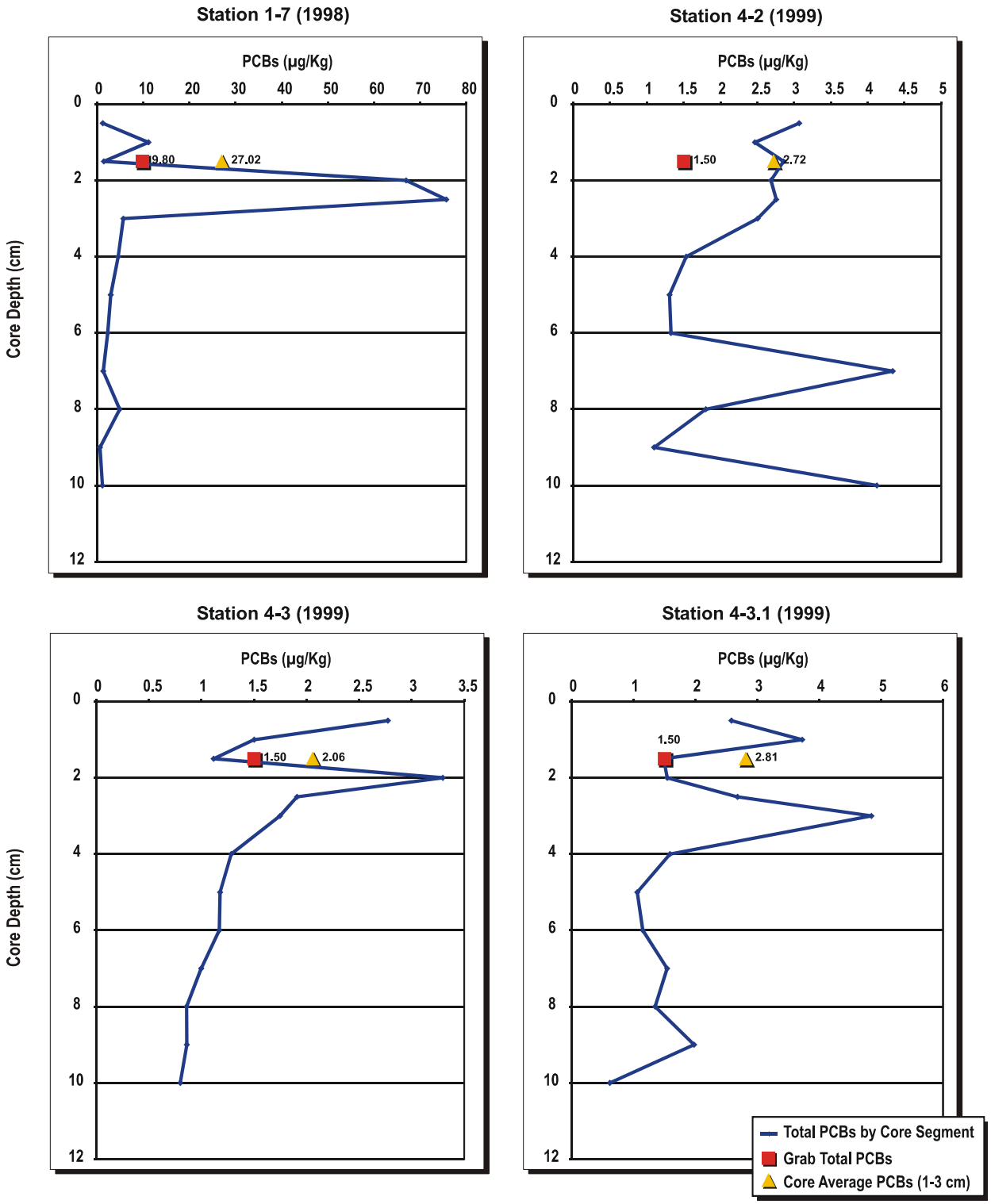


Figure 3-30. Vertical distribution of PCBs at Stations 1-7, 4-2, 4-3 and 4.3.1.

In summary, the origins of high concentrations of PCBs found in lower layers of Stations 1-5/6/7 and 1-7 are not known. It is unclear how the two high spikes of contamination at 2-3 cm below the surface, and several lesser spikes below 5 cm, could have originated from the ex-

AGERHOLM, except as speculated above. Some kind of sediment perturbation is suggested, but cannot be explained by the study results.

3.2.2.1.4. Sediment PCB Summary Comments

Total PCB concentrations did not significantly differ between Inner and Outer Rings, and thus the established null hypothesis could not be disproved. To further evaluate sediment chemical evaluations, concentrations were compared to several guideline concentrations used in various sediment quality programs (Table 3-19). These were the ER-L (Effects Range-Low) and ER-M (Effects Range- Median) from Long and Morgan (1991) and Long et al. (1995), the TEL (Threshold Effects Level) and PEL (Probable Effects Level) from FDEP (1994), and the AET (Apparent Effects Threshold) from Barrick et al., 1988. The potential impact to sediment biota from possible toxic effects of chemicals was first evaluated by comparing Inner Ring sediment concentrations to the Outer Ring stations, where potential impact was assumed to be insignificant. If the comparison (GLM or Kruskal-Wallis analysis) indicated no significant difference between the Inner Ring and Outer Ring stations, then possible impact was considered insignificant. If the difference was statistically different ($p<0.05$), then the Inner Ring concentrations were compared to the most common low (ER-L) and intermediate (ER-M) effects levels. Several of the less common guidelines (TEL, PEL and AET) are also considered for additional points of comparison, but are not used in the decision matrix.

Table 3-19. Sediment quality benchmark values (µg/kg dry weight).

Chemical Name	Threshold Effects Level (TEL)	Effects Range Low (ERL)	Effects Range Median (ERM)	Probable Effects Level (PEL)	Apparent Effects Threshold (AET)
Total PCB	21.55	22.7	180	188.79	130
Total PAH	1684.06	4022	44792	16770.4	N/A
Cadmium	676	1200	9600	4210	3000
Chromium	52300	81000	370000	160000	62000
Copper	18700	34000	270000	108000	390000
Lead	30240	46700	218000	112180	400000
Mercury	130	150	710	696	410
Nickel	15900	20900	51600	42800	110000
Silver	730	1000	3700	1770	3100
Zinc	124000	150000	410000	271000	410000

The concentration of the sum of all 26 PCB congeners measured in this study averaged 4.0 µg/kg, with a maximum of 14.3 µg/kg. Average Ring 1 and Ring 4 concentrations are below the ER-L of 22.7 µg/kg PCB (Long et al., 1995). Both the mean and maximum values are below the consensus based threshold effect concentration (TEC) of 48 µg/kg total PCB (dry wt.) developed by MacDonald et al. (2000) for marine and estuarine sediments. These TECs represent levels below which adverse effects are unlikely to occur, the lowest guideline used in the consensus estimates.

3.2.2.2. Comparison Between Field Data and Modeling Estimates

Figure 3-31 shows the sediment PCB concentration predictions from the modeling described previously (2.3.3.6), with an overlay of the actual PCB measurements made from samples collected at the reference stations and inner ring. Because the field measurements inherently

contain a historical accumulation of background PCBs, the concentrations for both ship and reference site are at least two orders of magnitude greater than their respective predicted levels, based on a PCB background of zero. If the model is accurate (it cannot be verified without performing a cost-prohibitive, prospective risk assessment), it is clear that the incremental sediment PCB loading from the AGERHOLM (i.e., that loading which is in addition to the background or reference condition/site) is insignificant relative to the background levels of PCBs found in sediments from both sites.

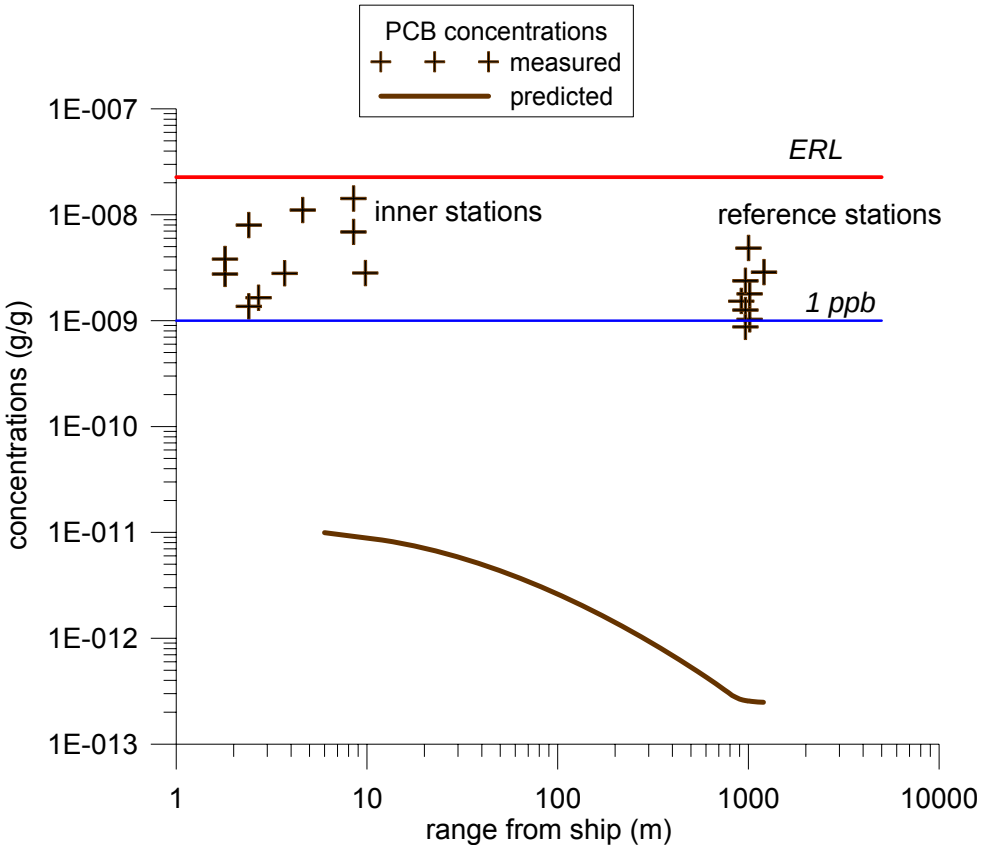


Figure 3-31. Measured and predicted total PCB concentrations in sediment.

3.2.2.3. Measured concentrations of PAHs in sediment

In general, total PAH concentrations were low and somewhat uniformly distributed between stations, rings and years (see Appendix H). Summary statistics for total PAH based on 41 compounds (total PAH41) and for the 16 EPA priority pollutant PAHs (total PAH16) are shown in Table 3-20. Total PAH41 ranged from 8.8 to 303.9 µg/kg for all samples. Results were extremely low (<50 µg/kg total PAH41) for all stations except 1-7, which showed inputs of weathered diesel and at the maximum concentration of 303.9 µg/kg (7.7% of the ERL value of 4,022 µg/kg). The spatial distribution of total PAH is shown in Figure 3-32 and Figure 3-33.

Inference testing was performed on the larger PAH set (PAH41), and on petrogenic and pyrogenic groups (discussed in Section 3.2.2.3.1) to determine potential sources. There were no significant differences (i.e., $p>0.05$) in total PAH41 between Ring 1 and Ring 4. Inference testing using a 2-way analysis of variance (ANOVA, under the GLM model) was performed using log-transformed and log (TOC) transformed data, as well as untransformed data. In

general, PAH data were log-normally distributed, and met test assumptions with normal distributions once transformed. Results from inference testing are summarized in Table 3-21.

In addition to inference testing, Pearson correlations between PAH and physical characteristics commonly associated with PAH were performed. Again, no statistically meaningful relationships were found for total PAH41 and TOC, calcium carbonate ($CaCO_3$) or grain size (percent fines, gravel, sand).

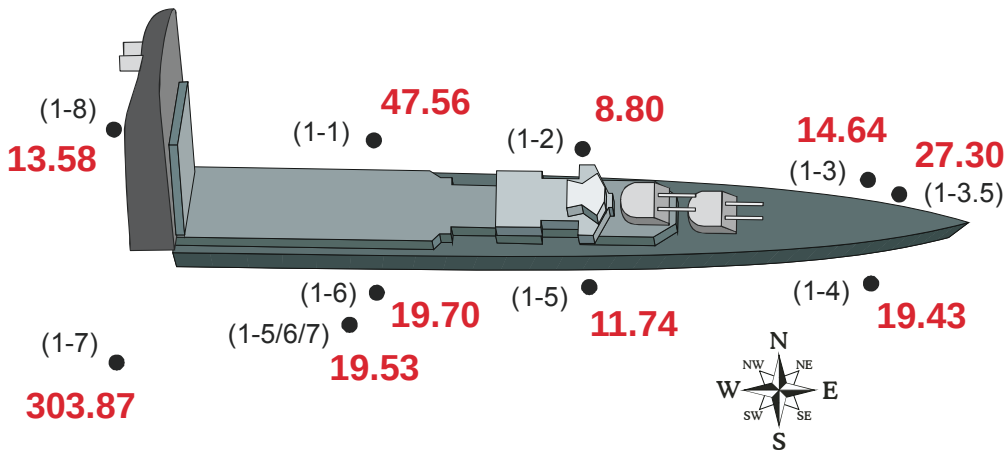


Figure 3-32. Ring 1 total PAH (µg/kg dry weight). Station numbers in parentheses.

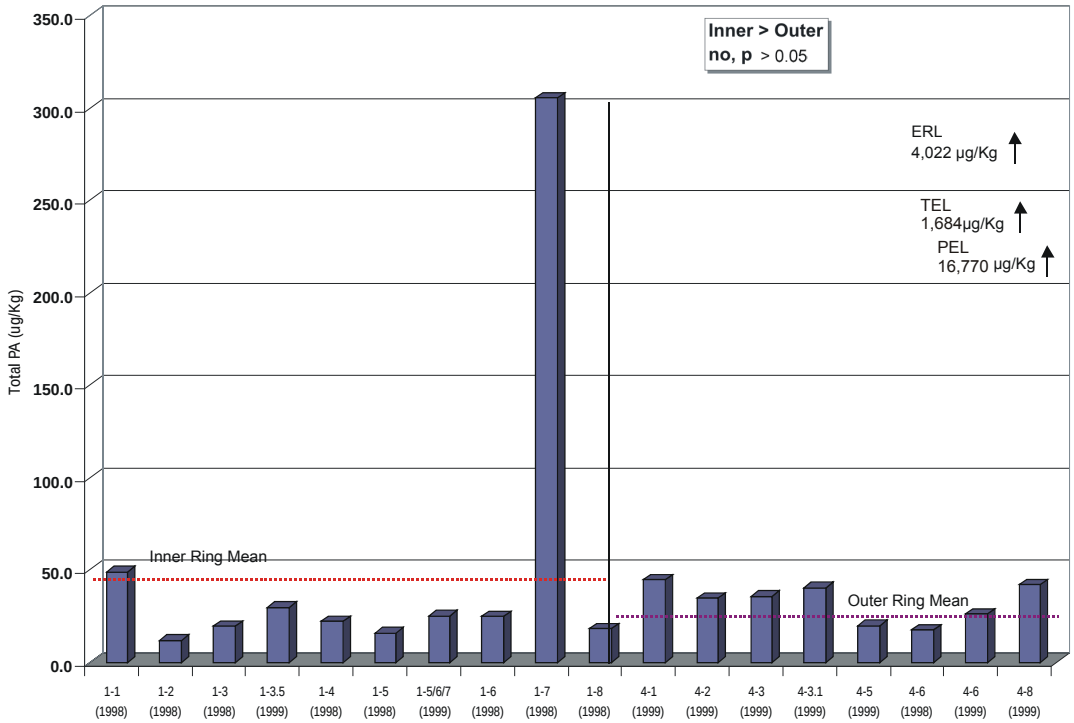


Figure 3-33. Total PAH sediment concentrations (µg/kg dry weight).

3.2.2.3.1. Source identification of PAH

In general, PAHs measured in SINKEX sediments are divided into two general categories - petrogenic and pyrogenic - based on their origin of formation. Petrogenic PAHs are derived from low temperature/high pressure processes occurring over geologic time frames. Petrogenic PAHs are found in fossil fuels, such as petroleum, petroleum products and coal. Pyrogenic PAHs are formed under medium to high temperature processes, such as combustion of fossil fuels or other carbon sources (e.g., forest fires). There are other origins of PAH, including perylene, which is naturally occurring or biogenic.

Predominant petrogenic PAHs vary depending on the nature and type of petroleum product, but, typically include naphthalene, C1-C4 naphthalenes, acenaphthene, acenaphthylene, biphenyl, fluorene, C1-C3 fluorenes, phenanthrene, C1-C4 phenanthrenes, dibenzothiophene, C1-C4 dibenzothiophenes, C1-C3 fluoranthenes/pyrenes, and the C1-C4 chrysenes and generally lower levels of the 5- and 6-ring PAHS.

Pyrogenic PAHs distributions typically include fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[e]pyrene, benzo[a]pyrene, indeno[1,2,3,-c,d]pyrene, dibenz[a,h]anthracene and benzo[g,h,i]perylene, but may also include parent 2- and 3-ring PAHs (e.g., naphthalene and phenanthrene).

PAH distribution plots for SINKEX sediment samples were evaluated to determine the potential sources of PAHs present in these samples. Representative plots of predominately pyrogenic and petrogenic PAHs are shown in Figure 3-34 and Figure 3-35, respectively. Table 3-22 summarizes total PAH information for each station by total concentration and probable source.

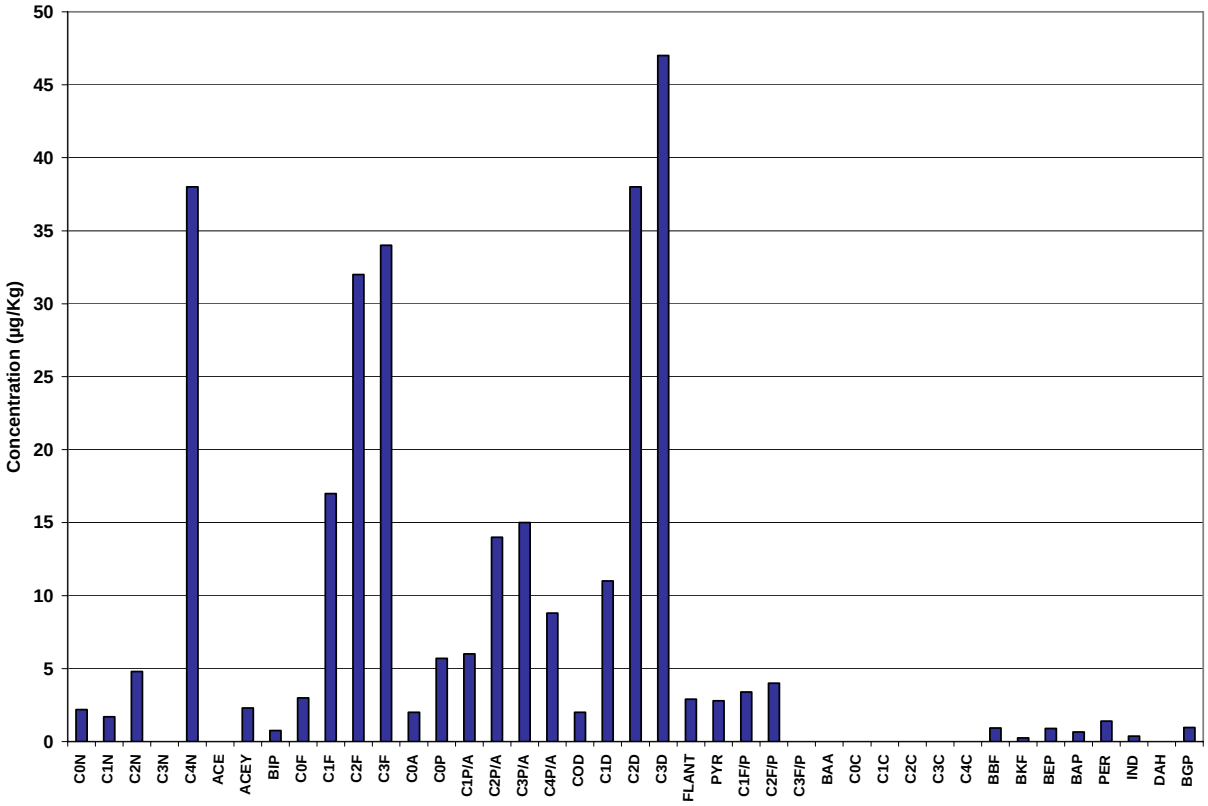


Figure 3-34. Station 1-7 (1998) - Representative of petrogenic (weathered diesel) input.

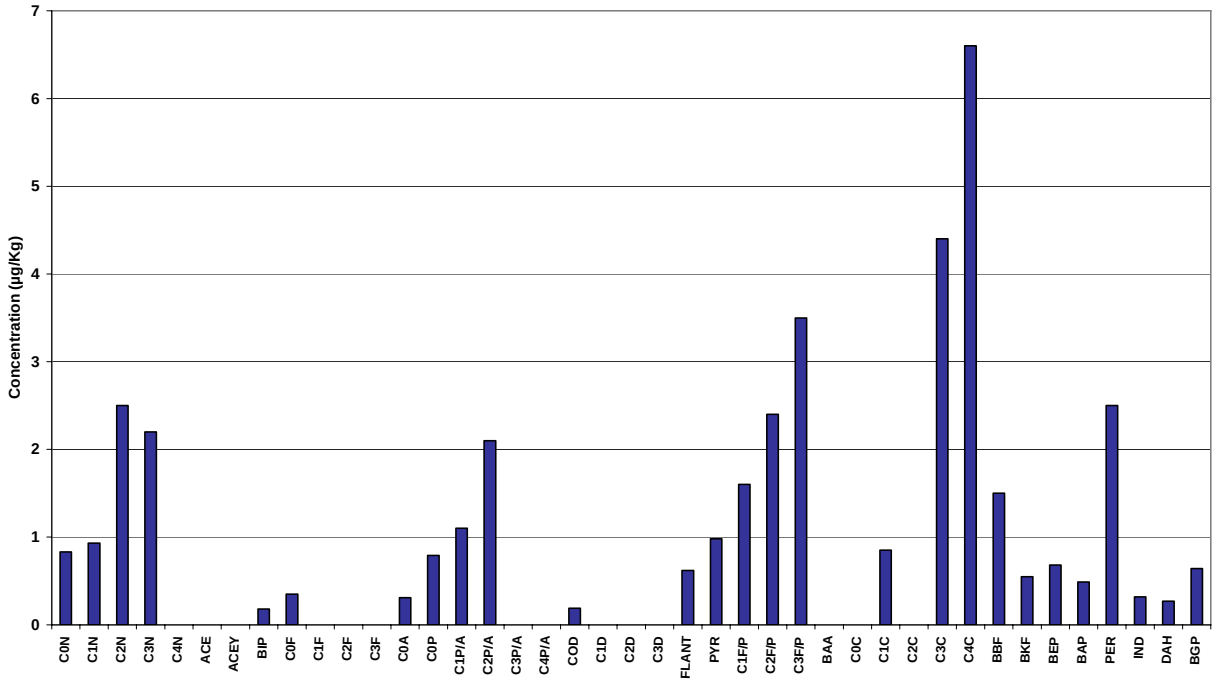


Figure 3-35. Station 4-8 (1999) - Representative of pyrogenic PAHs.

All sediment samples contained 4-, 5-, and 6-ring pyrogenic PAHs - benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[e]pyrene, benzo[a]pyrene, indeno[1,2,3,-c,d]pyrene, dibenz[a,h]anthracene and benzo[g,h,i]perylene. The most likely source of contamination is from deposition of atmospheric dust or particles containing combustion-related PAHs. Nonetheless, the samples can be further separated into three sub-groupings based on overall PAH distributions.

Group 1 - The sediment PAH distribution plots for Stations 1-2, 1-3, 1-4, 1-5, 1-6, 1-8, and 4-6 have similar distributions. These plots show the presence of the pyrogenic PAHs from atmospheric deposition and trace concentrations of several common laboratory contaminants (e.g., naphthalene, C1-naphthalene).

Group 2 - The sediment PAH distribution plots for Stations 1-5/6/7, 4-1, 4-2, 4-3, 4-3.1, 4-5 and 4-6 have similar distributions that vary slightly from the above mentioned sample group. In addition to the pyrogenic PAHs from atmospheric deposition and common laboratory contaminants, these samples also contain trace levels of a petroleum product as indicated by the presence of additional petrogenic PAHs (e.g., C2-phenanthrene, C3-phananthrene, C1-fluoranthenes/pyrenes, C2-fluoranthenes/pyrenes). Because petrogenic PAH distributions are consistent between samples, which were collected during the same survey, these samples may have been contaminated with low-level PAHs during sample collection and/or handling.

Group 3 - Four sediment samples had unique PAH distribution plots. Station 1-1 contains a trace concentration of a mid-range fuel with mid-level sulfur content (not atmospheric) and combustion related PAHs from atmospheric deposition. Station 1-3.5 contains trace amounts of a diesel-range fuel with low sulfur content and combustion related PAHs from atmospheric deposition. Station 1-7 contains trace levels of weathered diesel-range fuel with high sulfur content (not atmospheric) and combustion related PAHs from atmospheric deposition; and Station 4-8 contains trace concentrations of a weathered heavy fuel and combustion related PAH

content from atmospheric deposition. Although concentrations in these samples are elevated relative to other SINKEX samples, they are too low (i.e., maximum of 300 ppb total PAH) to identify a definitive petroleum source.

3.2.2.3.2. Sediment PAH Summary

Total PAH41 concentrations did not significantly differ between Inner and Outer Rings and thus the established null hypothesis could not be disproved. PAH41 concentrations ranged from 8.8 to 304 µg/kg, with only one station exceeding 50 µg/kg (Station 1-7). Most stations appear to contain pyrogenic products. Contamination at Station 1-7 indicates inputs of weathered diesel and, at the maximum concentration of 303.9 µg/kg, is 7.7% of the ERL value of 4,022 µg/kg, well below levels of concern.

Table 3-20. Summary statistics for PAH.

Analyte	Ring	N	Mean (µg/kg)	Min (µg/kg)	Max (µg/kg)	Range (µg/kg)	Std Dev (µg/kg)	CV (µg/kg)
Petrogenic PAH	ALL	18	31.53	4.65	292.65	288	65.69	208
Petrogenic PAH	1	10	40.69	4.65	292.65	288	88.85	218
Petrogenic PAH	4	8	20.08	7.63	30.83	23.2	7.71	38
Pyrogenic PAH	ALL	18	6.78	3.18	16.59	13.41	3.3	48
Pyrogenic PAH	1	10	6.54	3.18	16.59	13.41	4.03	61
Pyrogenic PAH	4	8	7.07	3.85	10.92	7.07	2.3	32
Total PAH (41 compounds)	ALL	18	39.97	8.8	303.87	295.07	66.83	167
Total PAH (41 compounds)	1	10	48.62	8.8	303.87	295.07	90.35	185
Total PAH (41 compounds)	4	8	29.17	14.1	41.23	27.13	10.47	35
Total PAH (16 compounds)	ALL	18	9.96	5.38	24.11	18.73	4.87	48
Total PAH (16 compounds)	1	10	10.67	5.49	24.11	18.62	6.22	58
Total PAH (16 compounds)	4	8	9.07	5.38	13.64	8.26	2.52	27

Table 3-21. Inference Testing: Probabilities (p) for GLM comparisons.

		GLM Hypothesis Test Results			Normality Test Results		
Total PCB Measurement	Transformation Standardization Method	Inter-Ring Comparison (different if <0.05)	Inter-Cruise Comparison (different if <0.05)	Ring-Cruise Interaction (important if <0.05)	Shapiro-Wilks Value	Probability of Non-Normal Distribution (non-normal if <0.05)	Distribution
Total PAH_{41}	Log Concentration	0.77	0.57	0.48	0.81		Non-Normal
	Log (Con/TOC)	**0.82**	**0.07**	**0.94**	**0.95**	**0.4973**	**Normal**
	None	0.73	0.89	0.62	0.39		Non-Normal
	TOC Standardized	0.71	0.63	0.75	0.56		Non-Normal
Petrogenic PAH	Log Concentration	0.68	0.43	0.52	0.79		Non-Normal
	Log (Con/TOC)	**0.74**	**0.06**	**0.93**	**0.95**	**0.3870**	**Normal**
	None	0.71	0.88	0.66	0.36		Non-Normal
	TOC Standardized	0.68	0.74	0.76	0.49		Non-Normal
Pyrogenic PAH	Log Concentration	**0.65**	**0.99**	**0.32**	**0.96**	**0.6033**	**Normal**
	Log (Con/TOC)	0.63	0.01	0.29	0.96		Normal
	None	0.75	0.98	0.29	0.89		Non-Normal
	TOC Standardized	0.60	0.01	0.69	0.94		Normal

Values in **bold** provide best normalization of raw data.

Table 3-22. Summary of total PAH concentrations and PAH sources by sample.

Station (Year)	Total PAH (µg/kg)	PAH Sources
1-1 (1998)	48	trace concentration of a mid-range fuel with mid-level sulfur content and combustion related PAHs from atmospheric deposition
1-2 (1998)	8.8	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants
1-3 (1998)	15	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants
1-4 (1998)	19	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants
1-5 (1998)	12	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants
1-6 (1998)	20	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants
1-7 (1998)	300	weathered diesel range fuel with high sulfur content and combustion related PAHs from atmospheric deposition
1-8 (1998)	14	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants
4-6 (1998)	14	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants
1-3.5 (1999)	27	trace concentration of a diesel range fuel with low sulfur content and combustion related PAHs from atmospheric deposition
1-5/6/7 (1999)	20	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants with a potential trace contribution from a fuel product

Station (Year)	Total PAH (µg/kg)	PAH Sources
4-1 (1999)	41	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants with a potential trace contribution from a fuel product
4-2 (1999)	31	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants with a potential trace contribution from a fuel product
4-3 (1999)	32	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants with a potential trace contribution from a fuel product
4-3.1 (1999)	37	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants with a potential trace contribution from a fuel product
4-5 (1999)	16	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants with a potential trace contribution from a fuel product
4-6 (1999)	23	combustion related PAHs from atmospheric deposition and traces of common laboratory contaminants with a potential trace contribution from a fuel product
4-8 (1999)	39	trace concentration of a weathered heavy fuel and combustion related PAHs from atmospheric deposition

3.2.2.4. Measured concentrations of metals in sediment

Eighteen sediment samples distributed according to Figure 2-18 and Figure 2-19 were examined for ten metals. Eight of the metals (cadmium, chromium, copper, lead, mercury, nickel, silver and zinc) are contaminants of concern and the ship is considered a potential source of these metals. Two of the metals, aluminum and iron, were measured for use as a standardizing tool in data analysis. Patterns and ratios of association with iron and aluminum are well documented in estuarine and coastal environments and are useful in assessing whether the source of elements is crustal or anthropogenic and their potential for environmental impacts. Table 3-23 summarizes SINKEX sediment metal measurements. Data from published literature is provided in the last column, indicating that reference site (Ring 4) concentrations of four metals (Cd, Cr, Ni, Ag) were 1.5 times higher than typical marine sediments; three others (Al, Pb, and Hg) were lower while the last two (Cu and Zn) were approximately the same.

Table 3-23. Summary statistics for ten metals in Ring 1, Ring 4 and all stations combined.

ANALYTE	SITE	Units	N	Mean	Minimum	Maximum	Standard Error	CV	Average Marine Sediment[9]
Aluminum	ALL		18	1.6	0.4	2.5	0.12	30	7.2
	1	%	10	1.5	0.4	2.1	0.17	37	
	4		8	1.8	1.3	2.5	0.13	19	
Cadmium	ALL		18	1.71	0.28	7.00	0.44	109	0.17
	1	µg/g	10	2.80	0.54	7.00	0.60	67	
	4		8	0.3	0.3	0.5	0.03	20	

[9] Wedepohl, K.H. 1995. The composition of the continental crust. Geochimica et Cosmochimica Acta. 79(7): 1217-1232.

ANALYTE	SITE	Units	N	Mean	Minimum	Maximum	Standard Error	CV	Average Marine Sediment[9]
Chromium	ALL		18	150.1	87.9	231.0	11.21	31	72.0
	1	µg/g	10	151.9	87.9	227.0	16.53	34	
	4		8	147.8	106.0	231.0	15.71	30	
Copper	ALL		18	52.2	23.9	137.0	7.58	61	33.0
	1	µg/g	10	71.0	32.4	137.0	10.29	45	
	4		8	28.7	23.9	33.9	1.02	10	
Iron	ALL		18	3.50	1.20	5.90	0.30	40	4.10
	1	%	10	3.00	1.20	5.90	0.50	54	
	4		8	4.00	2.90	5.60	0.30	20	
Lead	ALL		18	7.0	1.8	24.3	1.34	81	19.0
	1	µg/g	10	8.8	1.8	24.3	2.29	82	
	4		8	4.7	4.1	5.3	0.17	10	
Mercury	ALL		18	0.052	0.028	0.107	0.004	33	0.190
	1	µg/g	10	0.059	0.043	0.107	0.006	30	
	4		8	0.042	0.028	0.054	0.004	24	
Nickel	ALL		18	58.8	23.9	215.0	11.18	80	19.0
	1	µg/g	10	83.2	25.0	215.0	16.57	63	
	4		8	28.3	23.9	35.4	1.47	14	
Silver	ALL		18	0.30	0.10	1.60	0.10	98	0.06
	1	µg/g	10	0.50	0.20	1.60	0.10	78	
	4		8	0.20	0.10	0.20	0.00	19	
Zinc	ALL		18	110.2	61.4	256.0	13.57	52	95.0
	1	µg/g	10	139.4	77.2	256.0	20.22	45	
	4		8	73.7	61.4	85.2	2.68	10	

Complete tabular data is provided in Appendix I, a discussion of methods and QA/QC can be found in Section 2.5.6.5.

3.2.2.4.1. Aluminum and Iron

Aluminum concentrations averaged 1.6±0.5% for all stations with no significant difference in mean values between the Inner Ring (1.5% Al) and the Outer Ring (1.8% Al). These low Al values are somewhat higher (3.1±0.9%) when expressed on a carbonate-free basis. Nevertheless, all concentrations of Al in these sediments (even on a carbonate-free basis) are lower than typically found in marine sediments (7.2%).

In sharp contrast with Al, concentrations of Fe are much higher, averaging 3.5±1.4% (non-carbonate-free basis) and 6.5±2.5% on a carbonate-free basis. Such levels are more comparable, or even enriched, relative to typical marine sediments (4.1%). As a result, the Fe/Al ratio (Figure 3-36) for sediment samples from this study average 2.1±0.6, about 4 times higher than in general marine sediments.

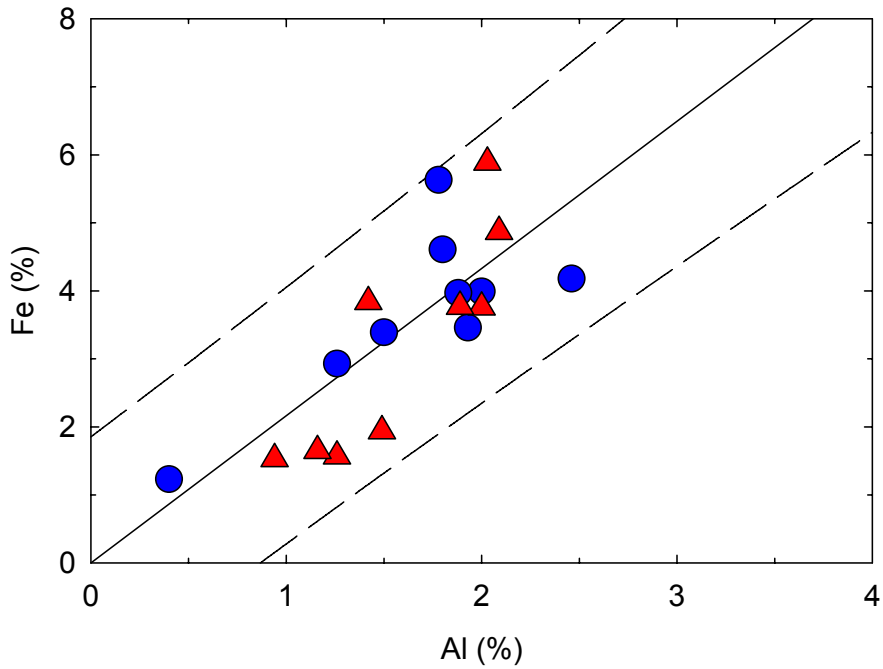


Figure 3-36. Concentrations of Al versus Fe for sediment samples from Ring 1 (red triangles) and Ring 4 (blue circles), dotted lines are 95% predictive intervals (r^2=0.60).

As explained earlier in Section 3.1.2, concentrations of Al, Fe and trace metals in sediments vary as a function of mineralogy, but often co-vary with sediment grain size and organic carbon content. For example, most trace metals, as well as Al and Fe, are present at very low levels in quartz or carbonate sand. In contrast to concentrations of most trace metals, Al and Fe are much higher in fine-grained aluminosilicates (clays). Aluminum and Fe are rarely introduced by anthropogenic processes in amounts that affect the background percent levels in sediment, relative to the much lower part per million or billion levels for trace metals. Thus, Al and Fe often provide a valuable standardization tool that can reduce selected variables that control metal concentrations to just one. In the ideal case, a good linear correlation is observed between concentrations of a trace metal and Al or Fe. Because Fe and Al are relatively well correlated in the sediments collected for this study (Figure 3-36), either metal can be used in the standardization process discussed below.

Measured concentrations of metals in sediments from Ring 1 were plotted against corresponding Al concentrations, as shown for Pb in Figure 3-37. Metals data from reference area sediments (Ring 4) were then plotted against Al using a linear regression equation and 95% prediction interval. In all cases, Ring 4 sediment data had a very limited range and thus each linear regression calculation was set to pass through the origin. Any positive deviations in concentrations of metals such as Pb, above the upper prediction limit, suggest the presence of anthropogenic inputs of that trace metal, most likely from the sunken vessel. The trends in Figure 3-37, for example, suggest that sediments from sites 1-2, 1-4, 1-7 and 1-3-5 have discernible anthropogenic inputs of Pb. This approach was used to supplement the inference test results in assessing whether other metals appear to be elevated relative to crustal metal background levels.

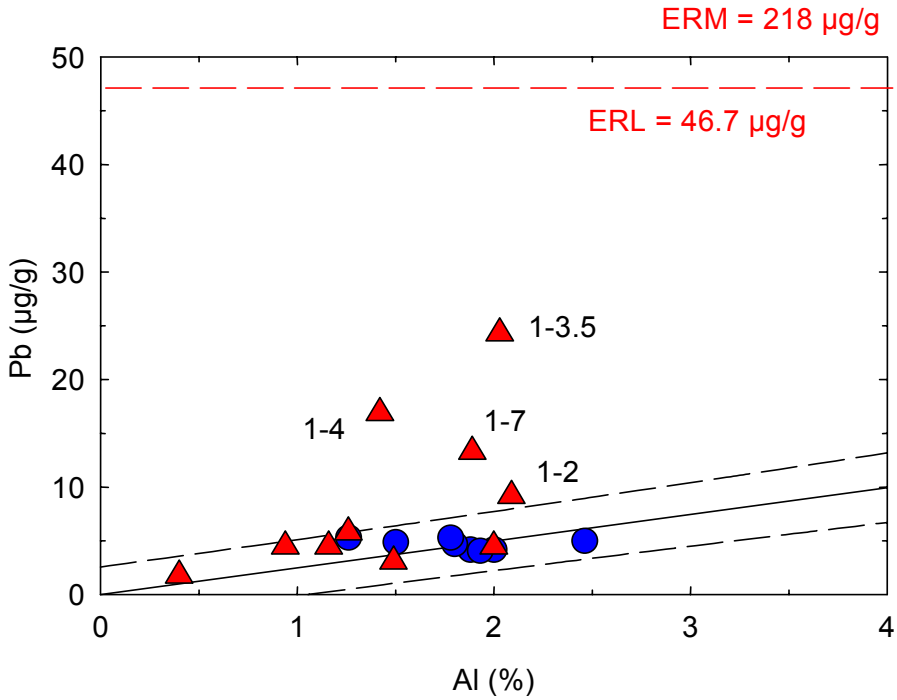


Figure 3-37. Concentrations of Al versus Pb for sediment samples from Ring 1 (red triangles) and Ring 4 (blue circles), dotted lines are 95% predictive intervals (r^2=0.635).

When appropriate, sediment quality criteria have been added to the metal/Al plots. Such criteria have been used extensively worldwide to assess possible adverse biological effects from trace metals and PAHs. Each of the trace metals investigated during this study has been assigned ERL and ERM concentrations by Long et al. (1995). All Pb values obtained for sediments from Ring 4 (Figure 3-37) are below the ERL and adverse effects would rarely occur.

3.2.2.4.2. Cadmium

The mean concentration for cadmium at all stations was 1.71µg/g dry wt., with a maximum of 7.0 µg/g (Station 1-7). Inner Ring data is shown in Figure 3-38 and all data are shown in Figure 3-39. Inner Ring mean concentrations differed significantly from the Outer Ring means (Table 3-24, log (concentration/TOC), $p<0.0008$). When the Cd data are plotted against Al, the enhanced levels of Cd are evident, with seven samples from Ring 1 having Cd values exceeding Ring 4 levels (Figure 3-40).

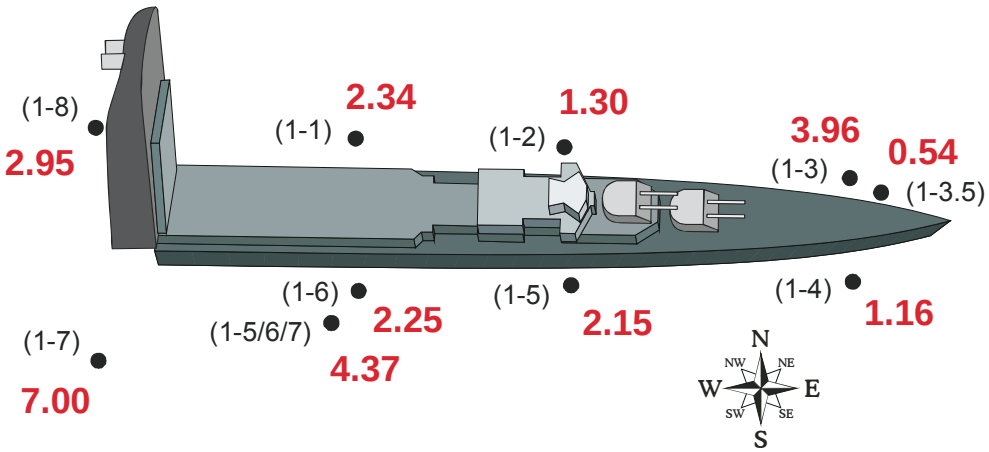


Figure 3-38. Ring 1 cadmium (µg/g dry weight). Station numbers in parentheses.

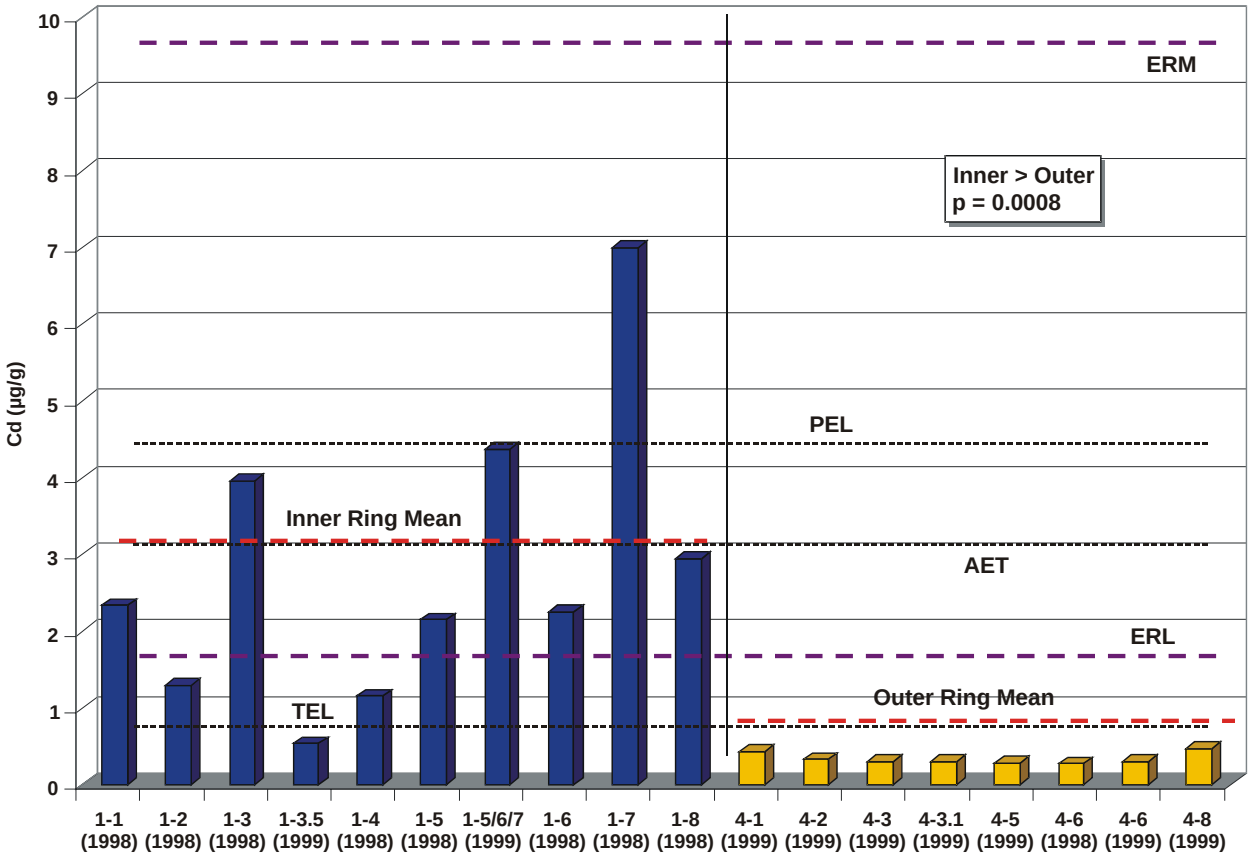


Figure 3-39. Cadmium concentrations (µg/kg dry weight).

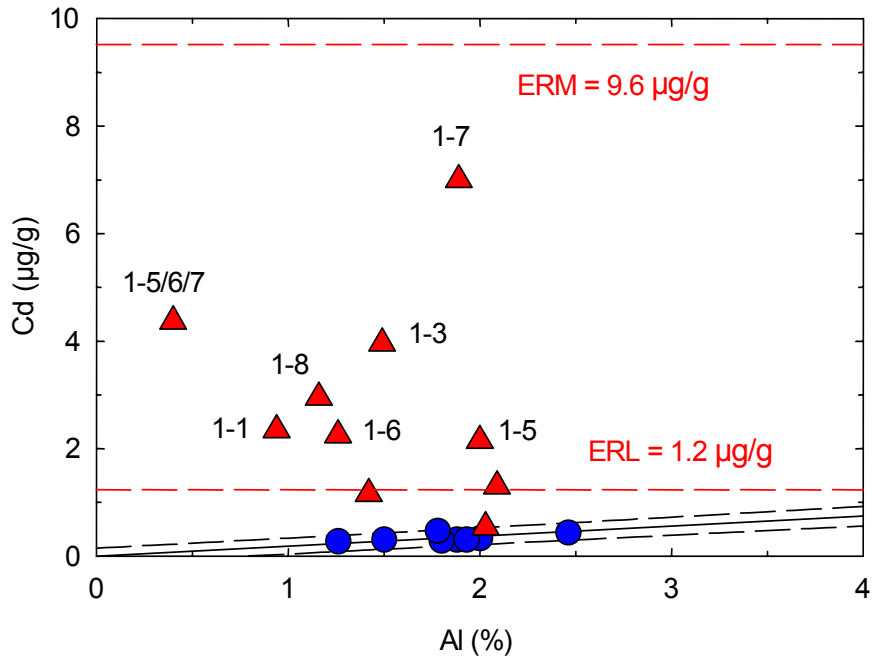


Figure 3-40. Concentrations of Al versus Cd for sediment samples from Ring 1 (red triangles) and Ring 4 (blue circles), dotted lines are 95% predictive intervals (r^2=0.791).

The average Inner Ring concentration is above the TEL and ER-L levels of 0.676 and 1.2 µg/g (respectively), and below the ER-M level of 9.6 µg/g. The PEL level of 4.2 µg/g is between the mean and maximum values for the Inner Ring stations. Based on these comparisons, the concentration of cadmium in the Inner Ring sediments is above the threshold concentration of effects, but on average below the level of probable effects.

3.2.2.4.3. Chromium

The mean concentration for chromium at all stations was 150.1 µg/g, with a maximum concentration of 231 µg/g (Station 4-8). Ring 1 average concentrations do not significantly differ from Ring 4 mean values (151.9 and 147.8 µg/g respectively, p=0.2934). Aluminum shows a strong linear relationship to chromium (Figure 3-41). Such a relationship is consistent with the presence of a Cr-rich (>200 µg/g) aluminosilicate that is diluted with varying amounts of carbonate and organic matter. Inner Ring data is shown in Figure 3-42.

Concentrations of Cr in quite a few SINKEX sediment samples, as well as the average concentration found in the continental crust (125 µg/g), exceed the value for the ERL (Figure 3-43). Similar observations are regularly made for Cr, most likely because the database compiled by Long et al. (1995) used Cr concentrations from an acid leach of the sediment rather than a total digestion. Only a small fraction (<25%) of the total Cr is removed by a strong acid leach (Trefry and Presley, 1976; Sinex et al., 1980). Thus, a leachable Cr value equal to the ERL level of 82 µg/g is more likely comparable with a total Cr level of > 200 µg/g, a value considerably higher than Cr values for continental crust or any samples from this study. The ERL and ERM values for Cr may need to be revised in future iterations of these sediment quality guidelines. No

data points exceed the established ERM for chromium. Chromium values appear to be normal for this area. Spatial representation of Inner and Outer Ring data is shown in Figure 3-43.

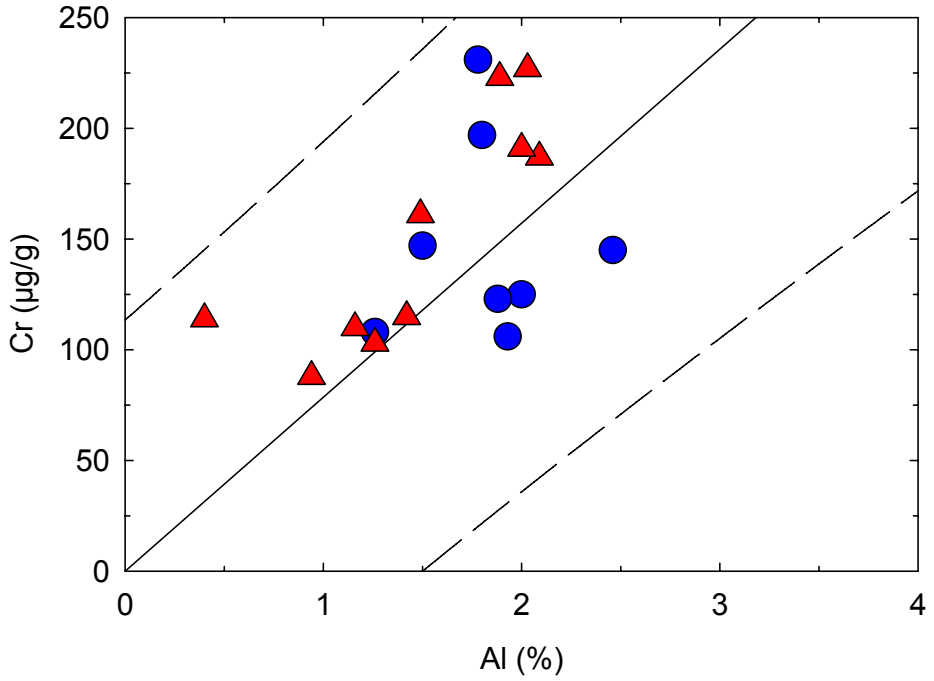


Figure 3-41. Concentrations of Al versus Cr for sediment samples from Ring 1 (red triangles) and Ring 4 (blue circles), dotted lines are 95% predictive intervals (r^2=0.475).

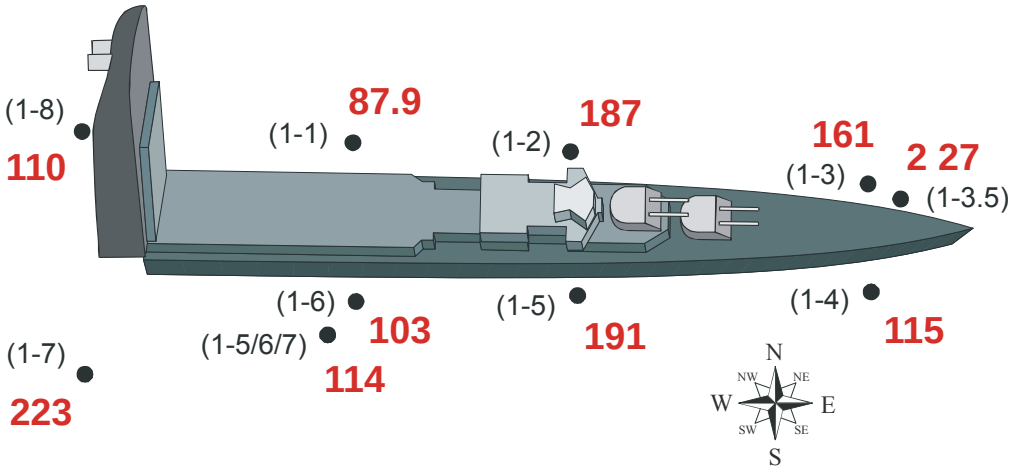


Figure 3-42. Ring 1 chromium (µg/g dry weight). Station numbers in parentheses.

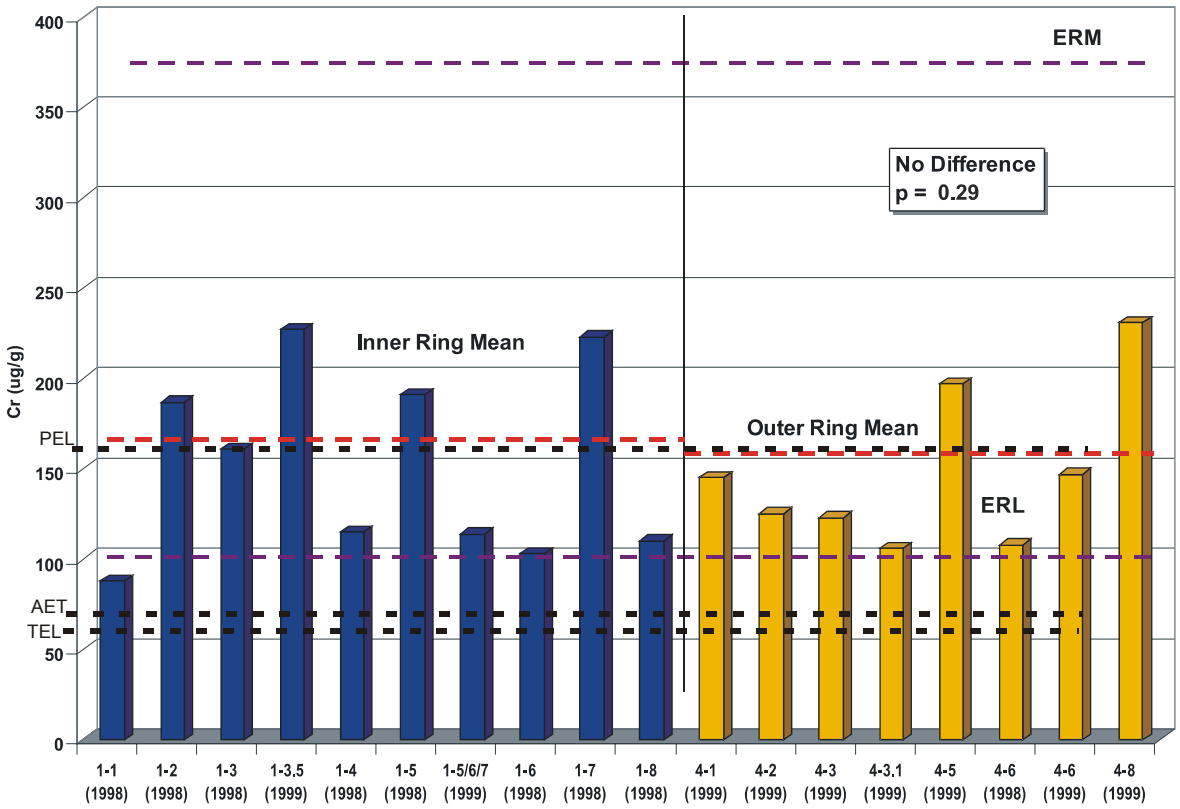


Figure 3-43. Chromium concentrations (µg/kg dry weight).

3.2.2.4.4. Copper

The mean concentration for copper at all stations was 52.2 µg/g, with a maximum concentration of 137 µg/g (Station 1-7). Inner Ring average concentrations are significantly higher than Ring 4 mean values, 71.0 and 33.9 µg/g respectively (Table 3-24, $p<0.0007$). All Inner Ring stations, with the exception of 1-4 exceed the 95% predictive interval pictured in Figure 3-44, indicating an anthropogenic source. The mean concentration is above both the TEL and ER-L levels of 18.7 and 34.0 µg/g (respectively), and below the ER-M level of 270 µg/g. The PEL level of 108.2 µg/g is between the mean and maximum values for the Inner Ring stations. Based on these comparisons, the concentration of copper in the Inner Ring sediments is above the threshold concentration of effects, but on average below the level of probable effects. Inner Ring data is shown in Figure 3-45. Spatial representation of the all data is shown in Figure 3-46.

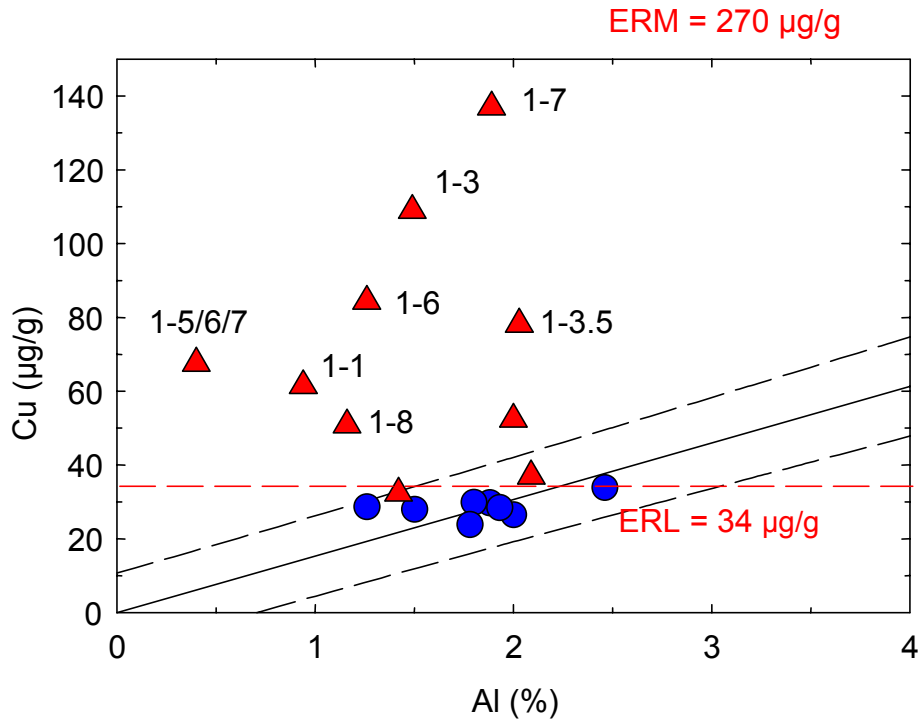


Figure 3-44. Concentrations of Al versus Cu for sediment samples from Ring 1 (red triangles) and Ring 4 (blue circles), dotted lines are 95% predictive interval (r^2=0.823).

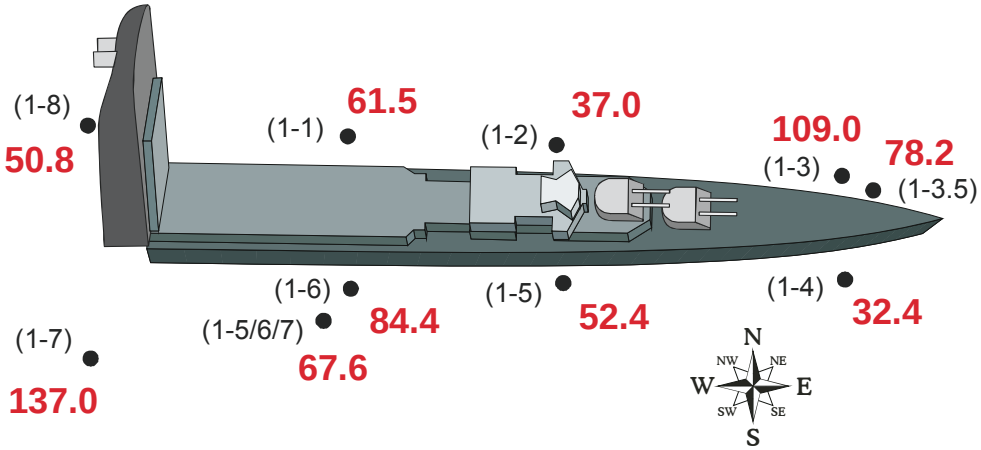


Figure 3-45. Ring 1 copper (µg/g dry weight). Station numbers in parentheses.

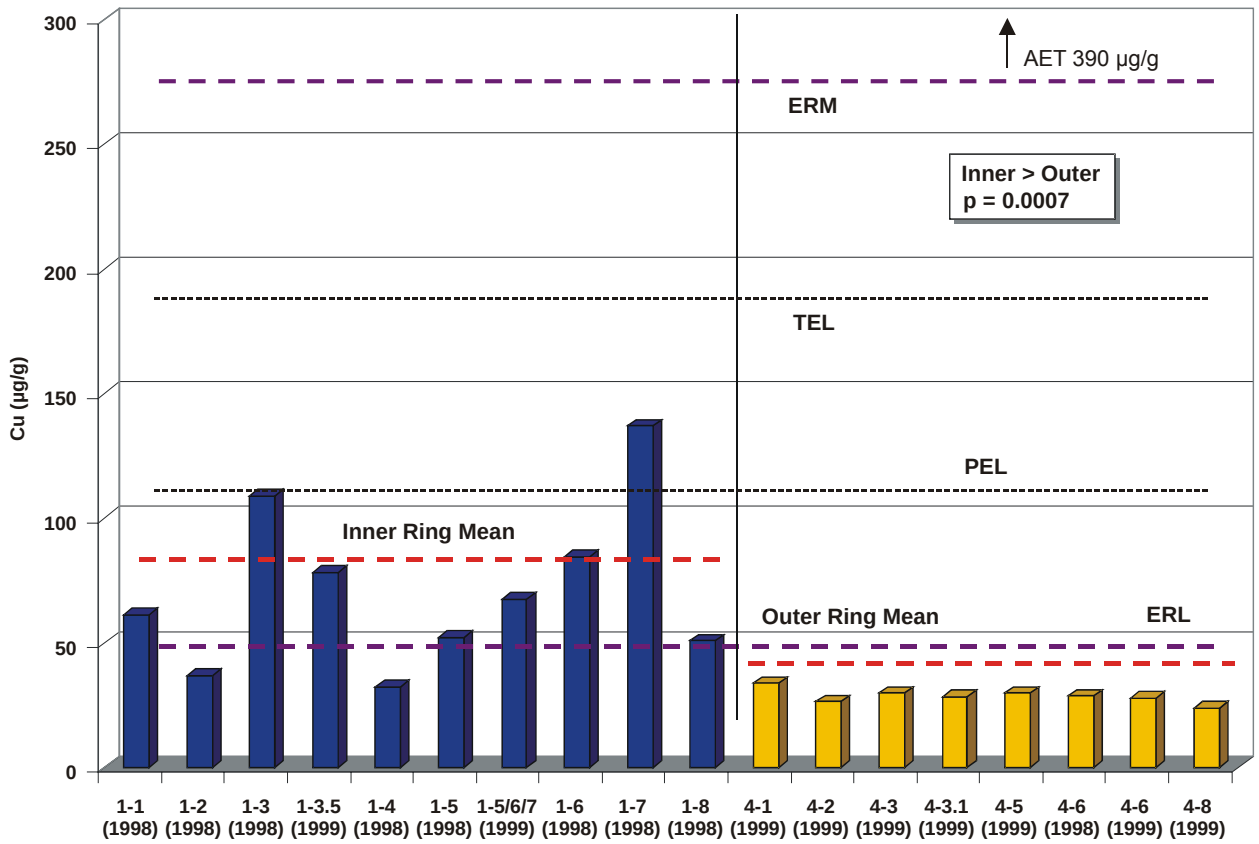


Figure 3-46. Copper concentrations (µg/g dry weight) in Ring 4.

3.2.2.4.5. Lead

The mean concentration for lead at all stations was 7.0 µg/g dry wt., with a maximum concentration of 24.3 µg/g (Station 1-3.5). Inner Ring average concentrations are not significantly higher than Ring 4 mean values, 8.8 and 4.7 µg/g respectively (Table 3-24, log(concentration/TOC), $p>0.6536$). The mean and maximum concentrations are also below the TEL and ER-L levels of 30.2 and 46.7 µg/g dry wt., respectively. The trends in Figure 3-37, suggest that sediments from sites 1-2, 1-4, 1-7 and 1-3-5 have discernible anthropogenic inputs of Pb above what would be predicted by the data from sediments from Ring 4. Inner Ring data is shown in Figure 3-47. Spatial representation of all data is shown in Figure 3-48.

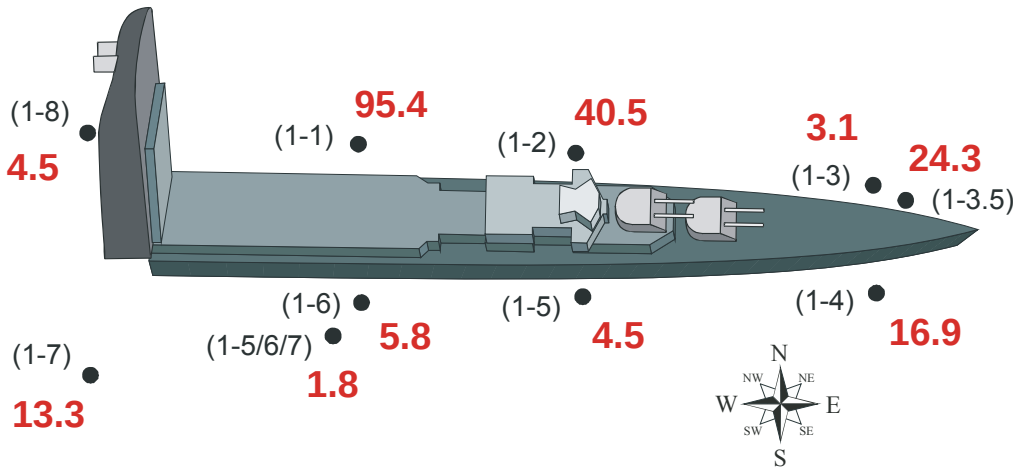


Figure 3-47. Ring 1 lead (µg/g dry weight). Station numbers in parentheses.

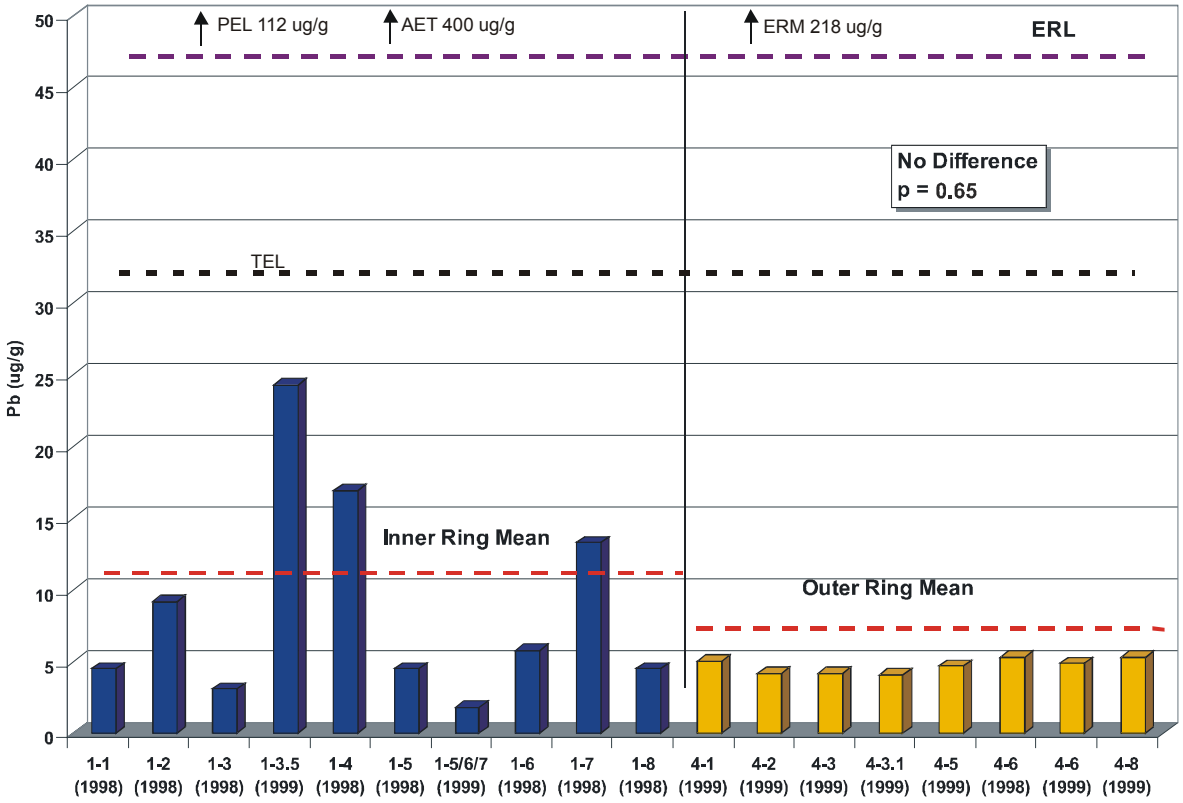


Figure 3-48. Lead concentrations (µg/kg dry weight).

3.2.2.4.6. Mercury

The mean concentration for mercury at all stations was 0.052 µg/g with a maximum concentration of 0.107 µg/g (Station 1-3). Inner Ring average concentrations do not significantly differ than Ring 4 mean values, 0.059 and 0.042 µg/g respectively (Table 3-24, TOC

standardized p=0.0875). Six Inner Ring stations lie outside the 95% predictive interval depicted in Figure 3-49, indicating possible anthropogenic contamination. The mean and maximum concentrations, however, are below the TEL and ER-L levels of 0.13 and 0.15µg/g dry wt. (respectively). Inner Ring data is shown in Figure 3-50. Spatial representation of all data is shown in Figure 3-51.

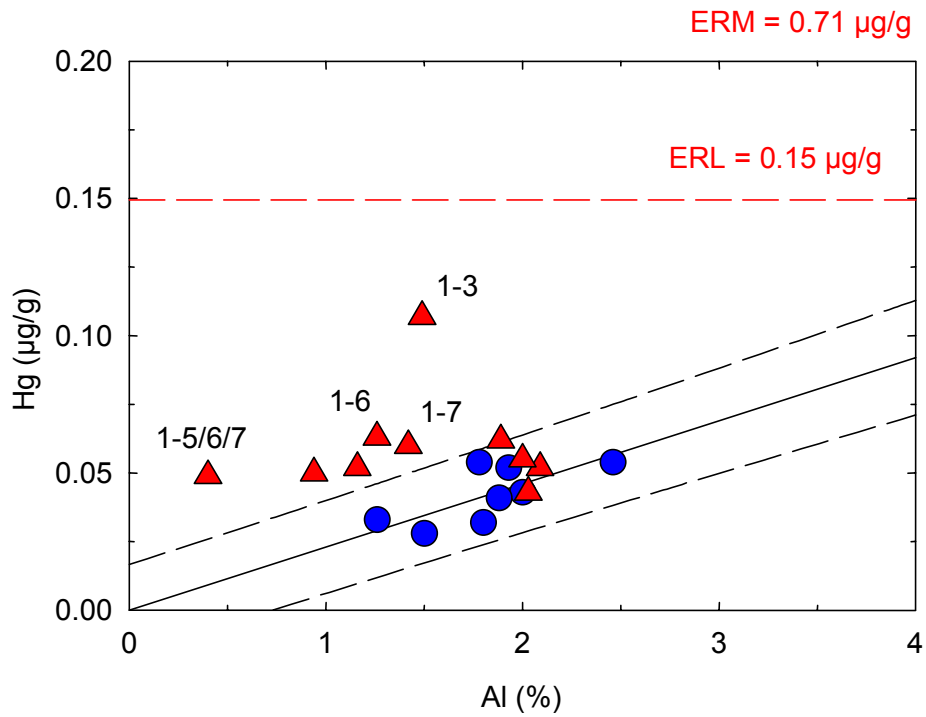


Figure 3-49. Concentrations of Al versus Hg for sediment samples from Ring 1 (red triangles) and Ring 4 (blue circles), dotted lines are 95% predictive intervals (r^2=0.832).

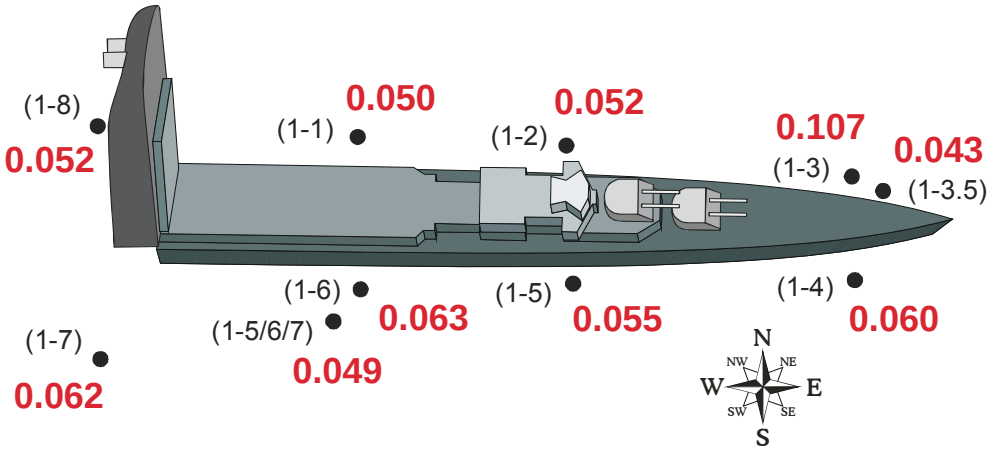


Figure 3-50. Ring 1 mercury (µg/g dry weight). Station numbers in parentheses.

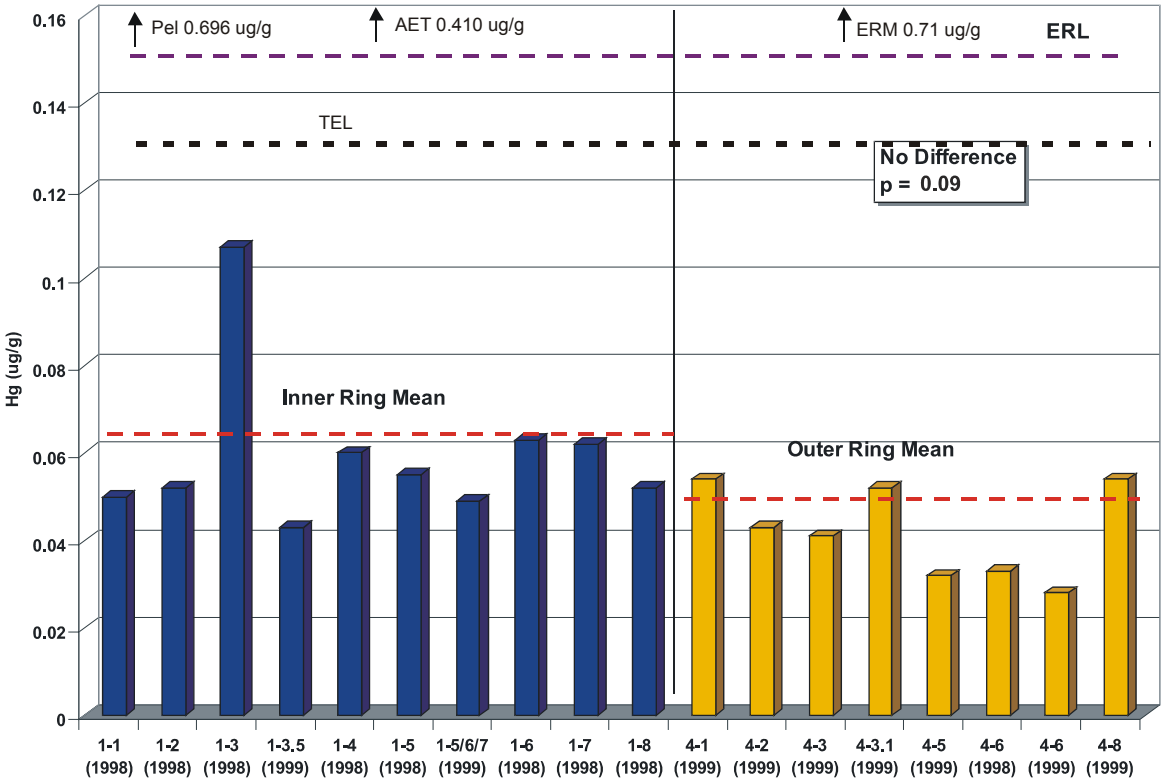


Figure 3-51. Mercury concentrations (µg/kg dry weight).

3.2.2.4.7. Nickel

The mean concentration for nickel at all stations was 58.8µg/g dry wt., with a maximum concentration of 215 µg/g (Station 1-3). Inner Ring average concentrations are higher than Ring 4 mean values, 83.2 and 28.3 µg/g respectively, and the difference is significant[10] (Kruskal-Wallis analysis p=0.003, Table 3-24). Eight Inner Ring stations lie outside the 95% predictive interval depicted in Figure 3-52, indicating possible anthropogenic contamination. The mean concentration is above the TEL and ER-L levels of 15.9 and 20.9µg/g dry wt., respectively. This concentration is greater than the ER-M (51.6 µg/g) and the PEL (42.8 µg/g). The mean Inner Ring value is, however, below the AET concentration (110 µg/g dry wt). Based on these comparisons, the concentration of nickel in the Inner Ring sediments is slightly above the level of probable effects, but below the AET threshold for effects. Inner Ring data is shown in Figure 3-53. Spatial representation of all data is shown in Figure 3-54.

[10] As explained in Methods section, when a distribution is best normalized by a rank transformation, statistical significance is determined using the p-value for the Kruskal-Wallis test rather than the p-value associated with the rank transformation.

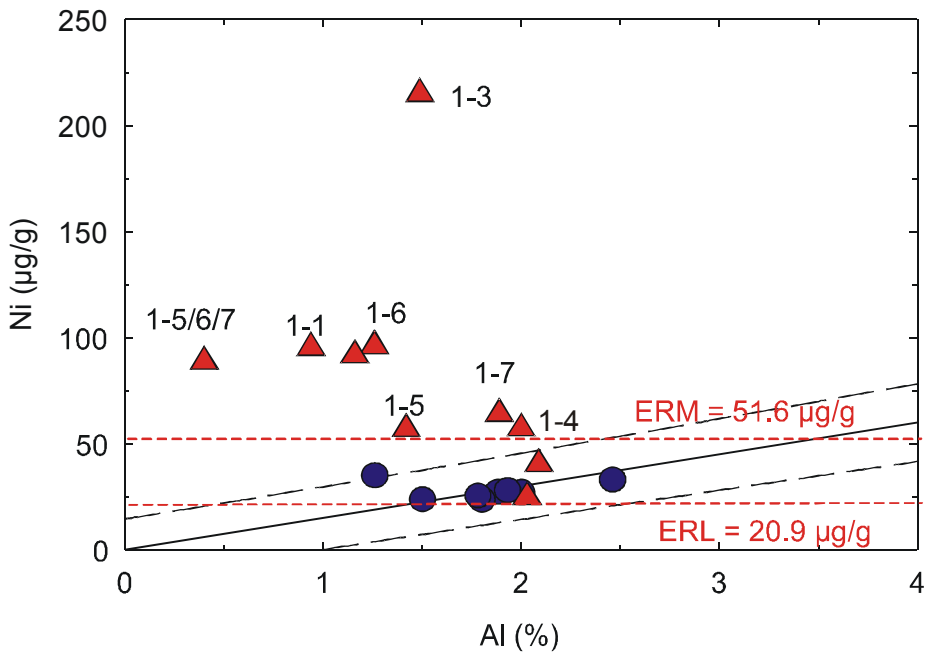


Figure 3-52. Concentrations of Al versus Ni for sediment samples from Ring 1 (red triangles) and Ring 4 (blue circles), dotted lines are 95% predictive interval (r^2=0.678).

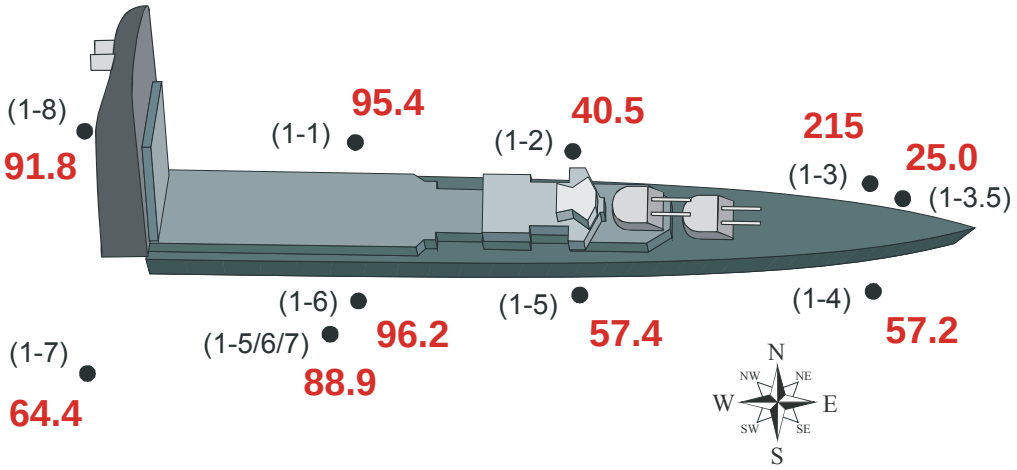


Figure 3-53. Ring 1 nickel (µg/g dry weight). Station numbers in parentheses.

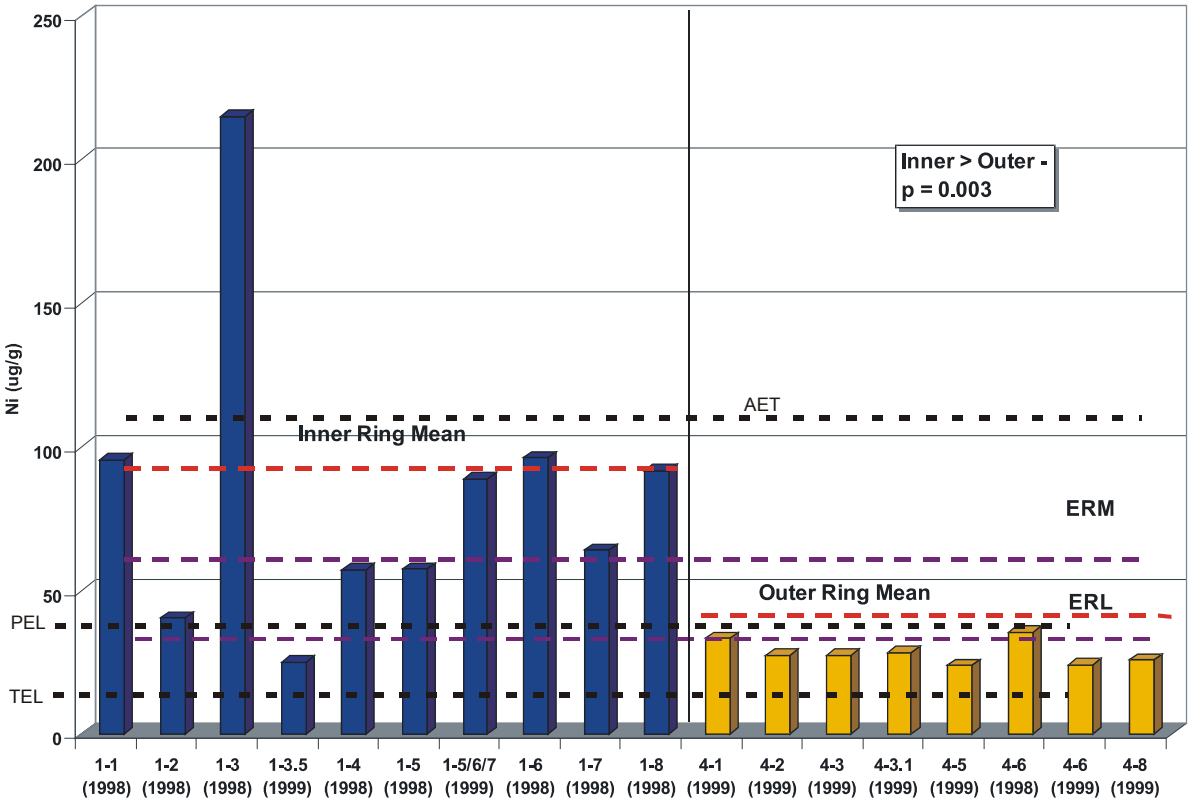


Figure 3-54. Nickel concentrations (µg/kg dry weight).

3.2.2.4.8. Silver

The mean concentration for silver at all stations was 0.3 µg/g, with a maximum concentration of 1.6 µg/g (Station 1-3). Inner Ring mean concentrations are significantly greater than Outer Ring means (Table 3-24, $p<0.0001$). Nine Inner Ring stations lie outside the 95% predictive interval depicted in Figure 3-55, indicating possible anthropogenic contamination. The mean Inner Ring concentration is below the TEL and ER-L levels of 0.73 and 1.0 µg/g dry wt., respectively. The maximum of 1.6 µg/g is above the TEL and ER-L concentrations, but below the ER-M level of 3.7 µg/g. and the PEL level of 1.77 µg/g. Inner Ring data is shown in Figure 3-56. Spatial representation of the all data is shown in Figure 3-57.

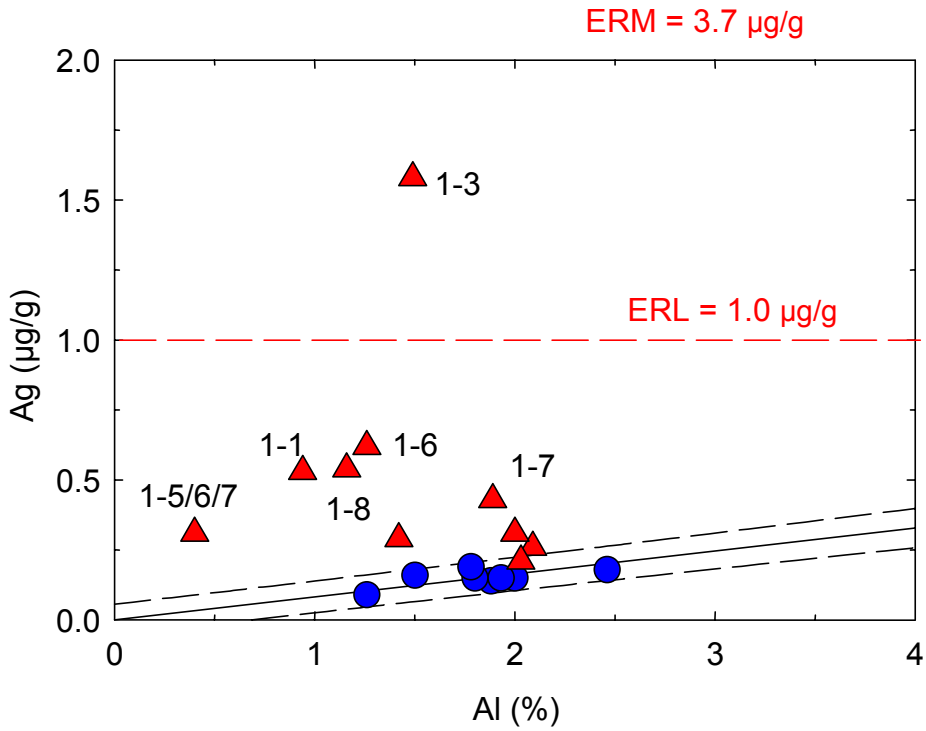


Figure 3-55. Concentrations of Al versus Ag for sediment samples from Ring 1 (red triangles) and Ring 4 (blue circles), dotted lines are 95% predictive intervals (r^2=0.839).

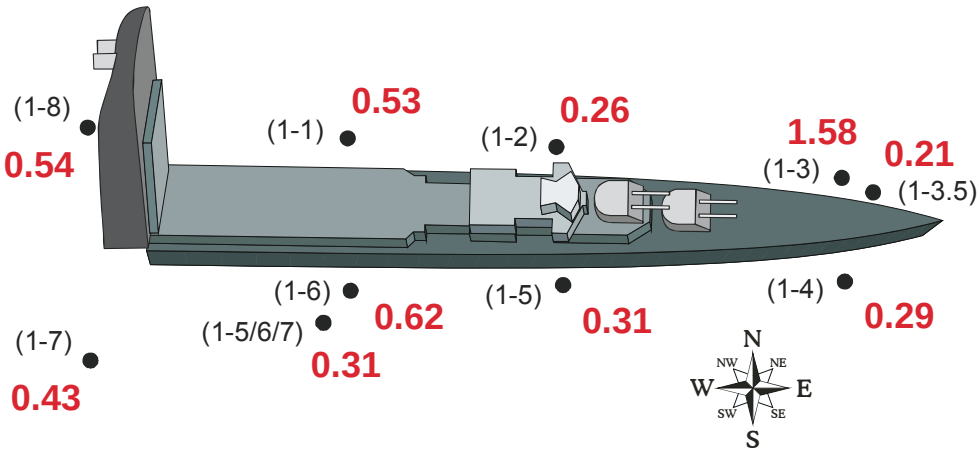


Figure 3-56. Ring 1 silver (µg/g dry weight). Station numbers in parentheses.

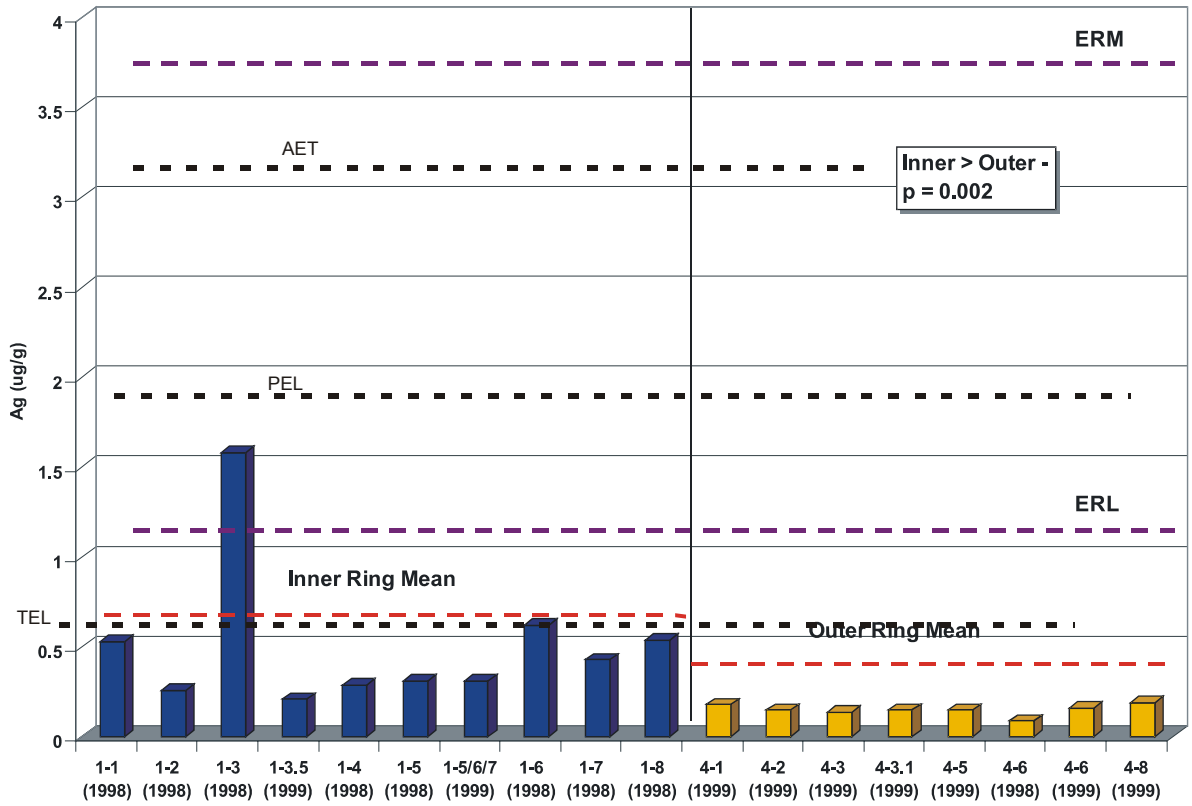


Figure 3-57. Silver concentrations (µg/kg dry weight).

3.2.2.4.9. Zinc

The mean concentration for zinc at all stations was 110.2 µg/g, with a maximum concentration of 256 µg/g (Station 1-4). The average Inner Ring concentration was not significantly greater than the mean Outer Ring value of 73.71 µg/g (log(concentration/TOC), p=0.1643). Seven Inner Ring stations lie outside the 95% predictive interval depicted in Figure 3-58 indicating possible anthropogenic contamination. The Inner Ring mean concentration is between the TEL of 124 µg/g and the ER-L of 150 µg/g. The maximum of 256 µg/g is below the PEL and ER-M concentrations of 271 and 410 µg/g, respectively. Inner Ring data is presented in Figure 3-59. Spatial representation of all data is shown in Figure 3-60.

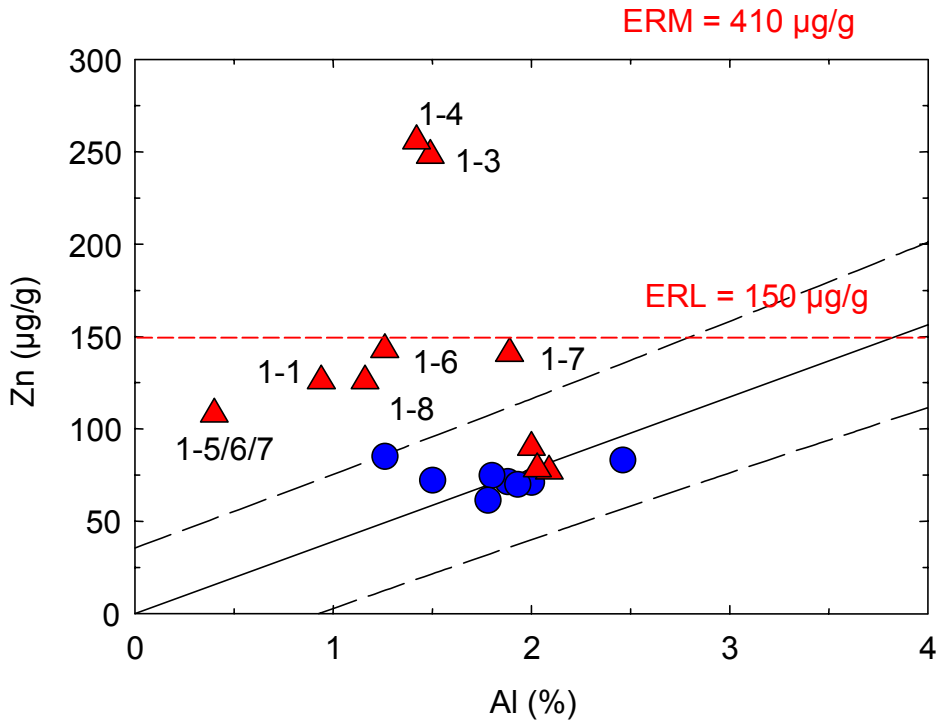


Figure 3-58. Concentrations of Al versus Zn for sediment samples from Ring 1 (red triangles) and Ring 4 (blue circles), dotted lines are 95% predictive intervals (r^2=0.708).

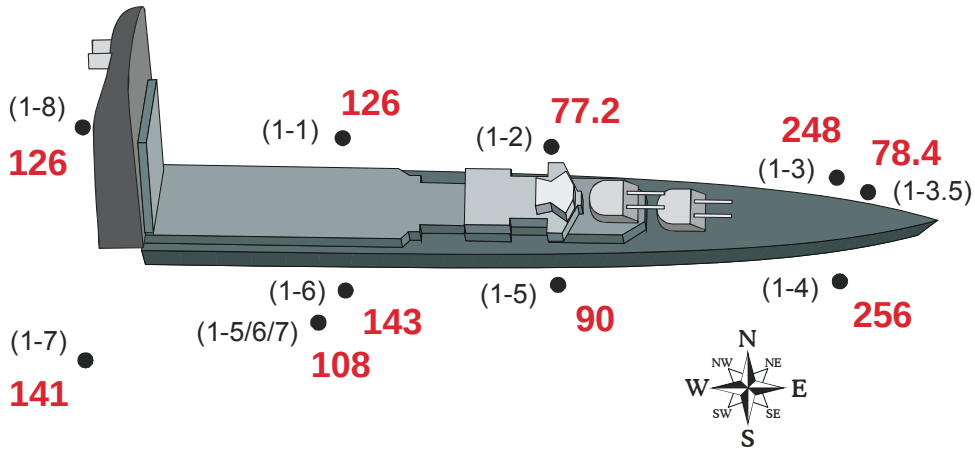


Figure 3-59. Ring 1 zinc (µg/g dry weight). Station numbers in parentheses.

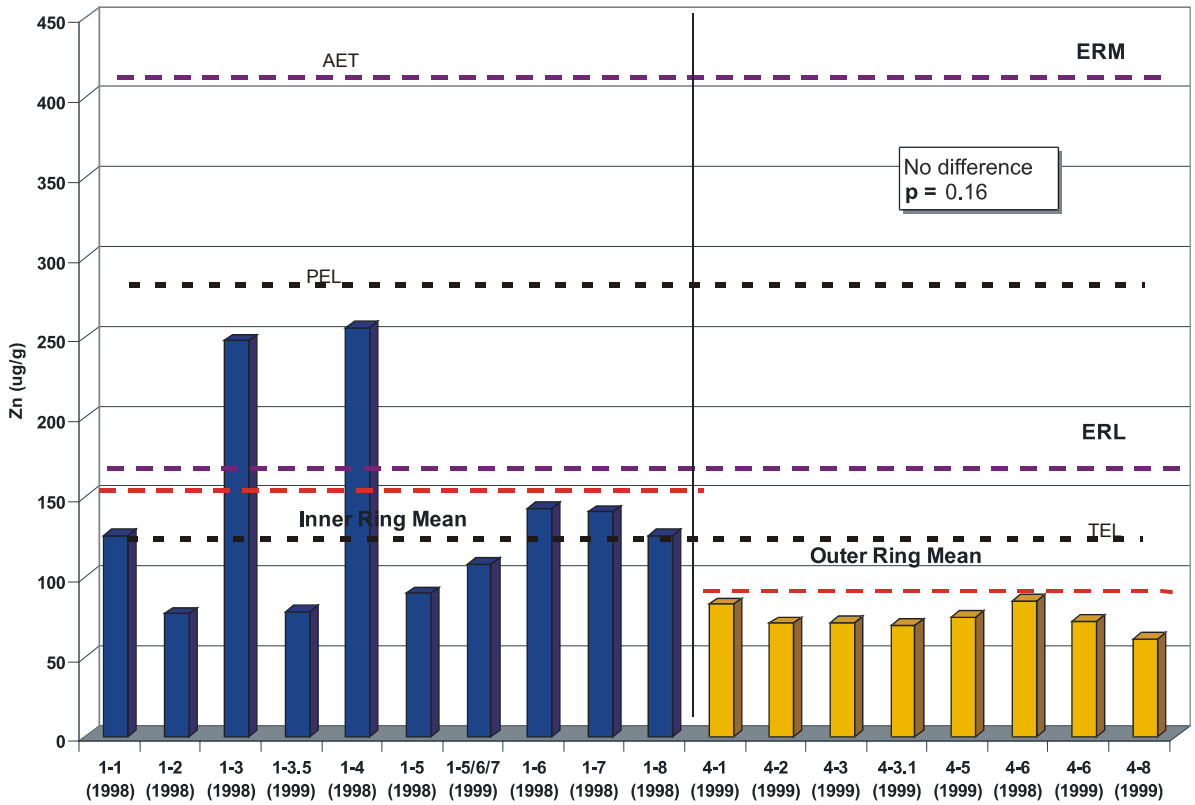


Figure 3-60. Zinc concentrations (µg/kg dry weight).

3.2.2.4.10. Sediment Metal Summary

Eight metals considered secondary chemicals of concern were evaluated for sediment concentration and statistical comparisons were made between Inner and Outer Rings. Cadmium, copper, nickel and silver were found to be significantly ($p \leq .05$) elevated at the Inner Ring. Average Inner Ring sediment concentrations for cadmium and copper were greater than the ERL and less than the ERM. Inner and Outer Ring mean concentrations for silver were below the ERL. The Inner Ring mean sediment concentration for nickel exceeded its ERM value.

As discussed earlier in greater detail in Section 2.3.1.3.1 (Source Characterization of Metals), sunken vessels provide a large reservoir of metals that could be potentially available to the environment, through slow leaching processes. One previous study of trace metals in metal pieces and paint from the ex-WILLIAM C. LAWE (Trefry and Trocine, 1998) determined concentrations of Ag, Cd, Cr, Cu, Hg, Ni, Pb, Sn and Zn in 34 different pieces of metal and Cr, Cu, Hg, Pb and Zn in 10 different samples of paint. For example, the results show that concentrations of Cr, Cu and Zn can exceed 100,000 µg/g in some pieces of sampled metal, while concentrations of Ni and Pb can exceed 20,000 µg/g. Each of these extreme levels approaches concentrations that are more than 1000 times greater than found in typical marine sediments. In samples of paint, concentrations of Zn, Pb and Hg were typically about 30,000; 5,000 and 0.5 µg/g, respectively. Thus, data from these ship components can be used to discuss specific sources for metal enrichment found in the sediment.

Despite the presence of such a large reservoir of metal, the results of this study show that very limited release of metals from the ship has occurred. The most contaminated sample of sediment is enriched in Cu and Ni by about 100 and 200 µg/g (a factor of 4-5), respectively, relative to local, natural sediments; yet the potential for increased metal levels could be many times greater when one considers the magnitude of the source loading. This observation certainly suggests that the rates of release of metals from the ship by dissolution, abrasion and other physical and biological processes are relatively slow.

Table 3-24. Two-way comparison of SINKEX rings (1 and 4 and cruises 1998-1999). Values in bold within shaded row provide best normalization of raw data (differences significant if p<0.05, noted in bold, same row unless normalization was through non-parametric rank correlation – noted in Kruskal-Wallis column).

Analyte	Transformation Standardization Method	Inter-Ring Comparison*	Inter-Cruise Comparison*	Ring-Cruise Interaction**	Shapiro-Wilks Value	Probability of Non-Normal Distribution†	Probability Kruskal-Wallis*	Distribution
Aluminum	Log Conc.	0.7558	0.6780	0.3407	0.9174	0.0000		Non-Normal
	None	**0.5103**	**0.6036**	**0.1443**	**0.9438**	**0.3359**	**0.3064**	**Normal**
	Rank	0.9647	0.2998	0.3312	0.9595	0.5923		Normal
	TOC Standardized	0.6763	0.0282	0.5983	0.9440	0.3389		Normal
	Log (Conc/TOC)	0.4622	0.0657	0.7511	0.9081	0.0795		Normal
Cadmium	Log Conc.	0.0000	0.5035	0.4604	0.8385	0.0000		Non-Normal
	None	0.0348	0.8599	0.8058	0.7841	0.0009	0.0004	Non-Normal
	Rank	0.0001	0.3760	0.2307	0.9569	0.5427		Normal
	TOC Standardized	0.0360	0.2119	0.2884	0.5567	0.0000		Non-Normal
	Log (Conc/TOC)	**0.0008**	**0.1104**	**0.9536**	**0.9060**	**0.0733**		**Normal**
Chromium	Log Conc.	0.1920	0.4138	0.3868	0.6751	0.0000		Non-Normal
	None	0.4159	0.3238	0.7463	0.8907	0.0396	0.8940	Non-Normal
	Rank	0.4556	0.2284	0.6137	0.9614	0.6294		Normal
	TOC Standardized	**0.2934**	**0.0051**	**0.4535**	**0.9419**	**0.3118**		**Normal**
	Log (Conc/TOC)	0.7092	0.0150	0.5706	0.9012	0.0604		Normal
Copper	Log Conc.	0.0159	0.4842	0.3080	0.5341	0.0000		Non-Normal
	None	0.0267	0.9487	0.9461	0.8110	0.0022	**0.0006**	Non-Normal
	Rank	0.0007	**0.8246**	**0.6735**	**0.9614**	**0.6294**		**Normal**
	TOC Standardized	0.0020	0.0027	0.0247	0.6793	0.0000		Non-Normal
	Log (Conc/TOC)	0.0074	0.0038	0.1657	0.8341	0.0048		Non-Normal
Iron	Log Conc.	0.9088	0.4098	0.5013	0.9273	0.0001		Non-Normal
	None	**0.7171**	**0.3092**	**0.7683**	**0.9382**	**0.2696**	**0.1684**	**Normal**
	Rank	0.8565	0.2550	0.5491	0.9614	0.6294		Normal
	TOC Standardized	0.9412	0.0190	0.8459	0.9455	0.3595		Normal
	Log (Conc/TOC)	0.5706	0.0565	0.7476	0.8827	0.0290		Non-Normal
Lead	Log Conc.	0.0332	0.9889	0.2002	0.6955	0.0000		Non-Normal
	None	0.1623	0.5370	0.4284	0.6791	0.0000	0.4487	Non-Normal
	Rank	0.9831	0.4474	0.6383	0.9576	0.5556		Normal
	TOC Standardized	0.1205	0.0931	0.1880	0.5983	0.0000		Non-Normal
	Log (Conc/TOC)	**0.6536**	**0.1912**	**0.5105**	**0.9670**	**0.7389**		**Normal**

Analyte	Transformation Standardization Method	Inter-Ring Comparison*	Inter-Cruise Comparison*	Ring-Cruise Interaction**	Shapiro-Wilks Value	Probability of Non-Normal Distribution†	Probability Kruskal-Wallis*	Distribution
Mercury	Log Conc.	0.5502	0.9467	0.9496	0.8092	0.0000		Non-Normal
	None	0.1265	0.7591	0.1937	0.8176	0.0027	0.0259	Non-Normal
	Rank	0.0909	0.5381	0.0640	0.9582	0.5664		Normal
	TOC Standardized	**0.0875**	**0.0012**	**0.4643**	**0.9491**	**0.4104**		**Normal**
	Log (Conc/TOC)	0.1840	0.0028	0.8206	0.9319	0.2099		Normal
Nickel	Log Conc.	0.0364	0.7980	0.4990	0.4862	0.0000		Non-Normal
	None	0.1418	0.4608	0.6557	0.7206	0.0001	**0.0029**	Non-Normal
	Rank	0.0708	**0.0470**	**0.7909**	**0.9614**	**0.6294**		**Normal**
	TOC Standardized	0.0234	0.0521	0.1354	0.4564	0.0000		Non-Normal
	Log (Conc/TOC)	0.0065	0.0390	0.3665	0.7051	0.0001		Non-Normal
Silver	Log Conc.	0.0204	0.8792	0.4898	0.9085	0.0000		Non-Normal
	None	0.1733	0.5621	0.3631	0.6438	0.0000	0.0004	Non-Normal
	Rank	**0.0001**	**0.8692**	**0.0437**	**0.9563**	**0.5315**		**Normal**
	TOC Standardized	0.0002	0.0016	0.5011	0.8417	0.0063		Non-Normal
	Log (Conc/TOC)	0.0000	0.0000	0.0630	0.8927	0.0429		Non-Normal
Zinc	Log Conc.	0.0918	0.9381	0.5849	0.6004	0.0000		Non-Normal
	None	0.1916	0.2818	0.4927	0.7334	0.0002	0.0016	Non-Normal
	Rank	0.0386	0.0386	0.6682	0.9607	0.6149		Normal
	TOC Standardized	0.0915	0.0372	0.3059	0.8704	0.0181		Non-Normal
	Log (Conc/TOC)	**0.1643**	**0.0310**	**0.4238**	**0.9607**	**0.6151**		**Normal**

*different if <0.05
**important if <0.05
***log of sediment concentration divided by fractional TOC
† non-normal if <0.05
Values in **bold** provide best normalization of raw data and indication of significance.

3.2.2.4.11. Acid-Volatile Sulfide/Simultaneously Extracted Metals

The underlying principle for this technique, which was developed for use in shallow coastal waters, is to use simultaneously extracted metals (SEM in µmoles/g) minus acid volatile sulfides (AVS µmoles/g) as an interpretative tool. In those instances where the value for SEM-AVS is negative, the amount of sulfide exceeds the total amount of metal and thus these metals should be bound in the sediments as a sulfide phase and not bioavailable (i.e. non-toxic). When the SEM-AVS value is positive, acid-leachable metals are present in the sediment in other than a sulfide phase and presumably bioavailable. The original use of AVS-SEM was to determine if metals present in sediment were bound by sulfide phases and therefore unavailable to the biota. The AVS/SEM technique is less meaningful in oxic sediments or where sediment AVS levels are less than 1 µmole/g (DiToro et al., 1990).

Concentrations of AVS and SEM were determined for 18 samples of surficial sediment (0 to 3 cm) collected during the 1998 and 1999 cruses. Analyses for SEM included the following metals: Cd, Cu, Ni, Pb, and Zn (Table 3-25). Concentrations of Fe were measured in the SEM solution to investigate possible leaching of Fe oxides, even though Fe is not included in the total SEM value. SEM-AVS values were positive for all sediments examined. Concentrations of AVS

were relatively low ranging from 0.0002 to 0.230 µmole/g. See also Appendix D for further information.

Low values are typical for oxidized surficial sediments and would be expected at sediment accumulation rates observed in the study area (approximately 0.03 cm/y). Reduced sulfur (i.e., AVS) does not form unless the sediments have been subjected to sulfate reduction; an observation not commonly observed in the top few centimeters of sediment from open ocean areas. Concentrations of SEM metals (Cd + Cu + Ni + Pb + Zn) released by the 1 N HCl treatment ranged from 0.389 to 9.737 µmole/g, considerably higher than the values for AVS. Values of SEM-AVS were all positive ranging from 0.089 to 9.527 µmoles/g. Under oxic conditions, SEM is most likely associated with iron oxides. Analysis of this important metal revealed a sizeable amount of Fe with the 1N HCl treatment. The SEM metals are most likely associated with this iron component and would not dissolve into solution, even at pH levels as low as 3. Such resistance to dissolution should help keep the SEM biologically unavailable, and thus non-toxic (Trefry and Metz, 1984).

Table 3-25. Summary of ASV/SEM sediment analysis.

Analyte	Site	n	Mean (µg/kg)	Minimum (µg/kg)	maximum (µg/kg)	Standard Error	CV
Cadmium	All	10	0.771	0.000	7.600	0.759	311.216
	1	9	0.857	0.010	7.600	0.843	295.186
	4	1	0.000	0.000	0.000		
Copper	All	18	0.539	0.014	1.690	0.125	97.941
	1	10	0.882	0.082	1.690	0.136	48.626
	4	8	0.111	0.014	0.740	0.090	228.084
Iron	All	9	0.522	0.220	0.810	0.063	36.203
	1	2	0.435	0.330	0.540	0.105	34.136
	4	7	0.547	0.220	0.810	0.076	36.883
Lead	All	18	0.043	0.010	0.092	0.006	58.833
	1	10	0.032	0.010	0.092	0.008	78.324
	4	8	0.057	0.020	0.073	0.007	32.975
Nickel	All	18	0.268	0.030	0.540	0.036	56.172
	1	10	0.314	0.030	0.510	0.048	48.514
	4	8	0.212	0.126	0.540	0.049	64.824
Zinc	All	18	2.599	0.207	7.290	0.560	91.366
	1	10	4.065	0.382	7.290	0.665	51.703
	4	8	0.767	0.207	3.370	0.373	137.566
AVS	All	18	0.117	0.000	0.400	0.031	111.945
	1	10	0.048	0.000	0.230	0.029	191.320
	4	8	0.204	0.001	0.400	0.044	61.165
SEM	All	18	3.879	0.389	9.737	0.761	83.210
	1	10	6.063	0.648	9.737	0.785	40.916
	4	8	1.148	0.389	4.670	0.505	124.497
SEM-AVS	All	18	3.762	0.089	9.527	0.781	88.119
	1	10	6.016	0.418	9.527	0.792	41.612
	4	8	0.944	0.089	4.670	0.536	160.482

3.2.3. Deep Ocean Tissue Exposure: measured concentrations of COCs in bioaccumulation studies

Efforts were made during several cruises to capture epibenthic infauna, but too few specimens were collected to quantitatively measure field bioaccumulation of PCBs in tissue. Consequently, this section presents results from bioaccumulation studies that compared tissue residue concentrations of PCOCs and SCOCs in surrogate test organisms exposed to Inner and Outer Ring sediments. Tissues of the bentnose clam *Macoma nasuta* and the polychaete worm *Nephtys caecoides* were chemically analyzed following 28-day laboratory exposures to surface sediments collected from each of nine near-ship stations (Ring 1) and five reference stations (Ring 4) in 1998 and 1999. The PCOCs and SCOCs known to bioaccumulate in food webs were examined: mercury, PCB and PAHs. Bioaccumulation data also are presented for cadmium, chromium, copper, lead, nickel, silver and zinc, which in general, can bioaccumulate but do not biomagnify in higher trophic levels. An overview of sediment and tissue relationships is presented in Section 3.2.3.1. Bioaccumulation of PCBs, PAHs and metals is discussed in Sections 4.2, 4.3 and 4.4, respectively. All chemical concentrations are reported on a dry weight basis. Where analytes were reliably detected at concentrations below the method detection limit (SINKEX "J" qualified data), reported laboratory concentrations were considered valid and used in subsequent statistical analyses. Dry weight tissue results are presented in Appendices M-O.

3.2.3.1. Introduction to bioaccumulation studies involving bentnose clam and polychaete worms

Evaluation of chemical bioaccumulation in clams and worms relies primarily on statistical comparisons of replicated station results. As was the case in the evaluation of sediment chemical data, stations within a common ring are considered replicates. Therefore, bioaccumulation data are compared between background Outer Ring #4 (5 stations) and ex-AGERHOLM Inner Ring #1 (9 stations), with the objective of discerning environmental perturbations as a result of SINKEX activities.

As explained in great detail earlier, Polychlorinated biphenyls (PCBs) were emphasized as the primary chemical of concern, since they are known ship-born contaminants because of their association with PCBs in solid materials. Polynuclear aromatic hydrocarbons (PAHs) were considered important indicators of hydrocarbon contamination sources, measured in sediments and potentially linked to the sunken ship because of their association with fuels and oils. Heavy and trace metals were measured in sediments and potentially linked to the sunken ship because of the fact that the ship is made of metal. Previously discussed in the Methods section, logistical considerations required separate field sampling and subsequent sediment processing into three distinct time periods: September 1998 (Cruise II), September 1999 (Cruise IV) and November 1999 (Cruise V). Bioaccumulation testing is used to evaluate uptake of contaminants by the population of organisms. However, condition and size of the test organisms, and season of testing may potentially influence the uptake of contaminants (Kennish 1992, Phillips 1986).Steps were taken to minimize the influence of testing over time, including controlling organism source, size and condition. However, since the bioaccumulation species examined were collected from the "wild", absolute test parity was not always possible and, as a result, special attention was given to control tissue (taken from test organisms exposed to native sediments) concentrations in the following report sections.

There are different ways to interpret bioaccumulation data, including the following: 1) an estimate of direct uptake from sediment into the test organism (surrogate); and 2) a predictive measure of transfer across trophic levels. When evaluating direct uptake from sediment (as in this study), tissue concentrations can be compared to various criteria that usually are established on a wet or dry weight basis, so data must be converted if they are reported in dissimilar units. In general, if tissue data are used to estimate the amount of contaminant in the test organism as a food source, then wet weight concentrations are generally used, as they represent the unadjusted concentration per mass of prey. Dry weight tissue concentrations are often used when comparing data across species or studies. In this study, summary statistics for bioaccumulation data are presented on a dry weight basis, to provide consistency with raw data presented in the appendices as reported by the laboratories, and to compare with several dry-weight values reported in the literature. Comparisons between Ring 1 and Ring 4 tissue concentrations are made using several standardization/normalization techniques including dry weight, dry-lipid weight basis following guidance from the US Army Corps of Engineers (McFarland, 1995) and log transforms of each. Moisture in the test clam tissue ranged from 83.1-90.44% with a mean of 88.8%; and dry weight lipids ranged from 3.4-10.7%, with a mean concentration of 6.0%. Moisture in the test worm tissue ranged from 81.7-87.4% with a mean of 83.7%; and dry weight lipids ranged from 4.9-12.8%, with a mean concentration of 9.5%. Data were also compared to concentrations reported by the NOAA National Status Program (also referred to as Mussel Watch program) for nearshore southern California mussel collection sites located external to bays and estuaries (NOAA, 1991). Four sites (Point Loma Light House, Point La Jolla, Oceanside Beach Jetty, and Santa Catalina Island Bird Rock) were chosen based on their location in relatively clean (i.e., light to non-industrial pollution sources) ocean areas. The sites chosen are believed to possess the largest number of routinely monitored tissue samples in Southern California and have the lowest concentrations of COCs, when compared to the other four local mussel watch sites. Therefore, these four sites are considered to be "clean local sites" for comparative purposes. These data comparisons are for general perspective, and are not used in hypothesis testing.

3.2.3.2. Overview of associations and correlations among variables

Chemical concentrations in clam and worm tissues were generally very low, as a function of chemical type, sample location and physical features of the exposure sediment. A Pearson's Correlation statistical test was used to examine relationships between sediment and tissue concentrations for PCB21 (21 Green Book, 0 for non-detected congeners), PCB21MDL (21 Green Book using ½ MDL for non-detected congeners), PCB26 (26 PCB congeners, 0 for non-detected congeners), PCB26MDL (26 PCB congeners, ½ MDL for non-detected congeners), high and low molecular weight PAHs, PAH16 (16 Green Book compounds), and PAH41 (sum of all PAH compounds examined). Original and lipid-standardized tissue concentrations for total PCBs and PAHs did not significantly correlate with untransformed sediment concentrations, TOC-normalized sediment concentrations or grain size, suggesting that neither COC concentrations nor sediment physical features influence the direct uptake of these chlorinated compounds or PAHs in biota (Table 3-26).

Sediment-tissue associations between eight metals (Cd, Cr, Cu, Hg, Pb, Ni, Ag, Zn) were similarly independent of the influence of sediment physical parameters. Only cadmium (clam tissue) showed significant correlation between tissue and TOC-standardized sediment concentrations (bold values in Table 3-27).

Table 3-26. Correlation results for organic COC concentrations in lipid-standardized tissue vs. TOC and fines standardized sediment - all data combined (n=16).

Species	Transform	Statistic	Low Molecular Weight PAH	High Molecular Weight PAH	Total PAH (Green Book)	Total PAH (41 compounds)
Macoma	None	r	0.41253	0.1594	0.28756	0.25193
		p	0.1123	0.5554	0.2802	0.3466
		r^2<.5, p≤0.05				
	Standardized (lipid & TOC)	r	0.31686	-0.44233	-0.19544	-0.06171
		p	0.2318	0.0862	0.4682	0.8204
		r^2<.5, p≤0.05				
	Standardized (lipid & fines)	r	0.16808	-0.14915	-0.09577	-0.10700
		p	0.5338	0.5814	0.7242	0.6933
		r^2<.5, p≤0.05				
Nephtys	None	r	-0.19635	-0.33011	-0.29699	-0.22342
		p	0.4661	0.2118	0.2640	0.4055
		r^2<.5, p≤0.05				
	Standardized (lipid & TOC)	r	-0.20105	-0.43531	-0.32853	-0.30040
		p	0.4553	0.0919	0.2141	0.2583
		r^2<.5, p≤0.05				
	Standardized (lipid & fines)	r	-0.07112	-0.15343	-0.13048	-0.16323
		p	0.7935	0.5705	0.6301	0.5458
		r^2<.5, p≤0.05				

Species	Transform	Statistic	Total PCB (Green Book, 0 for ND)	Total PCB (all congeners, 0 for ND)	Total PCB (Green Book, ½ MDL for ND)	Total PCB (all congeners, ½ MDL for ND)
Macoma	None	r	-0.40264	-0.40706	-0.32842	-0.27292
		p	0.1221	0.1176	0.2143	0.3064
		r^2<.5, p≤0.05				
	Standardized (lipid & TOC)	r	-0.14091	-0.16690	-0.12282	-0.11247
		p	0.6027	0.5367	0.6504	0.6784
		r^2<.5, p≤0.05				
	Standardized (lipid & fines)	r	0.13252	0.08729	0.37743	0.37096
		p	0.6247	0.7479	0.1495	0.1572
		r^2<.5, p≤0.05				
Nephtys	None	r	-0.17433	-0.14056	-0.05895	-0.02884
		p	0.5185	0.6036	0.8283	0.9155
		r^2<.5, p≤0.05				
	Standardized (lipid & TOC)	r	-0.29641	-0.27009	-0.27630	-0.25626
		p	0.2650	0.3117	0.3003	0.3381
		r^2<.5, p≤0.05				
	Standardized (lipid & fines)	r	-0.23710	-0.23353	-0.22377	-0.22439
		p	0.3766	0.3840	0.4048	0.4034
		r^2<.5, p≤0.05				

Bold=significant positive correlation at p<0.05.

Table 3-27. Correlation results for metal COC concentrations in lipid-standardized tissue vs. TOC and fines standardized sediment - all data combined (n=15).

Species	Transform	Statistic	Cadmium	Chromium	Copper	Mercury	Lead	Nickel	Silver	Zinc
Macoma	None	r	0.25401	0.62015	0.25098	0.40014	0.21357	0.30855	0.40477	0.01016
		p	0.3610	0.0136	0.3669	0.1394	0.4447	0.2632	0.1345	0.9713
		r^2>.5 and p≤0.05								
	Standardized (lipid & TOC)	r	**0.71521**	-0.05369	0.55376	-0.68323	-0.05852	0.09561	0.10522	0.63824
		p	**0.0027**	0.8493	0.0322	0.0050	0.8359	0.7346	0.7090	0.0105
		r^2>.5 and p≤0.05	**Yes**							
	Standardized (lipid & fines)	r	0.45836	0.21359	0.28034	-0.41946	-0.06077	0.12555	-0.13263	0.44065
		p	0.0857	0.4447	0.3115	0.1196	0.8297	0.6557	0.6375	0.1002
		r^2>.5 and p≤0.05								
Nephtys	None	r	0.18804	0.03196	0.03799	0.34508	0.28971	0.14810	-0.12601	-0.07508
		p	0.5021	0.9100	0.8931	0.2078	0.2949	0.5984	0.6545	0.7903
		r^2>.5 and p≤0.05								
	Standardized (lipid & TOC)	r	0.34347	-0.14128	0.61819	-0.38713	-0.02289	-0.17678	0.64352	0.01327
		p	0.2101	0.6155	0.0140	0.1540	0.9355	0.5285	0.0096	0.9626
		r^2>.5 and ≤0.05								
	Standardized (lipid & fines)	r	0.18975	0.21627	0.35805	0.03982	0.08158	0.08370	0.25684	-0.01891
		p	0.4982	0.4388	0.1901	0.8879	0.7726	0.7668	0.3555	0.9467
		r^2>.5 and p≤0.05								

Bold=significant positive correlation at p<0.05.

3.2.3.3. PCBs in Tissue

Results reported for total PCBs bioaccumulated depend largely on the analytical method and number of congeners analyzed. Total PCB has relied on several calculations, including the summation of Aroclors, subsets of congeners, homologs and the sum of all 209 congeners. Total Green Book PCBs represent the sum of 21 specific PCB congeners, with non-detected congeners given a value of zero. Table 3-28 summarizes the four different ways in which total PCBs were calculated. The least conservative measure in Table 3-28 (producing the lowest total concentration) is generated using Green Book criteria (PCB_{21}), since only 21 of 26 measured congeners are included, and non-detected congeners are excluded. The most conservative measure for total PCBs includes all 26 congeners (PCB_{26MDL}) measured during the SINKEX program and assigns a concentration of ½ the sample specific method detection limit (MDL) to non-detect congeners.

Post-test *Macoma* and *Nephtys* tissue concentrations (see Appendix J) were similar, averaging 20.85 µg/kg and 25.38 µg/kg total Green Book PCBs, respectively. These data are within the range of average concentrations reported by the NOAA National Status Program for nearshore southern California mussel collection sites located in relatively clean (i.e., light to non-industrial) areas (12.27 µg/kg Santa Catalina Island, 26.74 µg/kg Oceanside Breakwater). Figure 3-61 and Figure 3-62 present PCB21 and PCB26 MDL for the two species examined. Station 1-8 had the highest measured PCB21 concentrations measured for both *Macoma* and *Nephtys*, at 39.38 µg/kg and 136.68 µg/kg, respectively. Apparent in both figures is the impression that, with the exception of Station 1-8, there is little difference in total PCB concentration between Ring 1 and Ring 4.

Figure 3-63 and Figure 3-64 compare *Macoma* and *Nephtys* tissue concentrations from source material (Zero Time and Control) for each of the 26 PCB congeners examined. Data are presented for each of the three testing events and correspond to sediments collected during Cruises II, IV and V. Background concentrations (Zero Time) of several congeners are apparent in both test species, and generally appear to increase slightly during the 28-day exposure period with the uncontaminated control sediment. Increases in PCB congener concentrations during the testing procedure suggest two possible PCB contamination sources: 1) from the flow-thru seawater system, or 2) in the test control sediments. Congener concentrations in control sediments ranged from below detection limits to 0.14 µg/kg (PCB-44, Cruise IV) and 0.16 µg/kg (PCB-28, Cruise V), with the sum of all 26 congeners totaling less than 1.0 µg/kg for both control sets (Cruise II control sediments were not examined). These results suggest minimal PCB contribution from control sediments, at least for the sediments collected from the Outer Ring. Flowing seawater used during the test procedure is the most probable contamination source to control tissues. However, since control tissue concentrations are consistently less than test tissue concentrations and are assumed to impose uniform bias in their overall test distribution, they are not considered significant in the following statistical comparisons.

Table 3-28. Summary of total tissue PCBs in bentnose clam *Macoma nasuta* and the polychaete worm *Nephtys caecoides*.

Species	Analyte	Ring	n	Mean (µg/kg)	Minimum (µg/kg)	Maximum (µg/kg)	Standard Error	CV	Maximum Station
Macoma	Total PCB (Green Book 21, ND=0)	1	9	16.81	10.25	39.38	3.05	54.47	1-8 (1998)
		4	5	28.15	17.09	37.31	3.28	26.01	4-2 (1999)
		All	14	20.86	10.25	39.38	2.67	47.91	1-8 (1998)
	Total PCB (Green Book 21, ND=0.5*MDL)	1	9	35.22	21.86	53.03	3.36	28.59	1-8 (1998)
		4	5	37.41	32.67	42.64	2.00	11.93	4-6 (1998)
		All	14	36.00	21.86	53.03	2.23	23.19	1-8 (1998)
	Total Congeners (26, ND=0)	1	9	20.39	13.63	45.94	3.42	50.29	1-8 (1998)
		4	5	30.05	21.45	39.59	2.93	21.82	4-2 (1999)
		All	14	23.84	13.63	45.94	2.69	42.16	1-8 (1998)
	Total Congeners (26, ND=0.5*MDL)	1	9	41.08	23.93	59.59	3.96	28.95	1-8 (1998)
		4	5	40.76	35.03	49.10	2.66	14.59	4-6 (1998)
		All	14	40.96	23.93	59.59	2.64	24.16	1-8 (1998)
Nephtys	Total PCB (Green Book 21, ND=0)	1	9	29.43	9.51	136.68	13.56	138.19	1-8 (1998)
		4	5	18.10	14.42	24.92	1.79	22.08	4-3.1 (1999)
		All	14	25.38	9.51	136.68	8.68	127.93	1-8 (1998)
	Total PCB (Green Book 21, ND=0.5*MDL)	1	9	43.24	15.48	144.83	13.06	90.64	1-8 (1998)
		4	5	22.60	16.57	31.66	2.78	27.49	4-6 (1998)
		All	14	35.87	15.48	144.83	8.71	90.87	1-8 (1998)
	Total Congeners (26, ND=0)	1	9	34.99	12.15	160.78	15.90	136.34	1-8 (1998)
		4	5	20.15	15.61	27.05	1.91	21.16	4-3.1 (1999)
		All	14	29.69	12.15	160.78	10.21	128.73	1-8 (1998)
	Total Congeners (26, ND=0.5*MDL)	1	9	50.47	16.63	170.23	15.42	91.64	1-8 (1998)
		4	5	25.16	18.15	37.11	3.45	30.68	4-6 (1998)
		All	14	41.43	16.63	170.23	10.33	93.27	1-8 (1998)

CV= coefficient of variation

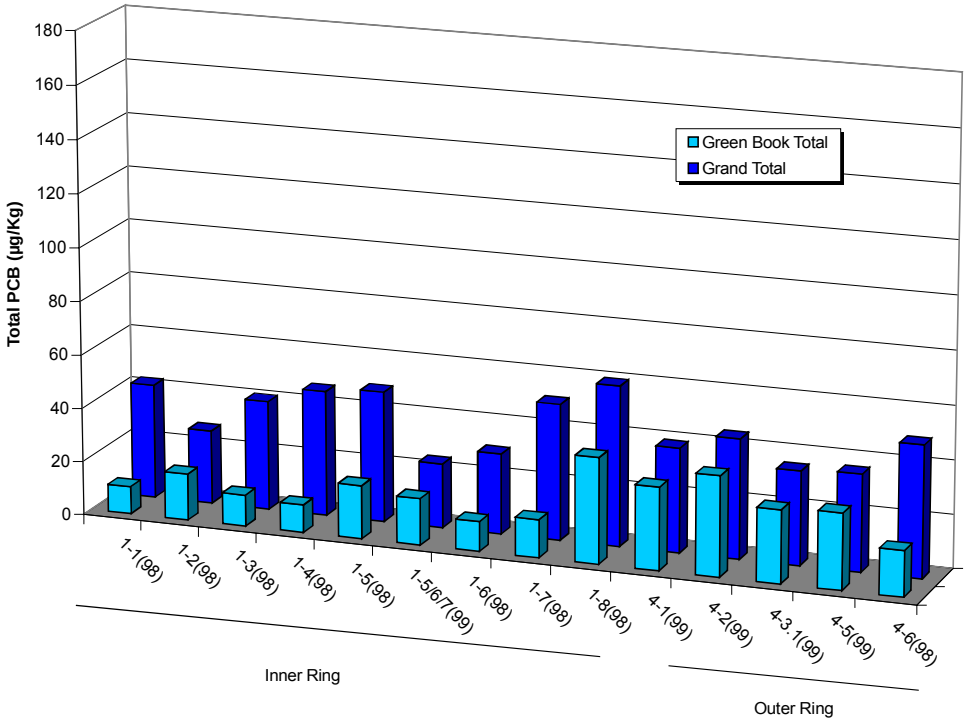


Figure 3-61. PCB_{21} (Green Book) and PCB_{26MDL} (Grand Total) congeners measured in *Macoma nasuta* tissues (µg/kg dry weight).

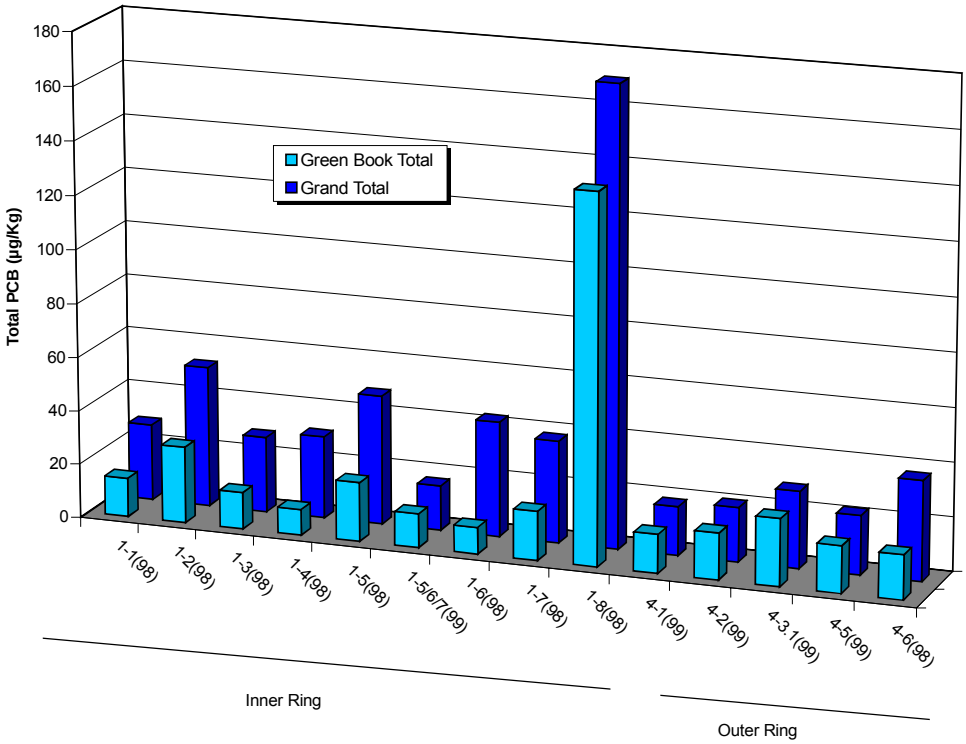
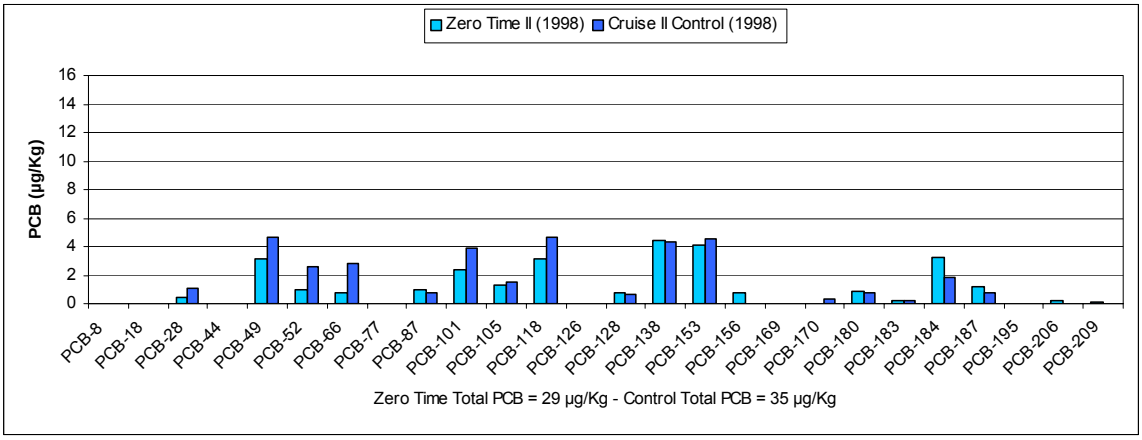
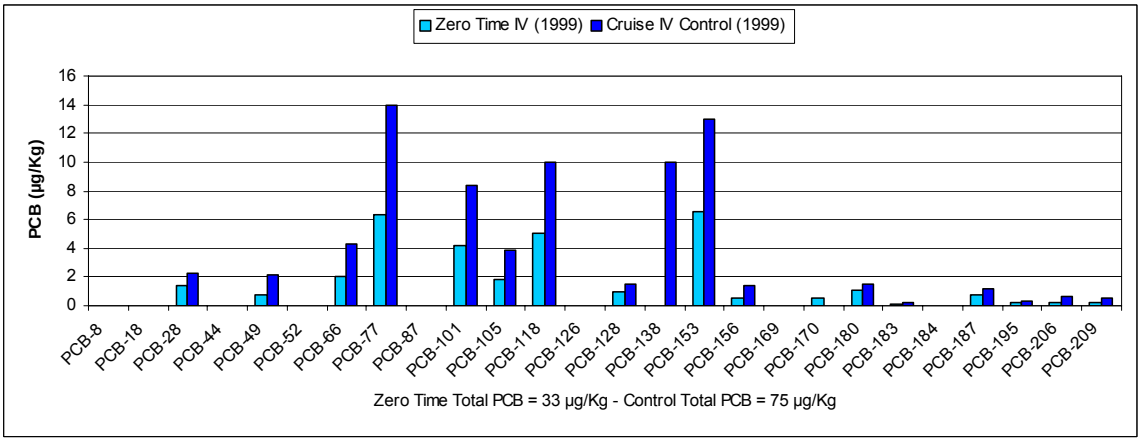


Figure 3-62. PCB_{21} (Green Book) and PCB_{26MDL} (Grand Total) congeners measured in *Nephtys caecoides* tissues (µg/kg dry weight).





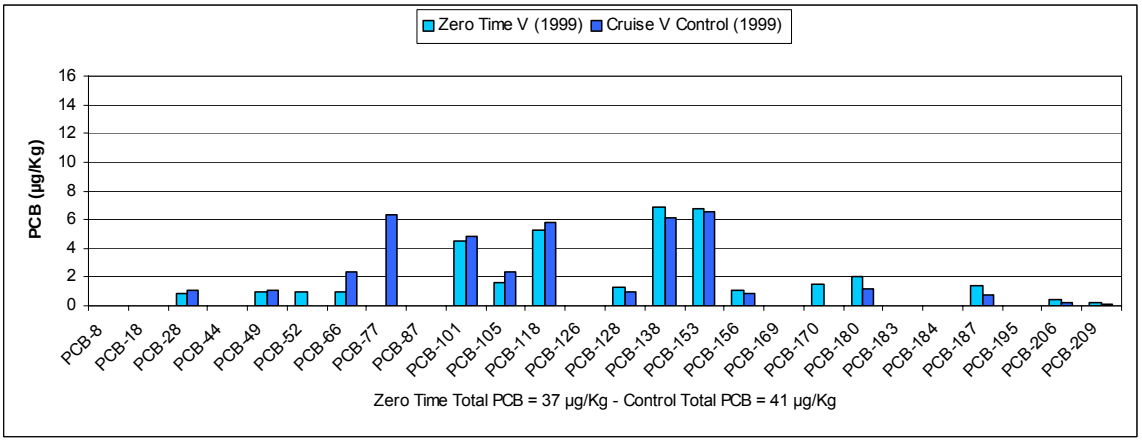


Figure 3-63. Comparison of *Macoma* tissue PCB congener distributions between zero time (initial) and control (28-day exposure) (µg/kg dry weight).

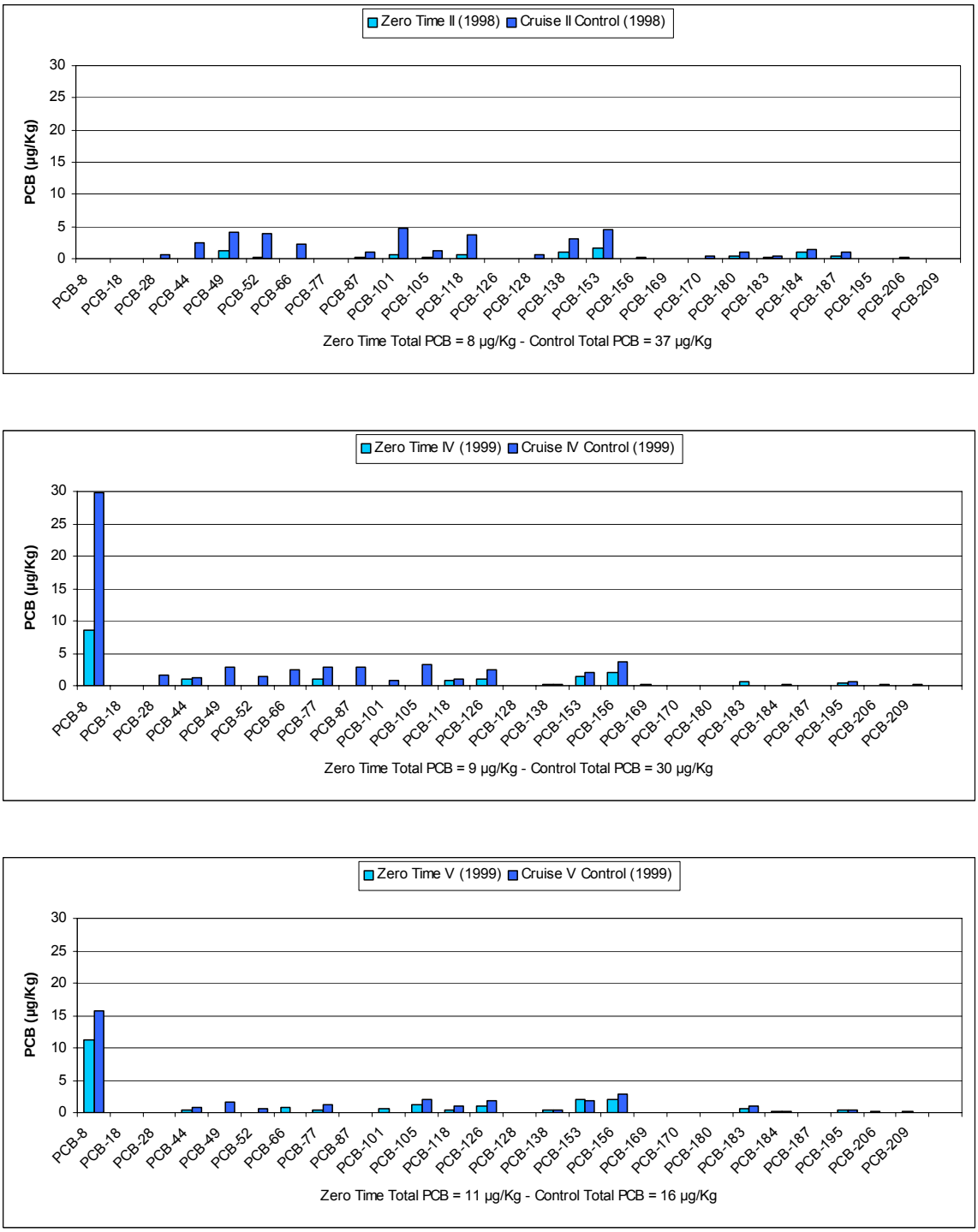


Figure 3-64. Comparison of *Nephtys* tissue PCB congener distributions between zero time (initial) and control (28-day exposure) (µg/kg dry weight).

For these bioaccumulation tests, the experimental design permitted a statistical comparison between near-ship samples (Inner-Ring) and reference samples (Outer-Ring). PCB_{21} and PCB_{26MDL} distributions were evaluated for normality using the Shapiro-Wilks test procedure prior to hypothesis testing. Non-normal distributions were subjected to three normalization/

standardization procedures (log transformation, ratio of total PCB to fractional lipid content, and the log of the ratio of total PCB to fractional lipid content). If the distribution remained "non-normal," a non-parametric hypothesis test was applied to the untransformed data. Table 3-29 and Table 3-30 present the results of normality and the 2-way ANOVA testing performed on PCB_{21} and PCB_{26MDL} measured in *Macoma* and *Nephtys* tissues.

The distribution of PCB concentrations for *Macoma* proved to be normal (Shapiro-Wilks p value > 0.05). Inter-Ring comparisons of PCB_{21} did not disprove the null hypothesis, and thus the ex-AGERHOLM and reference areas are not significantly different at the 0.05 level (p=0.316). Similarly, inference testing of differences between Rings 1 and 4 of PCB_{26MDL} proved insignificant at the p=0.05 level (p=0.178). The highest concentration measured for PCB_{26MDL}, was found at Inner-Ring station 1-8 (59.59 µg/kg). The second highest concentration occurred at Outer-Ring station 4-6, at 49.10 µg/kg.

Normality testing of *Nephtys* PCB totals indicates that Inner and Outer Ring tissue concentrations are non-normally distributed (Shapiro-Wilks, p<0.05). Additionally, log, lipid and log-lipid transformations failed to normalize the distributions, dictating the use of the non-parametric Kruskal-Wallis hypothesis test. Table 3-30 highlights the Kruskal-Wallis test on rank transformed data (raw rankings="Rank"). Inner-Ring comparisons of PCB_{21} did not disprove the null hypothesis, and thus, ex-AGERHOLM and reference areas are not significantly different at the 0.05 level. Mirroring the *Macoma* results, the highest *Nephtys* tissue concentration measured for PCB_{26MDL} was found at Inner-Ring station 1-8 (170.23 µg/kg). Also tracking *Macoma* results, the second highest concentration occurred at Outer-Ring Station 4-6, at 37.11 µg/kg.

Table 3-29. Two-way comparison of SINKEX Rings (1 and 4, cruises 1998-1999) for *Macoma* tissue PCBs.

	GLM Hypothesis Test Results				Normality Test Results		
Total PCB Measurement	Transformation Standardization Method	Inner-Ring Comparison (different if <0.05)	Inter-Cruise Comparison (different if <0.05)	Ring-Cruise Interaction (important if <0.05)	Shapiro-Wilks Value	Probability of Non-Normal Distribution (non-normal if <0.05)	Distribution
PCB_{21}	Lipid	0.2523	0.0845	0.4177	0.8623	0.0328	Non-Normal
	Log Concentration	0.2743	0.2546	0.4815	0.9216	0.2321	Normal
	Log Lipid	0.2622	0.0826	0.7859	0.9102	0.1589	Normal
	None	**0.3164**	**0.2989**	**0.3401**	**0.8869**	**0.0730**	**Normal**
	Rank	0.5060	0.1912	0.5902	0.9645	0.7964	Normal
PCB_{26MDL}	Lipid	0.2641	0.5948	0.8391	0.9505	0.5686	Normal
	Log Concentration	0.1035	0.0457	0.3874	0.9622	0.7592	Normal
	Log Lipid	0.2585	0.7164	0.9158	0.8972	0.1027	Normal
	None	**0.1784**	**0.0635**	**0.5475**	**0.9832**	**0.9891**	**Normal**
	Rank	0.1822	0.0460	0.8054	0.9645	0.7964	Normal

Table 3-30. Two-way comparison of SINKEX Rings (1 and 4, cruises 1998-1999) for *Nephtys* Tissue PCBs.

	GLM Hypothesis Test Results				Normality Test Results		
Total PCB Measurement	Transformation Standardization Method	Inner-Ring Comparison (different if <0.05)	Inter-Cruise Comparison (different if <0.05)	Ring-Cruise Interaction (important if <0.05)	Shapiro-Wilks Value	Probability of Non-Normal Distribution (non-normal if <0.05)	Distribution

PCB_{21}	Lipid	0.7513	0.6781	0.6073	0.6266	0.0001	Non-Normal
	Log Concentration	0.8886	0.7404	0.6236	0.7575	0.0016	Non-Normal
	Log Lipid	0.9021	0.6933	0.5435	0.8495	0.0219	Non-Normal
	None	0.8728	0.7661	0.7139	0.4449	0.0000	Non-Normal
	Rank	**0.4646**	**0.6988**	**0.3867**	**0.9645**	**0.7964**	**Normal**
PCB_{26MDL}	Lipid	0.6396	0.3022	0.5478	0.7479	0.0012	Non-Normal
	Log Concentration	0.9180	0.0777	0.5625	0.8657	0.0365	Non-Normal
	Log Lipid	0.8165	0.1198	0.5230	0.8409	0.0167	Non-Normal
	None	0.8483	0.4083	0.7144	0.5613	0.0000	Non-Normal
	Rank	**0.6536**	**0.0129**	**0.4187**	**0.9645**	**0.7964**	**Normal**

Values in **bold** provide best normalization of raw data.

3.2.3.4. PCB Congeners

Table 3-31 and Table 3-32 summarize *Macoma* and *Nephtys* tissue PCB congener concentrations for Ring 1, Ring 4 and both rings combined (All). Congeners not detected during analysis were assigned a value of one half the sample specific MDL. Sample-specific MDLs were very low, varying by species, sample and congener, ranging from 0.043 µg/kg (congener 49) to 35.0 µg/kg (congener 138) for *Macoma,* and 0.15 µg/kg (congener 49) to 31.0 µg/kg (congener 138) for *Nephtys*. Tissue method detection limits were higher than those for associated sediments due to the smaller amounts of the tissue samples. A discussion of detection limits is presented in Section 2.5.6.4.

A total of 728 individual congener-station combinations (26 congeners x 14 stations x 2 species) were examined. Even considering the low detection limits achieved during this investigation, 292 measurements (40%) fell below sample specific MDLs. Across all bioaccumulation stations, only eleven of the 26 congeners analyzed (congeners 44, 49, 52, 77, 87, 101, 105, 118, 138, 153, 209) had concentrations detected greater than 5.0 µg/kg. Individual congener tissue concentrations for *Macoma* exceeded 5.0 µg/kg at 4 station stations (1-8, 4-1, 4-2, 4-5), with the highest values detected at stations 1-8 (congener 153 or 2,2′,4,4′,5,5′-Hexachloropiphenyl - 6.2 µg/kg) and 4-2 (congener 77 or 3,3′,4,4′-Tetrachloro-biphenyl - 6.1 µg/kg). *Nephtys* exceeded 5.0 µg/kg at only one station (1-8), with the highest values detected of 29.0 µg/kg for congener 101 (i.e., 2,2′,4,5,5′-Pentachloropiphenyl).

Fourteen of the 209 PCB congeners, as defined by the World Health Organization (WHO, 1997) commonly termed “co-planar” or "dioxin-like" congeners, are suspected to exert toxicity similar to dioxins and furans (congeners 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, 170, 180, 189). Eight environmentally common co-planar congeners were examined in the tissues of *Macoma* and *Nephtys* (congeners 77, 105, 118, 126, 156, 169, 170, 180). Only four co-planar congeners were detected at concentrations above 5.0 µg/kg. Congener 77 was measured in *Macoma* at reference stations 4-1 and 4-2 (5.1 and 6.1 µg/kg, respectively); and also in *Macoma*, congener 118 was measured at 5.6 µg/kg. Of all Inner-Ring stations, only Station 1-8 had “co-planar” congeners elevated above 5.0 µg/kg (congener 105 at 11.0 µg/kg and congener 118 at 26 µg/kg).

Table 3-31. Summary statistics for 26 PCB congener concentrations in *Macoma nasuta* for Ring 1, Ring 4 and all stations combined (mg/kg).

Analyte	Ring	n	Mean (µg/kg)	Minimum (µg/kg)	Maximum (µg/kg)	Standard Error	CV	Maximum Station

Analyte	Ring	n	Mean (µg/kg)	Minimum (µg/kg)	Maximum (µg/kg)	Standard Error	CV	Maximum Station
8 - 2,4'-Dichlorobiphenyl	1	9	1.07	0.55	1.40	0.10	28.94	1-4 (1998)
	4	5	0.82	0.65	1.35	0.13	36.23	4-6 (1998)
	ALL	14	0.98	0.55	1.40	0.09	32.54	1-4 (1998)
18 - 2,2',5-Trichlorobiphenyl	1	9	1.31	0.24	1.85	0.20	45.60	1-4 (1998)
	4	5	0.72	0.44	1.80	0.27	84.45	4-6 (1998)
	ALL	14	1.10	0.24	1.85	0.17	58.92	1-4 (1998)
28 - 2,4,4'-Trichlorobiphenyl	1	9	0.67	0.18	1.65	0.20	88.05	1-7 (1998)
	4	5	1.29	0.54	2.10	0.25	43.69	4-2 (1999)
	ALL	14	0.89	0.18	2.10	0.17	71.59	4-2 (1999)
44 - 2,2',3,5'-Tetrachlorobiphenyl	1	9	0.88	0.40	1.40	0.14	46.85	1-5 (1998)
	4	5	1.52	1.45	1.60	0.03	3.75	4-1 (1999)
	ALL	14	1.11	0.40	1.60	0.12	40.92	4-1 (1999)
49 - 2,2',4,5'-Tetrachlorobiphenyl	1	9	1.30	0.48	1.90	0.14	33.15	1-8 (1998)
	4	5	1.11	0.78	1.90	0.21	42.34	4-6 (1998)
	ALL	14	1.23	0.48	1.90	0.12	35.45	1-8 (1998)
52 - 2,2',5,5'-Tetrachlorobiphenyl	1	9	1.62	1.00	2.80	0.23	42.24	1-8 (1998)
	4	5	1.41	1.20	1.80	0.12	19.49	4-6 (1998)
	ALL	14	1.55	1.00	2.80	0.15	36.77	1-8 (1998)
66 - 2,3',4,4'-Tetrachlorobiphenyl	1	9	0.81	0.39	1.60	0.12	45.53	1-5/6/7 (99)
	4	5	1.60	1.30	1.90	0.11	15.31	4-2 (1999)
	ALL	14	1.10	0.39	1.90	0.14	46.17	4-2 (1999)
77 - 3,3',4,4'-Tetrachlorobiphenyl	1	9	2.44	0.90	3.30	0.29	35.02	1-4 (1998)
	4	5	4.78	3.20	6.10	0.47	22.11	4-2 (1999)
	ALL	14	3.28	0.90	6.10	0.39	44.65	4-2 (1999)
87 - 2,2',3,4,5'-Pentachlorobiphenyl	1	9	0.73	0.32	1.40	0.12	48.88	1-8 (1998)
	4	5	0.59	0.50	0.65	0.02	9.28	4-1 (1999)
	ALL	14	0.68	0.32	1.40	0.08	42.51	1-8 (1998)
101 - 2,2',4,5,5'-Pentachlorobiphenyl	1	9	2.60	1.40	5.20	0.41	47.22	1-8 (1998)
	4	5	3.36	2.60	4.40	0.31	20.36	4-2 (1999)
	ALL	14	2.87	1.40	5.20	0.29	38.38	1-8 (1998)
105 - 2,3,3',4,4'-Pentachlorobiphenyl	1	9	0.75	0.36	1.20	0.10	39.92	1-8 (1998)
	4	5	1.54	1.00	1.80	0.15	21.34	4-3.1 (99)
	ALL	14	1.03	0.36	1.80	0.13	47.61	4-3.1 (99)
118 - 2,3',4,4',5-Pentachlorobiphenyl	1	9	2.01	1.20	4.30	0.31	45.74	1-8 (1998)
	4	5	4.12	2.60	5.60	0.48	26.28	4-2 (1999)
	ALL	14	2.76	1.20	5.60	0.38	50.92	4-2 (1999)
126 - 3,3',4,4',5-Pentachlorobiphenyl	1	9	1.97	0.29	2.70	0.27	41.70	1-4 (1998)
	4	5	0.77	0.31	2.60	0.46	132.37	4-6 (1998)
	ALL	14	1.54	0.29	2.70	0.28	67.73	1-4 (1998)
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	1	9	3.15	0.58	4.35	0.43	40.55	1-4 (1998)
	4	5	1.33	0.48	4.20	0.72	120.63	4-6 (1998)
	ALL	14	2.50	0.48	4.35	0.43	64.68	1-4 (1998)
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	1	9	2.28	1.00	4.20	0.33	43.34	1-8 (1998)
	4	5	4.08	2.80	5.00	0.42	23.15	4-2 (1999)
	ALL	14	2.92	1.00	5.00	0.35	44.33	4-2 (1999)
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	1	9	3.13	2.00	6.20	0.43	41.64	1-8 (1998)
	4	5	5.08	3.20	6.00	0.53	23.18	4-1 (1999)
	ALL	14	3.83	2.00	6.20	0.42	40.56	1-8 (1998)
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	1	9	1.35	0.16	2.20	0.28	63.19	1-4 (1998)
	4	5	0.79	0.36	2.10	0.34	95.58	4-6 (1998)
	ALL	14	1.15	0.16	2.20	0.22	72.86	1-4 (1998)
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	1	9	1.61	0.35	2.20	0.21	39.71	1-4 (1998)
	4	5	0.72	0.37	2.10	0.34	106.70	4-6 (1998)
	ALL	14	1.29	0.35	2.20	0.21	61.32	1-4 (1998)
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	1	9	1.26	0.21	2.15	0.28	67.41	1-4 (1998)
	4	5	0.78	0.44	2.10	0.33	94.81	4-6 (1998)
	ALL	14	1.09	0.21	2.15	0.22	75.14	1-4 (1998)
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	1	9	1.43	0.36	3.45	0.43	89.92	1-4 (1998)
	4	5	0.86	0.39	1.30	0.15	38.45	4-1 (1999)

Analyte	Ring	n	Mean (μg/kg)	Minimum (μg/kg)	Maximum (μg/kg)	Standard Error	CV	Maximum Station
	ALL	14	1.23	0.36	3.45	0.28	86.73	1-4 (1998)
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	1	9	1.16	0.22	2.20	0.29	75.03	1-4 (1998)
	4	5	0.31	0.28	0.36	0.02	13.08	4-6 (1998)
	ALL	14	0.86	0.22	2.20	0.21	93.68	1-4 (1998)
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	1	9	1.33	0.27	1.90	0.18	40.94	1-5 (1998)
	4	5	0.55	0.28	1.60	0.26	106.73	4-6 (1998)
	ALL	14	1.05	0.27	1.90	0.18	63.00	1-5 (1998)
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	1	9	1.10	0.35	3.50	0.32	86.63	1-8 (1998)
	4	5	0.56	0.34	0.95	0.11	42.23	4-2 (1999)
	ALL	14	0.91	0.34	3.50	0.22	88.94	1-8 (1998)
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	1	9	1.99	0.39	3.05	0.34	51.80	1-4 (1998)
	4	5	0.88	0.16	2.95	0.52	132.38	4-6 (1998)
	ALL	14	1.59	0.16	3.05	0.31	73.61	1-4 (1998)
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	1	9	1.64	0.14	2.55	0.30	54.72	1-4 (1998)
	4	5	0.71	0.25	2.45	0.44	137.03	4-6 (1998)
	ALL	14	1.31	0.14	2.55	0.27	76.49	1-4 (1998)
209 - Decachlorobiphenyl	1	9	1.49	0.08	5.60	0.53	107.25	1-8 (1998)
	4	5	0.48	0.14	1.35	0.23	106.43	4-6 (1998)
	ALL	14	1.13	0.08	5.60	0.37	122.14	1-8 (1998)

NOTE: Non-detected congeners processed as ½ MDL. Congeners used in calculating PCB21 are shaded.

Table 3-32. Summary statistics for 26 PCB congener concentrations in *Nephtys caecoides* for Ring 1, Ring 4 and all stations combined (mg/kg).

Analyte	Ring	N	Mean (µg/kg)	Minimum (µg/kg)	Maximum (µg/kg)	Standard Error	CV	Maximum Station
8 - 2,4'-Dichlorobiphenyl	1	9	0.90	0.44	1.25	0.10	32.05	1-2 (1998)
	4	5	0.43	0.22	0.85	0.11	59.59	4-6 (1998)
	ALL	14	0.73	0.22	1.25	0.09	48.61	1-2 (1998)
18 - 2,2',5-Trichlorobiphenyl	1	9	1.16	0.29	1.80	0.19	48.01	1-8 (1998)
	4	5	0.37	0.14	1.10	0.18	111.16	4-6 (1998)
	ALL	14	0.88	0.14	1.80	0.17	71.80	1-8 (1998)
28 - 2,4,4'-Trichlorobiphenyl	1	9	0.63	0.17	1.55	0.15	73.39	1-6 (1998)
	4	5	0.77	0.24	1.60	0.23	68.16	4-3.1 (1999)
	ALL	14	0.68	0.17	1.60	0.13	69.11	4-3.1 (1999)
44 - 2,2',3,5'-Tetrachlorobiphenyl	1	9	1.64	0.34	6.90	0.67	122.86	1-8 (1998)
	4	5	1.94	1.30	3.20	0.35	40.36	4-3.1 (1999)
	ALL	14	1.75	0.34	6.90	0.44	94.28	1-8 (1998)
49 - 2,2',4,5'-Tetrachlorobiphenyl	1	9	1.65	0.15	5.40	0.50	90.94	1-8 (1998)
	4	5	0.91	0.43	1.60	0.19	47.91	4-6 (1998)
	ALL	14	1.39	0.15	5.40	0.34	90.71	1-8 (1998)
52 - 2,2',5,5'-Tetrachlorobiphenyl	1	9	3.10	0.80	15.00	1.50	145.08	1-8 (1998)
	4	5	1.73	0.42	4.30	0.71	91.46	4-3.1 (1999)
	ALL	14	2.61	0.42	15.00	0.99	141.63	1-8 (1998)
66 - 2,3',4,4'-Tetrachlorobiphenyl	1	9	1.10	0.56	2.00	0.19	53.07	1-8 (1998)
	4	5	1.53	0.96	2.60	0.30	43.25	4-3.1 (1999)
	ALL	14	1.26	0.56	2.60	0.17	49.83	4-3.1 (1999)
77 - 3,3',4,4'-Tetrachlorobiphenyl	1	9	2.06	0.65	2.90	0.25	36.21	1-2 (1998)
	4	5	1.16	0.32	2.00	0.31	59.36	4-6 (1998)
	ALL	14	1.74	0.32	2.90	0.22	47.64	1-2 (1998)
87 - 2,2',3,4,5'-Pentachlorobiphenyl	1	9	2.19	0.32	14.00	1.48	203.51	1-8 (1998)
	4	5	0.56	0.22	0.69	0.09	35.31	4-3.1 (1999)
	ALL	14	1.61	0.22	14.00	0.96	223.20	1-8 (1998)
101 - 2,2',4,5,5'-Pentachlorobiphenyl	1	9	5.70	1.40	29.00	2.93	154.34	1-8 (1998)
	4	5	3.62	3.00	4.30	0.26	16.27	4-3.1 (1999)
	ALL	14	4.96	1.40	29.00	1.87	140.93	1-8 (1998)
105 - 2,3,3',4,4'-Pentachlorobiphenyl	1	9	2.07	0.37	11.00	1.12	162.62	1-8 (1998)
	4	5	0.79	0.51	1.10	0.11	30.39	4-5 (1999)
	ALL	14	1.62	0.37	11.00	0.73	168.51	1-8 (1998)
118 - 2,3',4,4',5-Pentachlorobiphenyl	1	9	4.64	1.00	26.00	2.70	174.35	1-8 (1998)
	4	5	1.80	1.40	2.30	0.15	18.84	4-3.1 (1999)
	ALL	14	3.63	1.00	26.00	1.74	179.42	1-8 (1998)
126 - 3,3',4,4',5-Pentachlorobiphenyl	1	9	1.66	0.20	2.40	0.23	42.22	1-2 (1998)
	4	5	0.45	0.10	1.65	0.30	150.77	4-6 (1998)
	ALL	14	1.23	0.10	2.40	0.24	73.18	1-2 (1998)
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	1	9	2.45	0.36	3.85	0.45	54.57	1-6 (1998)
	4	5	0.72	0.13	2.65	0.48	149.76	4-6 (1998)
	ALL	14	1.83	0.13	3.85	0.40	80.87	1-6 (1998)
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	1	9	4.36	0.94	22.00	2.23	153.60	1-8 (1998)
	4	5	1.80	1.40	2.00	0.11	13.61	4-6 (1998)
	ALL	14	3.45	0.94	22.00	1.45	156.93	1-8 (1998)

Analyte	Ring	N	Mean (µg/kg)	Minimum (µg/kg)	Maximum (µg/kg)	Standard Error	CV	Maximum Station
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	1	9	3.53	0.91	9.00	0.79	67.24	1-8 (1998)
	4	5	2.58	2.00	3.20	0.24	20.58	4-5 (1999)
	ALL	14	3.19	0.91	9.00	0.52	60.94	1-8 (1998)
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	1	9	1.55	0.23	3.10	0.26	51.22	1-8 (1998)
	4	5	0.46	0.22	1.35	0.22	108.62	4-6 (1998)
	ALL	14	1.16	0.22	3.10	0.23	75.04	1-8 (1998)
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	1	9	1.35	0.24	1.95	0.18	39.73	1-6 (1998)
	4	5	0.41	0.12	1.35	0.24	128.33	4-6 (1998)
	ALL	14	1.01	0.12	1.95	0.19	68.32	1-6 (1998)
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	1	9	1.05	0.19	2.00	0.26	74.55	1-8 (1998)
	4	5	0.43	0.14	1.30	0.22	115.27	4-6 (1998)
	ALL	14	0.83	0.14	2.00	0.20	89.40	1-8 (1998)
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	1	9	1.17	0.43	4.80	0.46	117.29	1-8 (1998)
	4	5	0.45	0.27	0.74	0.09	42.45	4-6 (1998)
	ALL	14	0.91	0.27	4.80	0.30	124.97	1-8 (1998)
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	1	9	0.59	0.17	1.95	0.23	114.92	1-6 (1998)
	4	5	0.20	0.14	0.29	0.03	28.94	4-6 (1998)
	ALL	14	0.45	0.14	1.95	0.15	125.86	1-6 (1998)
184 – 2,2',3,4,4',6,6'-Heptachlorobiphenyl	1	9	1.25	0.19	1.80	0.16	37.34	1-6 (1998)
	4	5	0.43	0.09	1.60	0.29	154.34	4-6 (1998)
	ALL	14	0.96	0.09	1.80	0.18	69.02	1-6 (1998)
187 – 2,2',3,4',5,5',6-Heptachlorobiphenyl	1	9	1.20	0.26	4.50	0.43	108.53	1-8 (1998)
	4	5	0.51	0.32	0.92	0.11	48.57	4-6 (1998)
	ALL	14	0.95	0.26	4.50	0.29	113.97	1-8 (1998)
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	1	9	1.70	0.05	2.70	0.32	57.10	1-2 (1998)
	4	5	0.48	0.06	1.85	0.34	158.99	4-6 (1998)
	ALL	14	1.26	0.05	2.70	0.28	83.91	1-2 (1998)
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	1	9	1.03	0.13	2.25	0.24	69.28	1-6 (1998)
	4	5	0.36	0.05	1.55	0.30	182.68	4-6 (1998)
	ALL	14	0.79	0.05	2.25	0.20	94.39	1-6 (1998)
209 - Decachlorobiphenyl	1	9	0.73	0.13	1.25	0.11	45.32	1-6 (1998)
	4	5	0.27	0.03	0.85	0.15	126.19	4-6 (1998)
	ALL	14	0.57	0.03	1.25	0.11	69.89	1-6 (1998)

NOTE: Non-detected congeners processed as ½ MDL. Congeners used in calculating PCB_{21} are shaded.

3.2.3.5. Tissue PCB Summary

Laboratory bioaccumulation tests were conducted on two common benthic test species, the bentnose clam *Macoma nasuta* and the polychaete worm *Nephtys caecoides*. Comparisons of PCB total dry weight tissue burdens did not significantly differ between Inner and Outer Rings ($p>0.05$). To place the average tissue PCB concentrations in a local context, several long-term coastal monitoring programs were also reviewed. Since 1986, the Mussel Watch Project of the National Oceanic and Atmospheric Administration (NOAA) has chemically analyzed mussels and oysters collected annually from coastal sites throughout the nation. Tissue body burdens in the bivalve *Mytilus* sp. have been monitored at four nearshore southern California sites located within 125 miles of the ex-AGERHOLM since 1986 (Point Loma Light House, Point La Jolla, Oceanside Beach Jetty, Santa Catalina Island Bird Rock). As explained previously, these

locations provide background or reference bioaccumulation levels for southern California areas exposed to the open ocean and not confined within the immediate vicinity of industrial waste streams.

Table 3-33 summarizes Mussel Watch data for eight of the total PCBs (sum of 18 congeners, 0 for non-detected values), total PAH compounds (sum of 16 compounds, 0 for non-detected values) and seven metals for samples collected between 1986 and 1998. Total PCBs calculated for the Mussel Watch Project contain Green Book congeners summed in PCB21, with the exception of congeners 77, 126 and 169. Considering all four monitoring sites, total PCB minimum and maximum dry weight concentrations ranged from 0.4 (Santa Catalina Island) to 98.9 µg/kg (Point Loma Lighthouse) and encompass all measured Green Book total PCB *Macoma* and *Nephtys* tissue concentrations for Inner Ring and Outer Ring SINKEX stations, with the exception of Station 1-8 (138.19 µg/kg, Table 3-28). Average Mussel Watch total PCBs ranged from 12.3 (Santa Catalina Island) to 60.6 µg/kg (Point Loma Lighthouse) and also encompassed all PCB_{21} concentrations, with the exception of *Nephtys* tissue results at Station 1-8.

Station 1-8 is located at the north-west quadrant of the sampling grid in close proximity to an incomplete separation of the rear 1/3 of the vessel from the rest of the hull, where approximately 30 meters of the aft portion rises vertically from the seafloor. It is not unexpected that the highest concentrations of tissue PCBs would be found in this area, since the very large opening in the hull provides increased ocean access to the internal structures. Of note, Station 1-7, also near the hull breach, has PCB concentrations much closer to Ring 4 stations. There are two possible and related explanations for the observed difference in these proximally close tissue PCB stations: 1) the heterogeneous distribution of PCBs within the sediment, and 2) sediment sampling methods. PCBs associated with the ex-AGERHOLM are sequestered in what appear to be reasonably stable "compartments" (e.g., PCB-saturated felt gasket material used to isolate large steal bulkheads within the hull).

When the ex-AGERHOLM hit the seafloor and broke open the after part of the ship, it is possible that some of this material fragmented and was buried in the sediments closest to the ship, creating isolated hotspots. Sediments obtained for chemical analyses at each station were taken from a single bottom sample, using a 0.05 m^2 sampling device. Sediment samples taken for bioaccumulation testing were collected with the same device; however, since a minimum of 11L of material for the bioaccumulation testing was required, 10 to 14 composite samples (grabs) were taken at each station. Assuming that some portion of the PCBs near the hull break are associated with PCB "enriched" ship fragments, there is a reasonable expectation that bioaccumulation organisms will be exposed to material not present in the single sediment chemistry sample.

Lastly, as recommended in both the Green Book (USEPA/USACE, 1991) and the Inland Testing Manual (USEPA/USACE, 1998), a comparison to U.S. Food and Drug Administration (FDA) action levels for poisonous and deleterious substances in fish and shellfish for human food (21 CFR 109.30), shows tissue dry weight measurements for total PCBs to be substantially below the FDA tolerance level of 2.0 mg/kg (2000 µg/kg wet weight, using a typical multiple of 5, estimated to be 10,000 µg/kg dry weight). The highest tissue value recorded (Station 1-8, 136.68 µg/kg wet weight) is only 1.4 percent of the FDA limit as dry weight).

In summary, 1) there is no statistically significant difference between Inner-Ring and Outer-Ring stations; 2) 27 of 28 bioaccumulation tests are within the range of published values for relatively low impact coastal areas; and 3) the highest measured PCB concentrations that have bioaccumulated are well below FDA shellfish limits.

Table 3-33. Summary of NOAA Status and Trends Mussel Watch tissue data for selected analytes, 1986-1999.

Mussel Watch Site	Analyte	Units	Number of Samples	Minimum	Maximum	Mean	Standard Deviation
Point La Jolla	Total PCB	µg/kg (dry)	14	10.100	54.760	20.557	12.618
	Total PAH	µg/kg (dry)	14	16.000	59.000	31.749	13.214
	Cadmium	mg/kg (dry)	21	0.518	3.000	1.735	0.595
	Copper	mg/kg (dry)	22	5.200	12.400	7.952	1.505
	Chromium	mg/kg (dry)	22	0.965	4.010	1.755	0.718
	Lead	mg/kg (dry)	22	1.004	7.000	3.512	1.771
	Mercury	mg/kg (dry)	22	0.065	0.420	0.182	0.087
	Nickel	mg/kg (dry)	22	0.842	2.200	1.366	0.388
	Silver	mg/kg (dry)	6	0.670	7.636	3.074	2.480
	Zinc	mg/kg (dry)	22	73.202	200.000	148.282	35.640
Point Loma Lighthouse	Total PCB	µg/kg (dry)	15	35.690	98.900	60.562	15.966
	Total PAH	µg/kg (dry)	14	35.390	2070.000	328.166	561.978
	Cadmium	mg/kg (dry)	21	0.740	2.400	1.550	0.450
	Copper	mg/kg (dry)	21	9.880	19.000	12.771	2.586
	Chromium	mg/kg (dry)	21	1.100	3.570	2.197	0.869
	Lead	mg/kg (dry)	21	0.900	4.100	2.242	0.919
	Mercury	mg/kg (dry)	21	0.200	0.760	0.372	0.177
	Nickel	mg/kg (dry)	21	0.400	4.800	2.254	1.043
	Silver	mg/kg (dry)	5	4.940	33.757	18.295	12.144
	Zinc	mg/kg (dry)	21	118.065	280.000	202.765	34.751
Oceanside Beach Jetty	Total PCB	µg/kg (dry)	16	15.830	43.600	26.737	8.508
	Total PAH	µg/kg (dry)	11	20.050	115.600	48.021	25.183
	Cadmium	mg/kg (dry)	22	0.445	2.900	1.583	0.843
	Copper	mg/kg (dry)	22	4.900	13.800	7.944	2.118
	Chromium	mg/kg (dry)	22	0.880	4.250	1.900	0.773
	Lead	mg/kg (dry)	22	0.390	2.600	0.980	0.532
	Mercury	mg/kg (dry)	20	0.010	0.150	0.064	0.030
	Nickel	mg/kg (dry)	21	0.820	5.200	1.941	1.321
	Silver	mg/kg (dry)	5	0.092	0.281	0.210	0.072
	Zinc	mg/kg (dry)	22	74.050	240.000	152.320	37.594
Santa Catalina Island Bird Rock	Total PCB	µg/kg (dry)	13	0.400	31.480	12.274	8.678
	Total PAH	µg/kg (dry)	14	5.610	1560.000	243.427	478.908
	Cadmium	mg/kg (dry)	19	1.200	7.700	4.026	1.626
	Copper	mg/kg (dry)	19	4.500	8.700	6.003	1.193
	Chromium	mg/kg (dry)	19	1.000	4.710	1.777	0.767
	Lead	mg/kg (dry)	19	0.800	3.530	1.883	0.706
	Mercury	mg/kg (dry)	19	0.010	0.210	0.117	0.055
	Nickel	mg/kg (dry)	19	0.920	7.100	2.605	1.923
	Silver	mg/kg (dry)	6	0.188	1.120	0.474	0.378
	Zinc	mg/kg (dry)	19	91.400	180.000	132.232	20.243

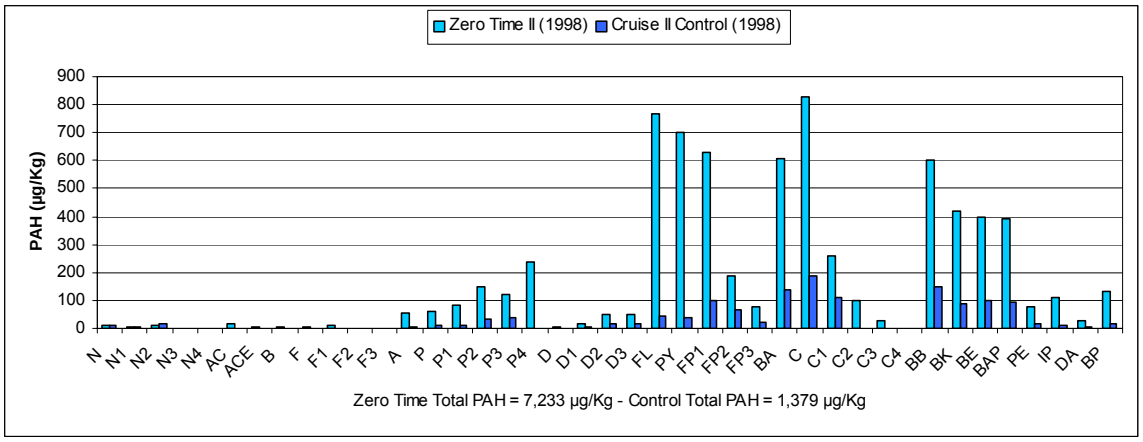
3.2.3.6. PAHs in Tissue

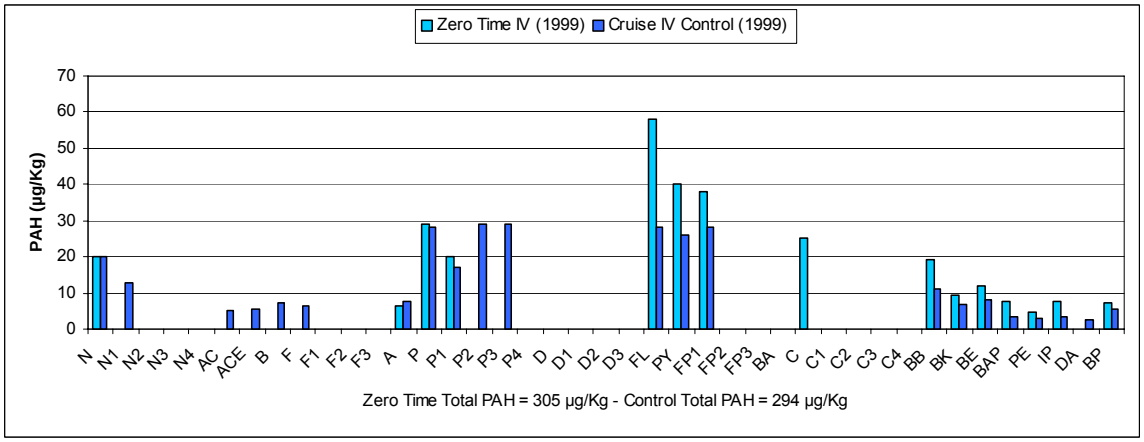
Like total PCBs, results reported for total PAHs depend on the number of PAH compounds analyzed. Total PAH estimates have relied on several calculations, including the summation of 16 EPA priority pollutant compounds used in the Green Book (PAH_{16}), 13 NOAA compounds (Long et al. 1995) and 41 compounds for the SINKEX Program (PAH_{41}). Tissue PAH compounds reported in this section are examined in four basic groups (petrogenic, pyrogenic, PAH_{16}, PAH_{41}) as described in Section 3.2.2.3. Compounds reported below the method detection limit were excluded from summed totals. All concentrations are reported on a tissue dry-weight basis.

Similar to PCBs in this bioaccumulation study, PAHs were measured in source tissues from tissues taken prior to test initiation (Zero Time) and at test conclusion in tissues exposed to native sediment ("uncontaminated" Controls) for both tested organisms. Figure 3-65 and Figure 3-66 compare individual PAH concentrations in *Macoma* and *Nephtys*, respectively, for Zero Time and Control for each of the three cruises. Both test species display elevated concentrations of predominately high molecular weight PAH compounds in source tissues at time zero, however the issue is not as severe in the *Nephtys* tissue as it is in the *Macoma* tissues. Nearly all concentrations decrease over the 28-day exposure period, indicating initial contamination of both the clam and worm stock test organisms. PAH Zero Time concentrations in *Nephtys* source tissue are very low; and only one compound (naphthalene, N in Figure 3-66) continues to exceed 20 µg/kg after the 28-day exposure. These concentrations are considered trivial, well within ranges observed in "wild" populations harvested for bioaccumulation testing, and do not influence statistical evaluations. Therefore, source concentrations in *Nephtys* were not subjected to adjustment prior to statistical testing.

Macoma source tissue concentrations, however, present a much different problem. The highest PAH_{41} concentration measured was 7,233 µg/kg in Zero Time Cruise II samples. Most individual PAH concentrations decreased after 28-days of control sediment exposure, with PAH_{41} concentrations dropping by 4%, 37% and 81%, respectively, in Cruises IV, V and II, showing that *Macoma* depurate these compounds over the 28-day sediment exposure. The distribution of individual PAH compounds in these tissues is characteristic of background material dominated by atmospheric combustion residues. For comparison, Figure 3-67 displays the PAH distribution for National Institute of Standards and Technology atmospheric dust (NIST-1649 SRMa). This distribution is dominated by 4-6 ring parent (non-alkylated) PAH compounds (e.g., pyrene). Generally the major PAHs associated with the combustion of fossil fuels are fluoranthene, pyrene, chrysene and benzo(b)fluoranthene with lesser amounts of naphthalene, fluorene and phenanthrene (2-3 ring PAHs).

A second indicator of combustion source material is parent PAH dominance over corresponding alkylated homologs. In combustion sources parent concentrations are much greater than alkylated daughter compounds, generally following the pattern: PAH (C0) >>C1>C2>C3>C4 PAH. This appears to be the case for chrysene and its alkylated homologs measured in the *Macoma* Zero Time tissue samples.





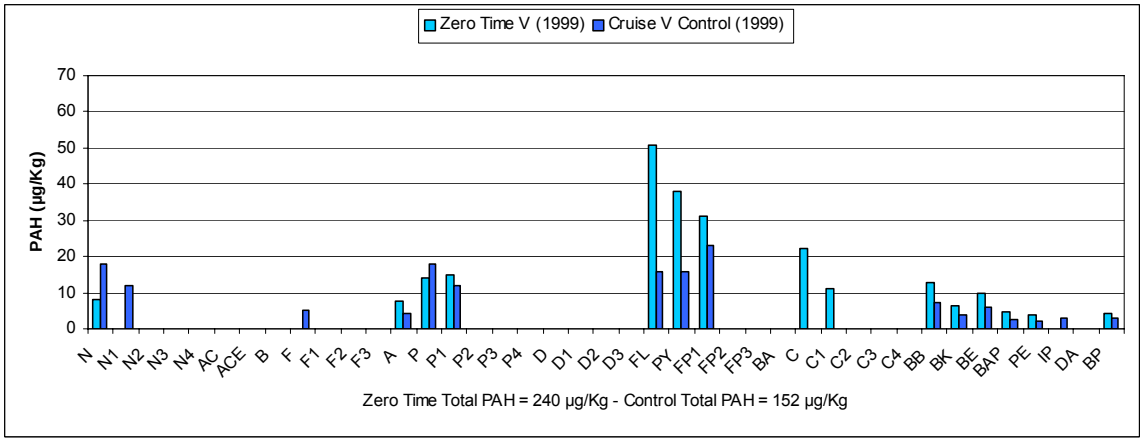
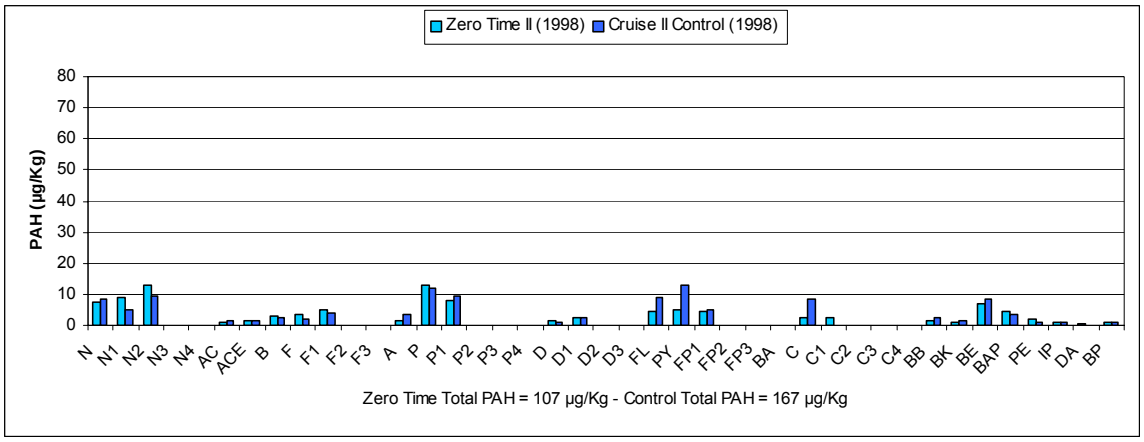
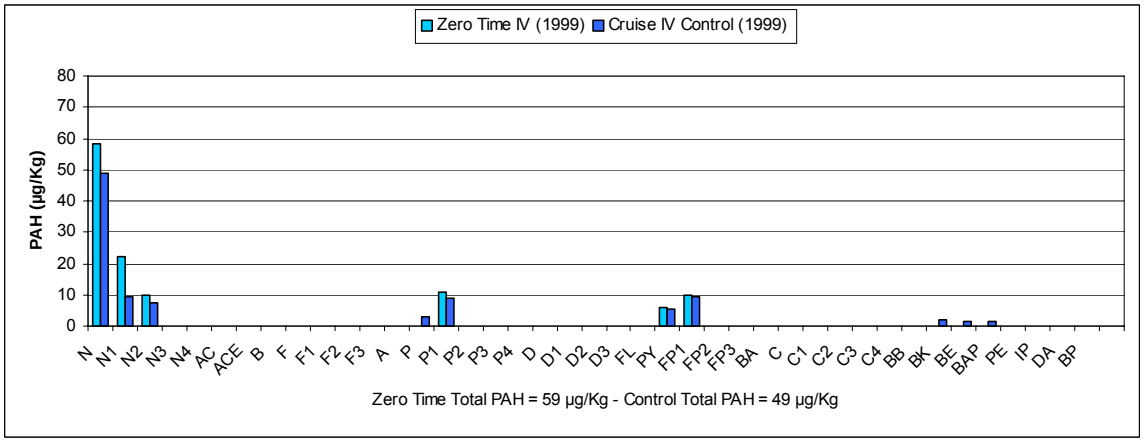


Figure 3-65. Comparison of *Macoma* tissue PAH distributions between zero time (initial) and control (28-day exposure) (µg/kg dry weight).





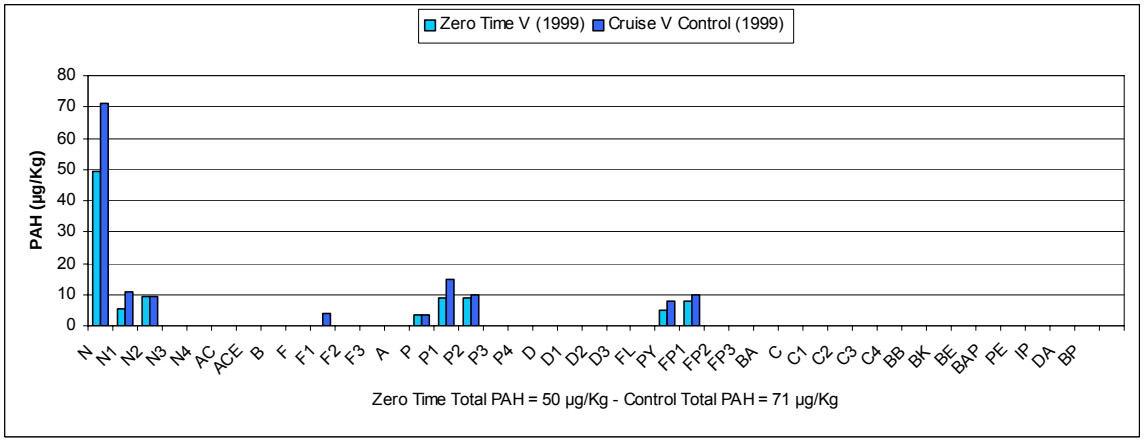


Figure 3-66. Comparison of *Nephtys* tissue PAH distributions between zero time (initial) and control (28-day exposure) (µg/kg dry weight).

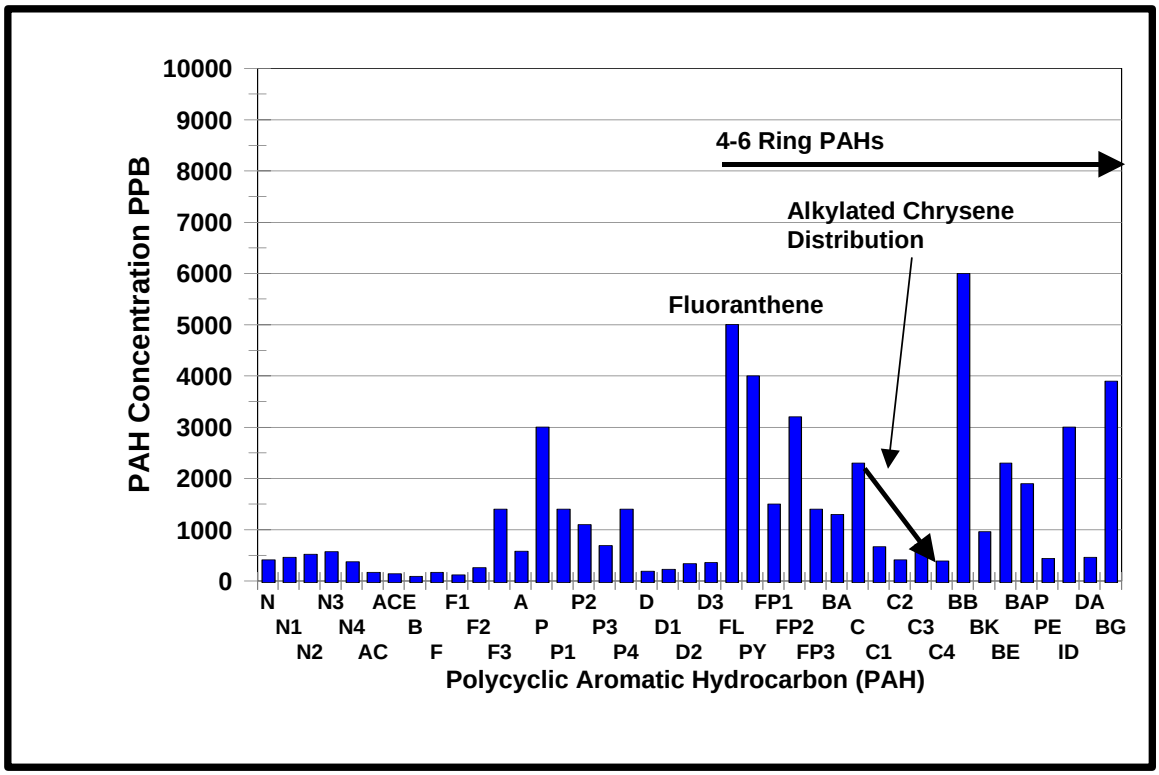


Figure 3-67. PAH distribution in NIST atmospheric dust.

All three bioaccumulation testing events (samples from Cruises II, IV and V) were performed by the same laboratory, under identical procedures and written protocols (MEC 2002). The greatest contamination is associated with bioaccumulation testing performed using Cruise II sediments. The bulk of these sediments were collected from Ring 1 nearest the ex-AGERHOLM, potentially skewing data analysis by artificially "increasing" *Macoma* PAH burdens near the hulk with their zero-time PAH concentrations. To address this issue issue for the *Macoma* tissues only, individual PAH compounds detected in Zero Time tissues for each cruise were flagged during data processing. Individual PAH concentrations measured in corresponding Control samples were then subtracted from each of the test tissue concentrations prior to statistical analysis. Subtractions resulting in negative PAH concentrations were assigned a concentration of 0.0 µg/kg. Again, "Zero Time" adjustments were only applied to *Macoma* tissues, with subsequent summary statistics and hypothesis testing applied to the adjusted values. Unadjusted concentrations are reported in Appendix K.

In the associated results, total PAH concentrations varied between stations, rings and years (cruises). Summary statistics for total PAHs based on PAH_{16}, PAH_{41}, petrogenic and pyrogenic groupings are shown in Table 3-34. Tissue PAH_{41} concentrations for *Macoma* and *Nephtys* ranged from 4.8 to 659.7 µg/kg and 43.5 to 1144.9 µg/kg, respectively. Total PAH_{16} concentrations are smaller, due to the lower number of analytes summed to create totals, ranging from 0.0 to 77.1 µg/kg for *Macoma,* and 35.1 to 365.7 µg/kg for *Nephtys*. Total PAH_{16} minimum concentrations of 0.0 µg/kg reported for *Macoma* are artifacts created by the removal of Control tissue contamination. Figure 3-68 and Figure 3-69 present total dry weight tissue PAH concentrations for *Macoma* and *Nephtys*, respectively.

Table 3-34. Summary of total tissue PAHs in bentnose clam *Macoma nasuta* and the polychaete worm *Nephtys caecoides*.

Species	Analyte	Ring	n	Mean (µg/kg)	Minimum (µg/kg)	Maximum (µg/kg)	Standard Error	CV	Maximum Station
Macoma	Total Green Book PAH (16 compounds)	1	9	31.49	0.40	77.10	7.86	74.84	1-5/6/7 (1999)
		4	5	8.34	0.00	27.30	5.27	141.27	4-6 (1998)
		All	14	23.22	0.00	77.10	6.08	97.91	1-5/6/7 (1999)
	Total PAH (41 compounds)	1	9	241.56	4.80	659.70	81.61	101.36	1-8 (1998)
		4	5	26.84	11.00	63.80	9.60	79.98	4-6 (1998)
		All	14	164.87	4.80	659.70	58.82	133.48	1-8 (1998)
	Petrogenic PAH (30 compounds)	1	9	234.30	4.80	647.10	80.87	103.54	1-8 (1998)
		4	5	24.36	11.00	63.80	9.94	91.27	4-6 (1998)
		All	14	159.32	4.80	647.10	58.11	136.46	1-8 (1998)
	Pyrogenic PAH (12 compounds)	1	9	7.26	0.00	33.00	3.66	151.28	1-5/6/7 (1999)
		4	5	2.48	0.00	12.40	2.48	223.61	4-3.1 (1999)
		All	14	5.55	0.00	33.00	2.52	170.21	1-5/6/7 (1999)
Nephtys	Total Green Book PAH (16 compounds)	1	9	145.45	36.05	365.70	46.42	95.75	1-5 (1998)
		4	5	50.16	35.10	82.60	9.46	42.18	4-6 (1998)
		All	14	111.42	35.10	365.70	31.98	107.40	1-5 (1998)
	Total PAH (41 compounds)	1	9	442.39	80.39	1144.90	161.46	109.49	1-5 (1998)
		4	5	73.40	43.50	153.60	21.20	64.60	4-6 (1998)
		All	14	310.61	43.50	1144.90	112.99	136.11	1-5 (1998)
	Petrogenic PAH (30 compounds)	1	9	305.10	57.10	862.80	109.59	107.76	1-6 (1998)
		4	5	51.38	30.70	110.30	15.20	66.14	4-6 (1998)
		All	14	214.49	30.70	862.80	76.90	134.15	1-6 (1998)
	Pyrogenic PAH (12 compounds)	1	9	137.29	17.00	400.90	55.41	121.09	1-5 (1998)
		4	5	22.02	12.50	43.30	6.20	62.92	4-6 (1998)
		All	14	96.12	12.50	400.90	38.13	148.41	1-5 (1998)

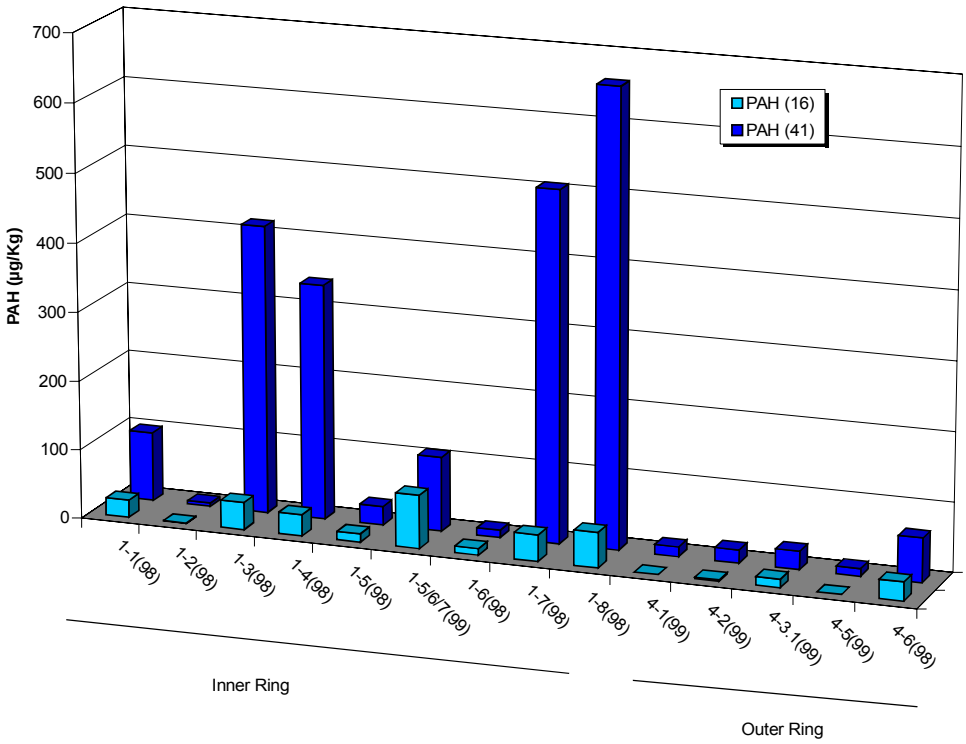


Figure 3-68. PAH_{16} and PAH_{41} measured in *Macoma* tissues (µg/kg dry weight).

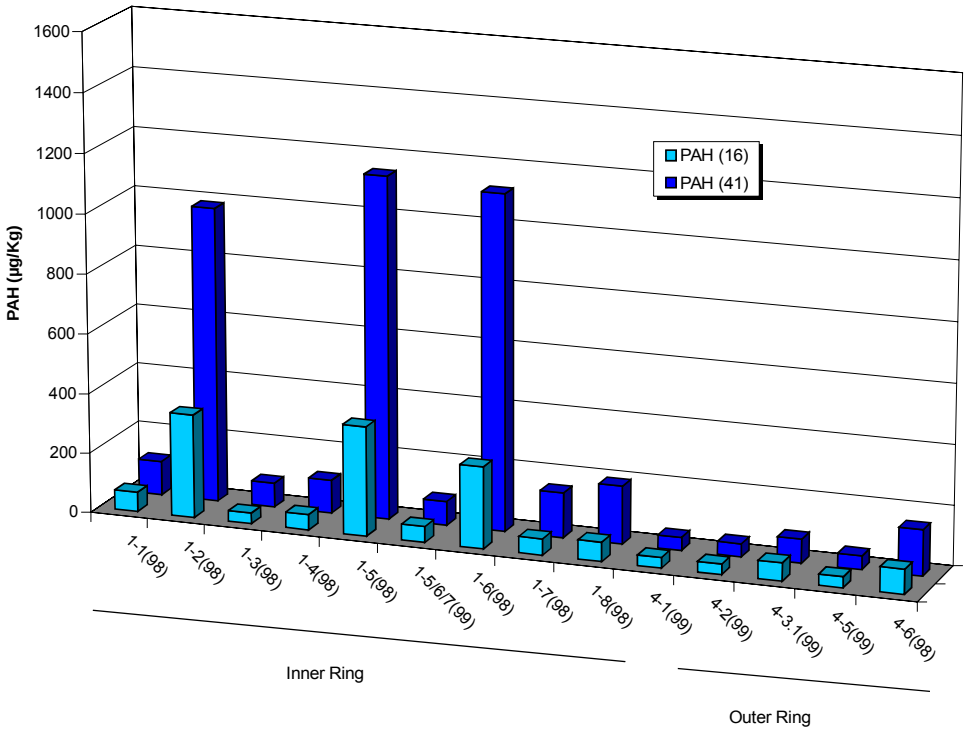


Figure 3-69. PAH_{16} and PAH_{41} measured in *Nephtys* tissues (µg/kg dry weight).

Total PAH_{41} and PAH_{16} tissue concentrations for *Macoma* show similar distribution patterns among stations (Figure 3-68). Stations near the break in the hull (1-7 and 1-8) produced the highest PAH_{41} concentrations (507 and 660 µg/kg, respectively) followed by the forward-most hull Stations 1-3 and 1-4 (417 and 340 µg/kg, respectively). The distribution of individual PAH compounds at these stations appear to be dominated by petrogenic PAHs (Figure 3-70), but source identification is biased due to the mathematical adjustment performed because of control tissue contamination. Of the Outer Ring stations, tissues from Station 4-6 yielded the highest total PAH concentrations (27 µg/kg PAH_{16}, 64 µg/kg PAH_{41}) and also appeared to be dominated by pyrogenic compounds.

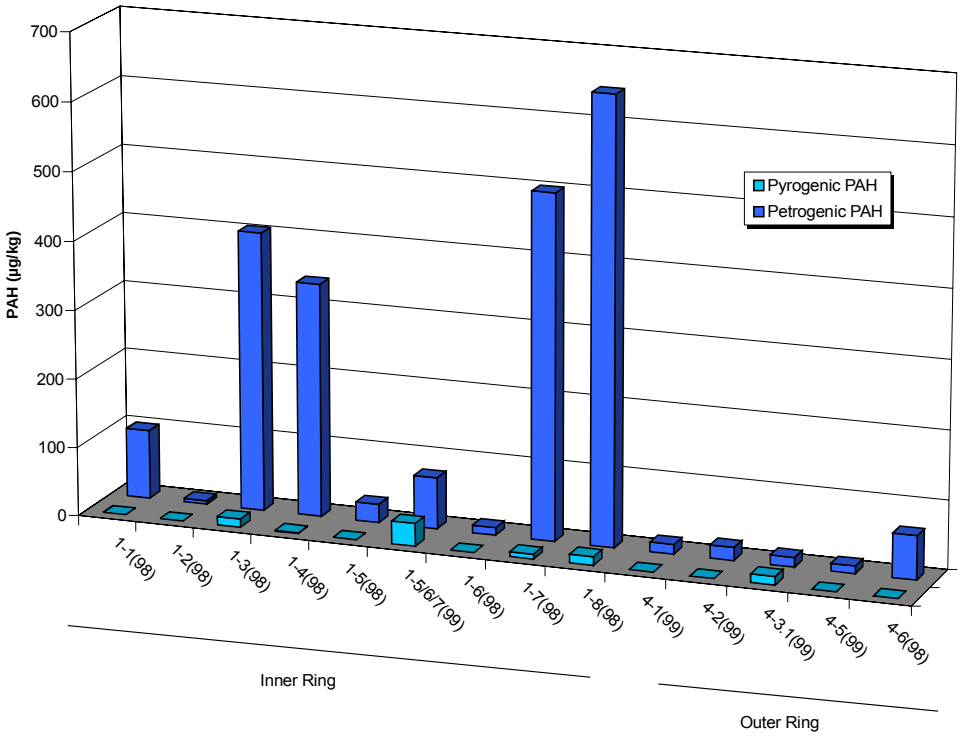


Figure 3-70. Petrogenic and pyrogenic PAHs measured in *Macoma* tissues (µg/kg dry weight).

Nephtys PAH_{41} and PAH_{16} tissue concentrations did not mirror *Macoma* tissue results (Figure 3-69). The highest concentrations occurred forward of the hull break and were distributed on both sides of the hull. Stations 1-5 (starboard amidships), 1-6 (starboard amidships) and 1-2 (port amidships) tissue concentrations exceeded 900 µg/kg (1145, 1120 and 986 µg/kg, respectively). In general, pyrogenic and petrogenic PAHs were evenly distributed among stations, with slightly higher concentrations reported for petrogenic sources (Figure 3-71). Source identification, however, is very limited since, unlike the bivalve *Macoma*, polychaetes possess mixed function oxidase systems (MFO) that actively and selectively degrade PAH compounds (Kennish 1992). Station 4-6, like *Macoma*, had the highest *Nephtys* tissue concentrations for PAH41 and PAH16 (154 and 83 µg/kg, respectively). Interestingly, PAH16 tissue concentrations at Station 4-6 exceeded those at Inner-Ring stations 1-7 and 1-8 (55 and 65 µg/kg, respectively).

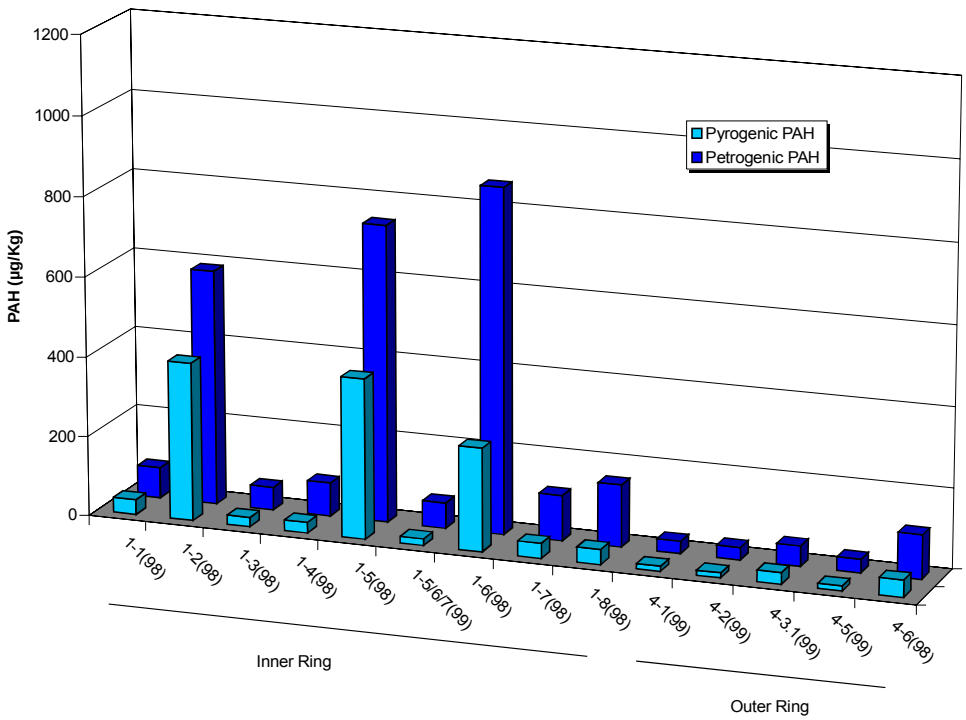


Figure 3-71. Petrogenic and pyrogenic PAHs measured in *Nephtys* tissues (µg/kg dry weight).

Mean total PAH tissue concentrations between grouped Inner- and Outer-Ring samples were compared using a general linear model (GLM) (see Section 2.5.12.3). Prior to hypothesis testing, total PAH distributions for the two groups were evaluated for normality using the Shapiro-Wilks test procedure. Non-normal distributions were subjected to three normalization/standardization procedures (log transformation, ratio of total PAHs to fractional lipid content, and the log of the ratio of total PAHs to fractional lipid content) and re-tested. A non-parametric hypothesis test was applied to the original (untransformed) data, in the event that transformed data failed the normality test. Table 3-35 and Table 3-36 present the results of normality and GLM hypothesis testing performed on PAH16, PAH41, petrogenic and pyrogenic totals measured in *Macoma* and *Nephtys* tissues, respectively. The only significant difference between mean tissue concentrations ($p=0.0104$) was reported for PAH16 in *Macoma*, where Inner-Ring stations (mean=31.49 µg/kg) were significantly higher than Outer-Ring stations (mean=8.34 µg/kg).

For *Macoma*, PAH_{41} and petrogenic PAHs were successfully normalized with a simple log transformation and neither PAH grouping showed significant differences between rings or cruises (i.e., $p>0.05$). Pyrogenic PAH data resisted normalization and hypothesis testing was performed on ranked data. The non-parametric Kruskal-Wallis hypothesis test performed on straight ranked data (Table 3-35 "Rank") was not significant (i.e., $p>0.05$).

None of the four total PAH groups were normally distributed for *Nephtys* (Shaprio-Wilks > 0.05); and log, lipid, log-lipid transformations failed to yield normal distributions (Table 3-36). As a result, the non-parametric Kruskal-Wallis hypothesis test was applied to simple rank data. There were no significant differences ($p>0.05$) between rings for any of the four total PAH groups tested.

Table 3-35. Two-way comparison of SINKEX Rings (1 and 4, cruises 1998-1999) for *Macoma* Tissue PAHs.

Macoma	GLM Hypothesis Test Results				Normality Test Results		
Total PAH Measurement	Transformation Standardization Method	Inner-Ring Comparison (different if <0.05)	Inter-Cruise Comparison (different if <0.05)	Ring-Cruise Interaction (important if <0.05)	Shapiro-Wilks Value	Probability of Non-Normal Distribution (non-normal if <0.05)	Distribution
Total PAH (Green Book)	Lipid	0.0051	0.0830	0.0039	0.8377	0.0152	Non-Normal
	Log Concentration	0.1344	0.6095	0.0623	0.8746	0.0487	Non-Normal
	Log Lipid	0.1893	0.8508	0.0730	0.9125	0.1714	Normal
	None	**0.0104**	**0.2557**	**0.0084**	**0.8960**	**0.0986**	**Normal**
	Rank	0.0795	1.0000	0.0573	0.9541	0.6261	Normal
Total PAH (41 compounds)	Lipid	0.3841	0.6302	0.8086	0.7407	0.0010	Non-Normal
	Log Concentration	**0.3759**	**0.6289**	**0.5770**	**0.9222**	**0.2365**	**Normal**
	Log Lipid	0.4531	0.8228	0.5603	0.9333	0.3396	Normal
	None	0.4093	0.5644	0.7583	0.7442	0.0011	Non-Normal
	Rank	0.3643	0.7484	0.4668	0.9645	0.7964	Normal
Petrogenic PAH	Lipid	0.4404	0.5457	0.7319	0.7225	0.0006	Non-Normal
	Log Concentration	**0.4186**	**0.4913**	**0.6300**	**0.9069**	**0.1419**	**Normal**
	Log Lipid	0.4967	0.6847	0.6066	0.9158	0.1910	Normal
	None	0.4573	0.4960	0.6960	0.7325	0.0008	Non-Normal
	Rank	0.4357	0.6261	0.5747	0.9630	0.7720	Normal
Pyrogenic PAH	Lipid	0.0011	0.0014	0.0046	0.6232	0.0001	Non-Normal
	Log Concentration	0.0698	0.1252	0.3447	0.7340	0.0009	Non-Normal
	Log Lipid	0.1077	0.1076	0.3381	0.8571	0.0278	Non-Normal
	None	0.0034	0.0048	0.0157	0.6577	0.0001	Non-Normal
	Rank	**0.0809**	**0.1670**	**0.4353**	**0.7550**	**0.0015**	**Non-Normal**

Values in **bold** provide best normalization of raw data.

Table 3-36. Two-way comparison of SINKEX Rings (1 and 4, cruises 1998-1999) for *Nephtys* Tissue PAHs.

Nephtys	GLM Hypothesis Test Results				Normality Test Results		
Total PAH Measurement	Transformation Standardization Method	Inner-Ring Comparison (different if <0.05)	Inter-Cruise Comparison (different if <0.05)	Ring-Cruise Interaction (important if <0.05)	Shapiro-Wilks Value	Probability of Non-Normal Distribution (non-normal if <0.05)	Distribution
Total PAH (Green Book)	Lipid	0.6163	0.5473	0.6687	0.5815	0.0000	Non-Normal
	Log Concentration	0.6937	0.2982	0.9811	0.7819	0.0030	Non-Normal
	Log Lipid	0.6551	0.4274	0.9416	0.7003	0.0004	Non-Normal
	None	0.6550	0.4575	0.7426	0.6409	0.0001	Non-Normal
	Rank	**0.9486**	**0.0975**	**0.4840**	**0.9645**	**0.7964**	**Normal**
Total PAH (41 compounds)	Lipid	0.5858	0.5295	0.6185	0.5589	0.0000	Non-Normal
	Log Concentration	0.5015	0.1513	0.9230	0.8359	0.0144	Non-Normal
	Log Lipid	0.5174	0.2577	0.8801	0.7610	0.0017	Non-Normal
	None	0.5854	0.4473	0.6427	0.6213	0.0001	Non-Normal
	Rank	**0.4391**	**0.0217**	**0.4071**	**0.9645**	**0.7964**	**Normal**
Petrogenic PAH	Lipid	0.5832	0.5320	0.6288	0.5914	0.0000	Non-Normal
	Log Concentration	0.4404	0.1496	0.9930	0.8669	0.0380	Non-Normal
	Log Lipid	0.4706	0.2584	0.9300	0.7875	0.0035	Non-Normal
	None	0.5758	0.4466	0.6594	0.6546	0.0001	Non-Normal
	Rank	**0.3052**	**0.0303**	**0.5616**	**0.9645**	**0.7964**	**Normal**
Pyrogenic PAH	Lipid	0.6142	0.5496	0.6197	0.5748	0.0000	Non-Normal
	Log Concentration	0.6904	0.1542	0.7705	0.8142	0.0075	Non-Normal
	Log Lipid	0.6594	0.2521	0.7676	0.7506	0.0013	Non-Normal
	None	0.6310	0.4799	0.6328	0.6029	0.0000	Non-Normal
	Rank	**0.8975**	**0.0179**	**0.6627**	**0.9541**	**0.6261**	**Normal**

Values in **bold** provide best normalization of raw data.

3.2.3.7. Tissue PAH Summary Comments

Macoma bioaccumulation test results, while confounded due to contamination in the source clams, suggests differences in PAH_{16} between Inner- and Outer-rings; however, PAH_{41} totals did not significantly differ between rings. *Nephtys* average PAH_{16} concentrations were not significantly different.

Comparisons of average Ring 1 and Ring 4 PAH16 tissue data to NOAA Mussel Watch Status and Trends *Mytilus* sp. PAH body burdens from four nearshore southern California sites located within 125 miles of the ex-AGERHOLM (Table 3-30) show SINKEX averages bounded by this range of values (31.8 µg/kg Point La Jolla to 328.2 µg/kg Point Loma Lighthouse). The highest Mussel Watch data reported at these stations (1560 µg/kg Santa Catalina and 2070 µg/kg Point Loma Lighthouse) exceed the highest Ring 1 tissue concentrations for both *Macoma* (660 µg/kg) and *Nephtys* (1145 µg/kg).

In summary, only the mean concentration for PAH_{16} appears to be significantly elevated in Inner-Ring *Macoma* tissues compared to Outer-Ring tissues. The broader and more complete measure of total PAHs comprised of 41 compounds (PAH_{41}) did not significantly differ between rings. While not statistically different, all other mean total PAH concentrations are highest at the Inner-Ring compared to the Outer-Ring, suggesting PAH enrichment due to the presence of the

ex-AGERHOLM. It is important to note that all bioaccumulated PAH_{16} are within the range of published concentrations for exposed "clean" nearshore areas.

3.2.3.8. Metals in Tissue

Fourteen SINKEX sediment samples distributed according to Figure 2-18 and Figure 2-19, were examined for bioaccumulation of eight metals in the same organisms processed for PCB and PAH contaminants (the bentnose clam *Macoma nasuta* and the polychaete worm *Nephtys caecoides*). Cadmium, chromium, copper, lead, mercury, nickel, silver and zinc are secondary contaminants of concern and the ex-AGERHOLM is considered a potential source of these metals. Table 3-37 summarizes SINKEX dry weight tissue metal measurements. Complete tabular data is provided in Appendix L.

Unlike the PCBs and some of the PAHs examined in Sections 3.2.3.3 and 3.2.3.6, all the metals of interest naturally occur in the marine environment. Many metals are important micronutrients and essential to metabolism (Rainbow 1985), the most notable of the SINKEX metals in this regard are copper and zinc (Stickney 1984). Other metals, such as cadmium and lead, appear to have no biological function. A consequence of the natural ubiquity of metals and their value as micronutrients is the development of specialized enzyme systems that actively regulate uptake and intracellular distribution. The biochemistry of marine organisms thus plays a vital role in susceptibility to metal toxicity. Heavy metals (atomic weights ranging from 63 to 200), when in excess, are often detoxified by lysosomes and metallothionein sequestering systems (Viarengo, et al 1985). Metallothionein-like proteins have been shown to associate with cadmium, mercury and silver in *Macoma balthica* (Mouneyrac et al., 2000). As expected, all eight metals were found in source tissues (Zero Time and Control) of both tested organisms. Figure 3-72 and Figure 3-73 graphically compare *Macoma* and *Nephtys* tissue concentrations from source material for each of the metals examined. Data are presented for each of the three testing events and correspond to sediment collections in September 1998 (Cruise II), September 1999 (Cruise IV) and November 1999 (Cruise V - Zero Time tissues (*Macoma* and *Nephtys*) were not processed by the metals laboratory due to a clerical error in the subsampling/shipping process. Background concentrations of all metals are apparent in both test organisms and most decrease in concentration during the 28-day exposure period, indicating depuration of the stock test organisms. Contamination in both *Macoma* and *Nephtys* tissue is very small. *Nephtys* chromium and nickel concentrations were elevated in Cruise II zero time samples compared controls after 28-days of exposure. Control concentrations are considered trivial, well within bounds of "wild" populations harvested for bioaccumulation testing and do not influence further statistical evaluations.

Table 3-37. Summary of total tissue metals in bentnose clam *Macoma nasuta* and the polychaete worm *Nephtys caecoides*.

Species	Analyte	Ring	n	Mean (mg/kg)	Minimum (mg/kg)	Maximum (mg/kg)	Standard Error	CV	Maximum Station
Macoma	Cadmium	1	9	0.76	0.34	1.76	0.14	54.87	1-5/6/7 (1999)
		4	5	0.51	0.46	0.65	0.04	16.03	4-6 (1998)
		All	14	0.67	0.34	1.76	0.09	52.71	1-5/6/7 (1999)
	Chromium	1	9	8.35	3.60	22.10	1.94	69.58	1-7 (1998)
		4	5	3.21	2.60	4.54	0.35	24.27	4-6 (1998)
		All	14	6.51	2.60	22.10	1.40	80.49	1-7 (1998)
	Copper	1	9	98.20	54.00	172.00	10.66	32.57	1-5/6/7 (1999)
		4	5	60.44	44.40	95.90	9.27	34.29	4-6 (1998)
		All	14	84.71	44.40	172.00	8.92	39.41	1-5/6/7 (1999)
	Mercury	1	9	0.79	0.28	1.01	0.07	27.58	1-5 (1998)
		4	5	0.36	0.21	0.84	0.12	76.01	4-6 (1998)
		All	14	0.64	0.21	1.01	0.08	49.35	1-5 (1998)
	Lead	1	9	4.36	3.27	5.13	0.24	16.48	1-3 (1998)
		4	5	3.08	2.12	5.29	0.57	41.11	4-6 (1998)
		All	14	3.91	2.12	5.29	0.30	28.26	4-6 (1998)
	Nickel	1	9	13.99	9.97	23.70	1.62	34.70	1-7 (1998)
		4	5	9.20	7.90	11.60	0.65	15.70	4-6 (1998)
		All	14	12.28	7.90	23.70	1.22	37.15	1-7 (1998)
	Silver	1	9	0.43	0.24	0.54	0.03	19.73	1-6 (1998)
		4	5	0.33	0.27	0.50	0.04	28.61	4-6 (1998)
		All	14	0.39	0.24	0.54	0.03	24.46	1-6 (1998)
	Zinc	1	9	270.11	231.00	322.00	10.89	12.10	1-4 (1998)
		4	5	256.20	243.00	288.00	8.36	7.30	4-5 (1999)
		All	14	265.14	231.00	322.00	7.62	10.75	1-4 (1998)
Nephtys	Cadmium	1	9	1.46	0.68	3.18	0.25	50.81	1-5/6/7 (1999)
		4	5	1.13	0.79	1.30	0.09	18.27	4-3.1 (1999)
		All	14	1.34	0.68	3.18	0.16	45.77	1-5/6/7 (1999)
	Chromium	1	9	4.28	0.50	15.90	1.63	114.43	1-4 (1998)
		4	5	1.64	0.30	6.92	1.32	179.42	4-6 (1998)
		All	14	3.34	0.30	15.90	1.17	131.09	1-4 (1998)
	Copper	1	9	16.42	5.31	78.50	7.82	142.83	1-5/6/7 (1999)
		4	5	8.59	8.05	9.30	0.25	6.41	4-3.1 (1999)
		All	14	13.62	5.31	78.50	5.03	138.05	1-5/6/7 (1999)
	Mercury	1	9	0.12	0.04	0.18	0.01	36.48	1-1 (1998)
		4	5	0.07	0.06	0.12	0.01	35.20	4-6 (1998)
		All	14	0.10	0.04	0.18	0.01	42.07	1-1 (1998)
	Lead	1	9	0.95	0.27	2.27	0.20	64.11	1-5 (1998)
		4	5	0.31	0.21	0.38	0.03	20.71	4-3.1 (1999)
		All	14	0.72	0.21	2.27	0.15	79.48	1-5 (1998)
	Nickel	1	9	4.96	2.00	9.98	0.92	55.51	1-4 (1998)
		4	5	2.84	1.80	6.12	0.82	64.60	4-6 (1998)
		All	14	4.20	1.80	9.98	0.70	62.06	1-4 (1998)
	Silver	1	9	0.09	0.03	0.29	0.03	86.42	1-5/6/7 (1999)
		4	5	0.08	0.04	0.09	0.01	25.37	4-1 (1999)
		All	14	0.08	0.03	0.29	0.02	72.93	1-5/6/7 (1999)
	Zinc	1	9	237.00	193.00	360.00	16.74	21.18	1-5/6/7 (1999)
		4	5	213.20	189.00	227.00	6.61	6.93	4-3.1 (1999)
		All	14	228.50	189.00	360.00	11.21	18.35	1-5/6/7 (1999)

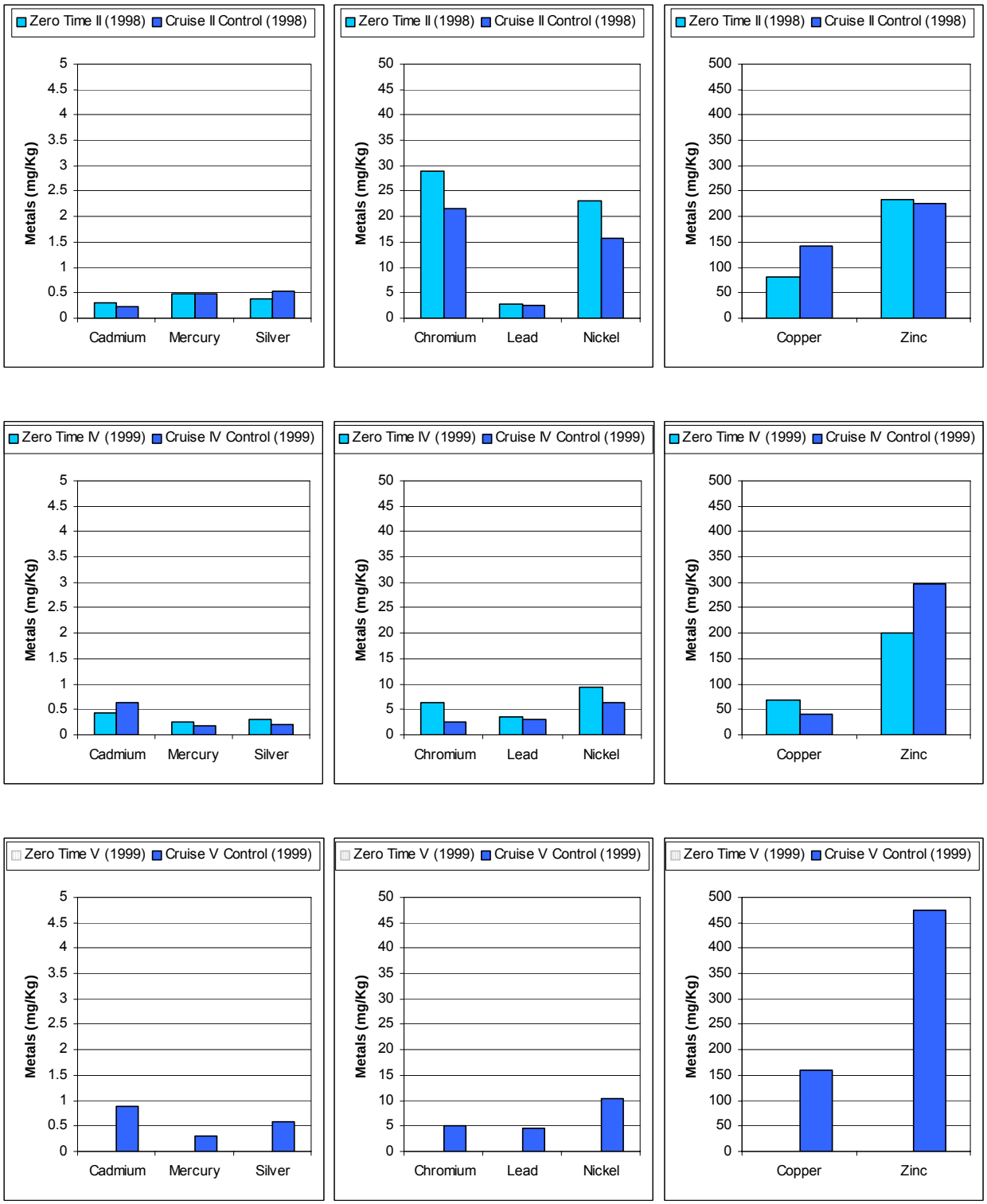


Figure 3-72. Comparison of Zero Time and Control *Macoma* tissue concentrations for eight heavy metals (Zero Time Cruise V data not processed).

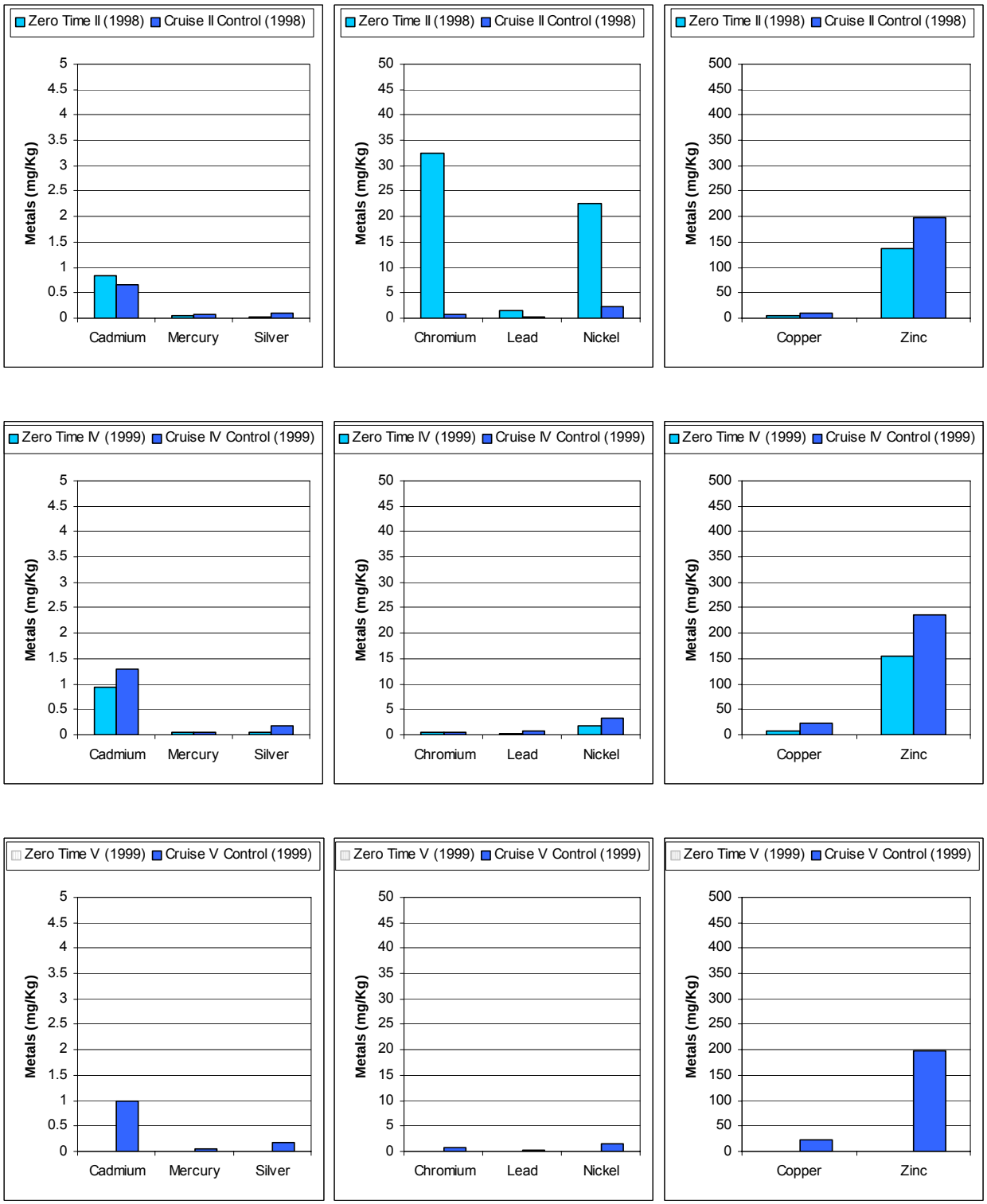


Figure 3-73. Comparison of Zero Time and Control *Nephtys* tissue concentrations for eight heavy metals (Zero Time Cruise V data not processed).

3.2.3.8.1. Tissue Cadmium

Cadmium concentrations varied between stations, rings and years. Summary statistics are shown in Table 3-37. Total tissue concentrations for *Macoma* and *Nephtys* ranged from 0.34 to 1.76 mg/kg and 0.68 to 3.18 mg/kg respectively. Figure 3-74 and Figure 3-75 graphically present total dry weight tissue cadmium concentrations. Station 1-5/6/7 had the highest tissue concentrations for *Macoma* (1.67 mg/kg) and *Nephtys* (3.18 mg/kg). *Nephtys* tested in sediments from Station 1-6, like Station 1-5/6/7 located starboard amidships, accumulated the second highest cadmium concentration (1.98 mg/kg). Cadmium concentrations standardized to lipid fraction significantly correlated with TOC (Table 3-27; r=0.795, p=0.004) for *Macoma* suggesting a possible ingestion accumulation pathway for this species.No strong (as used in the tissue metals presentation, a strong correlation is when $p<0.05$ and $r^2>0.5$) correlations were noted for *Nephtys*.

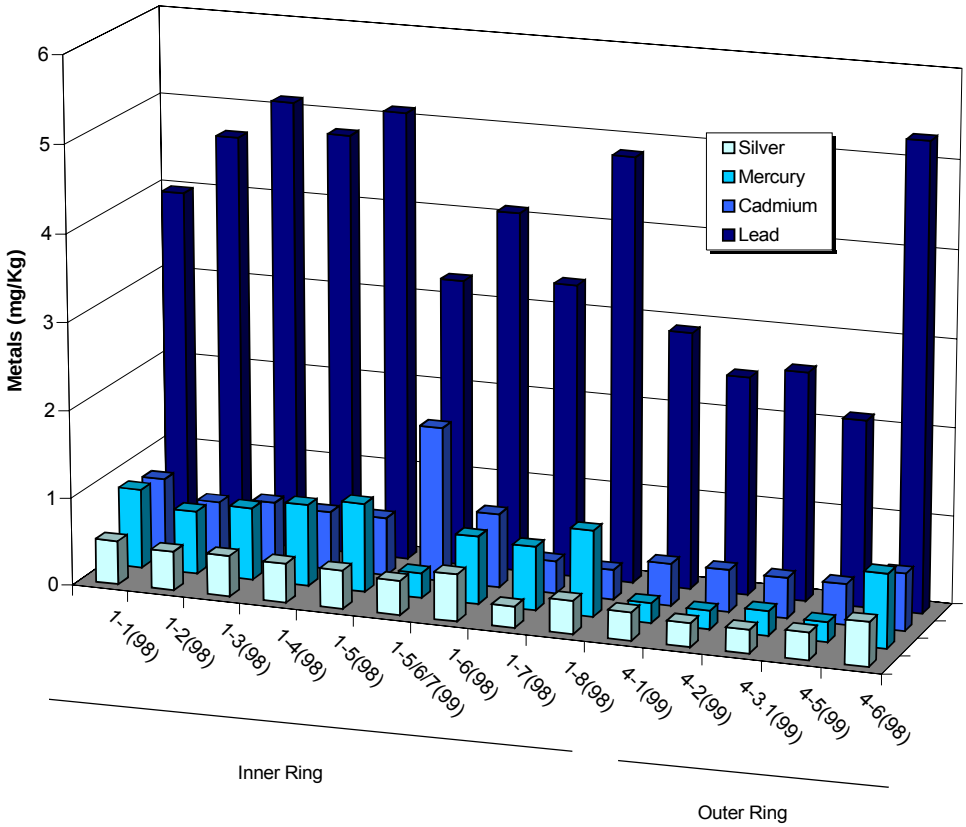


Figure 3-74. Dry weight *Macoma* tissue concentrations for cadmium, lead, mercury and silver.

Inner and Outer Ring cadmium means were compared using GLM techniques (Section 2.5.12.3). Data was successfully normalized for both species by simple log transformations and the results of GLM hypothesis testing are presented in Table 3-38 and. Significant differences between Inner and Outer Rings are apparent in both species ($p \leq 0.05$), with Inner Ring stations showing higher concentrations than Outer Ring stations.

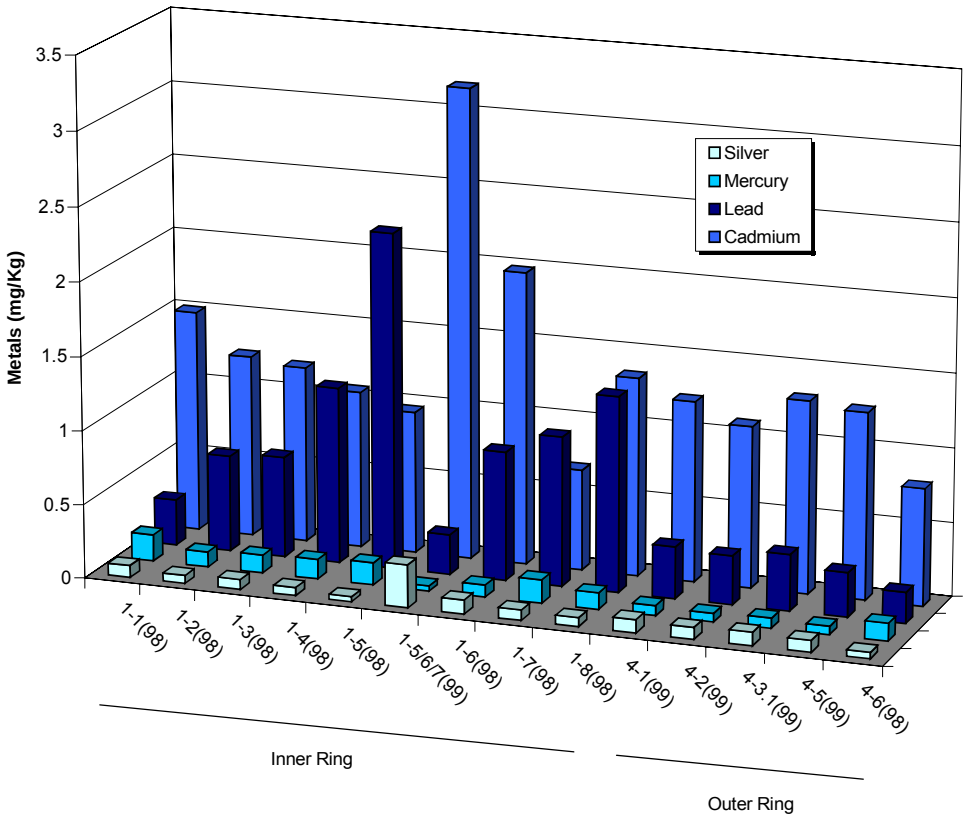


Figure 3-75. Dry weight *Nephtys* tissue concentrations for cadmium, lead, mercury and silver.

Comparisons of average Ring 1 and Ring 4 cadmium tissue data to NOAA Mussel Watch Project Status and Trends *Mytilus* sp. body burdens from four nearshore southern California sites located within 125 miles of the ex-AGERHOLM (Table 3-33) show SINKEX averages to be bounded by this range of exposed nearshore concentration (1.55 mg/kg Pt Loma Lighthouse to 4.03 mg/kg Santa Catalina). The highest Mussel Watch data reported was from Santa Catalina Island (7.7 mg/kg) and is 2.4 times higher than the highest SINKEX tissue value reported (Nephtys 3.18, Station 1-5/6/7). There are no FDA shellfish tissue guidelines for cadmium.

In summary**,** there is a significant difference between Inner Ring and Outer Ring stations for cadmium in *Macoma* and *Nephtys* tissues. Considering the very large reservoir of metals that could be potentially available to the environment due to the presence of the ex-AGERHOLM, it is not surprising that the test species were able to bioaccumulate cadmium in the presence of the ex-AGERHOLM. However, it is important to note that all bioaccumulated cadmium concentrations measured fall within the range of published values for exposed nearshore Mussel Watch areas.

3.2.3.8.2. Tissue Chromium

Chromium followed patterns similar to cadmium, with concentrations varying among individual stations, rings and years. Summary statistics are shown in Table 3-37. Total tissue concentrations for *Macoma* and *Nephtys* ranged from 2.60 to 22.10 mg/kg and 0.30 to 15.90 mg/kg respectively. Comparisons of average Ring 1 and Ring 4 chromium tissue data to NOAA Mussel Watch Project Status and Trends body burdens (Table 3-33) shows Inner Ring SINKEX averages to be

slightly greater than those reported for local exposed nearshore sites (ranging from 1.76 to 4.03 mg/kg). All Outer Ring station averages fall within the background coastal range. Station 1-7 located near the major hull break (starboard side) accumulated the highest concentrations for *Macoma* and were approximately 5 times the concentration reported for the highest clean Mussel Watch site (Santa Catalina Island). Inner and Outer Ring means were compared using GLM techniques (Section 2.5.12.3). Data was successfully normalized for both species by a simple log transformation and hypothesis test results are presented in Table 3-38 and Table 3-39. No significant differences between the Inner and Outer Rings are apparent in either species examined ($p>0.05$). No strong, significant correlations were noted between chromium and TOC or fine particles (Table 3-27).

Comparisons of average Ring 1 and Ring 4 chromium tissue data to NOAA Mussel Watch Project Status and Trends body burdens (Table 3-30) shows Inner Ring SINKEX averages to be slightly greater than those reported for local exposed nearshore sites (ranging from 1.76 to 4.03 mg/kg). All Outer Ring station averages fall within the background coastal range. Station 1-7 located near the major hull break (starboard side) accumulated the highest concentrations for *Macoma* and were approximately 5 times the concentration reported for the highest clean Mussel Watch site (Santa Catalina Island).

In summary, there are no significant differences between chromium concentrations of Inner Ring and Outer Ring stations for both *Macoma* and *Nephtys* tissues.

3.2.3.8.3. Tissue Copper

Copper concentrations varied between stations, rings and years. Summary statistics are shown in Table 3-37. Total tissue concentrations for *Macoma* and *Nephtys* ranged from 44.40 to 172.00 mg/kg and 5.31 to 78.50 mg/kg respectively. Figure 3-76 and Figure 3-77 graphically present total dry weight tissue concentrations. *Macoma* bioaccumulated copper to higher levels than did *Nephtys* in both the Inner and Outer Ring stations. Station 1-5/6/7 had the highest tissue concentration for both *Macoma* (172.00 mg/kg) and *Nephtys* (78.50 mg/kg). Of the Outer Ring stations, 4-5 accumulated the highest concentration for *Macoma* (95.90 mg/kg) and 4-3.1 for *Nephtys* (9.30 mg/kg). No strong or significant (Table 3-27) correlations to total organic carbon or fine sediment particles were noted for *Macoma* or *Nephtys*.

Inner and Outer Ring means were compared using GLM techniques (Section 2.5.12.3). Data was successfully normalized for *Macoma* species by a simple log transformation. *Nephtys* tissue data resisted normalization and hypothesis tests focused on ranked data. Hypothesis test results are presented in Table 3-38 and Table 3-39. *Macoma* tissue concentrations were significantly greater ($p=0.0047$) at the Inner Ring stations, when compared to Outer Ring stations. No significant differences were identified for *Nephtys* ($p<0.05$).

Comparisons of average Ring 1 and Ring 4 copper tissue data to NOAA Mussel Watch Project Status and Trends body burdens (Table 3-30) show Inner Ring SINKEX averages to be much greater than those reported for local exposed nearshore sites (ranging from 6.0 to 12.8 mg/kg). Outer Ring station averages for *Macoma* also exceed these background coastal values suggesting higher natural ambient copper levels at these offshore sites.

In summary, *Macoma* tissue copper concentrations are significantly greater at the Inner Ring than the Outer Ring reference area. Figure 3-76 shows Inner Ring sediment copper concentrations to be enriched above reference Outer Ring levels. Copper is abundant in

shipboard materials (e.g., antifouling paints, piping) and is suspected to be a contributor to the high tissue concentrations observed near the hulk.

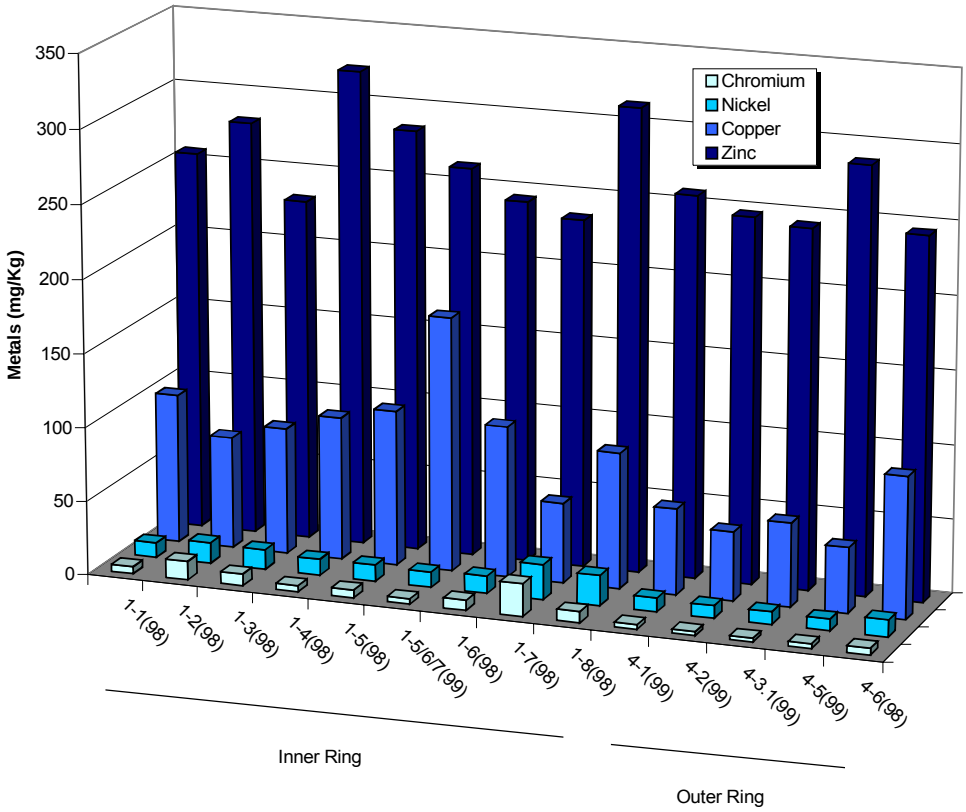


Figure 3-76. Dry weight *Macoma* tissue concentrations for chromium, copper, nickel and zinc.

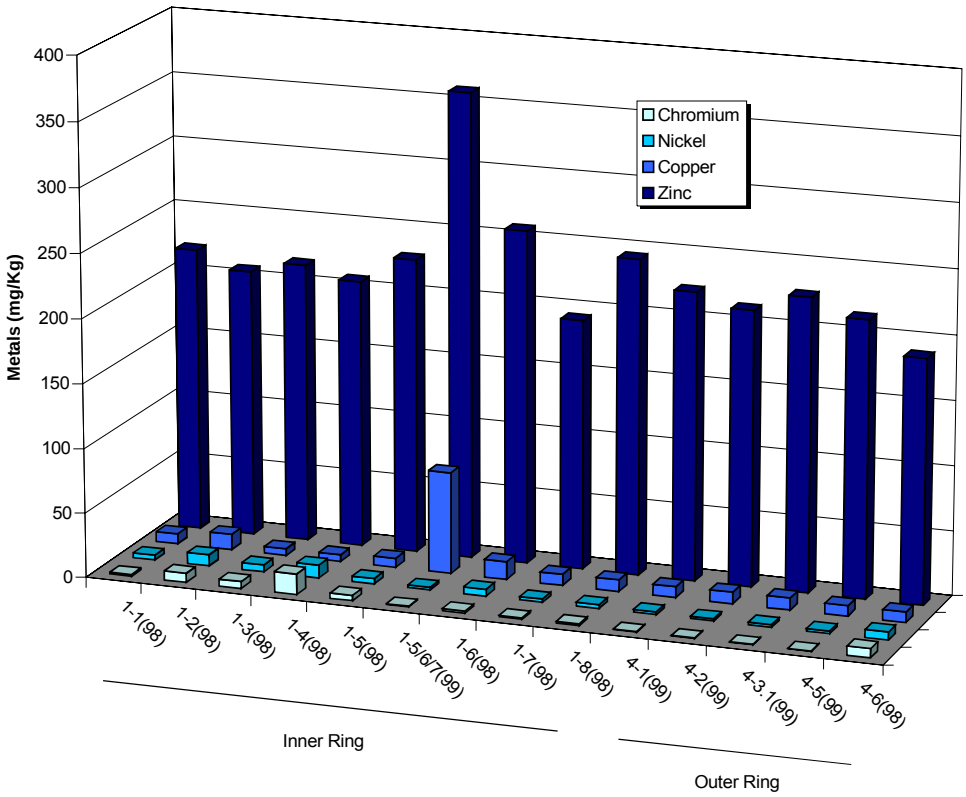


Figure 3-77. Dry weight *Nephtys* tissue concentrations for chromium, copper, nickel and zinc.

3.2.3.8.4. Tissue Lead

Summary statistics for lead are presented in Table 3-37. Total tissue concentrations for *Macoma* and *Nephtys* ranged from 2.12 to 5.29 mg/kg and 0.21 to 2.27 mg/kg respectively. Figure 3-74 and Figure 3-75 graphically present total dry weight tissue concentrations. *Macoma* bioaccumulated lead to higher levels than did *Nephtys* in both the Inner and Outer Ring stations. Station 1-3 had the highest tissue concentration for *Macoma* (5.29 mg/kg). On the opposite side of the hulk, Station 1-5 produced the highest tissue lead concentration for *Nephtys* (2.27 mg/kg). Of the Outer Ring stations, 4-6 accumulated the highest concentration for *Macoma* (5.29 mg/kg) and 4-3.1 for *Nephtys* (0.38 mg/kg). No strong or significant correlations to total organic carbon or fine sediment particles were noted for either organism (Table 3-27).

Inner and Outer Ring concentrations means were compared using GLM techniques (Section 2.5.12.3). *Macoma* data was normally distributed and *Nephtys* concentrations were successfully normalized for by a simple log transformation (Table 3-38 and Table 3-39). Hypothesis test results are presented in Table 3-38 and Table 3-39. *Macoma* and *Nephtys* tissue concentrations did not significantly differ between the Inner Ring stations and Outer Ring stations.

Comparisons of average Ring 1 and Ring 4 lead tissue data to NOAA Mussel Watch Project Status and Trends body burdens (Table 3-30) show Inner Ring SINKEX averages to be slightly greater than those reported for local exposed nearshore sites (ranging from 0.98 to 3.12 mg/kg). Outer Ring station averages were bounded by the clean background coastal values.

In summary, lead tissue concentrations in Macoma and Nephtys were not significantly different between Inner Ring hulk stations and Outer Ring reference stations. Additionally,

bioaccumulated lead concentrations are very close to ranges of values published for local exposed nearshore areas.

3.2.3.8.5. Tissue Mercury

Table 3-37 presents summary statistics for mercury concentrations in the tissues of *Macoma* and *Nephtys*. Total tissue concentrations for *Macoma* and *Nephtys* ranged from 0.21 to 1.01 mg/kg and 0.04 to 0.18 mg/kg respectively. Figure 3-74 and Figure 3-75 graphically present total dry weight tissue concentrations. *Macoma* bioaccumulated mercury to higher levels than did *Nephtys* in both the Inner and Outer Ring stations. Station 1-5 (starboard amidships) had the highest tissue concentration for *Macoma* (1.01 mg/kg). On the side opposite of the hulk, Station 1-1 produced the highest tissue mercury concentration for *Nephtys* (0.18 mg/kg). Of the Outer Ring stations, 4-6 accumulated the highest concentration for *Macoma* (0.84 mg/kg) and *Nephtys* (0.12 mg/kg). Of the Outer Ring stations, 4-6 accumulated the highest concentration for *Macoma* (0.84 mg/kg) and *Nephtys* (0.12 mg/kg). No strong or significant correlations (Table 3-27) to sediment fine material or TOC were noted for *Macoma* and *Nephtys*.

Inner and Outer Ring concentrations means were compared using GLM techniques (Section 2.5.12.3). Table 3-38 presents two additional "standardizations" for mercury, lipid standardization (tissue concentrations divided by fractional lipid weight) and log lipid standardization (log of tissue concentrations divided by fractional lipid weight), because unlike most other metals, mercury has been known to positively correlate with tissue lipids. Non-normal *Macoma* data were successfully normalized using a log transformation on lipid standardized concentrations (Table 3-38). *Nephtys* mercury concentrations were normalized by a simple log transformation (Table 3-39). Hypothesis test results are presented in Table 3-38 and Table 3-39. *Macoma* and *Nephtys* tissue concentrations did not significantly differ between the Inner Ring stations and Outer Ring stations ($p<0.05$).

Comparisons of average Ring 1 and Ring 4 lead tissue data to NOAA Mussel Watch Project Status and Trends body burdens (Table 3-33) show Inner Ring SINKEX *Macoma* mercury averages to be approximately twice that of the highest exposed nearshore area (Point Loma Lighthouse, 0.372 mg/kg). *Macoma* Outer Ring and, both Inner and Outer Ring *Nephtys* averages were within the range reported for local exposed nearshore Mussel Watch sites (0.64 to 3.72 mg/kg). FDA shellfish tissue guidelines for mercury are 1.0 mg/kg wet weight tissue, equating to approximately 10 mg/kg dry weight. The highest tissue mercury valued recorded was measured in *Macoma* from Station 1-5 (1.01 mg/kg dry weight) and is well below the current FDA action level for shellfish.

In summary, mercury tissues concentrations in *Macoma* and *Nephtys* do not significantly differ between Inner Ring hulk stations and Outer Ring reference stations. Additionally, bioaccumulated mercury concentrations are very close to ranges of values published for local exposed nearshore areas and all measured values are well below current FDA shellfish action levels.

3.2.3.8.6. Tissue Nickel

Table 3-37 presents summary statistics for nickel concentrations in the tissues of *Macoma* and *Nephtys*. Total tissue concentrations for *Macoma* and *Nephtys* ranged from 7.90 to 23.70 mg/kg and 1.80 to 9.98 mg/kg respectively. Figure 3-76 and Figure 3-77 graphically present total dry weight tissue concentrations. *Macoma* bioaccumulated nickel to higher levels than did *Nephtys*

in both the Inner and Outer Ring stations. Station 1-7 had the highest tissue concentration for *Macoma* (23.70 mg/kg). At the starboard bow, Station 1-4 produced the highest tissue nickel concentration for *Nephtys* (9.98 mg/kg). Of the Outer Ring stations, 4-6 accumulated the highest concentration for *Macoma* (11.60 mg/kg) and *Nephtys* (6.12 mg/kg). For both test organisms, nickel concentrations did not significantly correlate with either TOC or with fine sediments (Table 3-27).

Inner and Outer Ring concentration means were compared using GLM techniques (Section 2.5.12.3). Non-normal *Macoma* and *Nephtys* data were successfully normalized using a simple log transformation (Table 3-38 and Table 3-39). Hypothesis test results are presented in Table 3-38 and Table 3-39. *Macoma* and *Nephtys* tissue concentrations did not significantly differ between the Inner Ring stations and Outer Ring stations ($p<0.05$).

Macoma and *Nephtys* average Ring 1 and Ring 4 nickel tissue data exceed average NOAA Mussel Watch Project Status and Trends body burdens (Table 3-30) for clean local coastal sites (range 1.366 to 2.254 mg/kg). As noted in the sediment section (3.2.2.4.7), eight Inner Ring sediment stations lie outside the 95% predictive interval depicted in Figure 3-52, indicating nickel enrichment from the ex-AGERHOLM and possibly explain the reasonably high tissue levels at Stations 1-7 and 1-8.

In summary, nickel tissues concentrations in *Macoma* and *Nephtys* do not significantly differ between Inner Ring hulk stations and Outer Ring reference stations. Bioaccumulated nickel concentrations are generally greater than ranges of values published for local exposed nearshore areas.

3.2.3.8.7. Tissue Silver

Tissue concentrations of silver were low across all test stations and organisms. Summary statistics are shown in Table 3-37. Total tissue concentrations for *Macoma* and *Nephtys* ranged from 0.39 to 0.54 mg/kg and 0.03 to 0.29 mg/kg respectively. Figure 3-74 and Figure 3-75 graphically present total dry weight tissue silver concentrations. For both test organisms, reference and hulk station sediments produced low bioaccumulation concentrations. Station 1-6 had the highest tissue concentrations for *Macoma* (0.54 mg/kg) and Station 1-5/6/7 produced the highest *Nephtys* concentration (0.29 mg/kg). Outer Ring station 4-6 was again the high reference station with 0.50 mg/kg. No strong or significant correlations were noted between tissue concentrations, TOC and fine sediments for either bioassay organism (Table 3-27).

Inner and Outer Ring silver means were compared using GLM techniques (Section 2.5.12.3). Macoma data was normally distributed; however, *Nephtys* tissues could not be normalized and a rank hypothesis test was necessary. Results of GLM hypothesis testing are presented in Table 3-38 and Table 3-39. Significant differences between Inner and Outer Rings are not apparent in either *Macoma* or *Nephtys* ($p\geq0.05$).

Comparisons of average Ring 1 and Ring 4 silver tissue data to NOAA Mussel Watch Project Status and Trends *Mytilus* sp. body burdens from four exposed nearshore southern California sites located within 125 miles of the ex-AGERHOLM (Table 3-30) show SINKEX averages to be within the range of exposed reported nearshore concentrations (0.21 mg/kg Oceanside Beach Jetty to 18.23 mg/kg Point Loma Lighthouse). The highest Mussel Watch data reported was from the Point Loma Lighthouse area (33.75 mg/kg) and is 62.5 times greater than the highest SINKEX tissue value reported.

In summary, there are no significant differences between Inner Ring and Outer Ring stations for silver in *Macoma* or *Nephtys* tissues. All bioaccumulated silver concentrations fall within the range of published values for exposed nearshore areas.

3.2.3.8.8. Tissue Zinc

Typical of marine bioaccumulation studies, zinc was the numerically dominant metal. Summary statistics for zinc are shown in Table 3-37. Total tissue concentrations for *Macoma* and *Nephtys* ranged from 231 to 322 mg/kg and 189 to 360 mg/kg respectively. Figure 3-76 and Figure 3-77 graphically present total dry weight tissue zinc concentrations. For both test organisms, reference and hulk station sediments produced similar bioaccumulation concentrations. Average tissue concentrations in *Macoma* were approximately equal to those of *Nephtys* for both Ring 1 and Ring 4. Station 1-4 had the highest tissue concentrations for *Macoma* (322 mg/kg) and Station 1-5/6/7 produced the highest *Nephtys* concentration (360 mg/kg). Tissue concentrations for the Outer Ring were greatest at Stations 4-5 and 4-3.1 (288 and 227 mg/kg respectively). No strong or significant correlations (Table 3-27) were noted between tissue concentrations, TOC and fine sediments for either bioassay organism.

Inner and Outer Ring zinc means were compared using GLM techniques (Section 2.5.12.3). *Macoma* data was normally distributed; however, *Nephtys* could not be normalized requiring ranks to be used for hypothesis testing. Results of GLM hypothesis testing are presented in Table 3-38 and Table 3-39. No significant differences between Inner and Outer Rings are apparent in either organism ($p \leq 0.05$).

Comparisons of average Ring 1 and Ring 4 zinc tissue data to NOAA Mussel Watch Project Status and Trends *Mytilus* sp. body burdens from four exposed nearshore southern California sites located within 125 miles of the ex-AGERHOLM (Table 3-33) show SINKEX averages to be very close to this range of exposed nearshore sites (132 mg/kg Santa Catalina to 203 mg/kg Point Loma Lighthouse).

In summary, there are no significant differences between Inner Ring and Outer Ring stations for zinc in *Macoma* and *Nephtys* tissues. All bioaccumulated zinc concentrations are close to the range of published values for clean local coastal areas.

3.2.4. Supplementary Exposure Data from Sablefish Analysis

The full report for the supplementary Sablefish Assessment is provided in its entirety as Supplement I. Excerpts from the text and some tables and figures have been brought forward to this main document (see Supplement I for full details). The sablefish tissue data were evaluated to determine statistical differences between the fish collected from the ship and reference locations. The data were tested to determine whether they conformed to a normal distribution.[11] A one-way analysis of variance (ANOVA, Analytical Software 1996) and a nonparametric Kruskal-Wallis X^2 test was conducted to test the null hypothesis that there was no difference between sites for total length (mm), total weight (g), dry weight (%), dry weight of lipids (%), sum of ECD congeners (ng/g dry), sum of ECD congeners per unit lipid (ng/g lipid), Total PCB (ng/g dry), Total PCB per unit lipid (ng/g lipid), Log(sum of ECD congeners (ng/g dry)), Log(sum of ECD congeners per unit lipid (ng/g lipid)), Log(Total PCB ng/g), and Log(Total

[11] The Lin-Mudhokar test for normality (Analytical Software 1996) was performed on the raw and $\log_{10}$-transformed raw data. The null hypothesis – The data are normally distributed – was rejected if the p-value was < 0.05.

PCB ng/g lipid) for both muscle and liver tissues. The null hypothesis was rejected if the p-value from either the ANOVA or nonparametric test was ≤0.05. Correlation and regression analysis of the sablefish data were conducted to evaluate the relationships between body weight, length, %lipid content, and Total PCB concentrations.

The sablefish from the ship site were similar in length (ship sablefish range 450-530 mm) with the largest fish being caught from the reference locations (reference sablefish range 455-635 mm). However, sablefish from the reference locations were significantly heavier by about 20% than sablefish from the ship site. There was a strong relationship between length and weight for both groups of sablefish, but there was no relationship between weight or length and Total PCB ng/g lipid in muscle and liver tissues. Sablefish from both sites had similar dry weight and lipid content. The lipid content in liver tissues was about twice as high as the lipid content in the muscle tissue.

There were significantly higher PCBs in the muscle tissues of the sablefish from the ship site than from the reference site. The sumECD congeners ng/g dry weight measured in sablefish were about 1.7 times higher (p=0.04) than the sablefish from the reference locations (Figure 3-78). When the sumECD was normalized to lipid content (ng/g lipid), stronger differences between sites were detected (p=0.01) but the magnitude of the difference remained about the same with the ship sablefish about 1.6 times higher than the reference sablefish.

The Total PCB ng/g lipid measured and calculated for sablefish muscle tissues from the ship site were also significantly higher than the reference locations by about a factor of 1.48. There were very similar concentrations of Total PCB ng/g lipid in the muscle and liver tissues measured from the same fish. The very small sample size of liver tissues from the ship site (n=4) limited the ability to detect differences in liver PCB concentrations between the sites. Overall larger fish were collected from the reference locations than from the ship site, but the ship site had specimens with higher PCB concentrations than samples from the reference location. The geometric mean and 95% confidence interval calculated for sablefish muscle and liver showed that the sablefish from the ship site had significantly higher PCB levels in muscle tissues than the sablefish from the reference locations.

Both the ECD and SIM methods detected very similar distributions of congeners in the muscle and liver, with no apparent differences between ship site and reference stations. Of the congeners measured, PCB153 and PCB138 were the most abundant, followed by PCB118, PCB180, PCB187, PCB101, PCB105, and PCB170. The major difference in the congener distributions was that the congeners in the sablefish collected from the ship site were higher than the congeners in sablefish from the reference locations. A similar pattern was also observed for the homologs. Hexacholorbiphenyl was the most abundant homolog, followed by penta- and heptachlorobiphenyl. In comparison to the homolog distribution observed in reef fish sampled during the REEFEX study (Johnston et al. 2005a), sablefish had slightly more higher chlorinated hexa- and heptachlorobipehnyl and less penta- and tetrachlorobiphenyl than the fish sampled at the shallow water ex-VERMILLION and reference reefs off the coast of South Carolina.

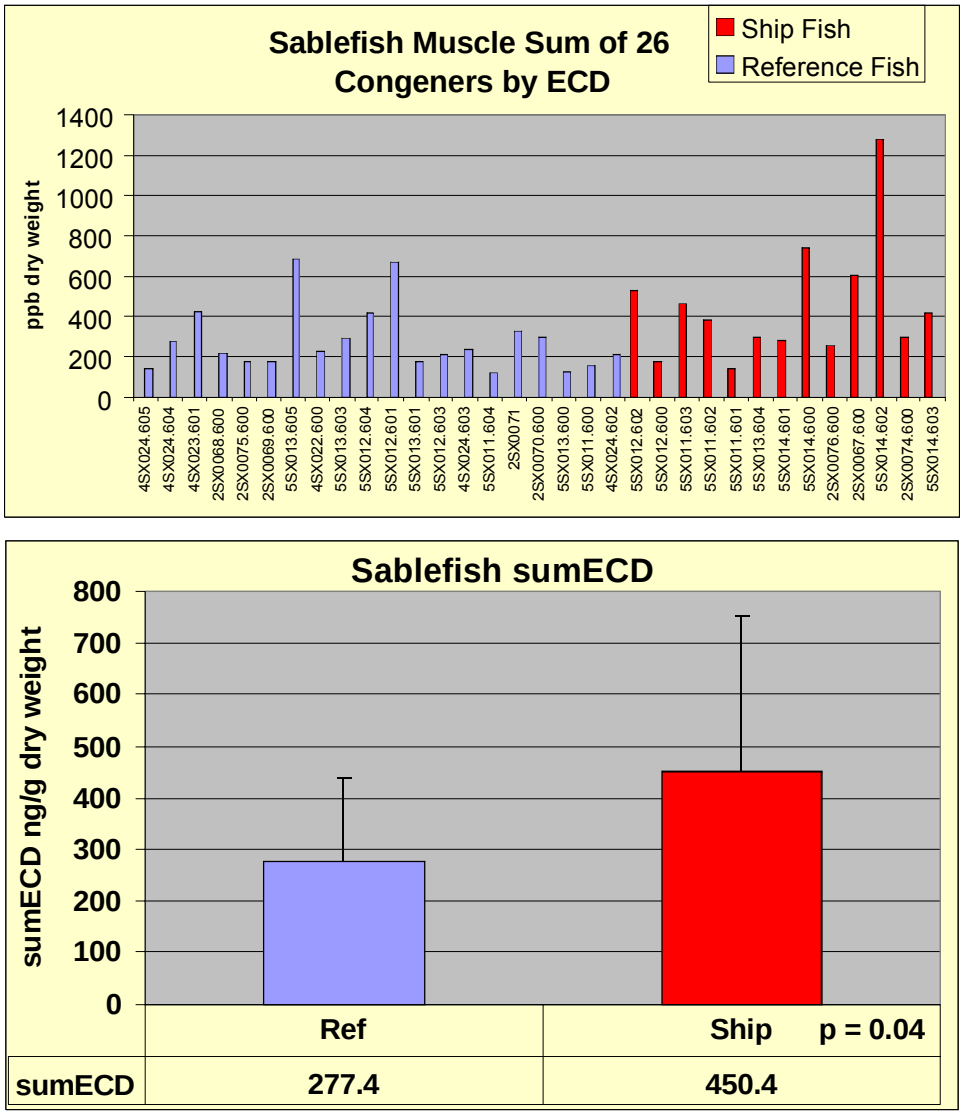


Figure 3-78. The concentration of PCB determined as the sum of 26 congeners measured by ECD (sumECD) in sablefish muscle for individual fish (Top) and the average and standard deviation (Bottom) obtained for fish collected from the reference and Ship locations. The sumECD measured in fish from the ship location was significantly ($p \leq 0.05$) higher than fish from the reference location.

In summary, the exposure assessment found that sablefish sampled from the ex-AGERHOLM (ship) site had statistically higher concentrations (by a factor of 1.4-1.5) of PCBs than sablefish sampled from reference locations about 4 nm away from the ship. Sablefish from the ship site were similar in length to sablefish from the reference locations, but the sablefish from the reference site were 20% heavier than sablefish from the ship site. There was no relationship between weight or length and Total PCB and sablefish from both sites had similar dry weight and lipid content. The distribution of congeners and homologs measured in the sablefish from both sites were very similar, with the main difference that congeners and homologs in sablefish from the ship site were about 1.5 times higher than levels measured in sablefish from the reference locations.

Table 3-38. Two-way comparison of SINKEX Rings (1 and 4, cruises 1998-1999) for *Macoma* Tissue Metals.

Macoma	GLM Hypothesis Test Results				Normality Test Results		
Total Metals Measurement	Transformation Standardization Method	Inner-Ring Comparison (different if <0.05)	Inter-Cruise Comparison (different if <0.05)	Ring-Cruise Interaction (important if <0.05)	Shapiro-Wilks Value	Probability of Non-Normal Distribution (non-normal if <0.05)	Distribution
Cadmium	**Log Concentration**	**0.0187**	**0.1231**	**0.0104**	**0.9065**	**0.1402**	**Normal**
	None	0.0004	0.0029	0.0003	0.7174	0.0006	Non-Normal
	Rank	0.1307	0.6931	0.1398	0.9565	0.6646	Normal
Chromium	**Log Concentration**	**0.3064**	**0.1132**	**0.6756**	**0.9204**	**0.2228**	**Normal**
	None	0.5166	0.3797	0.6400	0.7173	0.0006	Non-Normal
	Rank	0.1331	0.0206	0.8125	0.9624	0.7626	Normal
Copper	**Log Concentration**	**0.0047**	**0.8708**	**0.0017**	**0.9315**	**0.3206**	**Normal**
	None	0.0006	0.1217	0.0002	0.8735	0.0470	Non-Normal
	Rank	0.0419	0.8740	0.0348	0.9645	0.7964	Normal
Lead	Log Concentration	0.6877	0.0008	0.0802	0.9023	0.1220	Normal
	None	**0.9559**	**0.0009**	**0.1061**	**0.9116**	**0.1660**	**Normal**
	Rank	0.5627	0.0007	0.0626	0.9645	0.7964	Normal
Mercury	Lipid	0.9356	0.0076	0.5365	0.9175	0.2023	Normal
	Log Concentration	0.3599	0.0000	0.4051	0.7763	0.0026	Non-Normal
	Log Lipid	**0.8366**	**0.0013**	**0.3215**	**0.9218**	**0.2333**	**Normal**
	None	0.6932	0.0000	0.8348	0.8243	0.0101	Non-Normal
	Rank	0.5389	0.0098	0.5389	0.9645	0.7964	Normal
Nickel	**Log Concentration**	**0.3997**	**0.1733**	**0.9844**	**0.8951**	**0.0958**	**Normal**
	None	0.4960	0.2927	0.8622	0.7948	0.0043	Non-Normal
	Rank	0.3052	0.0303	0.5616	0.9645	0.7964	Normal
Silver	Log Concentration	0.7243	0.0876	0.1837	0.9157	0.1907	Normal
	None	**0.8149**	**0.0591**	**0.1821**	**0.9337**	**0.3431**	**Normal**
	Rank	0.9081	0.0566	0.1793	0.9610	0.7399	Normal
Zinc	Log Concentration	0.5425	0.8662	0.6752	0.9210	0.2274	Normal
	None	**0.5464**	**0.9031**	**0.6569**	**0.9054**	**0.1350**	**Normal**
	Rank	0.5459	0.6373	0.8923	0.9645	0.7964	Normal

Values in **bold** provide best normalization of raw data.

Table 3-39. Two-way comparison of SINKEX Rings (1 and 4, cruises 1998-1999) for *Nephtys* Tissue Metals.

Nephtys	GLM Hypothesis Test Results				Normality Test Results		
Total Metals Measurement	Transformation Standardization Method	Inner-Ring Comparison (different if <0.05)	Inter-Cruise Comparison (different if <0.05)	Ring-Cruise Interaction (important if <0.05)	Shapiro-Wilks Value	Probability of Non-Normal Distribution (non-normal if <0.05)	Distribution
Cadmium	**Log Concentration**	**0.0077**	**0.0065**	**0.2108**	**0.9171**	**0.1994**	**Normal**
	None	0.0008	0.0009	0.0137	0.7538	0.0014	Non-Normal
	Rank	0.0866	0.0658	0.8721	0.9645	0.7964	Normal
Chromium	**Log Concentration**	**0.7790**	**0.0027**	**0.3420**	**0.9133**	**0.1759**	**Normal**
	None	0.7635	0.1240	0.7239	0.7254	0.0007	Non-Normal
	Rank	0.9478	0.0034	0.2314	0.9367	0.3778	Normal
Copper	Log Concentration	0.0006	0.0004	0.0007	0.6639	0.0002	Non-Normal
	None	0.0000	0.0000	0.0000	0.4005	0.0000	Non-Normal
	Rank	**0.2736**	**0.1261**	**0.5316**	**0.9645**	**0.7964**	**Normal**
Lead	**Log Concentration**	**0.1381**	**0.3635**	**0.0518**	**0.9334**	**0.3400**	**Normal**
	None	0.3452	0.4239	0.2677	0.8139	0.0075	Non-Normal
	Rank	0.2065	0.3938	0.0174	0.9645	0.7964	Normal
Mercury	Lipid	0.7800	0.0535	0.3877	0.9161	0.1933	Normal
	Log Concentration	0.1612	0.0002	0.0949	0.9519	0.5903	Normal
	Log Lipid	0.6672	0.0017	0.1251	0.9828	0.9879	Normal
	None	**0.6914**	**0.0046**	**0.3825**	**0.9671**	**0.8357**	**Normal**
	Rank	0.7314	0.0038	0.3739	0.9645	0.7964	Normal
Nickel	**Log Concentration**	**0.6961**	**0.0107**	**0.7180**	**0.9131**	**0.1745**	**Normal**
	None	0.8184	0.0584	0.8293	0.8446	0.0188	Non-Normal
	Rank	0.4801	0.0021	0.6872	0.9610	0.7399	Normal
Silver	Log Concentration	0.0027	0.0003	0.0567	0.8651	0.0359	Non-Normal
	None	0.0000	0.0000	0.0000	0.5860	0.0000	Non-Normal
	Rank	**0.1495**	**0.0084**	**0.9608**	**0.9557**	**0.6526**	**Normal**
Zinc	Log Concentration	0.0004	0.0004	0.0208	0.7884	0.0036	Non-Normal
	None	0.0001	0.0001	0.0031	0.7025	0.0004	Non-Normal
	Rank	**0.6536**	**0.0129**	**0.4187**	**0.9645**	**0.7964**	**Normal**

Values in **bold** provide best normalization of raw data.

3.3. Characterization of Ecological Effects from COCs

Consistent with the Characterization of Exposure, the Characterization of Ecological Effects focused on whether there was a detectable effect from the vessel, and this question was evaluated by determining whether there was a statistically significant difference between Inner Ring and Outer Ring stations. The three lines of evidence from the Decision Matrix used in this section were (1) sediment toxicity, (2) sediment chemistry, and (3) sediment bioaccumulation. Sediment toxicity is discussed first and is a direct measurement of effects, while the other two lines reflect estimations of effects based on exposure data previously discussed.

3.3.1. Sediment Toxicity in Surrogate Benthic Organisms for Benthic Infauna Endpoint

Results of the solid phase bioassays for *Rhepoxynius abronius* (survival and reburial) and *Neanthes arenaceodentata* (survival and growth) are summarized in Table 3-40 and Table 3-41, respectively (presented at the end of this section due to their large size). Percent survival data for bioaccumulation test organisms are shown in Figure 3-79 (*Rhepoxynius abronius*) and Figure 3-80 (*Neanthes arenaceodentata*) and presented in Table 3-42. Evaluation of toxicity test results and determination of toxicity was based on criteria from the Ocean Testing Manual (USEPA/USACE, 1991). It was decided that sediment toxicity would be indicated if any one of the following three criteria was met:

- Mean survival of *Rhepoxynius abronius* for the Inner Ring (Ring 1) was statistically lower than that of the Outer reference Ring (Ring 4) **and** this mean survival at the Inner Ring was reduced by ≥20% relative to the mean survival observed for the Outer reference Ring:
- Mean survival of *Neanthes arenaceodentata* for the Inner Ring (Ring 1) was statistically lower than that of the Outer reference Ring (Ring 4) **and** this mean survival at the Inner Ring was reduced by ≥10% relative to the survival observed for the Outer reference Ring;
- Growth of *Neanthes arenaceodentata* was statistically lower for the Inner Ring (Ring 1) than that of the Outer reference Ring (Ring 4). Only statistical significance is addressed for *Neanthes* growth since the relationship between growth and biologically significant effects has not been established.

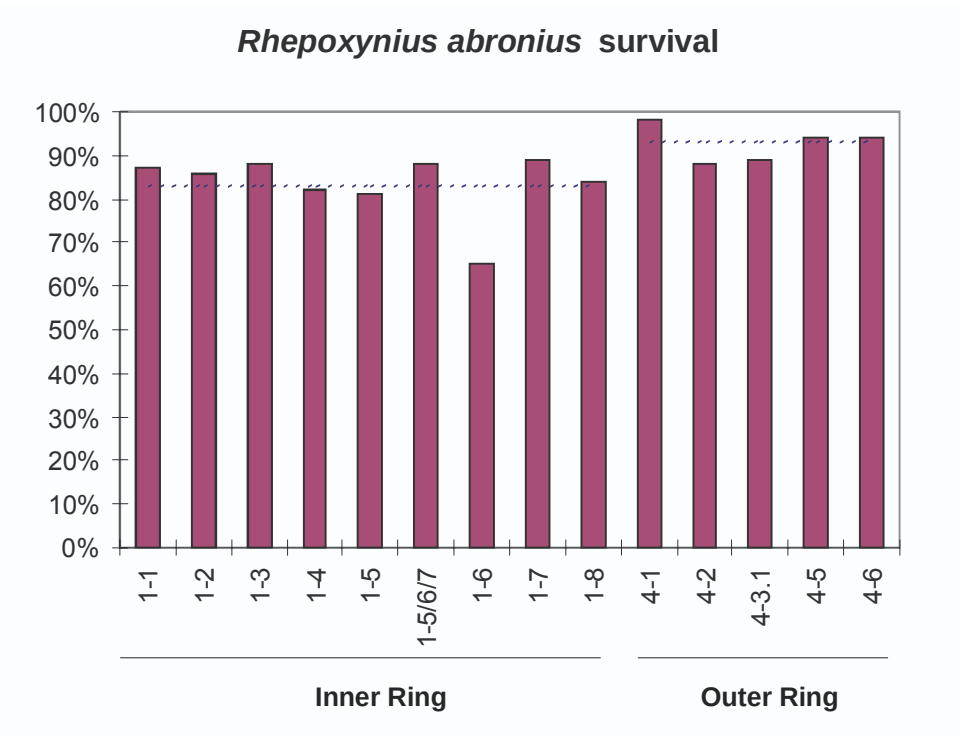


Figure 3-79. *Rhepoxynius abronius* survival.

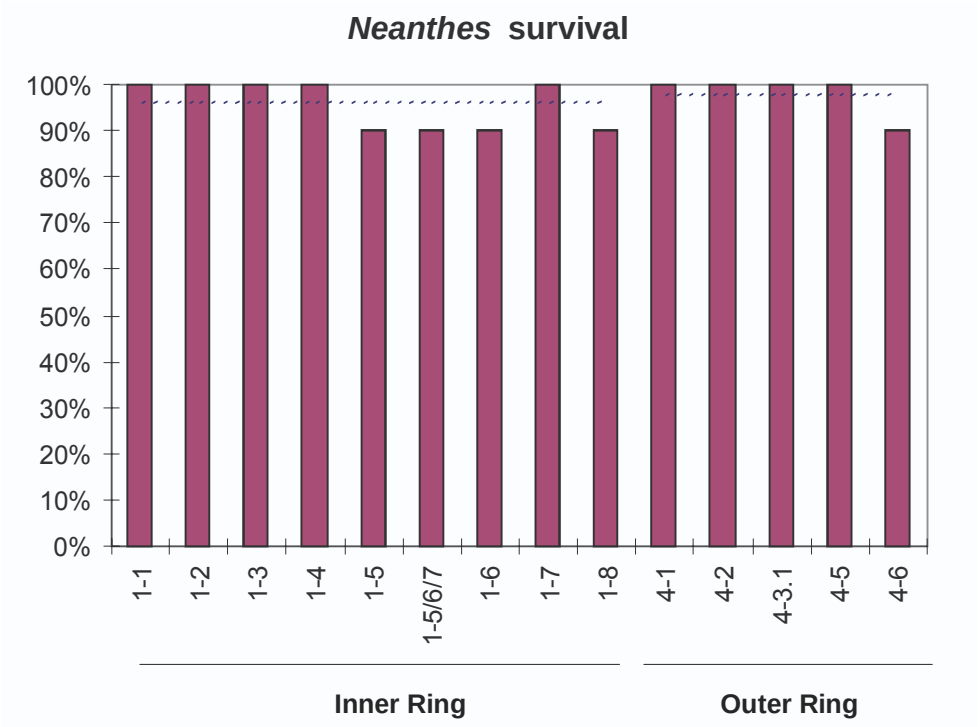


Figure 3-80. *Neanthes arenaceodentata* survival.

Nephtys caecoides and *Macoma nasuta* tests are used for the longer-term bioaccumulation analysis. Survival of these two organisms is not part of the decision criteria. Results indicated overlapping survival for *Nephtys* among Inner Ring stations (63-89% survival) compared with Outer Ring stations (64-94% survival) and narrower ranges for *Macoma*, 93-100% and 97-100% survival for Inner and Outer Ring, respectively.

The results of the statistical testing for toxicity and an analysis of the criteria used to indicate toxicity are presented in Table 3-43 and Table 3-44. Based upon Criteria 1-3 there were no statistically AND biologically significant responses when comparing Inner Ring versus Outer Ring for any of the biological toxicity measurements, except for the growth of *Neanthes arenaceodentata*. This indicates that the test organisms in sediments 1-2 m from the target vessel responded similarly to the organisms in sediments 1 km away from the target vessel in all cases except for reduced growth rates of the polychaete *Neanthes*. Issues related to growth measurements during this test are discussed below.

Also, based upon Criteria 1-3 there were no statistically and biologically significant responses among the measurement endpoints of *Rhepoxynius* reburial after 10 days of sediment exposure or *Neanthes* survival after 28 days of exposure or for *Macoma* and *Nephtys* survival during the bioaccumulation testing utilizing sediments from any of the stations within 1-2 m of the target vessel. *Rhepoxynius* reburial is used as an indicator that the amphipods survival test was conducted properly and not for determining toxicity (i.e., failure to rebury would invalidate the toxicity test). There was however, a single instance of obvious reduction in *Rhepoxynius* survival at one station (Station 1-6). *Neanthes* growth in sediments from all of the Inner Ring stations was also statistically significantly lower than the growth of these organisms when exposed to

sediments from the Outer Ring stations (Table 3-43 and Table 3-44). Both of these endpoint results can be explained by reasons other than toxicity.

The observed difference in growth for *Neanthes* exposed to sediments from the Inner Ring compared to those exposed to sediment from the Outer Ring should not be considered indicative of sediment toxicity for four main reasons:

1. There was a statistically significant difference in the starting mean weight of *Neanthes* available for conducting growth tests for the different SINKEX cruises. *Neanthes* growth is measured through increases in worm biomass as opposed to worm length. There is only one supplier of *Neanthes* for bioassay studies; consequently availability and size of organisms is variable, which resulted in different initial worm weights for the two chronic toxicity-testing events in this study. Additionally, the decision was made to collect reference sediments for each cruise from the same location, so that test comparisons to reference sediments could be made. Unfortunately, difficulties in sediment sampling around the ex-AGERHOLM (e.g., phosphatized sediments and gravels made sample collection difficult, additional factors included heavy seas, equipment problems, cycle time for sample collection, and time to relocate to new sampling area) precluded a balanced sampling design over time. During Cruise II in September 1999 eight Inner Ring stations and one Outer Ring station were sampled and the sediments tested and the average initial weight of the worms was 0.1015 mg, Cruise IV September 1999 sampled four Outer Ring stations and the average initial worm weight for the first chronic test was 0.298 mg (almost three times the average weight of worm used to characterize the Inner Ring during Cruise II), and Cruise V November 1999 sampled only one Inner Ring station and the average initial weight of the worms for the second chronic test was 0.555 mg (considerably larger than either of the two earlier cruises). A t-Test on mean size for Cruise II (mostly Inner Ring stations) and Cruise IV (exclusively Outer Ring stations) found that the worms used for Cruise IV growth tests were significantly larger ($p<0.014$) than those used for Cruise II tests. Thus, when comparing Inner Ring stations (mostly Cruise II small worms) against Outer Ring stations (mostly Cruise 4 medium size worms) it is not surprising that the Outer Ring ended up with larger worms and larger daily growth rates and produced a statistically significant result indicating greater growth for the Outer Ring. This significant result for greater growth for the Outer Ring was because significantly larger worms (greater biomass) were used for Cruise IV Outer Ring sediments. Consequently, the statistical test for *Neanthes* growth between Inner and Outer Ring reflects the initial starting weight of the worms and is not an appropriate assessment of potential toxicity.
2. Figure 3-81 provides a graphical view of the differences in average growth rates for Cruises II and IV and shows the growth rates for all stations for all three cruises. This figure illustrates that the growth rates for all stations, both near and far field (except Station 1-4) were less than control growth rates. Furthermore, growth relative to control was depressed about 17% for the Inner Ring while Outer Ring growth rates were depressed by about 30%. Thus, the general conclusion is that when worm size is considered in the evaluation for potential sediment toxicity, the sediments from the Inner Ring appear to be less harmful than those at the Outer Ring, supporting the contention that there is no evidence of sediment toxicity for the sediments nearest the vessel. Additionally, *Neanthes* growth at Station 1-6, the station with reduced *Rhepoxynius*

abronius survival, was about average for the Inner Ring stations, but greater than the average for the Outer Rings stations. This would suggest that sediments at Station 1-6 exhibit no toxicity as indicated by *Neanthes* growth.

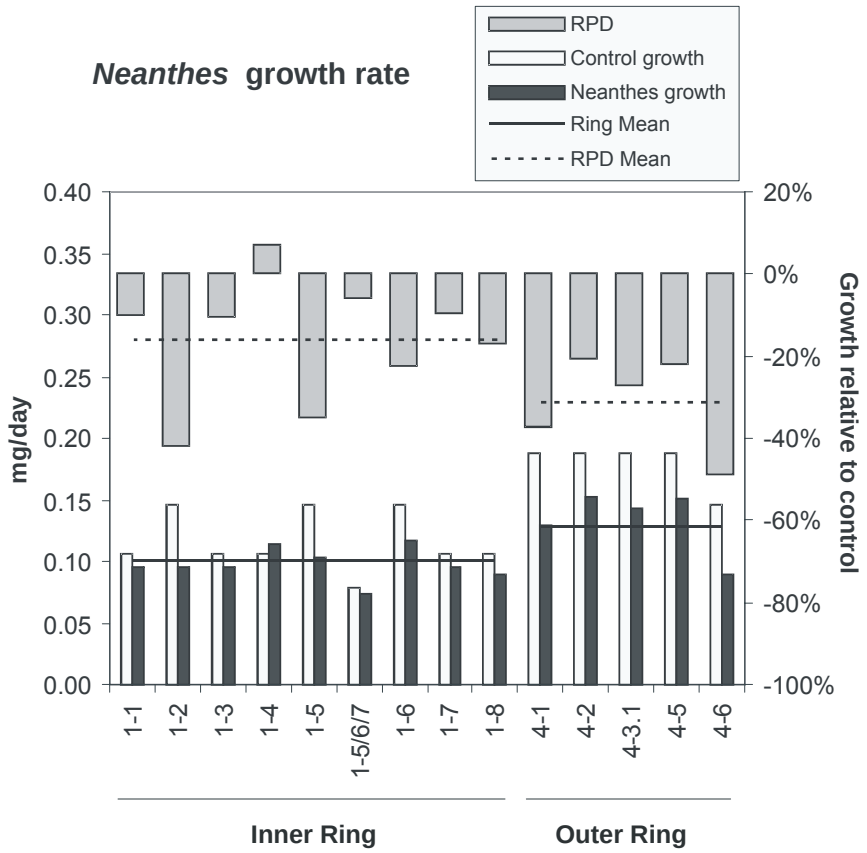


Figure 3-81. *Neanthes arenaceodentata* growth rates (RPD=Relative Percent Difference).

3. An analysis of the 8 Inner Ring stations of Cruise II against the one Outer Ring station sampled at the same time (Station 4-6, which for this test is considered to represent reference sediment) found no significant difference (p=0.569 Dunnett's T test) in growth for the individual Inner Ring stations compared to the one Outer Ring station 4-6. This comparison is valid as the starting weights of the worms from the Inner Ring were the same as that for the Outer Ring Station 4-6. This would indicate that the Inner Ring sediments did not significantly affect *Neanthes* growth and therefore, no toxicity is indicated.
4. *Neanthes* growth could be affected by differences in grain size, organic carbon content, or chemical contaminants in the sediments. There were significant differences between the Inner and Outer Ring for grain size measures and some sediment contaminants (Cd, Cu, Ni, Ag, in Table 3-24); therefore some differences in growth might be expected. Sediments for the Inner Ring stations had significantly more gravel (Figure 3-5), greater median grain size (mm) (Figure 3-8), significantly less silt (Figure 3-7) and sand (Figure 3-6), and significantly higher concentrations of Cd, Cu, Ni and Ag (3.2.2.4.10). That some of these differences could affect *Neanthes* growth is suggested by the regression analysis that shows significant and strong negative correlations with % gravel (r^2=0.58,

p=0.004), significant but weaker correlations with median grain size in mm (r^2=0.27, p=0.058), and a positive correlation with % silt (r^2=0.44, p=0.01) (Figure 3-82). As would be expected from these results the fine grain size PCA factor 2 was also weakly correlated (r^2=0.36, p=0.023) with *Neanthes* growth. Finally, PCA factor 3, which included Cd and Cu, and PCA factor 1, which included Ni, Ag, and TOC, were both poorly correlated with growth and non significant (r^2=0.10 and r^2=0.06, respectively).

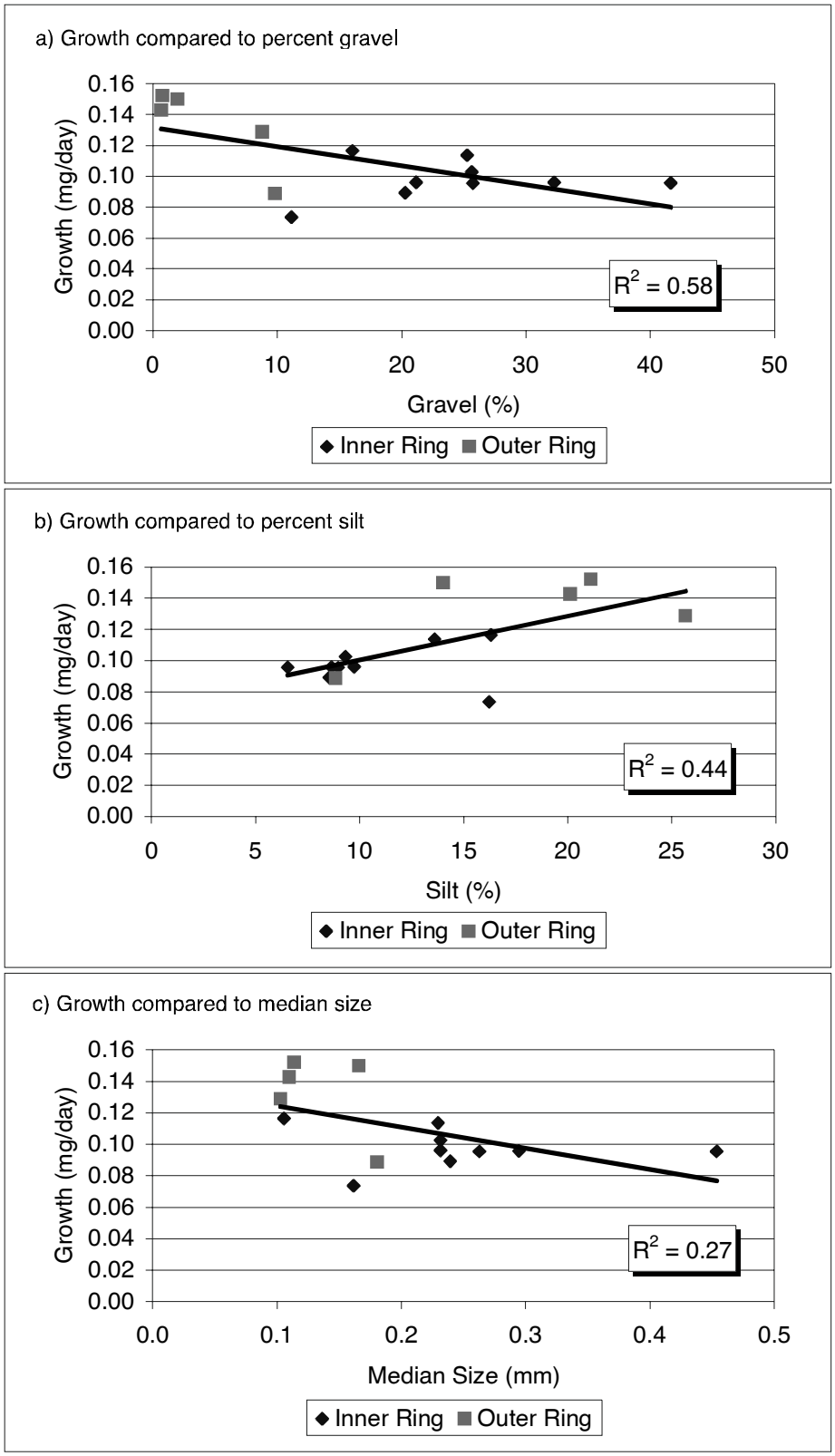


Figure 3-82. Regression of *Neanthes arenaceodentata* growth with grain size measures.

3.3.1.1. Confounding Factors Potentially Affecting Toxicity and Bioassay Results

Table 3-45 and Table 3-46 list the results of the laboratory analyses for grain size and carbon by station and survey, respectively. Table 3-47 presents the results for testing the grain size measures for compliance with ANOVA normality assumptions and whether the data was transformed. Table 3-48 presents the results of the statistical testing for grain size, organic and inorganic carbon parameters. The Inner Ring had significantly more gravel (24.4%) than the Outer Ring (4.4%) and consequently the median grain size for the Inner Ring was significantly greater, almost double that of the Outer Ring. Because the Inner Ring sediments had more gravel, they also had significantly less sand (51.7%) and silt (10.9%) than the Outer Ring (65.5% sand 18.0% silt). Thus, the Inner Ring sediments are coarser having larger particle sizes and fewer of the finer sediments than the Outer Ring. This represents a significant and striking physical difference between the sedimentary environments near the vessel compared to the reference areas. This physical difference in turn could affect the infaunal community, partitioning of contaminants to sediment particle surfaces, and bioavailability. Furthermore, total organic carbon (TOC) was about 2.1 times greater for the Inner Ring compared to the Outer Ring, and while this was not statistically significant ($p=0.078$) at the $p<0.05$ level, it is close to statistical significance and the higher TOC values for the Inner Ring may indeed have biological significance

What is unusual about these results is that typically TOC values are highly correlated with fine sediment fractions, but not at the ex-AGERHOLM study sites. This lack of a strong correlation between sediment fines and TOC concentrations is perhaps an indication that the TOC measured in this study was not detrital TOC closely associated with fine sediment particles but was, in part, a measure of the biomass of foraminifers, which comprised a significant portion of the sediment matrix. This is supported to some extent by the high proportion of $CaCO_3$ in the sediments, which appear to result from the abundant tests (i.e. shells) of both living and dead foraminifers. However, there was no statistical difference between the Inner and Outer Ring for CaCO3 content to support the hypothesis that higher TOC values in the Inner Ring were due to greater foraminifera biomass. It could be hypothesized that the Outer Ring had fewer live foraminifers than the Inner Ring, possibly accounting for the greater amount of TOC at the Inner Ring, but quantifying living foraminifera was not an objective of the study and this determination was not made.

The original and reanalyzed TOC values from Cruise II were also on the average fairly similar, except in the case of the larger TOC values noted in the original set of data (those exceeding ~6% in the original data). In the five cases of TOC values in excess of 6% during the original analysis, all were found to have less TOC during the reanalysis. During the review of the TOC data it was also noted that two different instruments with slightly different protocols had been used for determining TOC concentrations. Samples from Cruise II, September 1998, had utilized a Carlo-Erba analyzer that analyzed a 10 mg sediment sample. In contrast, samples from Cruises 4 and 5 utilized a Shimadzu analyzer that analyzed a 200 mg sediment sample. Reanalysis of sediment samples utilized only the Shimadzu analyzer and protocol. Reanalysis of sediment samples that were originally analyzed using the Shimadzu instrument found relatively minor differences, which generally were slightly lower (~94% of the original TOC determination). But reanalysis of the selected samples from Cruise II, which were originally analyzed using the Carlo-Erba instrument found larger differences. For example, three of the four samples that originally had TOC concentrations greater than 6% were found to have TOC concentrations that

were only 33-70% of the original values. However, in the fourth sample there was essentially no difference between the two methods (Table 3-49). In fact, the average concentration for the original measurements of TOC was ~142% of the reanalyzed values on the Shimadzu instrument. The smaller samples analyzed using the Carlo-Erba instrument appear to have overestimated the TOC concentrations, probably indicating a bias created from intersample variation and/or selection of more organically enriched particles during the initial measurements. Table 3-46 lists the TOC values used for the analyses in this report, and includes the remeasured values for most of the Inner Ring stations. However, a retesting for statistical differences between the Inner and Outer Ring was still not significant.

The results of ANOVA analysis testing for differences between Inner and Outer Ring stations showed no significant differences for the sediment measures percent clay, percent silt-clay, percent TOC, and percent calcium carbonate for the original sets of data (Table 3-48). Similarly, there was no difference between the rings for *Rhepoxynius abronius* reburial, *Neanthes arenaceodentata* survival, *Macoma nasuta* survival, or *Nephtys caecoides* survival (Table 2-11). However, the Inner Ring had significantly more gravel and less sand and silt, and this resulted in the median grain size being significantly greater for the Inner Ring. The Inner Ring also had significantly greater concentrations of Cd, Cu, Ni, and Ag, but showed no difference for total PCBs and PAHs. Differences in grain size measures as well as sediment contaminants can influence bioassay/toxicity endpoints because test organisms have preferred sediment sizes (e.g., DeWitt et al. 1988). In general, the test organism that was selected, *Rhepoxynius abronius*, seems to prefer coarser grained sediment. It generally demonstrates increased toxicity under finer grained sediment conditions. However, the sediments tested for this study are not only coarser but also contain higher levels of carbonate, whose effects have not been studied or addressed.

Neanthes arenaceodentata growth was statistically higher for the Outer Ring stations compared to the Inner Ring stations. As explained previously, the apparent higher growth rate for *Neanthes* in the Outer Ring was primarily due to utilizing different size worms and the differences in sediment grain size measures between the Inner and Outer Ring. When site worms were compared in relative ratios to their respective controls, the Inner Ring worms actually grew more than their Outer Ring counterparts. Further, when *Neanthes* growth for Cruise II was tested relative to reference (Stations 4-6) there was no statistical difference in growth.

Survival of *Rhepoxynius abronius* was significantly greater for the Outer compared to Inner Ring stations (Table 3-43 and Figure 3-79), but the difference in survival was only 10% and not 20%, which would indicate the lack of biologically significant toxicity as defined by Ocean Disposal criteria (Table 3-44). The lower survival of *Rhepoxynius abronius* could also be influenced by physical characteristics of the sediments. Outer Ring sediments have a higher proportion of sand and silt (83.5%), with generally low percentages of gravel (4.4%), while Inner Ring sediments have less sand and silt (62.6%) and significantly more gravel (24.4%) (Figure 3-83, Figure 3-84 and Table 3-45). The higher sand silt content and low proportion of gravel may be more favorable to survival of this species. *Rhepoxynius abronius* prefers sandy sediments and is known to be adversely affected by fine sediments with high clay content but the influence of higher percentages of gravel or carbonates content is unknown (DeWitt et al. 1988).

Regression analyses were run for *Rhepoxynius abronius* survival with grain size measures (percent gravel, sand, silt, clay, and median phi), as well as the PCA factors that included sediment contaminants. For the individual parameter regressions, *Rhepoxynius abronius* survival showed a significant and strong positive correlation (r^2=0.67**,** p=0.0003) with percent sand and a

significant but weak negative correlation (r^2=0.29, p=0.039) with percent clay (Figure 3-85). None of the PCA factors showed a significant correlation with survival. Station 1-6 (Cruise II) had the lowest *Rhepoxynius* survival (65%) of any station tested (Table 3-40). Station 1-6 also had the lowest proportion of sand (39.4%) and highest clay content (28.3%), compared with other stations (Table 3-45).

The regression analysis indicates that these grain size measures may affect *Rhepoxynius* survival. Survival in less than preferred sediment sizes may stress organisms and directly affect their survival or sensitivity to contaminants. Furthermore, the significant decrease in *Rhepoxynius* survival for the Inner Ring appears to be due to the low survival observed at Station 1-6. The low survival at Station 1-6 appears to result from the low proportion of preferred sandy sediments, which appears to be a consequence of Station 1-6 sediments having high proportions of gravel and silt, and not from toxicity due to contaminants. Survival may also be influenced by lack of food, such as not enough TOC or low TOC quality. The effect of high $CaCO_3$ in the sediments (particularly in the Outer Ring) on survival is unknown. Finally, we must consider that the *Rhepoxynius* toxicity at station 1-6 (65% survival) may be due to additive or synergistic toxicity from the combination of contaminants measured and those that were not investigated/suspected. However, though PCBs were elevated in the Inner Ring sediments, they were far below benchmark effects levels. Additional evidence supporting that PCBs did not cause toxicity at Station 1-6 includes the following facts: (1) there was no significant regression or correlation with PCA factor 3 (which included total PCB concentrations), (2) Station 1-7 (with 89% survival) appears to be more contaminated than 1-6, yet it does not meet the threshold for toxicity, and (3) the apparent lack of effects on *Neanthes* growth at this station.

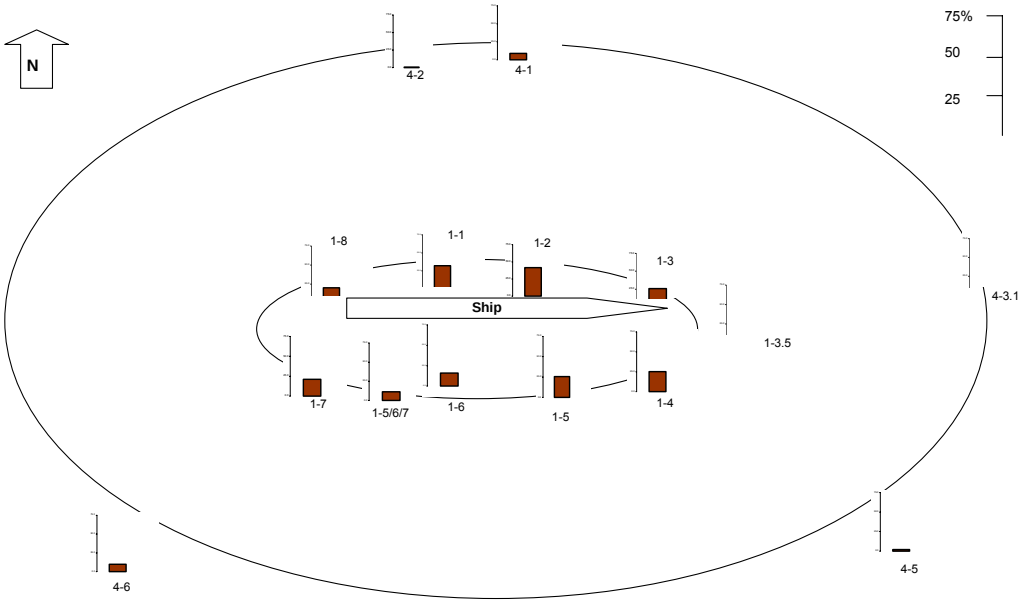


Figure 3-83. Sediment percent gravel content, Inner and Outer Ring stations.

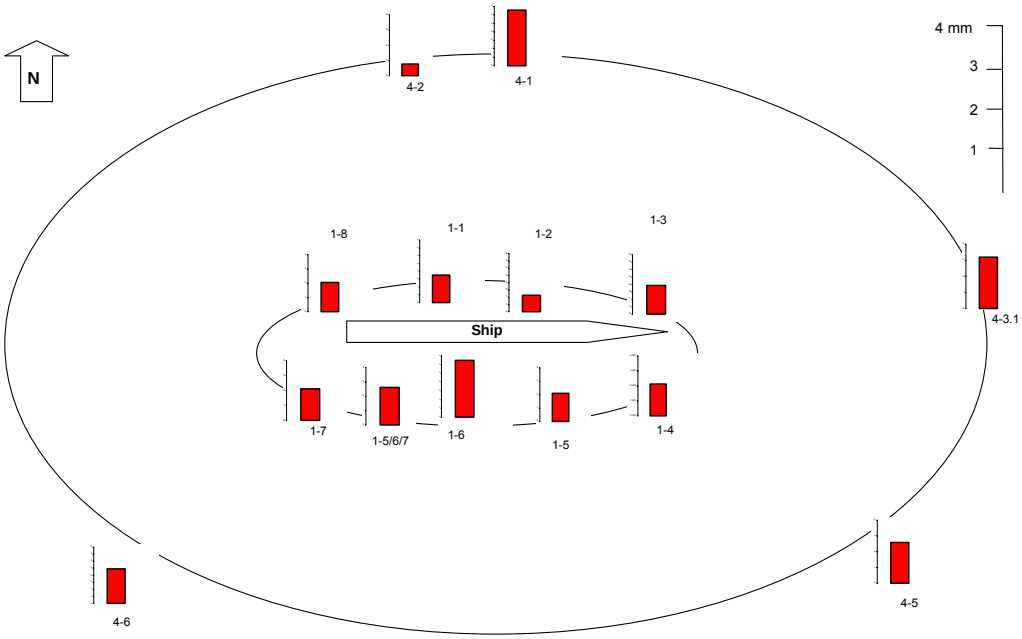


Figure 3-84. Sediment median grain size, Inner and Outer Ring stations.

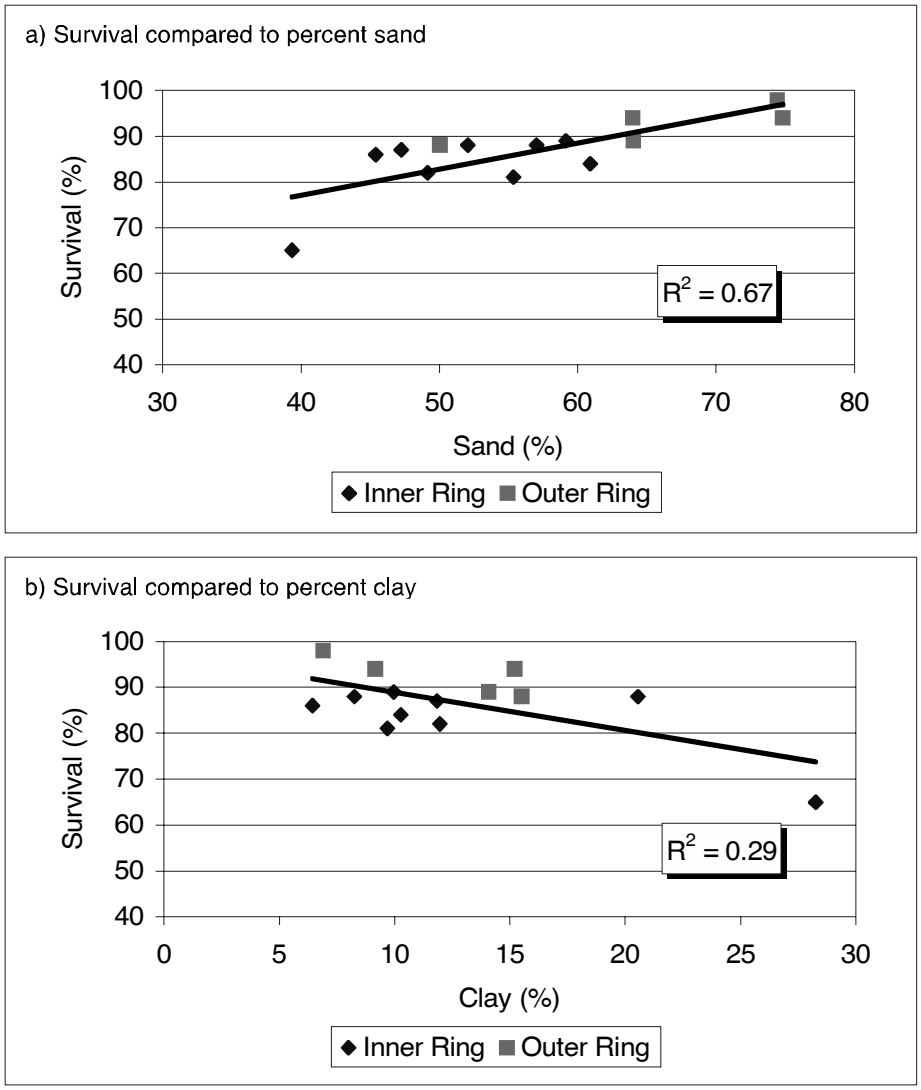


Figure 3-85. Regression of *Rhepoxynius abronius* survival with grain size measures.

One final concern about *Rhepoxynius* survival was the concentration of ammonia during the toxicity testing. Ammonia is of concern because at high concentrations this compound is toxic to most test organisms. Table 3-50 lists the initial and final interstitial porewater concentration for the Rhepoxynius bioassay. Both initial and final ammonia porewater concentrations were very low, as were ammonia concentrations in the overlying waters and, therefore, determined not to be a factor contributing to lower survival.

3.3.1.2. Conclusions

Significant differences were found between the Inner and Outer Ring stations relative to *Rhepoxynius abronius* survival, *Neanthes arenaceodentata* growth, grain size measures, and four sediment metals (Cd, Cu, Ni, and Ag). Sediments from the Inner Ring had significantly more gravel, greater median grain size, higher $CaCO_3$, less sand and silt, and higher concentrations for the four metals than the Outer Ring. These differences have the potential to affect the bioassay/toxicity results. *Rhepoxynius* survival was significantly lower for the Inner Ring compared to the Outer Ring, but survival was only 10% lower for the Inner Ring. Therefore,

based upon Ocean Disposal guidelines, this difference is not considered indicative of biologically significant toxicity. Survival would have to be 20% lower in order for the sediments to be considered to have biologically significant toxicity. Lower *Rhepoxynius* survival for the Inner Ring appears to be due mostly to a significant difference in grain size measures between the two Rings, and not to differences in sediment contaminants. Lower growth for *Neanthes* at the Inner Ring was due to differences in the size of worms available at the different times the tests were conducted. However, relative comparisons to controls showed better growth performance for the Inner Ring, and the Inner Ring comparison to the one Outer Ring station (4-6) that used the same worms showed no difference. There may also be a relationship between reduced growth rates and available TOC. Statistical testing for differences in growth relative to a reference station (Station 4-6) for Cruise II found no significant difference in *Neanthes* growth when exposed to Inner Ring sediments compared to Outer Ring Station 4-6. Based on these bioassay/toxicity results, the COCs associated with the ex-AGERHOLM appear not to pose a significant risk to biota in the area.

Considering the Infanual Community evaluation, stations having reductions in the number of species and/or abundance and decreased diversity or increased dominance by a few species could be a response to physical disturbance, toxicity, or reduction in food supply, and/or food quality and availability at these locations. To determine whether the decreased number of species and abundance at the Inner Ring are related to changes in food quality and availability, toxicity, or simply disturbance, the results of the toxicity and bioaccumulation studies were re-evaluated. The assumption was that if the infaunal community was being adversely impacted by sediment factors then the toxicity testing (to be reviewed in the following section) should correlate with infaunal community changes. However, the toxicity testing would not distinguish between the presence of toxicants in the sediment, or lack of food in these sediments, because either condition could result in adverse effects. There was, however, no demonstrated toxicity at these stations. The observation of reduced infaunal community measures (e.g., Stations 1-4, 1-3, 1-1, 1-5, and 1-8) coupled with the lack of toxicity (following section) for these stations is strongly suggestive that sediment contaminants are not causing these reductions and other factors related to site differences, such as grain size, TOC, sediment disturbances and “reef effects,” are more plausible explanations. Similarly, while laboratory toxicity was indicated for Station 1-6, there is no supporting evidence that the infaunal community at this station was affected. This supports the contention that the reduction of *Rhepoxynius abronius* survival at this location was most likely due to factors other than toxicity associated with the contaminant.

Table 3-40. Survival and reburial for *Rhepoxynius abronius*. Results are listed by station and replicate, with the mean and standard deviation for each station included.

Station	Replicate	Survival (%)			Reburial (%)		
		Survival	Mean	Standard Deviation	Reburial	Mean	Standard Deviation
Survey - September 1998							
1-1	1	80	87	0.091	100	100	0
	2	95			100		
	3	90			100		
	4	95			100		
	5	75			100		
1-2	1	80	86	0.065	100	99	0.024
	2	85			100		
	3	90			100		
	4	80			100		
	5	95			95		
1-3	1	100	88	0.091	100	96	0.069
	2	80			100		
	3	85			94		
	4	95			84		
	5	80			100		
1-4	1	95	82	0.140	100	100	0
	2	85			100		
	3	65			100		
	4	95			100		
	5	70			100		
1-5	1	65	81	0.119	92	98	0.034
	2	80			100		
	3	95			100		
	4	90			100		
	5	75			100		
1-6	1	60	65	0.146	100	94	0.055
	2	50			90		
	3	85			88		
	4	75			93		
	5	55			100		
1-7	1	95	89	0.108	100	100	0
	2	95			100		
	3	90			100		
	4	70			100		
	5	95			100		
1-8	1	90	84	0.065	100	100	0
	2	80			100		
	3	85			100		
	4	75			100		
	5	90			100		
4-6	1	100	98	0.045	100	100	0
	2	90			100		
	3	100			100		
	4	100			100		
	5	100			100		

(continued)

Table 3-40. Continued.

Station	Replicate	Survival (%)			Reburial (%)		
		Survival	Mean	Standard Deviation	Reburial	Mean	Standard Deviation
Survey - September 1999							
4-1	1	90	88	0.045	100	99	0.025
	2	90			100		
	3	90			100		
	4	90			94		
	5	80			100		
4-2	1	80	89	0.065	94	97	0.032
	2	90			100		
	3	95			100		
	4	85			94		
	5	95			95		
4-3.1	1	85	94	0.065	100	99	0.022
	2	95			100		
	3	90			100		
	4	100			100		
	5	100			95		
4-5	1	100	94	0.042	100	100	0
	2	90			100		
	3	95			100		
	4	90			100		
	5	95			100		
Survey - November 1999							
1-5/6/7	1	90	88	0.057	100	99	0.024
	2	80			100		
	3	85			100		
	4	95			95		
	5	90			100		

Table 3-41. Survival and Growth for *Neanthes arenaceodentata*. Results are listed by station and replicate, with mean and standard deviation for each station included.

Station	Replicate	Survival (%)			Growth (mg/day)		
		Survival	Mean	Standard Deviation	Growth	Mean	Standard Deviation
Survey - September 1998							
1-1	1	100	100	0	0.105	0.096	0.024
	2	100			0.062		
	3	100			0.073		
	4	100			0.108		
	5	100			0.072		
	6	100			0.092		
	7	100			0.134		
	8	100			0.124		
	9	100			0.094		
	10	100			--		
1-2	1	100	100	0	0.061	0.096	0.044
	2	100			0.171		
	3	100			0.107		
	4	100			0.147		
	5	100			0.120		
	6	100			0.022		
	7	100			0.076		
	8	100			0.072		
	9	100			0.069		
	10	100			0.112		
1-3	1	100	100	0	0.108	0.096	0.033
	2	100			0.060		
	3	100			0.103		
	4	100			0.063		
	5	100			0.112		
	6	100			0.129		
	7	100			0.146		
	8	100			0.081		
	9	100			0.111		
	10	100			0.043		
1-4	1	100	100	0	0.097	0.114	0.022
	2	100			0.100		
	3	100			0.109		
	4	100			0.095		
	5	100			0.101		
	6	100			0.156		
	7	100			0.130		
	8	100			0.087		
	9	100			0.129		
	10	100			0.133		
1-5	1	100	90	0.316	0.117	0.103	0.031
	2	100			0.084		
	3	100			0.068		
	4	0			--		
	5	100			0.089		
	6	100			0.111		
	7	100			0.123		
	8	100			0.141		
	9	100			0.137		
	10	100			0.053		

(continued)

Table 3-41. Continued.

Station	Replicate	Survival (%)			Growth (mg/day)		
		Survival	Mean	Standard Deviation	Growth	Mean	Standard Deviation
Survey - September 1998 (Continued)							
1-6	1	100	90	0.316	0.090	0.116	0.037
	2	100			0.172		
	3	100			0.076		
	4	100			0.135		
	5	100			0.074		
	6	100			0.167		
	7	0			--		
	8	100			0.123		
	9	100			0.119		
	10	100			0.093		
1-7	1	100	100	0	0.141	0.096	0.041
	2	100			0.130		
	3	100			0.109		
	4	100			0.080		
	5	100			0.149		
	6	100			0.064		
	7	100			0.037		
	8	100			0.032		
	9	100			0.112		
	10	100			0.108		
1-8	1	100	100	0	0.093	0.089	0.025
	2	100			0.060		
	3	100			0.058		
	4	100			0.119		
	5	100			0.106		
	6	100			0.049		
	7	100			0.097		
	8	100			0.116		
	9	100			0.098		
	10	100			0.095		
4-6	1	100	90	0.316	0.054	0.089	0.029
	2	100			0.047		
	3	100			0.100		
	4	100			0.086		
	5	100			0.083		
	6	100			--		
	7	100			0.090		
	8	0			--		
	9	100			0.130		
	10	100			0.121		
Survey - September 1999							
4-1	1	100	100	0	0.112	0.129	0.032
	2	100			0.110		
	3	100			0.163		
	4	100			0.097		
	5	100			0.085		
	6	100			0.125		
	7	100			0.118		
	8	100			0.172		
	9	100			0.180		
	10	100			0.126		

(continued)

Table 3-41. Continued.

Station	Replicate	Survival (%)			Growth (mg/day)		
		Survival	Mean	Standard Deviation	Growth	Mean	Standard Deviation
Survey - September 1999 (Continued)							
4-2	1	100	100	0	0.128	0.152	0.033
	2	100			0.111		
	3	100			0.148		
	4	100			0.176		
	5	100			0.097		
	6	100			0.200		
	7	100			0.174		
	8	100			0.148		
	9	100			0.180		
	10	100			0.160		
4-3.1	1	100	100	0	0.153	0.143	0.033
	2	100			0.163		
	3	100			0.102		
	4	100			0.150		
	5	100			0.138		
	6	100			0.184		
	7	100			0.090		
	8	100			0.193		
	9	100			0.115		
	10	100			0.141		
4-5	1	100	100	0	0.104	0.150	0.043
	2	100			0.108		
	3	100			0.142		
	4	100			0.168		
	5	100			0.131		
	6	100			0.149		
	7	100			0.146		
	8	100			0.228		
	9	100			0.111		
	10	100			0.214		
Survey - November 1999							
1-5/6/7	1	100	90	0.316	0.083	0.074	0.022
	2	100			0.065		
	3	100			0.090		
	4	100			0.070		
	5	100			0.104		
	6	100			0.090		
	7	100			0.027		
	8	100			0.063		
	9	0			--		
	10	100			0.069		

Table 3-42. Survival for *Macoma nasuta* and *Nephtys caecoides* by station and survey.

Station	Survival (%)	
	Macoma nasuta	*Nephtys caecoides*
Survey - September 1998		
1-1	97	89
1-2	97	72
1-3	93	84
1-4	93	86
1-5	100	67
1-6	100	71
1-7	93	84
1-8	97	63
4-6	97	64
Survey - September 1999		
4-1	97	87
4-2	100	92
4-3.1	100	94
4-5	100	79
Survey - November 1999		
1-5/6/7	93	82

Table 3-43. Results of ANOVAs tests for determination of toxicity. Inner (I) / Outer (O) stations.

Endpoint / Parameter	Data Transformation	p-value	Mean Values	Conclusion Summary
Macoma nasuta survival (%)	Ranks	0.074	I=96% O=99%	NS
Neanthes arenaceodentata growth (mg/day)	Non transformed	0.005*	I=0.10 O=0.13	O > I
Neanthes arenaceodentata relative to controls (%)	Non transformed	0.071	I=-16.1% O=-31.1%	NS
Neanthes arenaceodentata survival (%)	Ranks	0.400	I=96% O=98%	NS
Nephtys caecoides survival (%)	Arcsine	0.284	I=78% O=83%	NS
Rhepoxynius abronius reburial (%)	Ranks	0.784	I=98% O=99%	NS
Rhepoxynius abronius survival (%)	Arcsine	0.011*	I=83% O=93%	O > I

* $p<0.05$
NS=nonsignificant

Table 3-44. Analysis of criteria used to indicate toxicity.

Rhepoxynius abronius Survival		*Neanthes arenaceodentata* Survival		*Neanthes arenaceodentata* Growth	*Neanthes arenaceodentata* Growth Relative to Control
Inner Ring Significantly Lower than Outer Ring	**Inner Rng <20% Lower than Outer Ring**	**Inner Ring Significantly Lower than Outer Ring**	**Inner Ring <10% Lower than Outer Ring**	**Inner Ring Significantly Lower than Outer Ring**	**Inner Ring Significantly Lower than Outer Ring**
Yes	No	No	NA	Yes	No

NA=Not Applicable, no significant difference between Inner and Outer Rings.

Table 3-45. Grain size parameters by station and survey.

STATION	GRAVEL (%)	SAND (%)	SILT (%)	CLAY (%)	MEDIAN PHI	MEAN PHI
Survey - September 1998						
1-1	32.284	47.237	8.637	11.842	1.764	NC
1-2	41.618	45.404	6.531	6.447	1.141	NC
1-3	25.728	57.031	8.970	8.271	1.928	NC
1-4	25.268	49.148	13.608	11.976	2.123	1.816
1-5	25.624	55.364	9.319	9.693	2.111	0.840
1-6	16.056	39.356	16.317	28.271	3.244	4.728
1-7	21.154	59.153	9.736	9.957	2.110	0.833
1-8	20.288	60.908	8.533	10.271	2.063	0.867
4-6	9.812	74.431	8.846	6.911	2.470	2.649
Survey - September 1999						
4-1	8.802	50.033	25.651	15.514	3.283	4.373
4-2	0.764	64.037	21.108	14.092	3.135	4.454
4-3.1	0.670	63.990	20.124	15.216	3.188	4.662
4-5	1.993	74.818	14.004	9.185	2.593	3.495
Survey - November 1999						
1-5/6/7	11.157	52.058	16.226	20.560	2.630	4.766

Table 3-46. Percent total organic carbon and calcium carbonate by station and survey.

Station	Total Organic Carbon (%)	Calcium Carbonate (%)
Survey - September 1998		
1-1	5.06*	58.0
1-2	1.84*	22.2
1-3	9.08*	28.5
1-4	2.75*	41.7
1-5	2.32	29.3
1-6	4.93*	58.3
1-7	2.41*	35.9
1-8	4.43*	54.3
4-6	2.57	57.4
Survey - September 1999		
4-1	1.63	44.8
4-2	1.10	53.8
4-3.1	1.85	56.5
4-5	1.31	52.6
Survey - November 1999		
1-5/6/7	1.01	69.0

*Values are based upon reanalysis of sediment samples, see Section **Error! Reference source not found.**.

Table 3-47. Results of testing grain size measures, TOC, and $CaCO_3$ to comply with assumption of statistical testing. Bold indicates the transformation used in the analysis.

Test Parameter	Transformation	Shapiro-Wilk's Test			Levenes test for Equality of Variance
		W=Tendency for Normality	P=Probability for Normality	Distribution	
Percent gravel	**Arcsin**	**0.967**	**0.784**	**Normal**	**0.493**
Percent gravel	Non transformed	0.956	0.621	Normal	0.371
Percent sand	**Arcsin**	**0.939**	**0.389**	**Normal**	**0.441**
Percent sand	Non transformed	0.932	0.318	Normal	0.540
Percent silt	Arcsin	0.953	0.571	Normal	0.197
Percent silt	**Non transformed**	**0.964**	**0.741**	**Normal**	**0.095**
Percent clay	**Arcsin**	**0.914**	**0.176**	**Normal**	**0.586**
Percent clay	Non transformed	0.863	0.033	Non-normal	0.409
Percent silt-clay	**Arcsin**	**0.956**	**0.616**	**Normal**	**0.830**
Percent silt-clay	Non transformed	0.945	0.465	Normal	0.822
Median grain size in mm	**Log 10**	**0.965**	**0.762**	**Normal**	**0.846**
Median grain size in mm	Non transformed	0.867	0.038	Non-normal	0.371
Percent total organic carbon	Arcsin	0.939	0.308	Normal	0.012
Percent total organic carbon	Non transformed	0.872	0.023	Non-normal	0.012
Percent total organic carbon	**Ranks**	**NA**	**NA**	**NA**	**NA**
Percent calcium carbonate	Arcsin	0.969	0.780	Normal	0.166
Percent calcium carbonate	**Non transformed**	**0.971**	**0.806**	**Normal**	**0.159**

Table 3-48. Results of ANOVAs tests for grain size parameters, TOC, and $CaCO_3$. Significant differences between Inner (I) and Outer (O) Ring stations noted in bold.

Endpoint / Parameter	Data Transformation	p-value	Mean Values	Conclusion Summary
Percent gravel	**Arcsine**	**0.000**	**I=24.4% O=4.4%**	**I > O**
Percent sand	**Arcsine**	**0.011**	**I=51.7% O=65.5%**	**I < O**
Percent silt	**Non**	**0.021**	**I=10.9% O=18.0%**	**I < O**
Percent clay	Arcsine	0.869	I=13.0% O=12.2%	NS
Percent silt-clay	Arcsine	0.293	I=23.9% O=30.1%	NS
Median grain size mm	**Log 10**	**0.016**	**I=0.246 mm O=0.136 mm**	**I > O**
Percent TOC	Ranks	0.078	I=3.50% O=1.64%	NS
Percent $CaCO_3$	Non transformed	0.387	I=43.8% O=49.8%	NS

* $p \leq 0.05$
NS=nonsignificant

Table 3-49. Concentration of total organic carbon (%) and volatile solids (%) by two different analytical methods.

Sample Identification Number	Station Number	Original TOC Analysis 200-mg Shimadzu	Repeat TOC Analysis 200-mg Shimadzu	Volatile Solids	TOC Ratio (Orig/Repeat)
4SX002.23	4-3	1.27	1.96	6.3	0.65
4SX003.45	4-2	1.91	1.79	5.9	1.07
4SX005.24	4-2	2.03	2.29	6.9	0.89
4SX005.28	4-1	2.77	2.79	7.8	0.99
4SX005.28	4-1	2.98	NA	NA	1.07
4SX005.31	4-1	2.80	2.68	8.3	1.04
4SX006.03 #1	4-8	1.57	1.52	6.2	1.03
4SX006.03 #2	4-8	NA	1.53	6.0	1.03
5SX001.44	1-5/6/7	4.04	5.33	9.9	0.76
5SX001.48	1-5/6/7	6.07	5.94	11.1	1.02
5SX001.51	1-5/6/7	5.60	6.88	12.0	0.81
		Original TOC Analysis 10-mg Carlo-Erba	**Repeat TOC Analysis 200-mg Shimadzu**		
2SX0014 #1	1-3	12.1	8.90	17.6	1.36
2SX0014 #2	1-3	9.1	9.08	16.9	1.00
2SX0026 #1	1-6	7.24	4.93	9.9	1.47
2SX0026 #2	1-6	NA	4.57	10.2	1.58
2SX0043	1-8	3.91	4.43	8.3	0.88
2SX0049 #1	1-7	3.58	2.41	6.9	1.49
2SX0049 #2	1-7	NA	NA	7.1	NA
2SX0055	1-2	1.68	1.84	5.5	0.91
2SX0061	1-1	5.63	5.06	8.5	1.11
2SX0006 #1	1-4	8.12	2.75	7.5	2.95

Table 3-50. Initial and final ammonia interstitial porewater concentrations for solid-phase bioassay using *Rhepoxynius abronius*.

Station	Initial Ammonia (mg/L)	Final Ammonia (mg/L)
Survey - September 1998		
1-1	0.380	0.286
1-2	0.367	0.332
1-3	1.04	0.922
1-4	0.224	0.261
1-5	0.364	0.317
1-6	0.251	0.160
1-7	0.567	0.415
1-8	0.541	0.545
4-6	1.14	1.00
Survey - September 1999		
4-1	1.03	0.94
4-2	0.836	0.79
4-3.1	0.901	0.852
4-5	1.29	1.20
Survey - November 1999		
1-5/6/7	0.019	0.133

3.3.2. Estimation of Effects from Exposure Data

In addition to the direct measure of effects represented by the sediment toxicity line of evidence, it is possible to estimate effects based from two lines of evidence from Characterization of Exposure: Sediment Chemistry and Sediment Bioaccumulation. Both lines were evaluated in detail previously, in Sections 3.2.2 and 3.2.3 respectively, and are summarized here relative to their estimated effects.

3.3.2.1. Sediment Chemistry

In review, three successful sampling efforts took place between September 1998 and September/November 1999. Eighteen sediment stations were sampled (Figure 2-26, Table 2-8) on two rings located 1-3 meters from the ex-AGERHOLM and approximately 1,000 meters (reference stations) distant and examined for a wide variety of analytes. The project design included precision sediment sampling (±2 m), ultra-low chemical analyses of sediment and tissue samples, radiological dating of sediments, sedimentation rate sampling, toxicity and bioaccumulation testing utilizing two marine species, bottom current measurements and bottom water chemistry. Hypothesis testing was performed, taking advantage of the distribution of rings and stations. Stations are considered to be field replicates within a ring.

The null hypothesis is stated as follows: There is no significant difference in sediment chemistry samples, or in the toxicity or bioaccumulation of potentially toxic materials between the ex-AGERHOLM ship site (Inner Ring, Ring #1) and the reference site (Outer Ring, #4).

Table 3-51 summarizes ranges of concentrations of primary and secondary chemicals of concern, with significant differences ($p \leq 0.05$) highlighted. Total PCBs (Green Book) did not show significant differences between Inner and Outer Rings for sediments. Three of the nine secondary

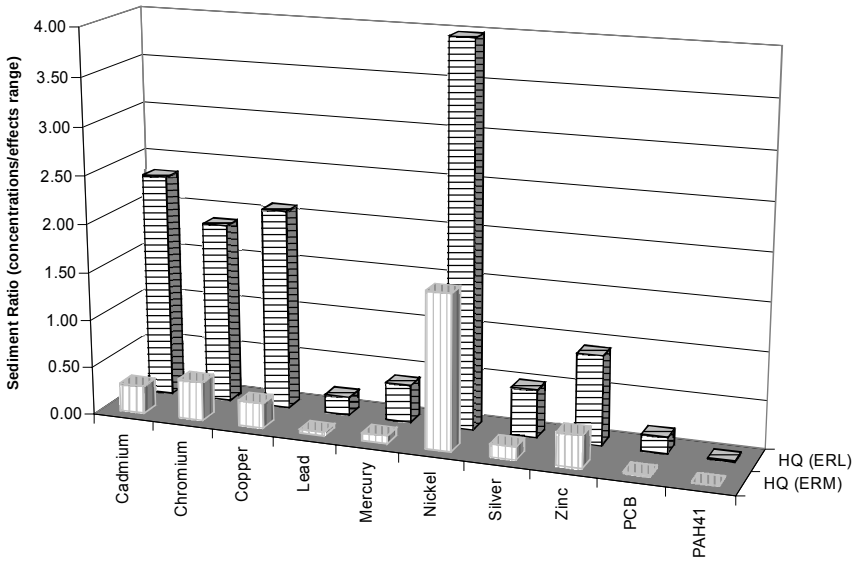
chemicals were significantly greater at the Inner Ring then the Outer Ring; cadmium, copper and silver. In general, secondary chemicals of concern were highly variable and numerically higher, though not often significantly different, in the Inner Ring indicating, as expected, a vessel signal.

Table 3-51. Mean, minimum and maximum concentrations of primary and secondary chemicals of concern examined in sediment and bioaccumulation samples associated with the ex-AGERHOLM site.

Primary Chemical of Concern	Sediment		*Macoma* Bioaccumulation		*Nephtys* Bioaccumulation	
	Inner Ring	Outer Ring	Inner Ring	Outer Ring	Inner Ring	Outer Ring
Total Green Book	3.9	1.5	16.81	28.15	29.43	18.1
PCBs	0.49-9.8	0.41-4.28	10.25-39.38	17.09-37.31	9.51-136.68	14.42-24.92
Secondary Chemicals of Concern	**Sediment**		***Macoma* Bioaccumulation**		***Nephtys* Bioaccumulation**	
	Inner Ring	**Outer Ring**	**Inner Ring**	**Outer Ring**	**Inner Ring**	**Outer Ring**
Total PAH_{41}	48.6	29.2	241	26	442	73
	8.8-303.9	14.1-41.2	4.8-659	11-64	80-1145	44-154
Cadmium	**2.8**	0.3	**0.76**	0.51	**1.46**	1.13
	0.54-7.00	0.30-0.50	**0.34-1.76**	0.46-0.65	**0.68-3.18**	0.79-1.30
Chromium	151.9	147.8	8.35	3.21	4.28	1.64
	87.9-227.0	106-231	3.6-22.1	2.6-4.54	0.50-15.90	0.30-6.92
Copper	**71.0**	28.7	**98.2**	60.4	16.42	8.59
	32.4-137.0	23.9-33.9	**54.0-172.0**	44.4-95.9	5.31-78.5	8.05-9.3
Lead	8.8	4.7	4.36	3.08	0.95	0.31
	1.8-24.3	4.1-5.3	3.27-5.13	2.12-5.29	0.27-2.27	0.21-0.38
Mercury	0.059	0.042	0.79	0.36	0.12	0.07
	0.043-0.107	0.028-0.054	0.28-1.10	0.21-0.84	0.04-0.18	0.06-0.12
Nickel	**83.2**	28.3	13.99	9.2	4.96	2.84
	25.0-215.0	23.9-35.4	9.97-23.7	7.9-11.6	2.00-9.98	1.8-6.12
Silver	**0.50**	0.20	0.43	0.33	0.09	0.08
	0.20-1.60	0.10-0.20	0.24-0.54	0.27-0.50	0.03-0.29	0.04-0.09
Zinc	139.4	73.7	270	256	237	213
	0.72-256.0	61.4-85.2	231-322	243-288	193-360	189-227

As discussed previously, few contamination "benchmarks" are recognized for marine sediments. Effects Range-Low (ER-L) and Effects Range-Median (ER-M) are commonly used for comparative purposes and often incorporated into ratios, where observed sediment concentrations are divided by the ER-L or ER-M. This ratio, sometimes referred to as a Hazard Quotient (HQ), is considered of interest when values greatly exceeding unity (one).

Figure 3-86 summarizes mean Inner Ring HQs for total PCB, total PAHs and eight metals. The HQ for the primary chemical of concern (total PCB) was well below unity for both ER-L and ER-M. The ER-M HQ exceeded unity for only nickel (HQ=1.61); however, it should be noted that Inner and Outer Ring concentrations did not differ significantly (Table 3-51), and along with mercury, the nickel ER-M is of questionable reliability in predicting adverse biological effects (Long et. al. 1995). Of the secondary chemicals of concern, both cadmium and copper have HQs slightly greater than one (2.33 and 2.09 respectively), and have concentrations significantly greater than the Outer Ring. Silver, which also exhibits an Inner Ring concentration greater than the Outer Ring, has an ER-L HQ of only 0.50, indicating that the greater concentrations near the ship pose no risk of effects.



	Cadmium*	Chromium	Copper*	Lead	Mercury	Nickel	Silver*	Zinc	PCB	PAH41
HQ (ERM)	0.29	0.41	0.26	0.04	0.08	1.61	0.14	0.34	0.02	0.00
HQ (ERL)	2.33	1.88	2.09	0.19	0.39	3.98	0.50	0.93	0.17	0.01

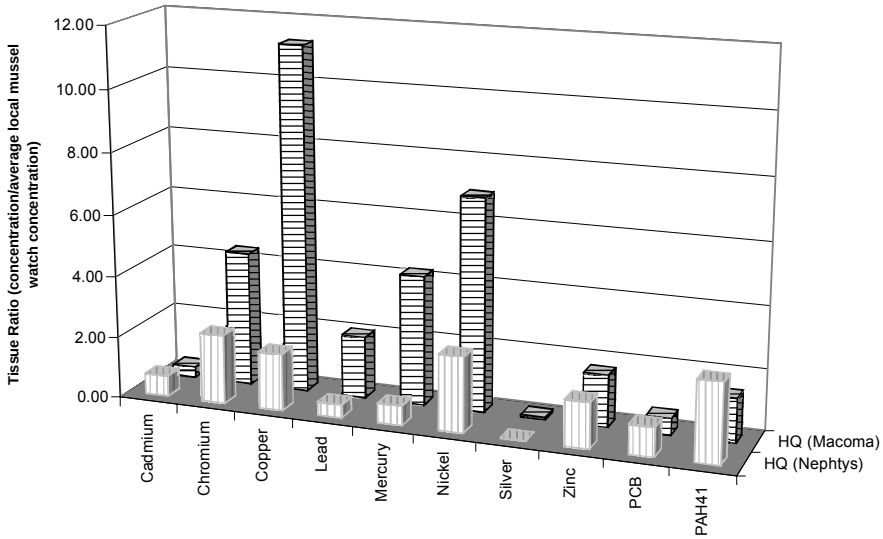
Figure 3-86. Inner Ring sediment Hazard Quotients (HQ) for primary and secondary chemicals of concern (* indicates Inner Ring significantly greater than Outer Ring at p≤0.05).

3.3.2.2. Tissue Bioaccumulation

Table 3-51 also summarizes tissue bioaccumulation ranges of concentrations for primary and secondary chemicals of concern (PCOCs and SCOCs, respectively) with significant differences ($p \leq 0.05$) highlighted. Green Book PCBs did not show significant differences between Inner and Outer Ring tissues in either Macoma or Nephtys tissues. Two of the nine SCOCs were significantly greater at the Inner Ring then the Outer Ring; cadmium (*Macoma* and *Nephtys*) and copper (*Macoma*). Like PCOCs, SCOCs were highly variable and numerically higher (though rarely significant) in the Inner Ring indicating, as expected, a vessel signal. Strong, significant correlations between sediment and tissue concentrations were apparent for only cadmium and *Nephtys* (Table 3-51), suggesting the absence of food chain biomagnification.

With the notable exception of FDA action levels for total PCB and mercury, almost no "benchmarks" are recognized for marine tissues. However; as discussed previously, data collected in support of the NOAA Status and Trends Mussel Watch Program provides valuable insight into local, ambient, nearshore marine tissue concentrations. A review of 16 years of southern California Mussel Watch data identified four areas located within 125 miles of the ex-AGERHOLM site. These areas are reasonably isolated from major industrial/municipal input, and for the purpose of this study are considered clean nearshore reference sites. Like the HQs generated for sediments using ER-Ls and ER-Ms, it is possible to generate hazard quotients from Mussel Watch averages (HQ_{MW}) for ex-AGERHOLM bioaccumulation tissues. The HQ_{MW} is created by dividing the observed dry tissue chemical concentration by the average local mussel watch "clean" tissue concentration. Figure 3-87 summarizes mean Inner Ring HQ_{MW} for total

PCB, total PAHs and eight metals, for Macoma and Nephtys. The HQ_{MW} for the primary chemical of concern, total PCB, is below unity for both species. HQ_{MW} for secondary chemicals of concern exceeded unity for chromium, copper, lead, mercury, nickel, zinc and PAHs in Macoma, and chromium, copper, nickel, zinc and PAHs in *Nephtys* (Figure 3-87). Again, it should be emphasized that there were only three significant Inner-Outer Ring differences identified in the bioaccumulation testing; cadmium (*Macoma* and *Nephtys*) and copper (*Macoma*).



	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Silver	Zinc	PCB	PAH41
HQ (Nephtys)	0.67 *	2.24	1.86	0.44	0.64	2.46	0.02	1.49	0.96	2.61
HQ (Macoma)	0.35 *	4.38	11.27 *	2.02	4.22	6.93	0.08	1.69	0.55	1.42

Figure 3-87. Inner Ring tissue Hazard Quotients (HQ) for primary and secondary chemicals of concern, values derived from local Mussel Watch data (* indicates Inner Ring significantly greater than Outer Ring at $p \leq 0.05$).

3.3.3. Supplementary Effects Data from Sablefish Analysis

The full report for the supplementary Sablefish Assessment is provided in its entirety as Supplement I. Excerpts from the text and some tables and figures have been brought forward to this main document[12]. The rationale for adding Sablefish (Figure 3-1) as supplementary effects data to this ecological risk assessment mirrors the rationale for use in the Human Health Risk Assessment, previously explained in 2.4.4.1:

- They are easier to collect than other deep sea fishes;
- Their high oil (lipid) content increases their affinity for PCBs;
- Their longevity – up to 90-100 years in age and relatively territorial feeding behavior means that body burdens can be related to site exposures; and

[12] References to Supplement I tables and figures are for information (these graphics not in main body of this report). Literature references are also contained in Supplement I and not in this report.

- Their high trophic level means that residue concentrations can be used to infer exposure and potential effects to other members of the deep-sea community, which cannot be sampled as easily.

The purpose of this supplementary analysis is to assess the ecological risk of PCB exposure to the deep-sea demersal community using sablefish as a surrogate receptor. This analysis provides a supplemental line of evidence for assessing the ecological risk of PCB exposure from the ex-AGERHOLM.

Tissue residue benchmarks were developed to evaluate potential effects from exposure (summarized in Section 3.2.4) to Total PCBs. The benchmarks for Total PCB were based on the tissue screening value (TSV), bioaccumulation critical value (B_{CV}), and critical body residues (CBR). These benchmarks (Table 3-52) are chemical residue thresholds, at or below which adverse toxicological effects would not be expected. Species sensitivity distributions (SSDs) of effects from critical body residues of PCBs in fish were also used to determine the probability of an effect to those endpoints.

Table 3-52. TSV, B_{CV}, and CBR Benchmark Effects Thresholds for Total PCBs in Sablefish[13].

Threshold type	Benchmark PCB Effects Concentration				Basis for Criterion
	wet weight	*dry weight*	*lipid weight*	*units*	Potential effects from bioaccumulation
TSV	436.8	1371.2	2345.1	ng/g	Tissue Screening Value (URS 1996, Dyer et al., 2000)
Bcv	1836.1	5764.0	10236.0	ng/g	Bioaccumulation Critical Value (see Table 5)
CBR, NOED	1500.0	4708.8	8362.1	ng/g	No Observed Effects Dose, sheepshead minnow (Hansen et al., 1975, Table 6)
CBR, LOED	2200.0	6906.3	12264.5	ng/g	Lowest Observed Effects Dose, pinfish (Hansen et al., 1974, Table 6)

Tissue screening values (TSV) are the most conservative of the three benchmarks used. The TSVs were originally developed for screening-level ecorisk assessments at Navy sites (URS 1996, 2002), and are based on water quality criteria (WQC), which were derived to protect 95% of aquatic organisms 95% of the time (U.S. EPA 1986, URS 1996, Shepard 1998). Because the TSV is equal to the "no adverse effect" tissue concentration, a single TSV applies to both freshwater and marine organisms (URS 1996). Therefore, numerical differences between freshwater and saltwater criteria are due to differential chemical uptake between freshwater and marine organisms (Shepard 1998). PCB residue levels below the TSV are assumed to pose little or no risk to aquatic biota (Shepard 1995, URS 1996, Dyer et al. 2000).

The bioaccumulation critical value (B_{CV}) is similar in concept to the TSV, but more relevant and less generic. The B_{CV} was calculated using the most recent saltwater quality criteria (U.S. EPA 1998b, 2005, Buchman 1999) and bioconcentration factors applicable to marine fish. The B_{CV} was defined as the tissue concentration that would occur if water exposure levels equaled the chronic or lowest available water benchmark effects concentration. The BCF for each homolog was estimated using the regression from Mackay (1982, cited in Petersen and Kristensen 1998). The B_{CV} for Total PCB accumulation in sablefish tissue was calculated using a BCF weighted by the fraction of Total PCB (f_{PCBi}) present in each homolog group measured in the sablefish samples in accordance with US EPA methods (U.S. EPA 1980, URS 1996).

[13] See Supplement I for details on benchmark threshold derivations and calculations.

Critical body residues (CBR) are defined as the threshold concentration of a contaminant in the tissue of an organism above which adverse effects could occur (McCarty et al. 1992, Pabst 1999). Generally, the effect occurs as a result of noncancerous effects and can result in death (mortality), or a reduction in fecundity, reproduction, or growth (chronic effects). Data from the US Army Corps of Engineers Environmental Residue-Effects Database (USACoE, 2002) were used to develop benchmarks for CBRs. The database was searched for effects on reproduction, growth and development, mortality and survival. Results that were based on adult exposure, whole body concentration, and ingestion or absorption were used, if available. Benchmarks were selected for highest no observed effect dose (NOED) and lowest observed effect dose (LOED) – for freshwater or saltwater fish species.[14] If the highest NOED was greater than the lowest LOED, then a NOED was selected that was lower than the lowest LOED (App.Q, Tables 8 and 9). The NOED and LOED benchmarks for fish and invertebrates were derived by multiplying the value obtained from ERED by an uncertainty factor (UF), if applicable. The LOED was based on 5% mortality observed in a 96 hr exposure to pinfish (*Lagodon rhomboides*, Hansen et al. 1974) and the NOED was based on no effect to sheepshead minnow (*Cyprinodon variegatus*) survival after a 28-day exposure (Hansen et al. 1975). Because these benchmarks were assumed to be directly applicable to sablefish, an UF=1 was used in calculating the NOED and LOED benchmarks.

Figure 3-88 compares the ship and reference sablefish residues, represented as probabilistic distribution curves, to the various effects threshold benchmark concentrations. With the exception of TSV, the most conservative benchmark – typically used for initial screening assessment, both distributions fall to the left indicating no adverse effects.

Ecological risk benchmarks are based on a single, species-level effect and do not necessarily reflect the wide range of species diversity and sensitivity to PCBs present in the ecosystem. One-way of addressing the broader implications of potential ecotoxicological risk from PCBs is to compare the PCB exposure distribution to species sensitivity distributions (SSD). Derived from toxicity data, SSDs are cumulative distribution functions, which describe the proportion of a class of organisms (i.e., fish) expected to be affected by a given level of exposure to a contaminant (Posthuma et al 2001, Maltby et al. 2005).

[14] NOED and LOED are used to be consistent with the ERED nomenclature, which defined "dose" as the whole body burden concentration. Values selected from the database were the no observed adverse effects (NOED) and lowest observed adverse effect (LOED), where adverse was defined as a negative impact to growth, development, reproduction, or survival.

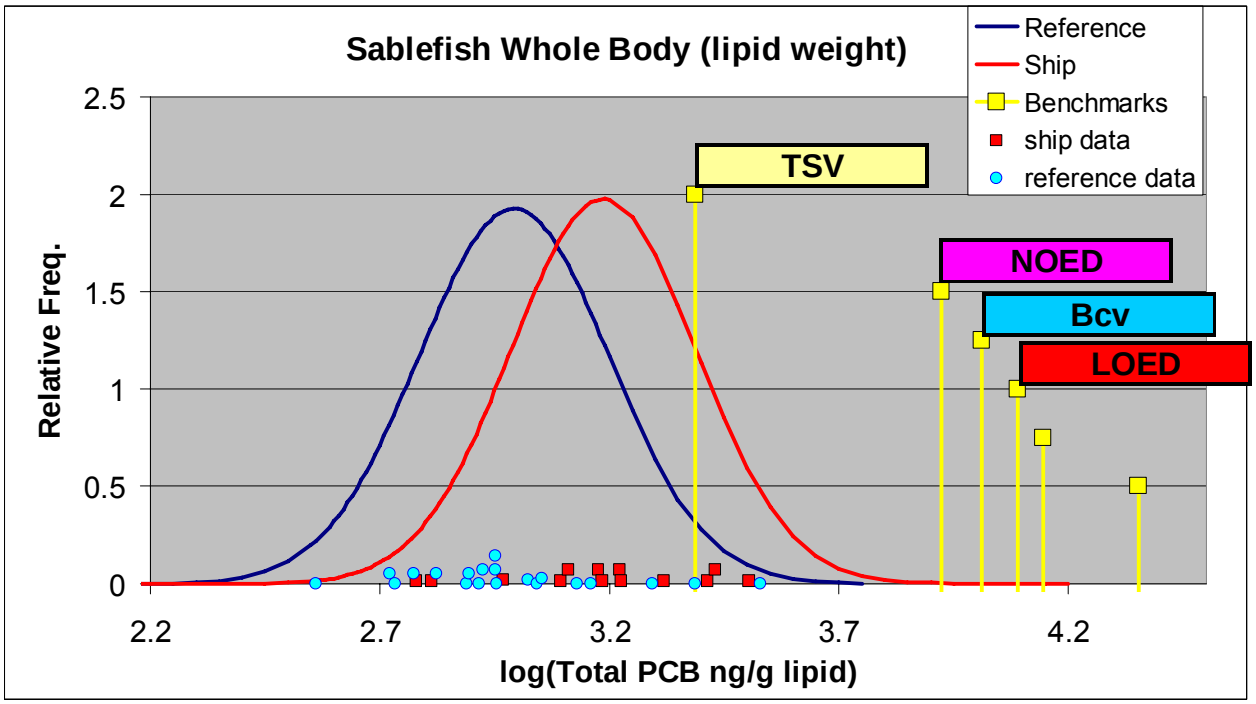


Figure 3-88. Comparison of PCBs in Ship and Reference site Sablefish to available threshold effects benchmark concentrations.

Data from the ERED database on effects of PCBs to fish (both fresh and saltwater species) were used to calculate SSDs for Total PCB residues in fish. The ERED data for effects to growth, mortality, reproduction, or survival from PCB residues in juvenile or adult fish were used to calculate the cumulative probability distributions for the No Observed Effect Dose (NOED) and the Lowest Observed Effect Dose (LOED), assuming that the toxicity data conformed to a lognormal distribution. The SSDs for the more conservative benchmark, the NOED, is shown in Figure 3-89, using Critical Body Residues (CBRs). The available toxicity data were mostly for freshwater species (lake trout, golden ide, catfish, etc), but the database also included saltwater species including sheepshead minnow, pinfish, salmonids, and other species. The SSD calculated from the ERED data are not based on genus-mean concentrations, but rather raw toxicity data. While genus-mean concentrations are more preferable for evaluating potential toxicity effects across a wide range of organisms (Posthuma et al 2001, Maltby et al. 2005), developing genus-mean effects levels were beyond the scope of this report. More research is required to develop genus-mean concentrations of effects from critical body residues in fish.

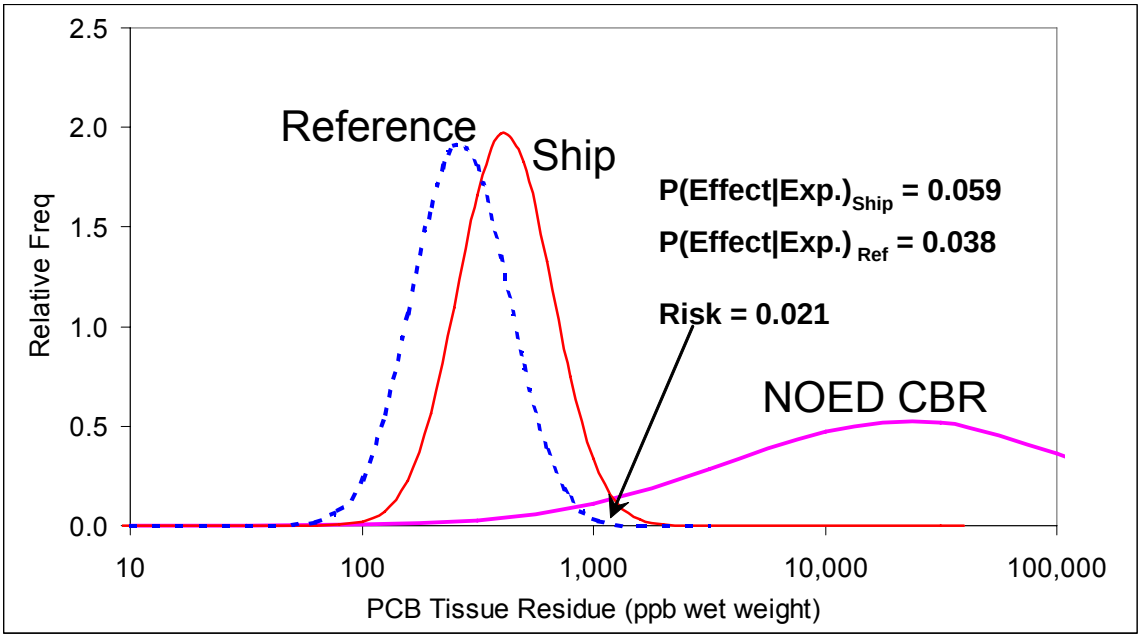


Figure 3-89. Total PCB species sensitive distribution (SSD) of NOEDs for Critical Body Residues(CBR) and exposure distributions for reference and ship shown as a probability density function.

3.4. References

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4.0. Risk Characterization

4.1. Overview of Study

The purpose of this study was to evaluate the marine environment in the vicinity of sunken Navy ships to determine whether potentially toxic materials from these ships were being released, and whether any releases might pose an adverse risk to the environment. Specifically, the objective was to investigate PCBs in solid materials onboard sunken Naval vessels, used for at-sea target training after they have been decommissioned and prepared in accordance with existing NAVSEA guidance for SINKEX vessels.

Polychlorinated biphenyls (PCBs), a group of synthetic halogenated aromatic hydrocarbons, which have been linked to environmental and public health issues, were the primary chemicals of concern (PCOCs) in this investigation. Total PCBs within this investigation were defined in two different ways: (1) the first approach was the sum of the 21 congeners (referred to as the "Green Book 21") used in the Ocean Testing Manual, which is joint guidance from the U.S. Environmental Protection Agency and U.S. Army Corps of Engineers provided in the publication "Evaluation of Dredge Material Proposed for Ocean Disposal, Testing Manual" commonly referred to as the Green Book (USEPA/USACE, 1991). (2) The second approach was based upon these 21 congeners and an additional 5 congeners important in environmental health assessments (referred to as the "SINKEX 26"). A set of secondary contaminants of concern (SCOCs), evaluated to assist in more thoroughly characterizing the site and explaining toxicity, if found, were also examined. These SCOCs included total polynuclear aromatic hydrocarbons (PAHs), cadmium, chromium, copper, lead, mercury, nickel, silver and zinc. Additionally, several other sediment properties were obtained to aid in data interpretation, including acid volatile sulfides/simultaneously extracted metals (AVS/SEM), particle size, total organic carbon (TOC) and calcium carbonate ($CaCO_3$).

Based on EPA guidance provided by the Office of Water and published documentation, as well as recommendations from a scientific peer review panel, the technical approach combined sediment-testing requirements from the Green Book and the US EPA's "Framework for Ecological Risk Assessment" (USEPA, 1992) and "Guidelines for Ecological Risk Assessment" (USEPA, 1998).

A standard two-sample hypothesis test was used, comparing conditions at the sunken vessel site to reference stations. The null hypothesis addressed by this investigation was: There is no significant statistical and adverse difference ($p<0.05$) between ship site and reference site results for any of the biological and chemical measurements performed.

Three successful sampling expeditions were completed between September 1998 and November 1999. A total of 41 stations were sampled for a suite of analytical measurements (see Table 2-8). The three primary lines of determination in this weight-of-evidence risk assessment focused on 18 of those stations located 2-4 meters from the ex-AGERHOLM (Figure 4-1) and reference stations located approximately 1,000 meters away (Figure 4-2). The stations were examined for a wide variety of analytes. The project design included precision sediment sampling, ultra-low chemical analyses of sediment and tissue samples, toxicity and bioaccumulation testing utilizing two marine species as surrogates for deep-water infauna, radiological dating of sediments, sedimentation rate sampling, and bottom current measurements.

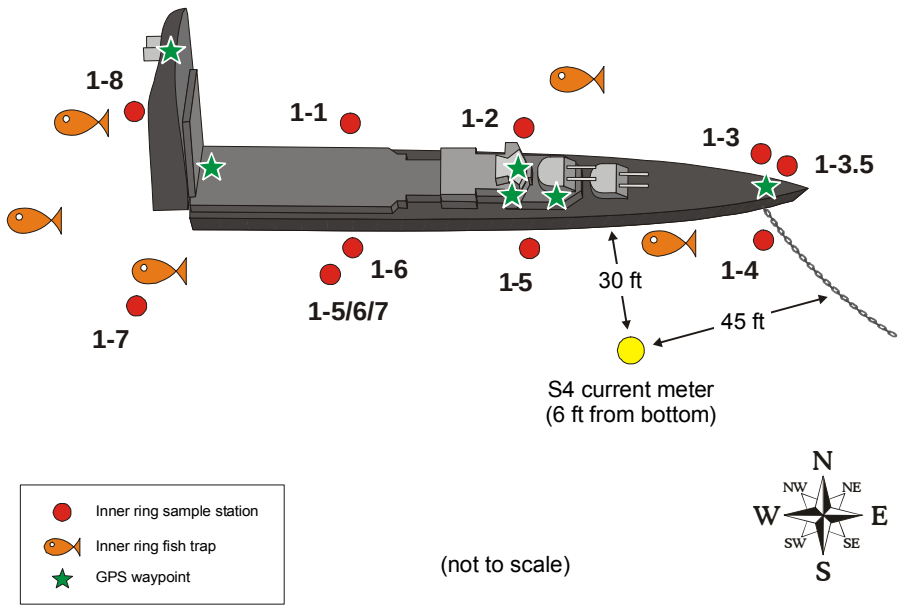
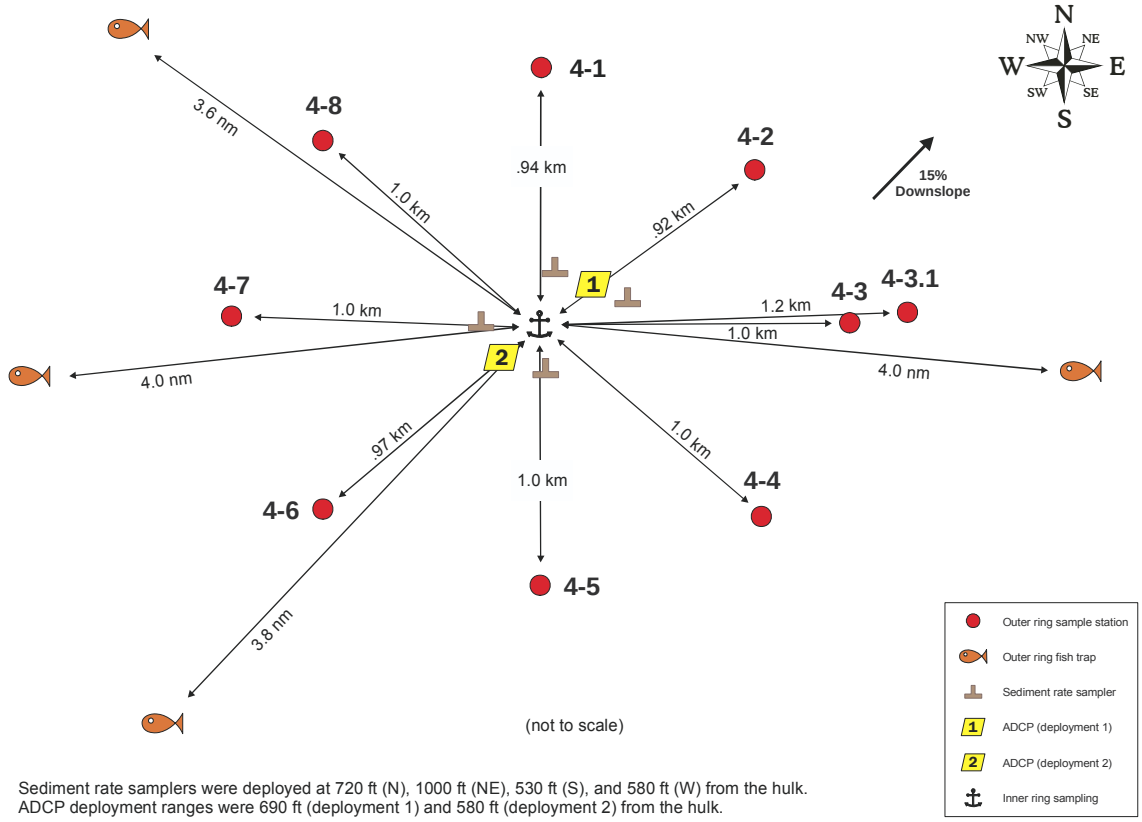


Figure 4-1. Sampling design for Inner Ring (SINKEX ship site).



Sediment rate samplers were deployed at 720 ft (N), 1000 ft (NE), 530 ft (S), and 580 ft (W) from the hulk.
ADCP deployment ranges were 690 ft (deployment 1) and 580 ft (deployment 2) from the hulk.

Figure 4-2. Sampling design for Outer Ring of reference stations.

4.2. Risk Estimation According to the Decision Matrix

The Decision Matrix developed in Section 2.4.4.4 is shown again in Table 4-1 to review the decision rules established prior to commencement of the field studies for the final evaluation of ecological risk due to PCBs from the sunken ship ex-AGERHOLM. This Decision Matrix was the result of study planning efforts conducted jointly with the EPA to ensure that the technical approach and final evaluation met the Green Book requirements for the testing of contaminated sediments prior to ocean disposal.

Table 4-1. Decision Matrix for sediment data.

Outcome Number	Sediment Chemistry	Acute/Chronic Toxicity	Laboratory Bioaccumulation	Benthic Community	Risk
1	-	-	-	±	None
2	+	-	-	±	None
3	-	+	-	±	Potential Local
4	+	+	-	±	Probable Local
5	+	-	+	±	Potential Food Chain
6	+	+	+	±	Probable Local and Potential Food Chain

+ Significant unfavorable difference for Inner Ring for any endpoints measured.
- No significant unfavorable difference for Inner Ring for endpoints measured.
± No effect on decision.

4.3. Assessment and Measurement Endpoints under the EPA's Framework for Ecological Risk Assessment

To evaluate the level of protection of the appropriate ecological communities at risk of contamination from PCBs onboard the sunken vessel, the risk assessment team followed the Framework for Ecological Assessment (USEPA, 1992). This framework requires a logical progression of study planning commencing with Problem Formulation, proceeding through an Analysis Phase, and ending with this Risk Characterization Phase. The Problem Formulation phase culminated with the development of a Site Conceptual Model (SCM) for expected environmental pathways of exposure and effects (Figure 4-3). An Analysis Plan was then developed based on the Assessment Endpoints and Measures of Exposure and Effects from the SCM. Table 4-2, first shown in Section 2.4.4.1, is provided below for the convenience of the reader.

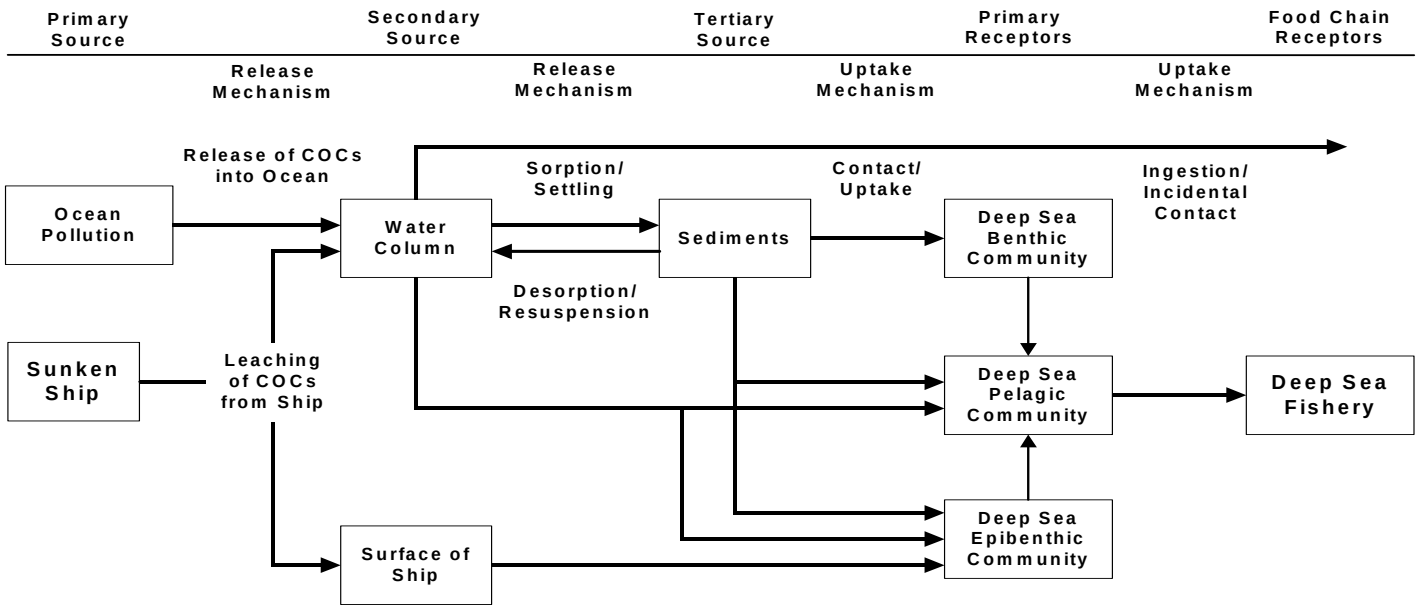


Figure 4-3. Site Conceptual Model (SCM) for SINKEX ecological risk assessment.

Table 4-2. Development of measures of exposure and effects from assessment endpoints and receptors of concern.

Assessment Endpoint	Test Species (surrogates for Receptors of Concern)	Measures of Exposure	Measures of Effects
Invertebrates of the benthic/epibenthic community and entry point to larger pelagic food chain	(1) Amphipod (*Rhepoxynius abronius*) (2) Polychaetes (*Nephtys caecoides* for exposure; *Neanthes arenaceodentata* for Effects) (3) Clam (*Macoma nasuta)*	(1) Tissue concentrations of PCBs, metals, and PAHs in surrogate clams (*Macoma nasuta)* and polychaetes (*Nephtys caecoides)*, after 28-day laboratory exposures to SINKEX sediments. (2) Sediment concentrations of PCBs, metals, and PAHs.	(1) 10-day acute toxicity test (survival & reburial) on surrogate amphipods (*Rhepoxynius abronius)*, exposed to SINKEX site and reference site sediments. (2) 28-day chronic toxicity test (growth and survival) on surrogate polychaetes (*Neanthes arenaceodentata*), exposed to SINKEX site and reference sediments.
Scavenging, demersal, and predatory fish in pelagic community	Sablefish/Black Cod (as primary consumer of contaminated sediment/water).	(1) Tissue concentrations of PCBs (2) Water column concentrations	Estimation from exposure
Humans	Sablefish/Black Cod (representing step in food chain)	Edible tissue (i.e. muscle) concentrations of PCBs	Human Health Risk Assessment (separate report)

Notes:
1. Efforts were made during several cruises to capture epibenthic infauna, but too few specimens were collected to address quantitatively.
2. Because PCB exposure to ecological receptors of concern through water absorption/consumption was considered to be a far less important pathway than sediment ingestion, and because sampling the water column in a robust manner to be statistically valid was problematic, water sampling was a very small component of the Analysis Plan (a few samples were taken to see if PCBs could even be detected) and was not part of the decision matrix. However, a short discussion on water sampling is found in the first part of Section 3.2 (Characterization of Exposure to COCs).

4.4. Summary of Exposure and Effects Results from the Decision Matrix

The preponderance of data indicate no significant elevation of PCBs in the sediment/or in tissue of benthic organisms since the sinking of the ex-AGERHOLM in 1982. Table 4-3 summarizes the overall results for the three "determining" lines of evidence, with a "NS" indicating that there was no statistical difference ($p<0.05$) found between the Inner Ring (ex-AGERHOLM site) and the Outer Ring (Reference site).

Table 4-3. Overall results from the three determining lines of evidence.

Line of Evidence	Overall Result	Results of Individual Tests			
PCB Chemistry	NS	21GB Congeners		26 SINKEX Congeners	
		NS		NS	
Toxicity	NS	10-day Rhepoxynius		28-day Neanthes	
		Survival*	Reburial	Survival	Growth+
		NS	NS	NS	NS
PCB Bioaccumulation	NS	28-day Nephtys		28-day Macoma	
		Tissue Concentration		Tissue Concentration	
		NS		NS	

Notes:

* 10% survival difference was statistically different, but did not meet Green Book species-specific requirements to show a 20% difference, based on inherent variability in response of this particular species. See Section 3.3.1.2.

+ Large difference in Zero Time size of test organisms caused uncertainty in results, but Inner Ring test worms grew faster, relative to control worms, than those of the Outer Ring. See Section 3.3.1.

To summarize, none of the three lines of evidence, using Green Book guidelines, showed any significant statistical differences between ship and reference site, indicating no adverse incremental risks posed to the marine environment from the presence of PCBs associated with the ex-AGERHOLM. The following sub-sections provide summaries of each Primary Line of Evidence used in the Decision Matrix

4.4.1. Primary Line of Evidence: PCB Chemistry in Sediments

Using both approaches for assessing PCBs in sediments, Figure 4-4 and Figure 4-5 show the mean PCB concentrations measured in the ocean sediments collected from each of the 10 Inner Ring and 8 Outer Ring stations. It appears that the values for the Inner Ring in both analyses are higher than those of the Outer Ring. Statistically, the p-values for the 21 Green Book and 26 SINKEX congeners are 0.13 and 0.07, respectively. The additional analytes and incorporation of one-half detection limit (rather than 0, for non-detects) caused the null hypothesis in the 26-congener analysis to be barely accepted ($p=0.07$, compared to the 0.05 criterion). All sediment PCB concentrations were below Effects Range-Low (ER-L), a threshold effects benchmark used in sediment assessment.

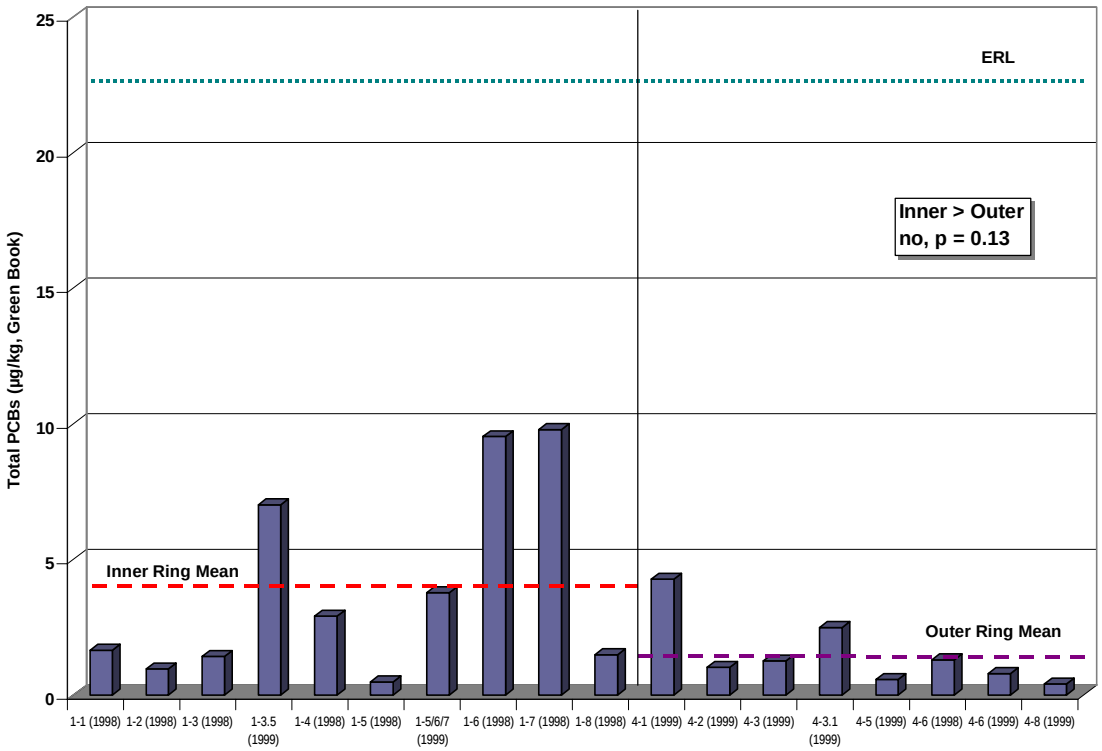


Figure 4-4. Total of 21 Green Book PCB Congeners measured at all SINKEX stations.

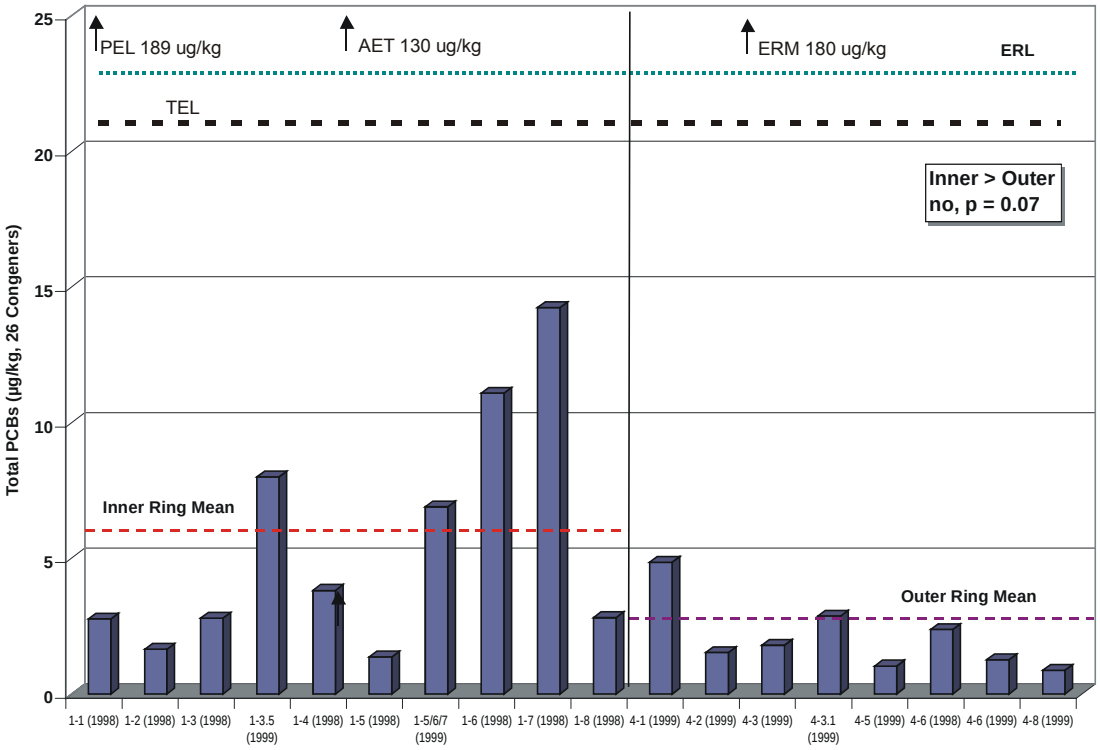


Figure 4-5. Total of 26 PCB Congeners measured at all SINKEX stations. One-half detection limit used when congener was not detected.

4.4.2. Primary Line of Evidence: Sediment Toxicity

Results for the three sediment toxicity tests selected for use in the Decision Matrix are shown in Figure 4-6, Figure 4-7, and Figure 4-8. Compared as Inner Ring (ship site) against the Outer Ring (reference site), the acute 10-day sediment toxicity test using *Rhepoxynius* (amphipod) resulted in mean survival rates of 83% and 93%, respectively. This 10% decrease at the Inner Ring was less than the Green Book criteria of 20%, so the overall result was "no significant difference." The mean Inner Ring survival of the amphipod was most influenced by Station 1-6 (65% survival), but this single case was best explained by other sediment differences (e.g., grain size, total organic carbon) between the two sites. In the chronic 28-day test using *Neanthes* (worm), mean survival for the Inner Ring (96%) was very close to that of the Outer Ring (98%) and statistically not different (p=0.4). Finally, in the 28-day *Neanthes* growth test, mean absolute growth rates were greater for the Outer Ring (0.13 mg/day) than for the Inner Ring (0.1 mg/day), but an opposite effect was observed when growth rates were compared to each control group. Outer Ring tests began with a batch containing larger worms, and the growth of these larger worms was less (17% less growth) than that of the Inner Ring worms (30 % growth suppression), relative to their respective controls (see Relative Percent Difference, Figure 4-8).

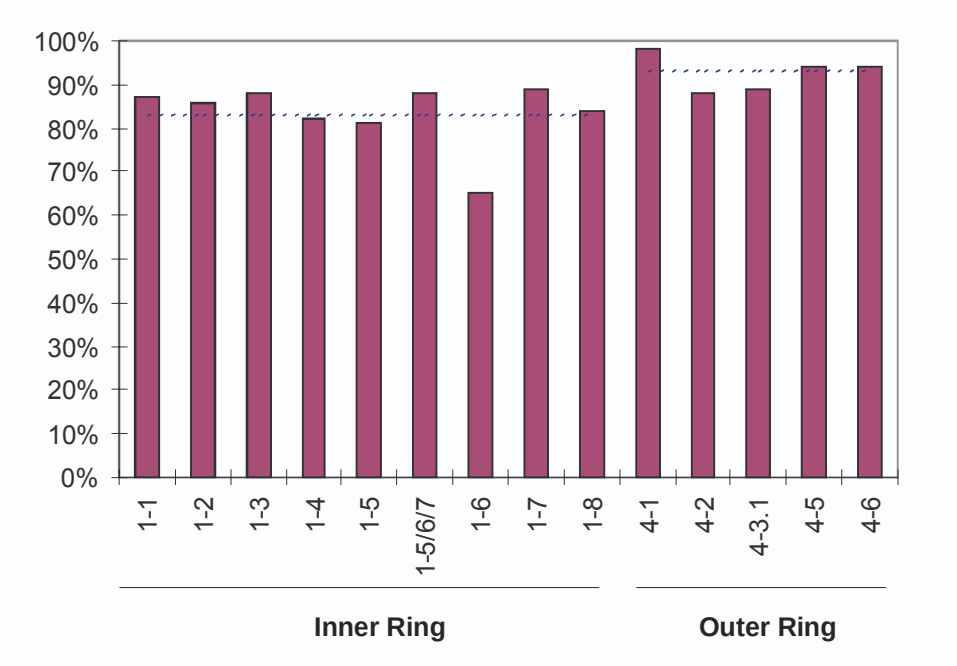


Figure 4-6. *Rhepoxynius abronius* survival.

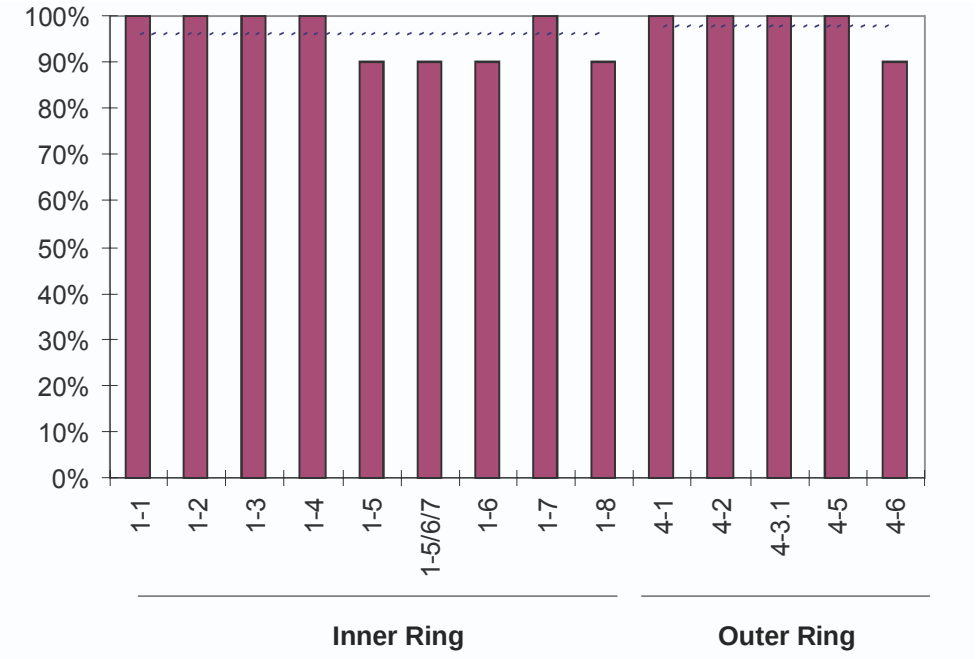


Figure 4-7. ***Neanthes arenaceodentata*** **survival.**

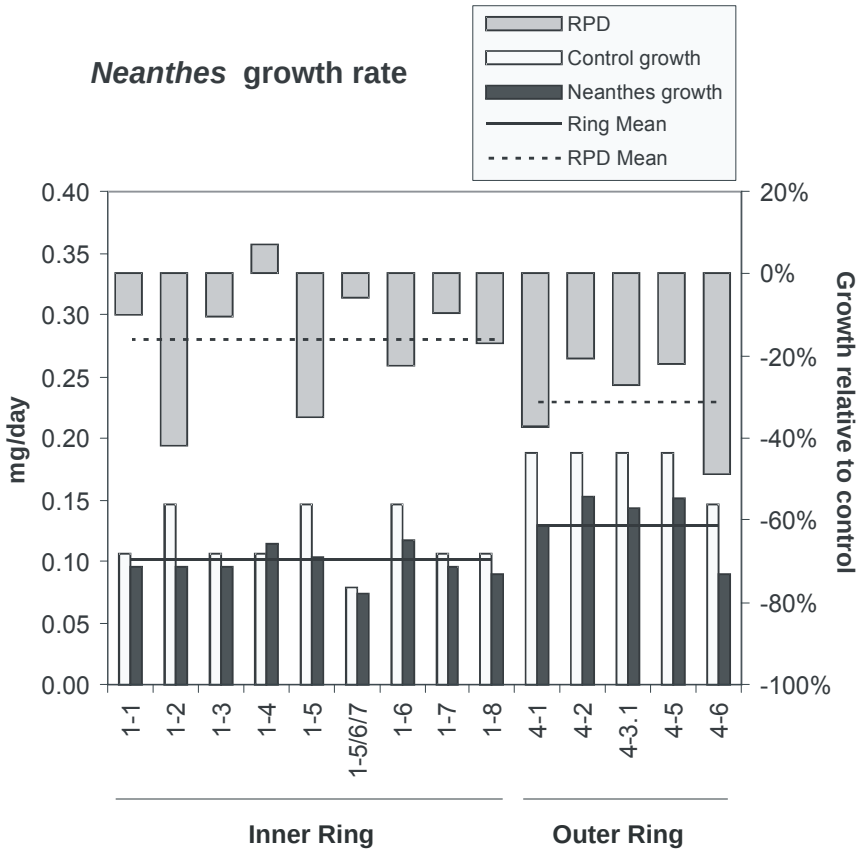


Figure 4-8. ***Neanthes arenaceodentata*** **growth rates (RPD=Relative Percent Difference).**

4.4.3. Primary Line of Evidence: PCB Bioaccumulation

Figure 4-9 and Figure 4-10 present the PCB bioaccumulation test results for *Macoma* and *Nephtys*, respectively, analyzing by the 21 Green Book congener convention (PCB21) and the more conservative (i.e., more analytes, non-detects assumed to be ½ of Detection Limit) 26 SINKEX congener convention (PCB26). Comparisons between Inner and Outer Ring means showed no statistical differences at the $p<0.05$ level for *Macoma* PCB21 ($p=0.3164$) and PCB26 ($p=0.1784$), or for *Nephtys* PCB21 (0.4646) and PCB26 ($p=0.6536$). However, Station 1-8 was notably elevated, with the highest PCB concentrations for both *Macoma* and *Nephtys*, at nearly four times the next highest station in *Macoma.* This and other apparent elevated results at the ship's stern are discussed in the following section.

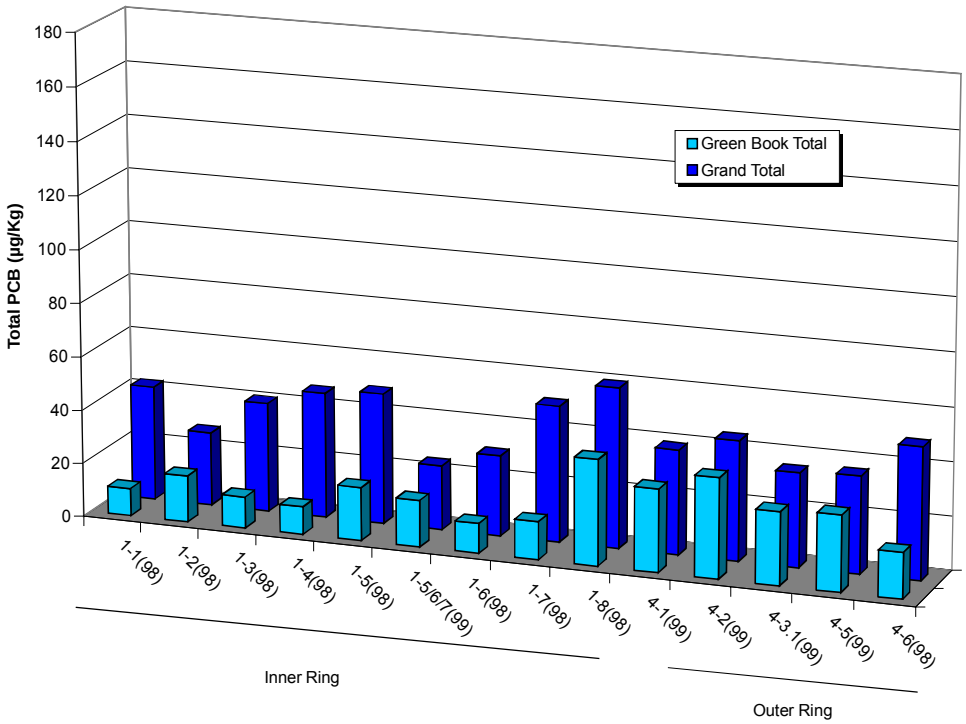


Figure 4-9. PCB_{21} (Green Book) and PCB_{26MDL} (Grand Total) congeners measured in *Macoma nasuta* tissues (µg/kg dry weight)

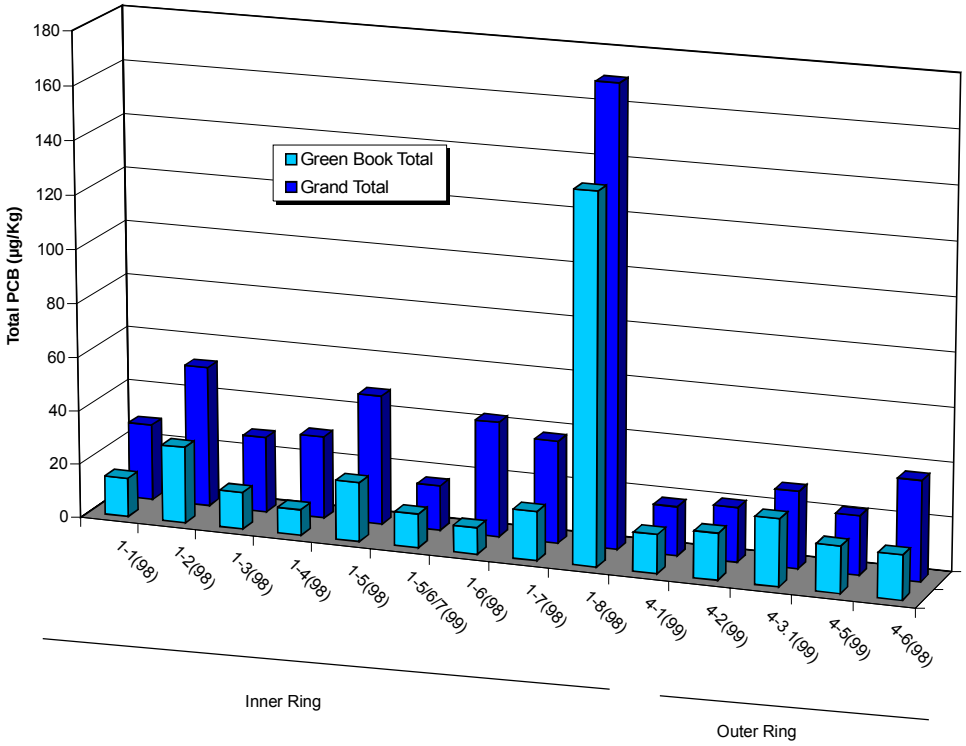


Figure 4-10. PCB_{21} (Green Book) and PCB_{26MDL} (Grand Total) congeners measured in *Nephtys caecoides* tissues (µg/kg dry weight).

4.5. Summary of Results from the Supplemental Lines of Evidence, Secondary COCs, and Individual Stations

4.5.1. Supplemental Lines of Evidence

Table 4-4 summarizes supplemental data, which were not included in the Decision Matrix. These data were important to collect and analyze for the following purposes:

- Characterize the relatively unknown environment at the deep ocean benthic site, using standard assessment techniques for benthic infaunal community structure.
- Provide additional information on potential sources of toxicity, other than PCBs (i.e., PAHs and metals).
- Use additional assessment techniques to analyze data, including the addition of Aroclors and congeners to total PCBs, core depth profiles of PCBs, and individual station comparisons.
- Evaluate sablefish data relative to ecological effects; sablefish was selected initially for the Human Health Risk Assessment (see Supplement II), but it was later decided that the data could also be used for this ERA.

Benthic Community: As evident in the decision matrix, results from the benthic community studies were not a factor considered in the overall risk, but were used to characterize the deep ocean environment at the site. Enumeration of 1,508 benthic specimens resulted in 240 uniquely identified taxa (exclusive of nematodes and calanoid copepods) collected in 23 boxcore samples, representing a fairly diverse deep ocean community, comparable to lower slope communities

measured elsewhere (Appendix A). There were no statistically significant differences in the whole community measures of diversity, richness, and abundance between the Inner (ex-AGERHOLM) and Outer (Reference site) Rings. There were differences in major taxonomic groups between the two sites, but these differences statistically (though weakly) correlated with differences in sediment (grain size and TOC, see 3.1.3).

Secondary COCs (SCOCs): As shown in Table 4-4, concentrations of cadmium, copper, nickel and silver were elevated at the hulk in one or more of three supplemental lines of evidence: sediment chemistry, *Macoma* bioaccumulation, or *Nephtys* bioaccumulation. They were considered in the analysis to be potential sources of toxicity, because the sunken ship is composed primarily of metals (e.g., previous work estimated over 200 tons of copper). Nevertheless, while concentrations for these four metals appear to be elevated in Inner Ring sediment and tissue (extrapolated from bioaccumulation tests on surrogate organisms), toxicity testing discussed previously as one of the primary lines of evidence, showed no significant differences between the ship and reference sites.

Table 4-4. Matrix indicating significant Inner and Outer Ring differences for secondary (PAHs and metals) chemicals of concern ("+" indicates difference significant at $p \leq 0.05$, "-" indicates rings not significantly different.).

Secondary Chemicals of Concern	Sediment Chemistry	Macoma Bioaccumulation	Nephtys Bioaccumulation
Total PAH_{41}	-	-	-
Cadmium	+	+	+
Chromium	-	-	-
Copper	+	+	-
Lead	-	-	-
Mercury	-	-	-
Nickel	+	-	-
Silver	+	-	-
Zinc	-	-	-

Sablefish Assessment: The exposure assessment found that sablefish sampled from the ex-AGERHOLM (ship) site had statistically higher concentrations (by a factor of 1.4 – 1.5) of PCBs than sablefish sampled from reference locations about 4 nm away from the ship. Tissue residue benchmarks were developed to evaluate potential effects from exposure to Total PCBs. The benchmarks for Total PCB were based on the tissue screening value (TSV), bioaccumulation critical value (B_{CV}), and critical body residues (CBRs). These benchmarks are chemical residue thresholds at or below which adverse toxicological effects would not be expected. The initial comparison of sablefish data to tissue residue benchmarks showed that there was low risk of exposure to primary and secondary consumers in the deep-sea pelagic community because Total PCBs in sablefish from the ship site were significantly higher than reference and three samples from the ship sites exceeded the conservative TSV benchmark used in screening analyses. However, as shown in Figure 4-11, no sample exceeded any of the more relevant benchmarks, suggesting it was unlikely that exposure would be harmful.

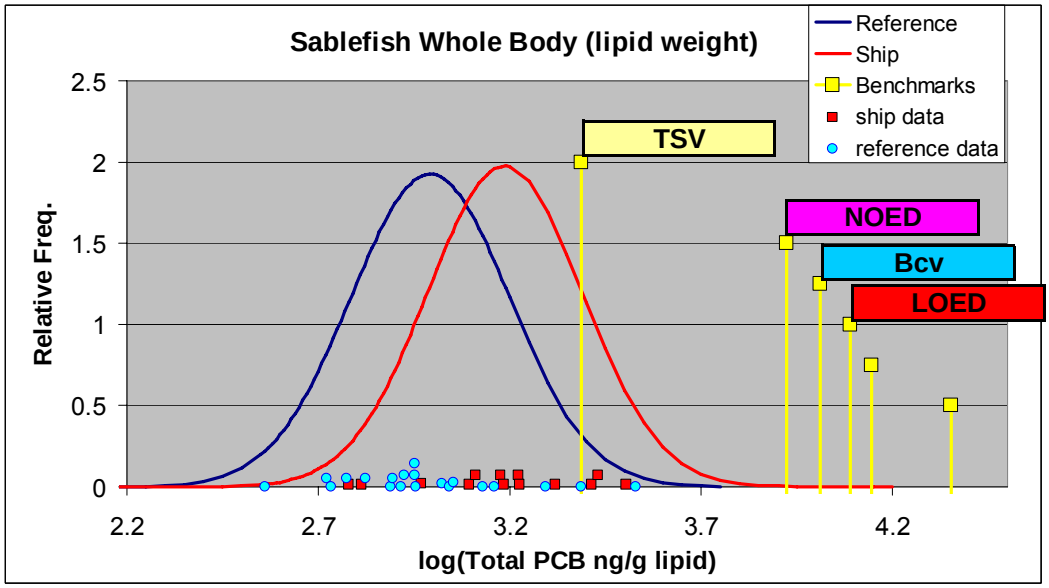


Figure 4-11. Comparison of PCBs in Ship and Reference site Sablefish to available threshold effects benchmark concentrations.

4.5.2. Examination of Secondary COCs

This section discusses COCs elevated at the ship site and also COCs evaluated to be elevated in the surrounding region. Hazard Quotients, which are simply ratios of the study data to standard criteria or benchmarks (the latter in this case), are presented for sediment chemistry and bioaccumulation to indicate the degree of ecological risk and tissue contamination associated with each metal, based on ERL/M and Mussel Watch (MW) data, respectively (Long et. al., 1995; NOAA, 1991). HQs provide a relative quantitative comparison by which an assessor can conclude that a given study COC is elevated or poses risk. As explained in previous sections, the ERL/M values are based on a compilation of sediment “effects” studies, whereas the Mussel Watch values are related to a set of “clean” coastal mussel sites selected for use by the SINKEX study from the entire Mussel Watch library of sites. The term “clean” refers to the relatively low tissue burdens measured in the selected data set, in comparison to all sites. While the MW HQs are not effects-based, they indicate a degree of contamination in tissue, as measured in the laboratory bioaccumulation tests, for the metals that were elevated at the ship site. It is probably more appropriate to refer to these tissue ratios as Bioaccumulation or Tissue Burden Quotients rather than Hazard Quotients, since the term “hazard” implies an adverse biological effect. In this SINKEX study, Long's "minimal-effects range" for concentrations below ER-L (Effects Range-Low) is considered to have "no risk" while the "possible-effects range" for concentrations between ER-L and ER-M (Effects Range-Median) is considered to have "low, but possible risk." Finally, if concentrations are above ER-M, in Long's "probable-effects range," this SINKEX study concludes that these COC levels pose a "probable risk."

4.5.2.1. COCs Elevated at Ship Site

Table 4-5 summarizes more detailed information to aid in defining the implications of the four metals showing elevation at the ship site, compared to the reference site.

Table 4-5. Hazard/Tissue Quotients for metals significantly elevated in ship site sediments, with quotients calculated relative to Long et al. (1995) ERLs/ERMs (sediment) and relatively clean Mussel Watch tissues (bioaccumulation).

Metal	Sediment Chemistry	*Macoma* Bioaccumulation	*Nephtys* Bioaccumulation
Cadmium	2.3 ERL	0.3 MW	0.7 MW
Copper	2.1 ERL	11 MW	–
Nickel	1.6 ERM	–	–
Silver	0.5 ERL	–	–

Note: (1) The symbol "-" denotes there was No Difference in this test between Ship and Reference sites (HQ-MWs were not listed for these).

(2) HQs are rounded to the nearest 0.1

Table 4-5 presents unclear outcomes across the three supplementary lines of evidence.

Nickel: Based on sediment chemistry, nickel appears to present the highest risk because its chemistry HQ is higher than ERM (as opposed to ERL), thus indicating a probable localized rather than possible risk. However, neither species showed elevated bioaccumulation at the ship site and it was pointed out in Section 3.3.2.1, the nickel ER-M is of questionable reliability in predicting adverse biological effects (Long et. al. 1995).

Copper: Based on sediment bioaccumulation, copper represents the highest tissue burden based on the *Macoma* MW HQ, a factor of 11 above the "clean" mussel average. However, its sediment value is only about two times above ERL, indicating low but possible risk.

Cadmium: Like copper, cadmium has a sediment HQ value about two times ERL, indicating low but possible risk, but the *Nephtys* and *Macoma* MW HQs are both less than unity, indicating no significant accumulation relative to clean MW sites.

Silver: Finally, silver poses the least risk of the elevated metals, with a sediment HQ of one-half, and a negligible HQ for *Nepthys*.

The bioaccumulation tests results for metals indicating no difference between ship and reference stations (i.e., annotated with "-" in Table 4-5) were also unclear. For copper and nickel, the levels accumulated in both the ship and reference site *Nephtys* test organisms were higher than the relatively clean Mussel Watch data. However, because the Inner/Outer ring comparisons concluded no difference between sites, the corresponding *Nephtys* ship site tissue/Mussel Watch quotients were not included in Table 4-5.

It should be noted that the ER-L/M dataset was never adjusted for confounding variables that can influence bioavailability, and ultimately ecological risk, including TOC and grain size, providing a potential explanation for the inconsistent results.

4.5.2.2. COCs Elevated in Region (High Background)

Two metals showed no difference between the ship and reference site for any chemical measurements (sediment or tissue bioaccumulation), yet had high background concentrations in the sediments relative to their respective effects benchmark levels. Chromium was higher than its ERL value, indicating low but possible effects, while lead was higher than its ERM value,

indicating probable effects. Since there was no statistical difference between the ship and reference sites for these two metals however, their elevation appears to be related to high background levels for that area of the ocean bottom.

Additionally, it should be noted that three metals showed high bioaccumulation levels, relative to the clean Mussel Watch stations. Zinc was elevated in both species, while mercury and lead were elevated in *Macoma*.

4.5.3. Discussion of Contaminant/Toxicity Results at Individual Sampling Stations – Beyond the Ship versus Reference Site Comparison

It was noted in the report that the highest chemical or biological signals in individual sampling stations on the Inner Ring (ex-AGERHOLM site) appeared to cluster near the large break in the hull at the rear of the ship (Figure 4-12). Table 4-6 summarizes where these signals were observed.

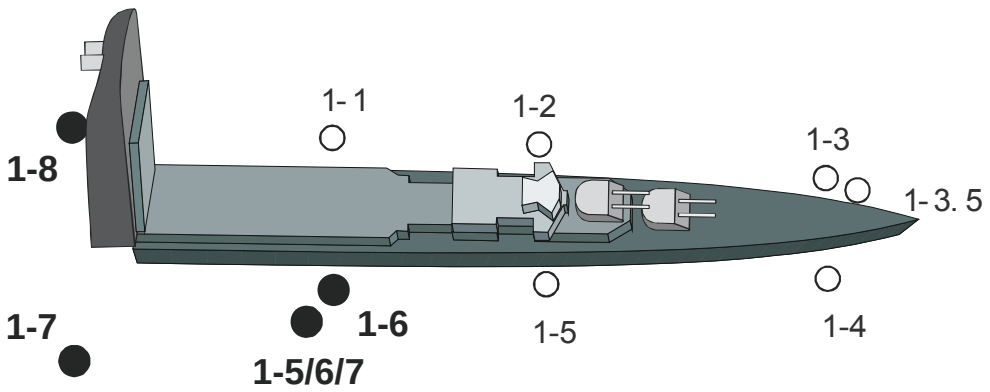


Figure 4-12. Inner Ring sediment sampling stations. Stations where elevated biological or chemical results appeared to be clustered, in bold.

Table 4-6. Clustering of individual sampling stations where elevated biological or chemical signals were observed qualitatively if not statistically.

	Station	Line of Evidence	Specific Test or COC	Remarks
Primary	1-6	Acute toxicity	amphipod survival	grain size effect
	1-6&7	PCB chemistry	by congeners	below ERL
	1-8	PCB bioaccumulation	*Nephtys*	higher than clean MW
Supplementary	1-7	PCB chemistry	by aroclor	sum of 3 Aroclors ~ 80% of total detected
	1-7	PAH chemistry	by 41 total PAHs	6x higher than all other stations, but below ERL
	1-7	chromium bioaccumulation	*Macoma*	higher than clean MW
	1-5/6/7	PCB chemistry	3 cm depth core measurement (1900 ppb)	anomalous data presumably due to sediment mixing during/after impact

These signals were not necessarily determined statistically but rather are apparent qualitatively. Furthermore, the signals could not be correlated station-to-station or among lines of evidence (i.e., chemistry, toxicity, and bioaccumulation). For example, Station 1-6 exhibited toxicity but no effects-inducing levels of PCBs or any other contaminant. Station 7 had higher chemistry levels for a number of constituents: chromium bioaccumulated in the *Macoma* clam, PCBs measured by congeners and Aroclors (the latter measurement representing about 80% of the total Aroclor concentration detected for the sum of all stations), and PAHs - the concentration for this station was at least 6 times higher than at any other station, but was still below effects-inducing levels. Sediment grain size characteristics of the ocean bottom appeared to be the most likely explanation for toxicity observed at Station 1-6. Qualitatively, these various individual station results did not appear to be correlated. Quantitatively, attempts to use statistics to probe for correlations did not reveal any relationships. As described in Section 2.5.12.4, Principal components analysis (PCA) uncovered the following factors and their corresponding contributions to the total variability (% in parentheses) in all SINKEX measurements:

- Hg, Ni, Ag, TOC (31%);
- Grain Size (28%);
- Cd, Cu, total PCBs (21%);
- CaCO3, Al, Cr, Fe (21%).

Each of the four factor groupings accounted for between one-fifth and one-third of the total variability. As described in Section 3.2.3.2, Pearson's correlation statistical tests were used to test for relationships between sediment and tissue concentrations, but none were found.

It is speculated that there were more contaminants released from the break in the ship and deposited into the sediments after the ship settled onto the ocean bottom, relative to the other parts of the hulk. The break in the hull, internal structures, and equipment at the rear of the ship may have caused an increase in exposed surface area of the inside of the hulk for leaching and/or particulate transfer of contaminants from shipboard materials into the environment.

In summary, based on the sampling design, in which individual stations represented replicates for both the ship and reference sites, the elevated contaminant and toxicity results observed did not contribute to an overall significant difference between ship and reference sites.

4.6. Risk Determination

4.6.1. Primary Lines of Evidence in Decision Matrix

Based on the analysis of the three primary lines of evidence (Table 4-1), which compared (1) PCB concentrations in sediment, (2) PCB concentrations in tissue, and (3) sediment toxicity between samples from the ex-AGERHOLM ship site (Inner Ring, #1) and those from the reference site (Outer Ring, #4), there is no significant risk to the marine environment from PCBs onboard the ex-AGERHOLM.

4.6.2. Supplementary Lines of Evidence

4.6.2.1. Analysis of Other PCB Data and Individual Station Comparisons

There are isolated individual spikes of sediment and tissue PCB concentrations, as well as toxicity, which were observed at individual sampling stations. However, no qualitative or quantitative correlations were detected among these signals with respect to specific location (i.e., individual sampling stations), line of evidence (i.e., sediment chemistry, toxicity, bioaccumulation), and environmental compartment (i.e., sediment vs. tissue concentrations).

4.6.2.2. Analysis of Secondary COCs

Based on the assessment of secondary COCs, there appears to be several metals (Ni, Cd, Cu, and Ag) that have been released from (presumably by corrosive/dissolution processes) the vessel accumulating in the sediment near the ship.

One of these metals (Ni) was present at sediment concentrations high enough to pose a localized risk to site ecology, while two others (i.e., Cu and Cd) were measured at sediment concentrations high enough to pose a low but possible risk. However, there is low confidence in the accuracy of the Nickel effects benchmark.

Three metals (Cd, Cu, and Ag) were shown in laboratory tests to bioaccumulate at higher levels in test organisms exposed to the ship sediments (relative to those exposed to reference site sediments). Copper was the only one that accumulated to levels much higher than observed in "clean" Mussel Watch stations.

Three other metals (Pb, Hg, and Zn) bioaccumulated in both ship and reference sediment laboratory tests – to levels higher than the "clean" Mussel Watch stations, indicating a possible elevated level of these metals in ocean sediments in this region.

4.6.2.3. Evaluation of Benthic Community and Grain Size

Relative to the available literature on lower slope communities off of Southern California, both the ex-AGERHOLM ship and reference sites appear to support diverse, but not abundant benthic communities. No differences in whole community ecological structure measures were observed between the two sites. Physical characteristics, especially grain size, which differed between the ship and reference sites, were weakly correlated, statistically, with differences observed among taxonomic sub-groupings of benthic community structure between the two sites (see 3.1.3). These physical differences may also explain the toxicity observed at Station 1-6.

4.6.2.4. Supplementary Sablefish Analysis

The overall risk determined by the probability of exceeding benchmarks for Total PCBs showed very low risk of potentially harmful exposure to the deep-sea pelagic community and negligible risk to sablefish from critical body residues of Total PCB.

4.7. Comparison with Global Marine PCB Data

Figure 4-13 summarizes numerous studies from the scientific literature where PCBs were detected in marine sediments bringing an additional global perspective to the ex-AGERHOLM study. Very limited amounts of data are available for deeper ocean benthic samples and, as a result, data from shallower samples are presented. Data are reported as dry weight or were

converted to dry weight concentrations if sufficient supporting data was provided. Wet weight concentrations that could not be converted have been incorporated as reported. Compared to information compiled in Figure 4-13, average Inner Ring PCB sediment concentrations at the ex-AGERHOLM site (3.9 µg/kg) are below approximately 50% of the concentrations reported in the general global sediment PCB literature.

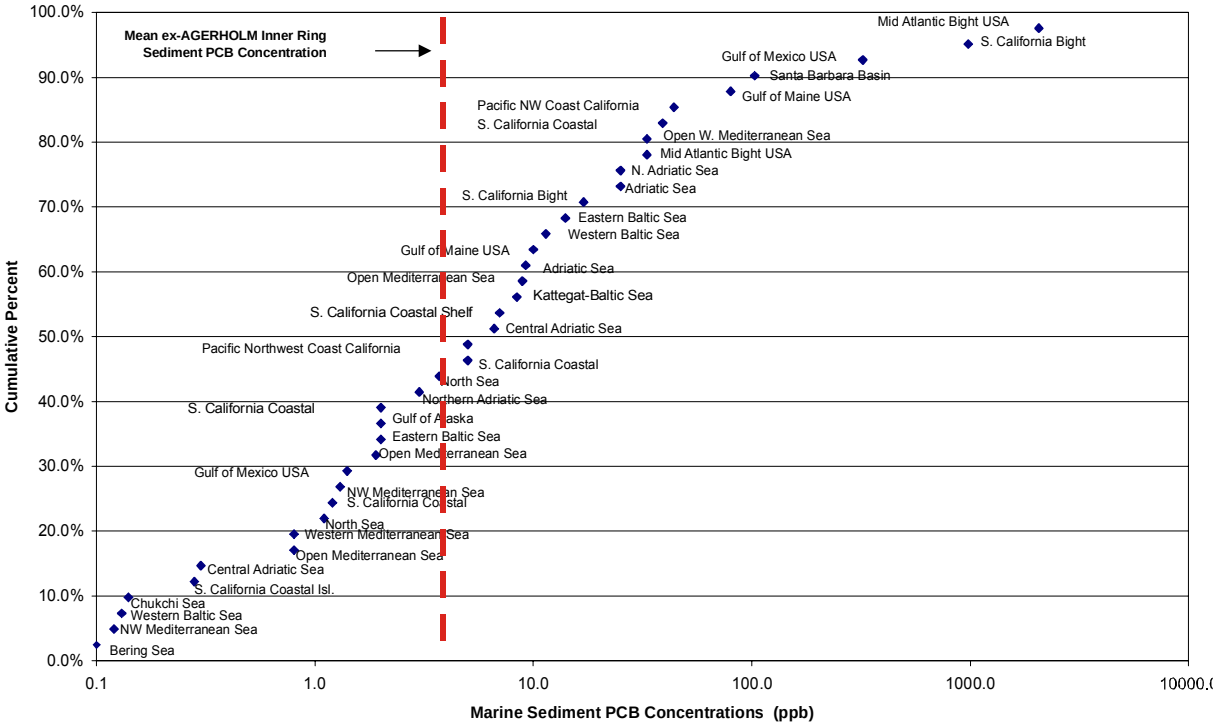


Figure 4-13. Global marine sediment PCB concentrations (from Section 2.3.4).

A review of the scientific literature (late 1970s to 2002, Appendix M) provided additional perspective on the potential effects of ex-AGERHOLM sediment PCB concentrations on resident biota. An attempt was made to locate a broad range of data focusing on open-ocean and deep ocean PCB sediment concentrations, and to relate these concentrations to published PCB toxicity values. Figure 4-14 summarizes biological effects of sediment PCB concentrations found in the scientific literature. Studies relating sediment PCB concentrations to effects on marine organisms vary in types of tests, organisms and sediments tested. Several types of toxicity tests of varying durations with several species and different life stages have been reported in the literature, possibly contributing to the wide range of PCB concentrations causing effects. These studies used natural or spiked sediments with results generally reported on a dry weight basis, and not normalized to total organic carbon content. A review of this figure indicates that the average sediment concentration measured at the ex-AGERHOLM site is much lower than all concentrations reported to have adverse effects on a wide variety of organisms. Additionally, it is noted that even the highest measured concentration (excluding sediment cores) in the ex-AGERHOLM study (9.8 µg/kg, Station 1-7) was below any of these measured effects concentrations.

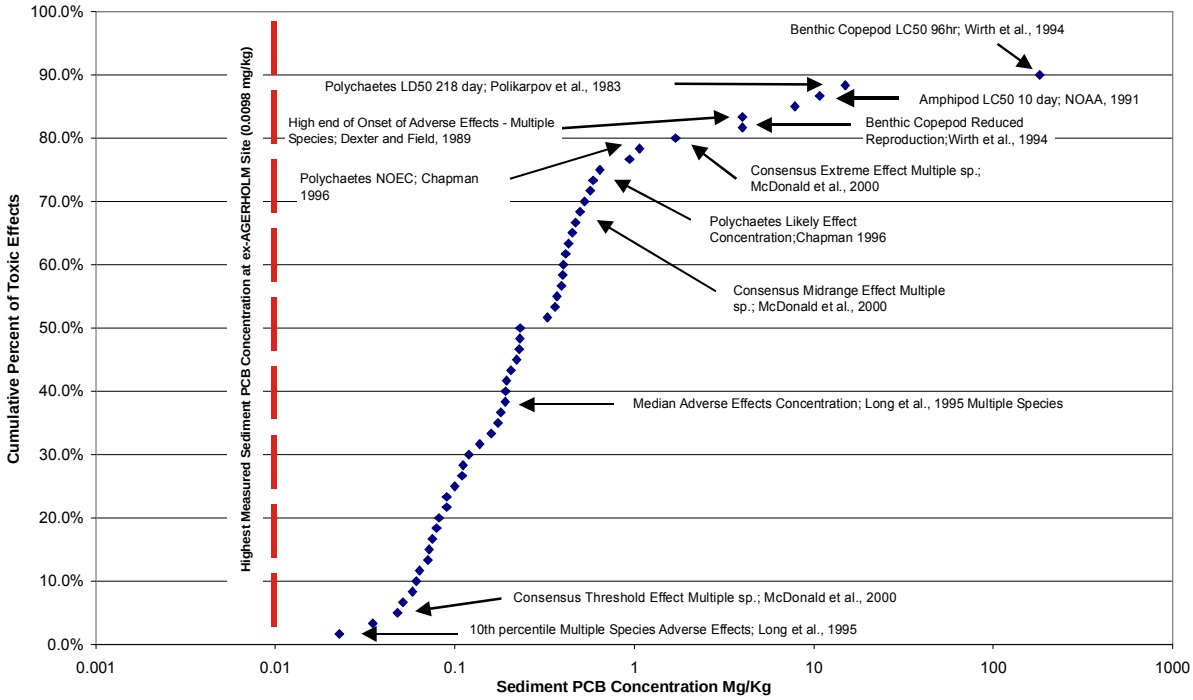


Figure 4-14. Sediment PCB toxicity reported from the literature, PCB concentrations reported may be as an Aroclor equivalent, or the sum of several congeners (from Section 2.3.4).

Figure 4-15 summarizes literature values documenting PCB concentrations in pelagic and benthic species tissue samples. It provides some indication of field concentrations in whole body and muscle tissues. Much of the data reported in the scientific literature is reported as lipid-normalized, or from other tissues such as liver. Tissue PCB concentrations vary over a wide range of concentrations (vertical bars) across the species measured. Average total PCB concentrations in ex-AGERHOLM bioaccumulation test organism tissues ranged from 16.81-21.85 µg/kg (Inner Ring-Outer Ring, *Macoma*) to 18.1-29.43 µg/kg (Inner Ring-Outer Ring, *Nephtys*) and fall in the lower 30-40 percent of data presented in Figure 4-15. Comparing all ex-AGERHOLM bioaccumulation data (ranges in Table 3-51) with the information presented in Figure 4-15 suggests that surrogate tissue concentrations from the SINKEX study fall within the lower range of concentrations reported in typical field-collected organisms.

More important than simple comparisons of tissue body burdens are those comparisons based on some kind of observed effects. Species sensitivity distributions (SSDs) of effects from critical body residues of PCBs in fish were developed for the supplementary Sablefish analysis to compare the PCB residues found in specimens collected from both the ship and reference sites to a larger data set obtained from the literature. Derived from toxicity data, SSDs are cumulative distribution functions, which describe the proportion of a class of organisms (i.e., fish) expected to be affected by a given level of exposure to a contaminant (Posthuma et al 2001, Maltby et al. 2005). The SSDs for the No Observed Effects Dose (NOED) is shown in Figure 4-16, indicating that the residues from the SINKEX study were well below observed effects levels for all species of fish from the literature.

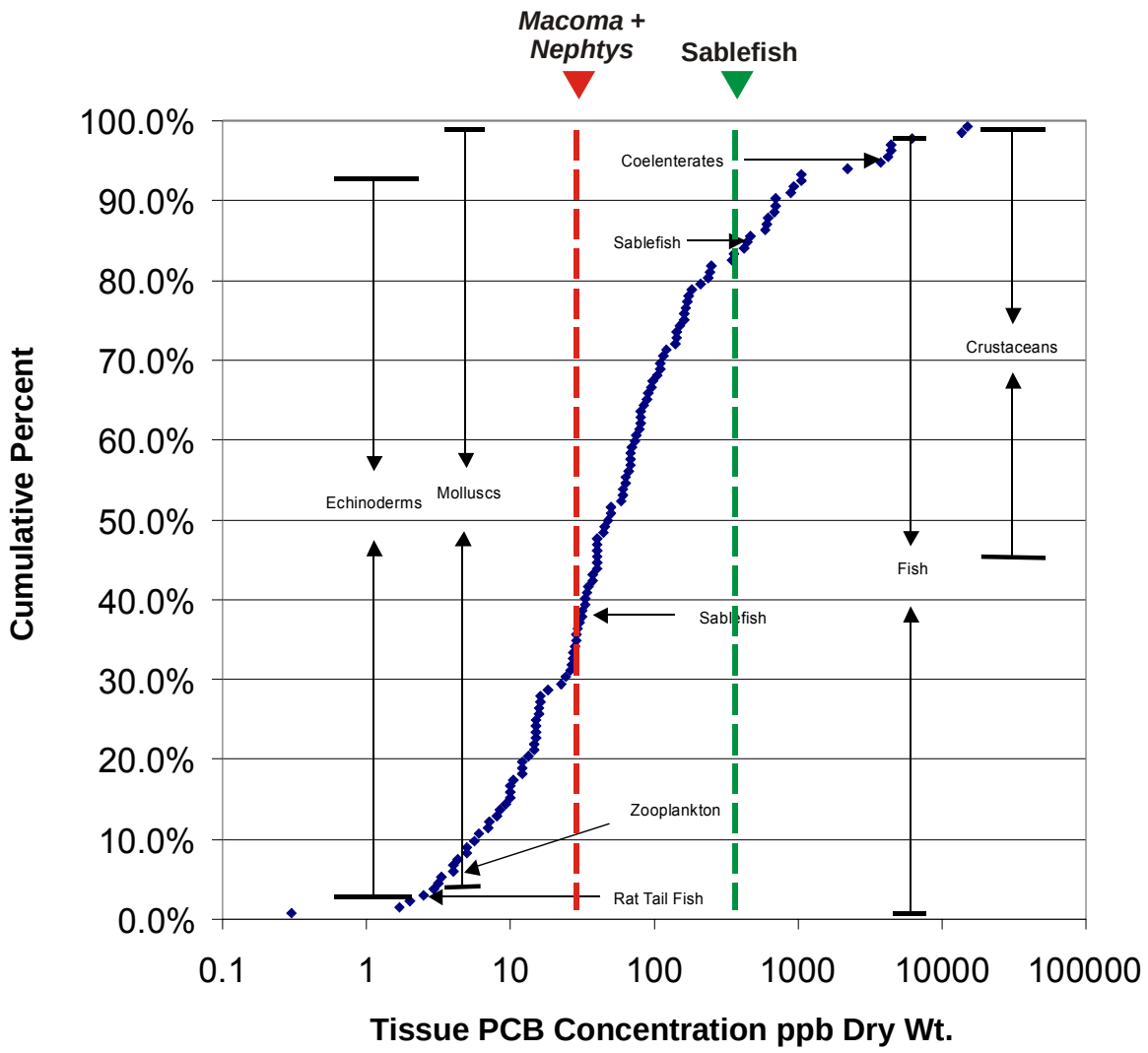


Figure 4-15. PCB concentrations in marine species whole body or muscle tissues (dry weight). PCB concentration may be as an Aroclor equivalent, or as the sum of congeners measured; single values, mean values, and extremes of ranges are included as single points (from Section 2.3.4). Mean combined *Macoma* + *Nephtys* (41 ppb) and mean sablefish (345 ppb) tissue PCB bioaccumulation concentrations for the ex-AGERHOLM site (combined Inner and Outer Rings) indicated by dashed lines.

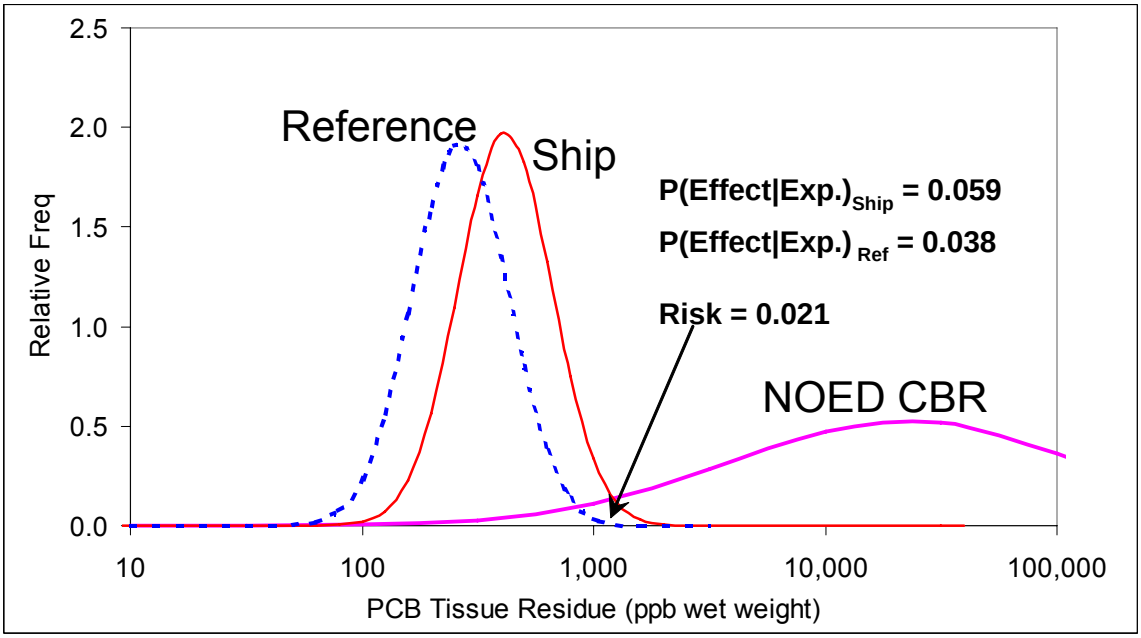


Figure 4-16. Total PCB species sensitive distribution (SSD) of NOEDs for Critical Body Residues (CBR) and exposure distributions for reference and ship shown as a probability density function.

4.8. Uncertainty Analysis

Sources of uncertainty in this study were as follows:

1. **Small number of samples:** Because of sampling difficulties, fewer samples than originally planned were collected, which resulted in a decrease of statistical power and confidence in the data. The decrease in the planned number of samples was due to difficulties in obtaining high-quality sediment samples from the heterogeneous (rock and sediment) bottom substrate, requiring a reduction in the scope of the sampling design from four concentric rings to one Inner Ring and one Outer Ring, thereby decreasing the total number of stations assessed and sediment samples retrieved.

2. **Long time lapse between field sampling events:** Samples were taken over a one-year period, which introduced the possibility of a time-dependent error that decreased the ability to statistically discriminate differences between ship and reference sites. Specifically, most ship stations were sampled in September 1998 and most of the reference stations were sampled in November 1999. This time lapse of 15 months between field collections was due to:

   - Scheduling difficulties associated with availability of ship time and ROV availability/downtime, and
   - Sampling difficulties in the hard bottom sediment matrix required modifications to the overall sampling plan, including the re-engineering of sampling equipment (e.g., geared closing mechanisms) and changes in the field sampling approach (e.g., reversion to the small boxcore after a developmental piston corer failed to obtain samples).

3. **Inconsistencies in lab analyses caused by time lapse (#2, above):** Samples taken over a year apart caused some inconsistencies in analytical procedures as itemized below:

- In a few cases, samples were analyzed by different laboratories, different personnel, different methods, and different equipment, however the precision and accuracy of the overwhelming majority of the results is not in question.
- For the bioaccumulation tests, different-sized worms (*Neanthes*) caused confusion in the comparison of growth rates. The different worm sizes were directly related to the specific seasons during which the test organisms were collected.

4. **Possible laboratory contamination:** There were two unexpected instances of possible external contamination of laboratory animals/equipment, leading to difficulties in interpretation:
   - An increase in tissue PCBs in *Macoma* and *Nephtys* during bioaccumulation tests, possibly due to contaminated lab water or sediments.
   - High initial body burdens of PAHs in *Macoma*, most likely from stock test organisms, which were not discovered until detailed analysis of the data was accomplished.
5. **Different sediment used for chemical and biological tests:** The complications in sampling design, caused by the challenges explained above, forced a two-pronged requirement at every sediment station to retrieve one small, undisturbed sample of sediment for chemical analysis and several larger samples from the same station composited as one large volume of sediment for bioassays (i.e., toxicity testing and bioaccumulation). Since the only effective sampling device for sediments proved to be the small boxcores, approximately 11-12 boxcore samples were required to produce a composite sample large enough to support bioassay testing of a single sampling station. Because sediments used for bioaccumulation testing were obtained and treated differently than sediment chemistry samples, and due to the apparent heterogeneity of contaminants, statistical testing attempting to link sediment and tissue concentrations was not successful.
6. **Only one study site used:** During initial planning stages, a scientific review panel thought that it was necessary to conduct investigations at more than just one SINKEX site. Although considerable time, resources and effort were spent attempting to find more than one SINKEX vessel, only the ex-AGERHOLM was located and subsequently utilized as a study site.

It was determined that the above uncertainties did not cause an incorrect conclusion in either the individual lines of evidence or in the overall assessment of risk.

4.9. Summary Conclusions

In conclusion, there is no evidence of adverse effects and no incremental risk to the marine environment associated with PCBs-ISM onboard the ex-AGERHOLM. The study followed an overall technical approach centered on a Weight-of-Evidence based decision matrix. This technical approach was the result of planning and consensus among Navy and US EPA risk assessors and managers, before the study plan was executed. The study adhered to principles laid out by the US EPA and the Army Corps of Engineers in the two documents that guide the conduct of ocean sediment testing and ecological risk assessment. A simple, 2-sample "study site" vs. "reference site" statistical testing design was used; however, the execution of deep ocean sampling over a period of one year required the use of a 2-way Analysis of Variance (ANOVA) rather than a simple "t-test." No significant differences were found between the study site and reference sites, for any of the three primary lines of evidence (sediment chemistry PCBs,

sediment toxicity, and PCB bioaccumulation). No differences in whole community ecological structure measures were observed between the two sites. Physical characteristics, especially grain size, which differed between the ship and reference sites, weakly correlated with, and may explain differences observed among taxonomic sub-groupings of benthic community structure between the two sites. These physical differences may also explain the apparent toxicity observed at Station 1-6. Isolated, elevated measurements for some COCs (PCBs, metals, PAHs) in sediment and tissue samples were observed for some stations, and these higher concentrations appeared to be clustered near the massive break in the ship structure. With the exception of three metals, the elevated COCs were not high enough to trigger statistical significance when combined as replicates in the Inner vs. Outer Ring comparisons. Nickel poses a probable localized risk while copper and cadmium pose a possible, but low localized risk. The supplemental sablefish analysis found low risk of potentially harmful exposure to the deep sea pelagic community and negligible risk to sablefish from critical body residues of Total PCBs. The human health risk assessment concluded there was negligible risk to the human consumers of sablefish though the commercial market basket pathway. Finally, when compared to global data on marine PCBs, the PCB concentrations found at the ex-AGERHOLM site are well below any observed effects levels.

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Risk Assessment of the Potential Release of PCBs and Other Contaminants from Sunken Navy Ships in the Deep Ocean: ex-AGERHOLM Case Study

March 2006
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PEO SHIPS

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INFAUNA

GROUP	SAMPLE ID	2SX0082	2SX0083	2SX0084	2SX0085	2SX0086	2SX0087	2SX0088	2SX0089	2SX0090	2SX0091	2SX0092	2SX0093	2SX0094	2SX0095	2SX0096	2SX0097	2SX0098	2SX0099	2SX0100	2SX0101	2SX0102	4SX001.05	4SX001.06	4SX001.07	4SX001.08	4SX003.05	4SX003.06	4SX003.07	4SX003.08	4SX004.05	4SX004.06
	STATION	1-8	1-8	1-2	1-2	1-3	1-3	4-6	4-6	1-6	1-6	1-7	1-7	1-7	1-7	1-1	1-1	1-4	1-4	1-5	1-5	1-1	4-5	4-5	4-5	4-5	4-2	4-3	4-4	4-5	4-3.1	4-3.1
	SCREEN SIZE	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0
	REPLICATE	1	1	1	1	1	1	1	1	1	1	1	1	2	2	1	1	1	1	1	1	1	1	1	2	2	1	1	2	2	1	1
GROUP	NAME																															
Annelida	Ampharetidae juv	2		1	1	1		1		3	10	2		1	2	1	3			1		8							1			
	Amphicteis scaphobranchiata																															
	Anobothrus bimaculatus	3		1	1	6						2						1														
	Anobothrus mancus	2			1		1			1			1	4	2			18	4			5			5	4	1	1		1	1	3
	Anobothrus sp																															
	Aphelochaeta monilaris																						2		2							
	Aphelochaeta spp		2		2	1						2	8	1	3	1	1			2						1	2				2	
	Aricidea (Acmira) rubra																															
	Aricidea (Acmira) simplex		1												1																	
	Aricidea (Aedicira) sp															1																
	Aricidea (Aedicira) sp 1																		1													
	Aricidea (Allia) sp 1		3		1													1									6					
	Aricidea (Allia) sp A (SCAMIT 1996)				3		1		2				1				2		1	1	4		1		2							
	Aricidea (Allia) spp														1		1															
	Aricidea (Aricidea) wassi			1	3				1		1		2								1		1		2		1					1
	Capitella "capitata"																									1						
	Capitellidae																														1	
	Chaetozone "setosa"																						1									
	Chaetozone sp 1					4																										
	Chaetozone sp 2												1																			
	Chaetozone spp				1				1				1							1					1							
	Chloeia pinnata																															
	Cirratulidae		1		1				1				1							1					1		2					
	Cirratulus sp																															
	Cirrophorus branchiatus																						1									
	Cirrophorus furcatus								1																							
	Clymaldane laevis							1																								
	Clymenella sp																											1				
	Clymenopsis californiensis							3																1								
	Cossura brunnea																			1			4				3					
	Cossura sp														2		1								3				1	2		
	Decamastus gracilis																					1										
	Dipolydora caulleryi		1			1						1			1			1														
	Eclysippe trilobata																															
	Ephesiella brevicapitis												2												1							
	Epidiopatra sp					1																										
	Euchone sp																															
	Euclymeninae sp																															
	Euclymeninae sp A (SCAMIT1987)																															
	Eunice segregata																															
	Eusyllis sp												1																			
	Exogone sp 1															1	1															
	Exogone sp 2	3								2	4	3	2																			
	Exogone sp 3																								2							
	Exogone sp 4																											1				
	Exogone sp 5																															
	Exogone sp A																															
	Exogone spp																															
	Fauveliopsis glabra		1									1						1							1							4
	Flabelligeridae				1																											
	Glycera branchiopoda																							1								
	Glycera nana				1							1			1															1		
	Glycera spp																															
	Glycera tesselata							1																								
	Goniada sp															1																
	Hesionidae																															
	Hesiospina sp 1		1		2								4				2				1											
	Idanthyrus armatus																															
	Idanthyrus sp			1																												
	Jasmineira sp B (SCAMIT1986)																															
	Lanassa sp		1																	1												
	Lanice conchilega																															

Appendix A: Benthic Infauna Data

		INFAUNA															MISCELLANEOUS INVERTEBRATES														
	SAMPLE ID	4SX004.07	4SX004.08	4SX005.05	4SX005.06	4SX005.07	4SX005.08	5SX001.05	5SX001.06	5SX001.07	5SX001.08	5SX002.05	5SX002.06	5SX003.05	5SX003.06	Total Infauna	ADCP	MISC	Misc Inverts 1	Misc Inverts 2	NWFT	WFT Crab	4-1	4-2	4.3.1	4-4	4-5	4-7	4-8	Total Misc. Inverts	Grand Total
	STATION	4-3.1	4-3.1	4-1	4-1	4-1	4-1	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-3.5	1-3.6	4-6	4-6																
	SCREEN SIZE	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	1.0	0.5	0.5	1.0	0.5	1.0																
	REPLICATE	2	2	1	1	2	2	1	1	2	2	1	1	1	1																
GROUP	NAME																														
Annelida	Ampharetidae juv							1	2	1			1	1		44														0	44
	Amphicteis scaphobranchiata															0		1												1	1
	Anobothrus bimaculatus									4	3		2			23														0	23
	Anobothrus mancus	2	1		2		1	2			2		7			72														0	72
	Anobothrus sp															0							1							1	1
	Aphelochaeta monilaris															4														0	4
	Aphelochaeta spp	5		2		3						1		9	1	49														0	49
	Aricidea (Acmira) rubra			1		2										3														0	3
	Aricidea (Acmira) simplex															2														0	2
	Aricidea (Aedicira) sp															1														0	1
	Aricidea (Aedicira) sp 1															1														0	1
	Aricidea (Allia) sp 1															11														0	11
	Aricidea (Allia) sp A (SCAMIT 1996)												2			20														0	20
	Aricidea (Allia) spp															2														0	2
	Aricidea (Aricidea) wassi	2		1				1				3		15		36														0	36
	Capitella "capitata"															1														0	1
	Capitellidae	2	1				1									5								1						1	6
	Chaetozone "setosa"				1									1		3														0	3
	Chaetozone sp 1															4							1							1	5
	Chaetozone sp 2															1														0	1
	Chaetozone spp											2	1			8								1						1	9
	Chloeia pinnata															0											1			1	1
	Cirratulidae			1								1		2		12														0	12
	Cirratulus sp	1														1											1			1	2
	Cirrophorus branchiatus															1														0	1
	Cirrophorus furcatus												1	2		4														0	4
	Clymaldane laevis															1										2	2			4	5
	Clymenella sp															1														0	1
	Clymenopsis californiensis															4														0	4
	Cossura brunnea			1												9														0	9
	Cossura sp	4		4		10					1	1		2		31														0	31
	Decamastus gracilis															1														0	1
	Dipolydora caulleryi															5														0	5
	Eclysippe trilobata								1				1			2							1		1					2	4
	Ephesiella brevicapitis			1										2		6									1					1	7
	Epidiopatra sp															1														0	1
	Euchone sp															0											1			1	1
	Euclymeninae sp						1									1														0	1
	Euclymeninae sp A (SCAMIT1987)		1													1														0	1
	Eunice segregata															0										1				1	1
	Eusyllis sp															1														0	1
	Exogone sp 1															2														0	2
	Exogone sp 2															14													1	1	15
	Exogone sp 3															2														0	2
	Exogone sp 4															1														0	1
	Exogone sp 5			1												1														0	1
	Exogone sp A													4		4														0	4
	Exogone spp										2	1				3														0	3
	Fauveliopsis glabra	2	1	1	1	3	3					1				20							6	1	2					9	29
	Flabelligeridae					2										3														0	3
	Glycera branchiopoda															1		1							1					2	3
	Glycera nana										1					5														0	5
	Glycera spp												1			1	1													1	2
	Glycera tesselata															1											1			1	2
	Goniada sp	2														3											1			1	4
	Hesionidae							1	1					1		3														0	3
	Hesiospina sp 1															10														0	10
	Idanthyrus armatus															0	16									1				17	17
	Idanthyrus sp															1														0	1
	Jasmineira sp B (SCAMIT1986)															0									1					1	1
	Lanassa sp															2														0	2
	Lanice conchilega														2	2														0	2

Appendix A: Benthic Infauna Data

INFAUNA

GROUP	SAMPLE ID	2SX0082	2SX0083	2SX0084	2SX0085	2SX0086	2SX0087	2SX0088	2SX0089	2SX0090	2SX0091	2SX0092	2SX0093	2SX0094	2SX0095	2SX0096	2SX0097	2SX0098	2SX0099	2SX0100	2SX0101	2SX0102	4SX001.05	4SX001.06	4SX001.07	4SX001.08	4SX003.05	4SX003.06	4SX003.07	4SX003.08	4SX004.05	4SX004.06
	STATION	1-8	1-8	1-2	1-2	1-3	1-3	4-6	4-6	1-6	1-6	1-7	1-7	1-7	1-7	1-1	1-1	1-4	1-4	1-5	1-5	1-1	4-5	4-5	4-5	4-5	4-2	4-3	4-4	4-5	4-3.1	4-3.1
	SCREEN SIZE	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0
	REPLICATE	1	1	1	1	1	1	1	1	1	1	1	1	2	2	1	1	1	1	1	1	1	1	1	2	2	1	1	2	2	1	1
GROUP	NAME																															
Annelida	Laonice sp			2	1	1							2		2		3			1	2	1										
	Leitoscoloplos mexicanus															1																
	Leitoscoloplos pugettensis																							1								
	Leitoscoloplos spp				1												1										1					
	Levinsenia gracilis																															
	Levinsenia sp																2															
	Lumbrineridae			1							5	1	1				1	2	1		1	3	1				1					1
	Lumbrineris cruzensis																															
	Lumbrineris limicola	2			1	1																										
	Lysippe sp																															
	Lysippe sp B																					1										
	Maldane sarsi											1																				
	Maldane sp									1	3			2		1					2											
	Maldanidae				2	1																		1			1		1			3
	Mediomastus spp										1											1				1	1					
	Melinna exila									3																						
	Metasychis sp																															
	Mexamage longibranchiata																															
	Monticellina cryptica																												1			
	Monticellina serratiseta																							1								
	Myriochele gracilis																															
	Myriochele heeri																											1				
	Myriochele pygidialis			1																												
	Myriochele spp																															
	Naineris uncinata																															
	Neomediomastus glabrus										1																					
	Nephtys cornuta										3		1				1		1													
	Ninoe longibranchiata										2																		1	1		
	Notomastus abyssalis																															
	Notomastus cinctus																															
	Notomastus spp																															
	Notoproctus pacificus																											1		1		
	Oligochaeta																															
	Ophelina sp													1	1																	
	Oweniidae																			1												
	Paradoneis sp																												1			
	Paralysippe annectens		3					5		1		1			1			1	1	2												
	Paraonidae													1																		
	Pherusa abyssalis																															
	Pherusa sp.																										1	3		1		
	Phyllochaetopterus limicolus	4			1					9			1	3				2		6		2						1		1	1	
	Polychaeta																								1							
	Polynoidae																															
	Polynoidae sp 1																															
	Polyophthalmus translucens																										2					1
	Rhodine sp									1																						
	Samytha californiensis																															
	Samytha sp																															
	Samythella pala																															
	Scolelepis sp 1																	1														
	Scoloplos mazatlanensis								1																							
	Sphaerodoropsis biserialis																								1							
	Sphaerodoropsis sp				1																											
	Sphaerodoropsis sphaerulifer																															
	Sphaerosyllis bilineata														1		1				1											
	Sphaerosyllis biserialis												1																			
	Spiophanes duplex																					1										
	Spiophanes fimbriata			1		1					1	1										2	1									
	Spiophanes pallidus														2																	
	Spiophanes spp		1														1															
	Sternaspis fossor																											3				

Appendix A: Benthic Infauna Data

		INFAUNA															MISCELLANEOUS INVERTEBRATES														
	SAMPLE ID	4SX004.07	4SX004.08	4SX005.05	4SX005.06	4SX005.07	4SX005.08	5SX001.05	5SX001.06	5SX001.07	5SX001.08	5SX002.05	5SX002.06	5SX003.05	5SX003.06	Total Infauna	ADCP	MISC	Misc Inverts 1	Misc Inverts 2	NWFT	WFT Crab	4-1	4-2	4-3.1	4-4	4-5	4-7	4-8	Total Misc. Inverts	Grand Total
	STATION	4-3.1	4-3.1	4-1	4-1	4-1	4-1	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-3.5	1-3.6	4-6	4-6																
	SCREEN SIZE	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	1.0	0.5	0.5	1.0	0.5	1.0																
	REPLICATE	2	2	1	1	2	2	1	1	2	2	1	1	1	1																
GROUP	NAME																														
Annelida	Laonice sp															15														0	15
	Leitoscoloplos mexicanus															1														0	1
	Leitoscoloplos pugettensis													1		2														0	2
	Leitoscoloplos spp			1		1		1								6														0	6
	Levinsenia gracilis			1												1														0	1
	Levinsenia sp															2														0	2
	Lumbrineridae		1	1									1			22														0	22
	Lumbrineris cruzensis		1													1														0	1
	Lumbrineris limicola															4														0	4
	Lysippe sp															0							1		1					2	2
	Lysippe sp B															1														0	1
	Maldane sarsi															1														0	1
	Maldane sp										1					10														0	10
	Maldanidae	2		3	4	1	3				1					23							1							1	24
	Mediomastus spp				1											5														0	5
	Melinna exila															3														0	3
	Metasychis sp															0									1					1	1
	Mexamage longibranchiata													1	2	3														0	3
	Monticellina cryptica															1														0	1
	Monticellina serratiseta															1														0	1
	Myriochele gracilis														1	1														0	1
	Myriochele heeri															1														0	1
	Myriochele pygidialis		1		2											4								2	2		2			6	10
	Myriochele spp			1									1			2							1							1	3
	Naineris uncinata															0										1				1	1
	Neomediomastus glabrus															1														0	1
	Nephtys cornuta										1					7														0	7
	Ninoe longibranchiata			1	1	1					1					8														0	8
	Notomastus abyssalis															0											1			1	1
	Notomastus cinctus															0											2			2	2
	Notomastus spp						1									1							2		1					3	4
	Notoproctus pacificus	2	2													6							1	3	2					6	12
	Oligochaeta													2		2														0	2
	Ophelina sp													1		3														0	3
	Oweniidae															1														0	1
	Paradoneis sp															1														0	1
	Paralysippe annectens															15														0	15
	Paraonidae	1									1					3														0	3
	Pherusa abyssalis															0									2					2	2
	Pherusa sp.								1							6														0	6
	Phyllochaetopterus limicolus		1			1				2	4		1			40														0	40
	Polychaeta															1														0	1
	Polynoidae					1			1			1				3														0	3
	Polynoidae sp 1															0			1											1	1
	Polyophthalmus translucens															3							1							1	4
	Rhodine sp															1														0	1
	Samytha californiensis									1						1														0	1
	Samytha sp															0											1			1	1
	Samythella pala														1	1														0	1
	Scolelepis sp 1															1														0	1
	Scoloplos mazatlanensis															1														0	1
	Sphaerodoropsis biserialis															1														0	1
	Sphaerodoropsis sp															1														0	1
	Sphaerodoropsis sphaerulifer					1										1														0	1
	Sphaerosyllis bilineata															3														0	3
	Sphaerosyllis biserialis															1														0	1
	Spiophanes duplex								1		1					3														0	3
	Spiophanes fimbriata															7														0	7
	Spiophanes pallidus															2														0	2
	Spiophanes spp															2														0	2
	Sternaspis fossor		1			1	1									6							2	1	4					7	13

Appendix A: Benthic Infauna Data

INFAUNA

GROUP	SAMPLE ID	2SX0082	2SX0083	2SX0084	2SX0085	2SX0086	2SX0087	2SX0088	2SX0089	2SX0090	2SX0091	2SX0092	2SX0093	2SX0094	2SX0095	2SX0096	2SX0097	2SX0098	2SX0099	2SX0100	2SX0101	2SX0102	4SX001.05	4SX001.06	4SX001.07	4SX001.08	4SX003.05	4SX003.06	4SX003.07	4SX003.08	4SX004.05	4SX004.06
	STATION	1-8	1-8	1-2	1-2	1-3	1-3	4-6	4-6	1-6	1-6	1-7	1-7	1-7	1-7	1-1	1-1	1-4	1-4	1-5	1-5	1-1	4-5	4-5	4-5	4-5	4-2	4-3	4-4	4-5	4-3.1	4-3.1
	SCREEN SIZE	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0
	REPLICATE	1	1	1	1	1	1	1	1	1	1	1	1	2	2	1	1	1	1	1	1	1	1	1	2	2	1	1	2	2	1	1
GROUP	NAME																															
Annelida	Streblosoma sp B											1																				
	Subadyte mexicana			1													1			1										1		
	Terebellidae																															
	Terebellides californica																					1										
	Terebellides reishi												1																			
	Terebellides spp				1										1		1															
	Thelepus sp																															
	Therochaeta pacifica																															
	Therochaeta sp											1																				
	Trichobranchus sp 2					1	1																									
Annelida Total		16	15	10	26	19	3	11	7	21	31	18	31	13	21	7	23	28	9	19	12	26	12	5	22	7	22	12	6	9	5	13
Cnidaria	Anthenaria													1																		
	Campanulinidae (colonial)																															
	Ceriantharia																									1						
	Ceriantharia sp 1 (Sinkex)																															1
	Edwardsia sp A													1																		
	Gorgonacea, Unidentified	1																														
	Halianthella sp A (SCAMIT)																															
	Halianthella sp C																															
	Hydrozoa									1							1															
	Lafoea sp																															
	Liponemo brevicornis																															
	Pavonaria finmarchica																															
Cnidaria Total		1	0	0	0	0	0	0	0	1	0	0	0	2	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Crustacea	Acidostoma hancocki																												1			
	Ampelisca plumosa	1																														
	Ampelisca pugetica																															1
	Ampelisca romigi																							1								
	Ampelisca sp 1																									1						
	Ampelisca spp																															
	Ampelisca unsocalae							2																								
	Araphura breviara																								1							
	Bathura luna																															
	Bathymedon vulpeculus							1																								
	Bebnectes sp																															
	Bonnierella linearis californica																															
	Byblis bathyalis										1	1																				
	Calastacas quinqueseriatus																															
	Campylapsis sp 1																															
	Campylapsis sp 2																											1				
	Campylaspis rubromaculata																											1				
	Caprella equilibra																															
	Caprella sp 1																															
	Caprellidae																															
	Carpoapseudes caraspinosus																															
	Chauliopleona dentata						1							1					1								2					
	Chorilia longipes																															
	Cirolana californiensis					1																				1						
	Cumacea																															
	Cypridinidae sp 1																															1
	Desmosomatidae sp 1																								2							
	Diastylis sp							1																								
	Dulichiopsis remis													2																		
	Eualus townsendi																															
	Eudorella sp							1						1																		
	Euphilomedes dentata																															
	Eurycope californiensis							4																			1	1				
	Eusiridae																										3	1				
	Eusirus longipes																											1				
	Gammaridea																		1													
	Gnathia sp																												1			

Appendix A: Benthic Infauna Data

		INFAUNA															MISCELLANEOUS INVERTEBRATES														
	SAMPLE ID	4SX004.07	4SX004.08	4SX005.05	4SX005.06	4SX005.07	4SX005.08	5SX001.05	5SX001.06	5SX001.07	5SX001.08	5SX002.05	5SX002.06	5SX003.05	5SX003.06	Total Infauna	ADCP	MISC	Misc Inverts 1	Misc Inverts 2	NWFT	WFT Crab	4-1	4-2	4-3.1	4-4	4-5	4-7	4-8	Total Misc. Inverts	Grand Total
	STATION	4-3.1	4-3.1	4-1	4-1	4-1	4-1	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-3.5	1-3.6	4-6	4-6																
	SCREEN SIZE	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	1.0	0.5	0.5	1.0	0.5	1.0																
	REPLICATE	2	2	1	1	2	2	1	1	2	2	1	1	1	1																
GROUP	NAME																														
Annelida	Streblosoma sp B															1														0	1
	Subadyte mexicana							1						4	1	10														0	10
	Terebellidae									1	1					2														0	2
	Terebellides californica															1		1												1	2
	Terebellides reishi															1														0	1
	Terebellides spp								1		1					5							1	1						2	7
	Thelepus sp														1	1														0	1
	Therochaeta pacifica															0													1	1	1
	Therochaeta sp															1														0	1
	Trichobranchus sp 2															2														0	2
Annelida Total		25	11	21	12	27	11	7	8	9	21	11	19	48	9	718	17	3	1	0	0	0	19	10	19	5	13	0	2	89	807
Cnidaria	Anthenaria					1										2											2			2	4
	Campanulinidae (colonial)									1						1										1				1	2
	Ceriantharia															1														0	1
	Ceriantharia sp 1 (Sinkex)												1			2											1			1	3
	Edwardsia sp A															1									4		2			6	7
	Gorgonacea, Unidentified															1														0	1
	Halianthella sp A (SCAMIT)						1									1							1		1					2	3
	Halianthella sp C															0							1							1	1
	Hydrozoa															2														0	2
	Lafoea sp														1	1														0	1
	Liponemo brevicornis															0		2												2	2
	Pavonaria finmarchica															0					1									1	1
Cnidaria Total		0	0	0	0	1	1	0	0	1	0	0	1	0	1	12	0	2	0	0	1	0	2	0	5	1	5	0	0	16	28
Crustacea	Acidostoma hancocki		1													2														0	2
	Ampelisca plumosa															1														0	1
	Ampelisca pugetica															1														0	1
	Ampelisca romigi				6											7							4				1			5	12
	Ampelisca sp 1															1									1					1	2
	Ampelisca spp			1	4											5														0	5
	Ampelisca unsocalae														1	3									1					1	4
	Araphura breviara													1		2														0	2
	Bathura luna															0									1	1				2	2
	Bathymedon vulpeculus															1														0	1
	Bebnectes sp			1												1														0	1
	Bonnierella linearis californica			1	2											3														0	3
	Byblis bathyalis														1	3														0	3
	Calastacas quinqueseriatus															0		1												1	1
	Campylapsis sp 1											2				2					7									7	9
	Campylapsis sp 2															1														0	1
	Campylaspis rubromaculata		1													2														0	2
	Caprella equilibra									2	4				1	7														0	7
	Caprella sp 1															0						1								1	1
	Caprellidae											2		2		4														0	4
	Carpoapseudes caraspinosus															0									1	1				2	2
	Chauliopleona dentata	1														6														0	6
	Chorilia longipes														1	1		2				1					1			4	5
	Cirolana californiensis														2	4									2		1			3	7
	Cumacea					1			1							2														0	2
	Cypridinidae sp 1			1												2														0	2
	Desmosomatidae sp 1													1		3														0	3
	Diastylis sp															1														0	1
	Dulichiopsis remis			1												3														0	3
	Eualus townsendi															0											1			1	1
	Eudorella sp													1		3														0	3
	Euphilomedes dentata			1												1														0	1
	Eurycope californiensis														1	7							1							1	8
	Eusiridae															4														0	4
	Eusirus longipes															1														0	1
	Gammaridea						1				1					3														0	3
	Gnathia sp															1														0	1

Appendix A: Benthic Infauna Data

INFAUNA

GROUP	SAMPLE ID	2SX0082	2SX0083	2SX0084	2SX0085	2SX0086	2SX0087	2SX0088	2SX0089	2SX0090	2SX0091	2SX0092	2SX0093	2SX0094	2SX0095	2SX0096	2SX0097	2SX0098	2SX0099	2SX0100	2SX0101	2SX0102	4SX001.05	4SX001.06	4SX001.07	4SX001.08	4SX003.05	4SX003.06	4SX003.07	4SX003.08	4SX004.05	4SX004.06
	STATION	1-8	1-8	1-2	1-2	1-3	1-3	4-6	4-6	1-6	1-6	1-7	1-7	1-7	1-7	1-1	1-1	1-4	1-4	1-5	1-5	1-1	4-5	4-5	4-5	4-5	4-2	4-3	4-4	4-5	4-3.1	4-3.1
	SCREEN SIZE	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0
	REPLICATE	1	1	1	1	1	1	1	1	1	1	1	1	2	2	1	1	1	1	1	1	1	1	1	2	2	1	1	2	2	1	1
GROUP	NAME																															
Crustacea	Hamatoscalpellum sanctipretense																															
	Harpacticoida																										1				1	
	Heterophoxus ellisi							1					2																			
	Hyperoche sp																															
	Ilyarachna sp 1					1																			1				2			
	Jassa slatteryi																								1							
	Lamprops sp														1																	
	Lebbeus washingtonianus																															
	Leptophoxus cf fultanus																								1				2		1	
	Leptostylis calva																		1									1				
	Leucon sp 1																															
	Leucon sp 2																									1						
	Leucon sp H																										1					
	Leucon subnasica																														1	
	Leuconidae																															
	Lithodes couesi																															
	Malacostraca																															
	Monoculodes glyconica			1				2							1																	
	Monoculodes latissimanus																															
	Mononuclodes necopinus																															
	Munidopsis quadrata																															
	Munnidae sp 1																															
	Munnopsurus sp A																															
	Nicippe tumida																															
	Notoxenodes sp																								1							
	Oediceriopsis elsula																															
	Oedicerotidae sp 1																															
	Orchomene sp																															
	Ostracoda sp 1												1												2							
	Paraleptognathia gracilis																															
	Paraphoxus "oculatus"				1					1																						
	Philomedidae sp 1				1				2				3														3		2			
	Podoceridae sp 1																															
	Prachynella lodo																															
	Procampylaspis sp		1									2																				
	Protellidae sp 1																															
	Pseudoharpinia excavata																															
	Pseudomma oculospinum																															
	Rhepoxinius cf. daboius							1																								
	Scoloura phillipsi																															
	Spirontocaris sp					1																										
	Stenothoidae																															
	Synaptotanais notabilis																															
	Tanaidacea sp 1		4		2						2		1								1						2					
	Tanaidae																															
	Tanaopsis cadieni																							1	1							
	Tritella tenuissima																															
	Typhlotanaidae sp 2																															
	Typhlotanais cf crassus												3																			
	Typhlotanais williamsi	2	4		2						1	2									1											
	Urothoe varvarini																										1	1		1		
Crustacea Total		3	9	1	6	3	1	13	2	1	4	5	10	4	2	0	0	0	3	0	2	0	0	2	10	3	14	7	8	1	3	2
Echinoderm	Allocentrotus fragilis																															
	Amphipholis pugetana																							1								
	Amphipholis sp											1				1				2					7	2						
	Amphiura arcystata																															
	Amphiuridae																			1					1	1		1				
	Asteroid																															
	Asteronyx longifissus																															
	Brisingid asteroid sp 1																															
	Brissopsis pacifica															1																

Appendix A: Benthic Infauna Data

		INFAUNA															MISCELLANEOUS INVERTEBRATES														
	SAMPLE ID	4SX004.07	4SX004.08	4SX005.05	4SX005.06	4SX005.07	4SX005.08	5SX001.05	5SX001.06	5SX001.07	5SX001.08	5SX002.05	5SX002.06	5SX003.05	5SX003.06	Total Infauna	ADCP	MISC	Misc Inverts 1	Misc Inverts 2	NWFT	WFT Crab	4-1	4-2	4-3.1	4-4	4-5	4-7	4-8	Total Misc. Inverts	Grand Total
	STATION	4-3.1	4-3.1	4-1	4-1	4-1	4-1	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-3.5	1-3.6	4-6	4-6																
	SCREEN SIZE	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	1.0	0.5	0.5	1.0	0.5	1.0																
	REPLICATE	2	2	1	1	2	2	1	1	2	2	1	1	1	1																
GROUP	NAME																														
Crustacea	Hamatoscalpellum sanctipretense															0								1						1	1
	Harpacticoida	1				20										23														0	23
	Heterophoxus ellisi													2		5											1			1	6
	Hyperoche sp										1					1														0	1
	Ilyarachna sp 1			1										1	2	8														0	8
	Jassa slatteryi										5			2		8														0	8
	Lamprops sp															1														0	1
	Lebbeus washingtonianus															0		3												3	3
	Leptophoxus cf fultanus	2														6														0	6
	Leptostylis calva				2				2				1			7														0	7
	Leucon sp 1												1			1														0	1
	Leucon sp 2															1														0	1
	Leucon sp H															1														0	1
	Leucon subnasica															1									1					1	2
	Leuconidae											1				1														0	1
	Lithodes couesi															0		2												2	2
	Malacostraca							1								1														0	1
	Monoculodes glyconica															4														0	4
	Monoculodes latissimanus															0											2			2	2
	Mononuclodes necopinus				1									1	1	3														0	3
	Munidopsis quadrata															0								1						1	1
	Munnidae sp 1											1				1														0	1
	Munnopsurus sp A													1		1														0	1
	Nicippe tumida															0											1			1	1
	Notoxenodes sp															1														0	1
	Oediceriopsis elsula		1													1														0	1
	Oedicerotidae sp 1			1												1														0	1
	Orchomene sp													1		1														0	1
	Ostracoda sp 1											1		2		6														0	6
	Paraleptognathia gracilis							1								1														0	1
	Paraphoxus "oculatus"			1												3														0	3
	Philomedidae sp 1			2												13														0	13
	Podoceridae sp 1				1											1														0	1
	Prachynella lodo	1														1														0	1
	Procampylaspis sp														1	4														0	4
	Protellidae sp 1			1						1	1					3														0	3
	Pseudoharpinia excavata															0									1					1	1
	Pseudomma oculospinum															0								1						1	1
	Rhepoxinius cf. daboius															1														0	1
	Scoloura phillipsi													1		1														0	1
	Spirontocaris sp															1														0	1
	Stenothoidae										1					1														0	1
	Synaptotanais notabilis										2	1		1		4														0	4
	Tanaidacea sp 1			1		3		5			2					23														0	23
	Tanaidae									1						1														0	1
	Tanaopsis cadieni															2														0	2
	Tritella tenuissima					1			1		6					8								1						1	9
	Typhlotanaidae sp 2													1		1														0	1
	Typhlotanais cf crassus							3			1	3				10														0	10
	Typhlotanais williamsi										2	2	2			18														0	18
	Urothoe varvarini															3														0	3
Crustacea Total		5	3	13	16	25	1	10	4	4	26	13	4	18	11	272	0	8	0	0	7	2	5	4	8	2	8	0	0	44	316
Echinoderm	Allocentrotus fragilis															0		1												1	1
	Amphipholis pugetana								1						7	9											7			7	16
	Amphipholis sp														3	16										1	2			3	19
	Amphiura arcystata									1						1														0	1
	Amphiuridae														3	7														0	7
	Asteroid															0									1					1	1
	Asteronyx longifissus															0				1										1	1
	Brisingid asteroid sp 1															0			1											1	1
	Brissopsis pacifica															1		1					1				2			4	5

Appendix A: Benthic Infauna Data

INFAUNA

GROUP	SAMPLE ID	2SX0082	2SX0083	2SX0084	2SX0085	2SX0086	2SX0087	2SX0088	2SX0089	2SX0090	2SX0091	2SX0092	2SX0093	2SX0094	2SX0095	2SX0096	2SX0097	2SX0098	2SX0099	2SX0100	2SX0101	2SX0102	4SX001.05	4SX001.06	4SX001.07	4SX001.08	4SX003.05	4SX003.06	4SX003.07	4SX003.08	4SX004.05	4SX004.06
	STATION	1-8	1-8	1-2	1-2	1-3	1-3	4-6	4-6	1-6	1-6	1-7	1-7	1-7	1-7	1-1	1-1	1-4	1-4	1-5	1-5	1-1	4-5	4-5	4-5	4-5	4-2	4-3	4-4	4-5	4-3.1	4-3.1
	SCREEN SIZE	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0
	REPLICATE	1	1	1	1	1	1	1	1	1	1	1	1	2	2	1	1	1	1	1	1	1	1	1	2	2	1	1	2	2	1	1
GROUP	NAME																															
Echinoderm	Chiridota sp																															
	Laetmophasma ? sp																															
	Luidiaster sp																															
	Ophiacantha phragma					1																										
	Ophiosphalma jolliense											1																				
	Ophiura leptoctenia																															
	Ophiura sp																															1
	Ophiuroidea		3		2			6	5		1	2	2								3				6	1	1					2
	Zoroaster evermanni																															
Echinoderm Total		0	3	0	2	1	0	6	5	0	1	4	2	0	0	2	0	0	0	3	3	0	0	1	14	4	1	1	0	0	0	3
Minor phyla	Chelysoma columbianum																															
	Clavipora occidentalis (colonial)																															
	Clavipora sp (colonial)																															
	Ectoprocta													1																		
	Lineidae			1								1			1	1																
	Lineus sp																															
	Nemertea				2																						1			1		
	Notoxenodes sp																															
	Paleonemertea	1										1					1	1			1											
	Porifera (colonial)																									1						
	Sipuncula																															
	Tealianthus sp A																											1		1		
	Tubulanus polymorphus/pellucidus		2		1																											
	Tubulnus nr. nothus																					1										
Minor phyla Total		1	2	1	3	0	0	0	0	0	0	2	0	1	1	1	1	1	0	0	1	1	0	0	0	1	1	1	0	2	0	0
Mollusca	Adontorhina cyclia																															
	Anatoma crispata																															
	Axinodon redondoensis																											1			4	
	Axinopsida serricata																															
	Bathybembix bairdi																															
	Bivalvia										4	2		1	1																	
	Borsonella omphale																									1						
	Bullomorpha sp A (MEC)																															
	Cadulus sp																															
	Colus aphelus																													1		
	Cuspidariidae																															
	Delectopecten sp											1																				
	Ennucula tenuis																															1
	Gadilidae																												3		2	
	Hanleya sp																							1		1						
	Lepidozona sp					1																										
	Limopsis panamensis																											1				
	Mollusca			1						1																					1	
	Musculus sp							1																								
	Neilonella ritteri	1			1						1			1		1			1		1											
	Neptuneidae																							1								
	Nuculana leonina																															1
	Odostomia sp																1															
	Oenopota sp																									1						
	Philine auriformis									1		1				1																
	Philine sp																											1				
	Pleurotomella herminea																															
	Polyschides tolmiei							1	1																							
	Scaphopoda									1									2													
	Solariella nuda							1																								
	Spathoderma californica				2		1	1	1		3		5		1		1				1				1				3		3	
	Trochidae																														1	
	Yoldiella sp														1								1	1	1							
Mollusca Total		1	0	1	3	1	1	4	2	3	8	4	5	2	3	2	2	0	3	0	2	0	1	3	2	3	0	3	6	1	11	2
Grand Total		22	29	13	40	24	5	34	16	26	44	33	48	22	27	12	27	29	15	22	20	27	13	11	48	19	38	24	20	13	19	21

Appendix A: Benthic Infauna Data

		INFAUNA																MISCELLANEOUS INVERTEBRATES														
	SAMPLE ID	4SX004.07	4SX004.08	4SX005.05	4SX005.06	4SX005.07	4SX005.08	5SX001.05	5SX001.06	5SX001.07	5SX001.08	5SX002.05	5SX002.06	5SX003.05	5SX003.06	Total Infauna	ADCP	MISC	Misc Inverts 1	Misc Inverts 2	NWFT	WFT Crab	4-1	4-2	4.3.1	4-4	4-5	4-7	4-8	Total Misc. Inverts	Grand Total
	STATION	4-3.1	4-3.1	4-1	4-1	4-1	4-1	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-3.5	1-3.6	4-6	4-6																
	SCREEN SIZE	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	1.0	0.5	0.5	1.0	0.5	1.0																
	REPLICATE	2	2	1	1	2	2	1	1	2	2	1	1	1	1																
GROUP	NAME																														
Echinoderm	Chiridota sp														1	1														0	1
	Laetmophasma ? sp															0		1												1	1
	Luidiaster sp															0				1										1	1
	Ophiacantha phragma															1		1												1	2
	Ophiosphalma jolliense														1	2		1							1		2			4	6
	Ophiura leptoctenia															0							4	3	8					15	15
	Ophiura sp				3											4							4							4	8
	Ophiuroidea		1	3	2		1	4	5			1	2	8	7	68														0	68
	Zoroaster evermanni															0		1												1	1
Echinoderm Total		0	1	3	5	0	1	4	6	1	0	1	2	8	22	110	0	6	1	2	0	0	9	3	10	1	13	0	0	45	155
Minor phyla	Chelysoma columbianum															0							2							2	2
	Clavipora occidentalis (colonial)					1										1														0	1
	Clavipora sp (colonial)				1											1														0	1
	Ectoprocta															1														0	1
	Lineidae															4														0	4
	Lineus sp									1					1	2											1			1	3
	Nemertea						1				1					6														0	6
	Notoxenodes sp								1							1														0	1
	Paleonemertea															5														0	5
	Porifera (colonial)															1							1				1	1		3	4
	Sipuncula											1				1														0	1
	Tealianthus sp A															2														0	2
	Tubulanus polymorphus/pellucidus									1						4														0	4
	Tubulnus nr. nothus															1														0	1
Minor phyla Total		0	0	0	1	1	1	0	1	2	1	1	0	0	1	30	0	0	0	0	0	0	3	0	0	0	2	1	0	6	36
Mollusca	Adontorhina cyclia				1					1					1	3														0	3
	Anatoma crispata				1											1														0	1
	Axinodon redondoensis															5														0	5
	Axinopsida serricata														1	1														0	1
	Bathybembix bairdi														2	2														0	2
	Bivalvia															8														0	8
	Borsonella omphale															1														0	1
	Bullomorpha sp A (MEC)														1	1														0	1
	Cadulus sp	1		5		4	1	4								15														0	15
	Colus aphelus															1														0	1
	Cuspidariidae				1											1														0	1
	Delectopecten sp			1			1									3														0	3
	Ennucula tenuis															1														0	1
	Gadilidae															5														0	5
	Hanleya sp															2										2	4			6	8
	Lepidozona sp															1														0	1
	Limopsis panamensis		1													2							2							2	4
	Mollusca										1			1		5	1													1	6
	Musculus sp															1														0	1
	Neilonella ritteri															7														0	7
	Neptuneidae							1								2														0	2
	Nuculana leonina	3		3	1	2										10								1	1					2	12
	Odostomia sp															1														0	1
	Oenopota sp												3			4														0	4
	Philine auriformis															3														0	3
	Philine sp				1											2									1					1	3
	Pleurotomella herminea															0								1						1	1
	Polyschides tolmiei								1						2	5								1	2		1			4	9
	Scaphopoda															3														0	3
	Solariella nuda															1														0	1
	Spathoderma californica			9		6		1				2				41														0	41
	Trochidae															1														0	1
	Yoldiella sp					1		2	1		1					9														0	9
Mollusca Total		4	1	18	5	13	2	8	2	1	2	2	3	1	7	148	1	0	0	0	0	0	2	3	4	2	5	0	0	17	165
Grand Total		34	16	55	39	67	17	29	21	18	50	28	29	75	51	1290	18	19	2	2	8	2	40	20	46	11	46	1	2	217	1507

Appendix B. Global Marine Sediment PCB Concentrations

Location/Depth	PCB Concentration(s)/Type	Reference
Santa Barbara Basin (West Coast USA) (595m) Southern California Bight	103 ppb (mid 1960s); as Aroclor 1254	Hom et al., 1974
Open Mediterranean Sea (1000-3840m; 0-1cm)	0.8-8.9 ppb dry wt (1981); as Phenoclor DP6	Geyer et al., 1984
Open Mediterranean Sea (2900m; 0-1cm)	1.9 ppb dry wt (1983); as Aroclor 1260	Burns and Villeneuve 1987
Open Western Mediterranean Sea	0.8-33 ng/gm dry wt; as Aroclor 1260	Tolosa et al., 1997
Gulf of Maine USA (132-330m)	10-80 ppb dry wt (1983); total PCBs	Larsen et al., 1985
North Sea (200-300m)	1.1 to 3.7 ppb dry wt (1980-85); PCB type not stated	Basford and Eleftheriou, 1988
Gulf of Alaska (27 – 249m)	2 ppb dry wt (1990); total PCBs	Iwata et al., 1994
Bering Sea (27 - 249m)	0.13 ppb dry wt (1990); total PCBs	Iwata et al., 1994
Chukchi Sea (27 – 249m)	0.14 ppb dry wt (1990); total PCBs	Iwata et al., 1994
Western Baltic Sea	0.12-11.4 ppb dry wt (1996) Sum of 23 congeners	Dannenberger et al., 1996
Bering – Chukchi Seas Depth not reported	8.5-16.5 ng/g dry wt. (1993) Sum of 113 congeners	Strachan et al., 2001
NW Mediterranean Sea (>1000 - >2000m)	0.1-1.3 ppb dry wt (1995) range of 12 individual congeners	Tolosa et al., 1995
Central Adriatic Sea	0.3-6.6 ppb dry wt (1993) As Aroclor 1254	Dujmov et al., 1993
Northern Adriatic Sea	3-25 ppb dry wt (1993) Total PCBs	Caricchia et al., 1993
Eastern Baltic Sea (approx. 50m)	2-14 ppb dry wt (1995) Total PCBs	Van Bavel et al., 1995
Adriatic Sea (100-138m)	9.22-24.67 ppb dry wt (1993) Total PCBs	Galassi et al., 1993
Southern California Rural Coastal Shelf (60m)	5-39 ppb dry wt (1977-85) Total PCBs	NOAA 1991
Southern California Coastal Islands	0.28-1.2 ppb dry wt (1985) Total PCBs	NOAA 1991

Appendix B: Global Marine Sediment PCB Concentrations

Location/Depth	**PCB Concentration(s)/Type**	**Reference**
Mid-Atlantic Bight USA	33-2063 ppb dry wt (1984-87) Total PCBs normalized to % fines	NOAA 1991
Pacific Northwest Coast California USA	5-44 ppb dry wt (1984-87) Total PCBs normalized to % fines	NOAA 1991
Southern California Bight California USA	17-977 ppb dry wt (1984-87)	NOAA 1991
Gulf of Mexico USA	1.4-322 ppb dry wt (1993-994) Total PCBs	Maruya et al., 1997

Appendix C. PCB Concentrations In Pelagic And Benthic Species Tissues

Species	Location	PCB Concentration	PCB Type	Reference
Plankton				
Zooplankton	Newfoundland	0.667 ppb wet weight (whole bodies)	sum of all ortho substituted congeners	Ray et al., 1999
Zooplankton	Newfoundland	85.7 ppb lipid wt (whole bodies)	sum of all ortho substituted congeners	Ray et al., 1999
Zooplankton	Arctic Canada	15.66-33.8 ppb dry wt (whole bodies)	sum of 89 congeners	Fisk et al., 2001
Coelenterates				
Sea Pansy	Gulf of Mexico	850 ppb wet wt (whole body tissue)	Aroclor 1254	Giam et al., 1971
Crustaceans				
Mesopelagic crustaceans	Atlantic Ocean	8.9-35 ppb wet wt (whole body tissue), 490-1040 ppb lipid wt		Harvey et al., 1971
Crustaceans	Suruga Bay, Japan	780-2800 ppb lipid wt (Whole body tissue)	Kenechlor mixture	Lee et al., 1997
Amphipods	Arctic Abyssal	4448-15093 ppb dry wt (whole body tissue)	Aroclor 1254	Hargrave et al., 1992
Amphipods	Northwest Territory, Canada	32-36 ppb wet wt (whole body)	sum of 47 congeners	Bright et al., 1995
Amphipods	Barents Sea	38-50 ppb lipid wt. (whole body)	sum of 28 congeners	Borga et al., 2001
Copepods	Barents Sea	44-46 ppb lipid wt. (whole body)	sum of 28 congeners	Borga et al., 2001
Euphausiids	Barents Sea	28-29 ppb lipid wt. (whole body)	sum of 28 congeners	Borga et al., 2001
Spiny lobster	Santa Catalina Island, California (1977)	10 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Lobster	Atlantic Coast USA	0.7-2.2 ppm dry wt (muscle tissue)	sum of 18 congeners	Gadbois et al., 1996

Appendix C: PCB Concentrations In Pelagic And Benthic Species Tissues

Species	**Location**	**PCB Concentration**	**PCB Type**	**Reference**
Ridgeback prawn	Santa Catalina Island, California (1976)	16 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Crabs	Gulf of Mexico	17 ppb wet wt (whole body tissue)	Aroclor 1254	Giam et al., 1971
Crab (*Macropipus tuberculatus*)	Western Mediterranean	16.5 ppb wet wt (whole body tissue)	sum of 7 congeners	Porte and Albaiges 1993
Lithodid crab	Suruga Bay, Japan	780 ppb lipid wt (muscle tissue)	Kenechlor mixture	Lee et al., 1997
Lithodid Crab	Suruga Bay, Japan	1000-1100 ppb lipid wt (misc organs)	Kenechlor mixture	Lee et al., 1997
Glyphocrangon shrimp	Suruga Bay, Japan	930 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Stout red shrimp	Suruga Bay, Japan	930 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Botan shrimp	Suruga Bay, Japan	1500 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Japanese lobster	Suruga Bay, Japan	1600 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Giant red shrimp	Suruga Bay, Japan	510 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Indian Ocean lobsterette	Suruga Bay, Japan	1100 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Jack-knife shrimp	Suruga Bay, Japan	1100 ppb lipid wt (whole body tissue); as Kenachlor mixture		Takahashi et al., 1998
Lithodid crab	Suruga Bay, Japan	780 ppb lipid wt (muscle tissue)	Kenachlor mixture	Takahashi et al., 1998
Lithodid crab	Suruga Bay, Japan	1100 ppb lipid wt (other organ tissue)	Kenachlor mixture	Takahashi et al., 1998
Lithodid crab	Suruga Bay, Japan	1000 ppb lipid wt (hepatopancreas)	Kenachlor mixture	Takahashi et al., 1998
Lithodid crab	Suruga Bay, Japan	1100 ppb lipid wt (gill tissue)	Kenachlor mixture	Takahashi et al., 1998
Isopoda	Suruga Bay, Japan	2800 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Echinoderms				
Colonial tunicate	Gulf of Mexico	139 ppb wet wt (whole body tissue)	Aroclor 1254	Giam et al., 1971

Appendix C: PCB Concentrations In Pelagic And Benthic Species Tissues

Species	**Location**	**PCB Concentration**	**PCB Type**	**Reference**
Sea star (*Acanthaster planci*)	South Pacific Islands	0.01-0.21 ppm wet wt (gonad tissue)		McCloskey and Deubert 1973
Sea stars and urchins	Suruga Bay, Japan	1400-1900 ppb lipid wt (Whole body tissue)	Kenechlor mixture	Lee et al., 1997
Sea urchin (*Strongylocentrotus droebachiensis*)	Northwest Territory, Canada	<1.0-210 ppb wet wt (whole body)	sum of 47 congeners	Bright et al., 1995
Sea cucumber (Holothurian)	Suruga Bay, Japan	380 ppb lipid wt (whole body tissue)	Kenechlor mixture	Lee et al., 1997
Sea cucumber (Holothurian)	Atlantic Ocean	0-0.5 ppb wet wt (whole body)		Grassle, et al., 1986
Sun starfish	Suruga Bay, Japan	1400 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Goniasterid sea star	Suruga Bay, Japan	1500 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Echinothurid sea urchin	Suruga Bay, Japan	1900 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Sea cucumber (Holothurian)	Suruga Bay, Japan	380 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Molluscs				
Mussel (*Mytilus edulis*)	U.S. National Grand Mean	0.015 ppm wet wt (whole body)	total PCBs	NOAA 1988
Mussel (*Mytilus galloprovincialis*)	Western Mediterranean	6.6 ppb wet wt (whole body tissue)	sum of 7 congeners	Porte and Albaiges, 1993
Mussel (*Mytilus* sp.)	Western Mediterranean	3-2700 ppb wet wt (whole body assumed)	Aroclor 1254 from 1972-1990	Tolosa et al., 1997
Mytilus californianus	Channel Islands, California	6-69 ppb wet wt (whole tissue)	Aroclor 1254	NOAA 1991
Mytilus californianus	Channel Islands, California	6-110 ppb dry wt (whole tissue)	Aroclor 1254	NOAA 1991
Mytilus californianus	Channel Islands, California (1986)	125 ppb wet wt (whole tissue)	Aroclor 1254	NOAA 1991
Mytilus californianus	Channel Islands, California (1988)	17 ppb wet wt (whole tissue)	Aroclor 1254	NOAA 1991
Scallop (*Hinnites* sp.)	Santa Catalina Island, California (1973-74)	3 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Scallop (*Hinnites sp.*)	Cortez Bank, California (1975)	7 ppb wet wt (edible tissue)	total PCBs	NOAA 1991

Appendix C: PCB Concentrations In Pelagic And Benthic Species Tissues

Species	Location	PCB Concentration	PCB Type	Reference
Cephalopods	Suruga Bay, Japan	1900 ppb lipid wt (whole body tissue)	Kenechlor mixture	Lee et al., 1997
Cuttlefish	Suruga Bay Japan	580 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Octopus	Suruga Bay, Japan	2100 ppb lipid wt (viscera tissue)	Kenachlor mixture	Takahashi et al., 1998
Octopus	Suruga Bay, Japan	670 ppb lipid wt (muscle tissue)	Kenachlor mixture	Takahashi et al., 1998
Japanese common squid	Suruga Bay, Japan	1900 ppb lipid wt (whole body tissue)	Kenachlor mixture	Takahashi et al., 1998
Squid	Southern hemisphere oceans	<10-25 ppb wet wt (liver tissue)	total PCB	Yamada et al., 1997
Squid	Northwestern Atlantic	110 ppb wet wt (liver tissue)	total PCB	Yamada et al., 1997
Market squid	Southern California coast (1980-81)	10 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Squid	Southern hemisphere	10-25 ppb wet wt (liver tissue)	Kenachlor mixture	Yamada et al., 1997
Squid	Japanese waters	50-310 ppb wet wt (liver tissue)	Kenachlor mixture	Yamada et al., 1997
Squid	Northwestern Atlantic	110 ppb wet wt (liver tissue)	Kenachlor mixture	Yamada et al., 1997
Squid	Bay of Biscay	280 ppb wet wt (liver tissue)	Kenachlor mixture	Yamada et al., 1997
Fish				
Mesopelagic fishes	North Atlantic	1-170 ppb wet wt; 40-38,000 ppb lipid wt		Grassle, et al., 1986
Mesopelagic fishes	Gulf of Mexico	2-926 ppb wet wt		Grassle, et al., 1986
Mesopelagic fish	Atlantic Ocean	10-59 ppb wet wt (whole body tissue); 1460-7300 ppb lipid wt		Harvey et al., 1971
Mesopelagic fish	Western Mediterranean	1.0-2.1 ppb wet wt. (muscle); 166-327 ppb wet wt (liver); 2.9-20.3 ppb wet wt (gills); 5.1-8.8 ppb wet wt (digestive tract)		Garcia et al., 2000

Appendix C: PCB Concentrations In Pelagic And Benthic Species Tissues

Species	Location	PCB Concentration	PCB Type	Reference
Mesopelagic fish	Western North Atlantic (300-1500 m)	2.3-5.7 ppb wet wt (whole body less liver)	Aroclor 1254	Stegeman et al., 2001
Myctophid fish	Mediterranean	50 ppb dry wt. (whole body)	Phenoclor DP-5	CRC, 1990
Myctophid fish	Western Mediterranean	240 ppb dry wt. (whole body)	Aroclor equivalent	Burns and Villeneuve, 1987
Myctophid fish	Central/Eastern Mediterranean	27-89 ppb dry wt. (whole body)	Phenoclor DP-5	Fowler and Elder, 1978
Myctophid fish	Open western North Pacific	20-370 ppb lipid wt. (whole body)	sum of 117 congeners	Takahashi et al., 2000
Mora moro	Northwest ern Mediterranean Sea (1000 m)	24 ppb wet wt. (muscle)	sum of 22 congeners	Sole et al., 2001
Mullus sp.	Western Mediterranean	17-1529 ppb wet wt (muscle tissue assumed)	Aroclor 1254 (from 1977-1990)	Tolosa et al., 1997
Antimora rostrata (rattail fish)	Atlantic Ocean	3.8-12.5 ppm lipid wt (liver tissue)	total PCBs	Barber and Walen, 1979
Rattail fish	Atlantic Ocean	0.5 ppb wet wt (muscle tissue); 340 ppb wet wt (liver tissue)		Grassle, et al., 1986
Rattail fish, *Coryphaenoides armatus*	Northwestern Atlantic	0.316-5.38 ppm wet wt (liver tissue)	Aroclor 1254	Grassle, et al., 1986
Rattail fish, *Coryphaenoides armatus*	Northwestern Atlantic	4;0.027-0.738 ppm wet wt (liver tissue)	Aroclor 1268	Grassle, et al., 1986
Rattail fish, *Coryphaenoides armatus*	Northwestern Atlantic	0.37-6.14 ppm wet wt (liver tissue)	total PCB	Grassle, et al., 1986
Antimora rostrata	Northwestern Atlantic	2.39-6.71 ppm wet wt (liver tissue)	Aroclor 1254	Grassle, et al., 1986
Antimora rostrata	Northwestern Atlantic	2.86-7.14 ppm wet wt (liver tissue)	total PCB	Grassle, et al., 1986
Coryphaenoides rupestris	Northwestern Atlantic	0.577-0.645 ppm wet wt (liver tissue)	Aroclor 1254	Grassle, et al., 1986
Coryphaenoides rupestris	Northwestern Atlantic	0.65-0.78 ppm wet wt (liver)	total PCB	Grassle, et al., 1986
Rattail fish, *Coryphaenoides armatus*	Northwestern Atlantic	215-3920 ppb wet wt (liver tissue)	total of 24 congeners	Grassle, et al., 1986
Rattail fish	Atlantic Coast USA	1.7-33.8 ppm dry wt (liver tissue)	sum of 18 congeners	Gadbois et al., 1996

Appendix C: PCB Concentrations In Pelagic And Benthic Species Tissues

Species	Location	PCB Concentration	PCB Type	Reference
Antimora rostrata	Cape Hatteras, North Carolina (2500 m)	3.8-12.5 ppm lipid wt	total PCBs	Risebrough et al., 1976
Anoplopoma fimbria (Sablefish)	Farallon Islands, California	56 ppm lipid wt (liver tissue); 7 ppm wet wt	total PCBs	Melzian et al., 1987
Sablefish (*Anoplopoma fimbria*)	Santa Monica Bay, California (1977)	11541 ppb wet wt (liver tissue)	Total PCBs	NOAA 1991
Microstomus pacificus (Dover sole)	Farallon Islands, California	16 ppm lipid wt (liver tissue); 1.6 ppm wet wt	total PCBs	Melzian et al., 1987
Brotulid fish	Atlantic Ocean	36 ppb wet wt (muscle tissue); 1200 ppb wet wt (liver tissue)		Grassle, et al., 1986
Black scabbard fish (*Aphanopus carbo*)	North Atlantic	5.8 ppm (liver tissue)	Aroclor 1260	Grassle, et al., 1986
Black Scabbard Fish (*Aphanopus carbo*)	North Atlantic	5800 ppm lipid wt (liver tissue)	sum of congeners 28, 52, 101, 138, 153, 180	Ballschmitter et al., 1997
Black Scabbard Fish (*Aphanopus carbo*)	Eastern North Atlantic	5800 ppb lipid wt (liver tissue)	total PCB	Kramer et al., 1984
Open water fishes	Gulf of Mexico	53-150 ppb wet wt (whole body tissue)	Aroclor 1254	Giam et al., 1971
Open water fishes	Gulf of Mexico	20-36 ppb wet wt (whole body tissue)	Aroclor 1260	Giam et al., 1971
Grouper	Gulf of Mexico	3-220 ppb wet wt (muscle tissue)	Aroclor 1260	Giam et al., 1971
Mullet (*Mullus barbatus*)	Western Mediterranean	27.3 ppb wet wt (muscle tissue)	sum of 7 congeners	Porte and Albaiges 1993
Mullet (*Mullus surmuletus*)	Western Mediterranean	30.1 ppb wet wt (muscle tissue)	sum of 7 congeners	Porte and Albaiges 1993
Tunafish (*Thunnus thynnus*)	Western Mediterranean	15.1 ppb Wet wt (muscle tissue)	sum of 7 congeners	Porte and Albaiges 1993
Scorpionfish	Suruga Bay, Japan	1200 ppb lipid wt (whole body tissue)	Kenechlor mixture	Lee et al., 1997
Scorpionfish	Suruga Bay, Japan	1200 ppb lipid wt (whole body tissue)	Kenechlor mixture	Takahashi et al., 1998
Dory	Suruga Bay, Japan	780 ppb lipid wt (whole body tissue)	Kenechlor mixture	Takahashi et al., 1998
Green-eye	Suruga Bay, Japan	720 ppb lipid wt (whole body tissue)	Kenechlor mixture	Takahashi et al., 1998

Appendix C: PCB Concentrations In Pelagic And Benthic Species Tissues

Species	Location	PCB Concentration	PCB Type	Reference
Dogfish shark	Suruga Bay,Japan	1000 ppb lipid wt (liver tissue)	Kenechlor mixture	Takahashi et al., 1998
Argentine fish	Suruga Bay, Japan	450 ppb lipid wt (whole body tissue)	Kenechlor mixture	Takahashi et al., 1998
Cusk eel	Suruga Bay, Japan	1900 ppb lipid wt (whole body tissue)	Kenechlor mixture	Takahashi et al., 1998
Four horn sculpin	Northwest Territory, Canada	7.3-230 ppb wet wt (whole body - less liver)	sum of 47 congeners	Bright et al., 1995
Short horn sculpin	Northwest Territory Canada	1.4-38 ppb wet wt (whole body - less liver)	sum of 47 congeners	Bright et al., 1995
Four horn sculpin	Northwest Territory, Canada	6.8-1300 ppb wet wt (liver)	sum of 47 congeners	Bright et al., 1995
Short horn sculpin	Northwest Territory, Canada	5.5-220 ppb wet wt (liver)	sum of 47 congeners	Bright et al., 1995
Tilefish	Middle Atlantic Bight, Lydonia Canyon	1.7-28.7 ppb dry wt (muscle tissue)	sum of 20 congeners	Steimle et al., 1990
Tilefish	Middle Atlantic Bight, Lydonia Canyon	12.1-322.9 ppb dry wt (liver tissue)	sum of 20 congeners	Steimle et al., 1990
Tilefish	Middle Atlantic Bight, Lydonia Canyon	10.3-282.9 ppb dry wt (gonad tissue)	sum of 20 congeners	Steimle et al., 1990
Tilefish	Middle Atlantic Bight, Lydonia Canyon	0.3-15 ppb dry wt (muscle tissue)	sum of 20 congeners	Steimle et al., 1990
Tilefish	Middle Atlantic Bight, Hudson Canyon	3.3-82.3 ppb dry wt (Liver tissue)	sum of 20 congeners	Steimle et al., 1990
Tilefish	Middle Atlantic Bight, Hudson Canyon	4.2-45.5 ppb dry wt (gonad tissue)	sum of 20 congeners	Steimle et al., 1990
Tilefish (*Lopholatilus chamaeleonticeps*)	Atlantic Coast, USA	0.2-0.73 ppm dry wt (liver tissue)	total PCBs	Steimle et al., 1990
Lizard fish	Gulf and Atlantic Coasts, USA	10-80 ppb dry wt (muscle tissue)	Aroclor 1254	Boehm and Hirtzer 1982
Lizard fish	Gulf and Atlantic Coasts, USA	100-2130 ppb dry wt (liver tissue)	Aroclor 1254	Boehm and Hirtzer 1982
Fish (*Lycodes frigidus*)	Arctic Abyssal	1300 ppb dry wt (liver tissue)	Aroclor 1254	Hargrave et al., 1992

Appendix C: PCB Concentrations In Pelagic And Benthic Species Tissues

Species	Location	PCB Concentration	PCB Type	Reference
Cod (*Gadus morhua*)	Northwest Atlantic Ocean	14-49 ppb wet wt (liver tissue)	Aroclor 1242	Hellou et al., 1993
Cod (*Gadus morhua*)	Northwest Atlantic Ocean	1 ppb wet wt (muscle tissue)	Aroclor 1254	Hellou et al., 1993
Cod (*Gadus morhua*)	Northwest Atlantic Ocean	86-260 ppb wet wt (liver tissue)	Aroclor 1254	Hellou et al., 1993
Cod (*Gadus morhua*)	Northwest Atlantic Ocean	3.2-9.0 ppb wet wt (ovary tissue)	Aroclor 1254	Hellou et al., 1993
Cod (*Gadus morhua*)	Northwest Atlantic Ocean	41-130 ppb wet wt (liver tissue)	Aroclor 1260	Hellou et al., 1993
Cod fish	Northwest Atlantic Ocean	154 ppb wet wt (liver tissue)	Aroclor 1254	Hellou et al., 1993
Cod fish	Northwest Atlantic Ocean	5.1 ppb wet wt (gonad tissue)	Aroclor 1254	Hellou et al., 1993
Cod fish	Barents Sea	205 ppb lipid wt. (liver)	sum of 28 congeners	Borga et al., 2001
Polar Cod fish	Barents Sea	108 ppb lipid wt. (liver)	sum of 28 congeners	Borga et al., 2001
Haddock	Barents-Greenland Seas	16 ppb dry wt (muscle tissue)	Aroclor 1254	Ali et al., 1997
Long rough dab	Barents-Greenland Seas	79 ppb dry wt (muscle tissue)	Aroclor 1254	Ali et al., 1997
Redfish	Barents-Greenland Seas	74 ppb dry wt (muscle tissue)	Aroclor 1254	Ali et al., 1997
Atlantic cod	Barents-Greenland Seas	461ppb dry wt (muscle tissue)	Aroclor 1254	Ali et al., 1997
Greenland halibut	Barents-Greenland Seas	249 ppb dry wt (muscle tissue)	Aroclor 1254	Ali et al., 1997
Capelin	Barents-Greenland Seas	170 ppb dry wt (muscle tissue)	Aroclor 1254	Ali et al., 1997
Capelin	Barents-Greenland Seas	120 ppb lipid wt (muscle tissue)	Aroclor 1254	Ali et al., 1997
Halibut	Barents-Greenland Seas	585 ppb dry wt (muscle tissue)	Aroclor 1254	Ali et al., 1997
Haddock, Redfish, Halibut, Long rough dab, Greenland halibut	Barents-Greenland Seas	200-300 ppb lipid wt (muscle tissue)	Aroclor 1254	Ali et al., 1997
Atlantic cod	Barents-Greenland Seas	500 ppb lipid wt (muscle tissue)	Aroclor 1254	Ali et al., 1997
Flounder	Gulf and Atlantic Coasts, USA	10-70 ppb dry wt (muscle tissue)	Aroclor 1254	Boehm and Hirtzer 1982

Appendix C: PCB Concentrations In Pelagic And Benthic Species Tissues

Species	Location	PCB Concentration	PCB Type	Reference
Flounder	Gulf and Atlantic Coasts, USA	40-980 ppb dry wt (liver tissue)	Aroclor 1254	Boehm and Hirtzer 1982
Turbot	Northwest Atlantic Ocean	30.4 ppb wet wt (liver tissue)	Aroclor 1254	Hellou et al., 1995
Turbot	Northwest Atlantic Ocean	24.4 ppb wet wt (muscle tissue)	Aroclor 1254	Hellou et al., 1995
Turbot	Northwest Atlantic Ocean	1.2 ppb wet wt (gonad tissue)	Aroclor 1254	Hellou et al., 1995
American plaice	Northwest Atlantic Ocean	16.2-70.2 ppb wet wt (liver tissue)	Aroclor 1254	Hellou et al., 1995
American plaice	Northwest Atlantic Ocean	11.2-14.8 ppb wet wt (muscle tissue)	Aroclor 1254	Hellou et al., 1995
American plaice	Northwest Atlantic Ocean	1.0-2.8 ppb wet wt (gonad tissue)	Aroclor 1254	Hellou et al., 1995
Yellowtail flounder	Northwest Atlantic Ocean	4.7-9.5 ppb wet wt (liver tissue)	Aroclor 1254	Hellou et al., 1995
Yellowtail flounder	Northwest Atlantic Ocean	0.8-1.8 ppb wet wt (muscle tissue)	Aroclor 1254	Hellou et al., 1995
Yellowtail flounder	Northwest Atlantic Ocean	1.3-2.3 ppb wet wt (gonad tissue)	Aroclor 1254	Hellou et al., 1995
Longfin sanddab	San Clemente Island, California (1985)	4362 ppb wet wt (liver tissue)	total PCBs	NOAA 1991
Pacific sanddab	Santa Monica Bay, California (1977)	22316 ppb wet wt (liver tissue)	total PCBs	NOAA 1991
Dover sole	Santa Catalina Island, California (1971-74)	40-90 ppb wet wt (muscle tissue)	total PCBs	NOAA 1991
Shark	Northwest African Atlantic	39.4-4,723 ppb wet wt (liver)	sum of 24 congeners	Serrano et al., 2000
Shark	Northwest African Atlantic	45.0-5,410 ppb lipid wt (liver)	sum of 24 congeners	Serrano et al., 2000
Grouper	Northwest African Atlantic	387 ppb wet wt (liver)	sum of 24 congeners	Serrano et al., 2000
Grouper	Northwest African Atlantic	1,610 ppb lipid wt (liver)	sum of 24 congeners	Serrano et al., 2000
Velvet belly shark	Nordfjord, Norway	1470-3870 ppb lipid wt (liver tissue)	sum of 31 congeners	Berg et al., 1998

Appendix C: PCB Concentrations In Pelagic And Benthic Species Tissues

Species	Location	PCB Concentration	PCB Type	Reference
Ling cod	Nordfjord, Norway	2370-3520 ppb lipid wt (liver tissue)	sum of 31 congeners	Berg et al., 1998
Round nose grenadier	Nordfjord, Norway	2400 ppb lipid wt (liver tissue)	sum of 31 congeners	Berg et al., 1998
Tusk	Nordfjord, Norway	3910-82000 ppb lipid wt (liver tissue)	sum of 31 congeners	Berg et al., 1998
Jelly wolf fish	West Coast, Greenland	110 ppb Lipid wt (muscle tissue)	total PCBs	Berg et al., 1998
Blue hake	West Coast, Greenland	1156 ppb Lipid wt (muscle tissue)	total PCBs	Berg et al., 1998
Flounder	Northeastern Newfoundland	<10 ppb Wet wt (muscle tissue)	total PCBs	Ray et al., 1998
Hake	Gulf and Atlantic Coasts, USA	16-90 ppb dry wt (muscle tissue)	Aroclor 1254	Boehm and Hirtzer 1982
Hake	Gulf and Atlantic Coasts, USA	40-410 ppb dry wt (liver tissue)	Aroclor 1254	Boehm and Hirtzer 1982
Blue hake	Atlantic Coast, USA	1.7-33.8 ppm dry wt (liver tissue)	sum of 18 congeners	Gadbois et al., 1996
Pacific hake	Southern California Coast (1980-81)	12 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Pacific sardine	Southern California Coast (1980-81)	105 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Northern anchovy	Southern California Coast (1980-81)	8 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Pacific bonito	Southern California Coast (1980-81)	29 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Pacific mackerel	Southern California Coast (1980-81)	26 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Jack mackerel	Southern California Coast (1980-81)	17 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Swordfish	Southern California Coast (1980-81)	20 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Thresher shark	Southern California Coast (1980-81)	15 ppb wet wt (edible tissue)	total PCBs	NOAA 1991

Appendix C: PCB Concentrations In Pelagic And Benthic Species Tissues

Species	Location	PCB Concentration	PCB Type	Reference
Mako shark	Southern California Coast (1980-81)	35 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
White shark	Southern California Coast (1980-81)	41 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Blue shark	Southern California Coast (1980-81)	16 ppb wet wt (edible tissue)	total PCBs	NOAA 1991
Coastal fish	U.S. National Coastal Average	<0.1 ppm wet wt (whole body)	total PCBs	NOAA 1988
Coastal fish	U.S. National Coastal Average	0.1-0.2 ppm wet wt (muscle tissue)	total PCBs	NOAA 1988
Coastal flatfish	Southern California	0.01-1.11 ppm wet wt (muscle tissue)	total PCBs	NOAA 1988
Nearshore and benthic fish	U.S. National Status and Trends Median concentration	0.58 ppm wet wt (liver tissue)	total PCBs	NOAA 1988

Values expressed as wet wt. and lipid wt. may vary greatly

Appendix D. Ancillary Sediment Measurements

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-7	1-8	4-6
Year	1998	1998	1998	1998	1998	1998	1998	1998	1998
Sediment Grain Size (%)									
Gravel	32.3	41.6	25.7	25.3	25.6	16.1	21.2	20.3	9.8
Sand	47.2	45.4	57.0	49.1	55.4	39.4	59.2	60.9	74.4
Silt	8.6	6.5	9.0	13.6	9.3	16.3	9.7	8.5	8.8
Clay	11.8	6.4	8.3	12.0	9.7	28.3	10.0	10.3	6.9
Calcium Carbonate (%)	58	22.2		41.7	29.3	58.3	35.9	54.3	57.4
Total Organic Carbon (%)	5.06	1.84	9.08	2.75	2.32	4.93	2.41	4.43	2.57
AVS (µmol/g)									
Cadmium	0.01	0.01		0.01	0.01	0.01	0.01	0.03	<0.002
AVS	0.0028	0.0044		<0.0002	<0.0002	0.0016	0.0008	0.0268	0.0005
Copper	0.95	1.07		1.69	0.78	0.61	0.64	1.28	0.74
Lead	0.03	0.03		0.02	0.02	0.01	0.01	0.03	0.02
Nickel	0.37	0.28		0.41	0.03	0.51	0.48	0.31	0.54
Zinc	5.68	3.64		5.15	3.54	4.57	4.67	4.87	3.37

Station	4-6	1-3.5	1-5/6/7	4-1	4-2	4-3	4-3-1	4-5	4-8
Year	1999	1999	1999	1999	1999	1999	1999	1999	1999
Sediment Grain Size (%)									
Gravel			11.2	8.8	0.8		0.7	2.0	
Sand			52.1	50.0	64.0		64.0	74.8	
Silt			16.2	25.7	21.1		20.1	14.0	
Clay			20.6	15.5	14.1		15.2	9.2	
Calcium Carbonate (%)	55.4	40.7	69	44.8	53.8	55.9	56.5	52.6	27.7
Total Organic Carbon (%)	1.63	1.21	1.01	1.63	1.1	1.46	1.85	1.31	1.57
AVS (µmol/g)									
Cadmium	0.003	0.005	0.068	0.003	0.003	0.003	0.003	0.004	0.004
Copper	0.03	0.08	0.96	0.01	0.02	0.02	0.02	0.02	0.02
Iron	0.55	0.54	0.33	0.22	0.52	0.37	0.7	0.66	0.81
Lead	0.07	0.06	0.09	0.04	0.07	0.07	0.06	0.07	0.05
Nickel	0.15	0.13	0.23	0.13	0.2	0.17	0.14	0.23	0.13
Sulfide	0.4	0.23	0.21	0.3	0.24	0.25	0.1	0.21	0.13
Zinc	0.41	0.38	0.86	0.21	0.47	0.35	0.38	0.5	0.45

Appendix E. Water PCB Data

Station	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	4-2	4-2	4-2	4-2	4-5	4-5	4-5	4-5
Replicate	**1**	**2**	**3**	**4**	**1**	**2**	**3**	**4**	**1**	**2**	**3**	**4**
Year	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**
PCBs (µg/kg)												
114 - 2,3,4,4',5-Pentachlorobiphenyl	<0.32	<0.34	<0.36	<0.38	<0.32	<0.32	<0.32	<0.34	<0.33	<0.33	<0.34	<0.33
123 - 2',3,4,4',5-Pentachlorobiphenyl	<0.36	<0.38	<0.41	<0.43	<0.37	<0.37	<0.37	<0.39	<0.38	<0.37	<0.39	<0.37
157 - 2,3,3',4,4',5'-Hexachlorobiphenyl	<0.28	<0.3	<0.32	<0.34	<0.29	<0.29	<0.29	<0.3	<0.3	<0.3	<0.3	<0.3
167 - 2,3',4,4',5,5'-Hexachlorobiphenyl	<0.3	<0.31	<0.33	<0.35	<0.3	<0.3	<0.3	<0.32	<0.31	<0.31	<0.32	<0.31
189 - 2,3,3',4,4',5,5'-Heptachlorobiphenyl	<0.33	<0.35	<0.37	<0.39	<0.34	<0.34	<0.34	<0.35	<0.35	<0.34	<0.35	<0.34
2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	<0.1	<0.11	<0.12	<0.12	<0.11	<0.11	<0.11	<0.11	<0.11	<0.11	<0.11	<0.11
2,2',3,3',4,4',5,6-Octachlorobiphenyl	<0.27	<0.29	<0.31	<0.32	<0.28	<0.28	<0.28	<0.29	<0.28	<0.28	<0.29	<0.28
2,2',3,3',4,4',5-Heptachlorobiphenyl	<0.3	<0.31	<0.33	<0.35	<0.3	<0.3	<0.3	<0.32	<0.31	<0.31	<0.32	<0.31
2,2',3,3',4,4'-Hexachlorobiphenyl	<0.12	<0.12	<0.13	<0.14	<0.12	<0.12	<0.12	<0.12	<0.12	<0.12	<0.12	<0.12
2,2',3,4',5,5',6-Heptachlorobiphenyl	<0.3	<0.32	<0.34	<0.36	<0.31	<0.31	<0.31	<0.33	<0.32	<0.32	<0.33	<0.32
2,2',3,4,4',5',6-Heptachlorobiphenyl	<0.17	<0.18	<0.19	<0.2	<0.18	<0.18	<0.18	<0.18	<0.18	<0.18	<0.18	<0.18
2,2',3,4,4',5'-Hexachlorobiphenyl	<0.26	<0.27	<0.29	<0.3	<0.26	<0.26	<0.26	<0.27	<0.27	<0.26	<0.27	<0.26
2,2',3,4,4',5,5'-Heptachlorobiphenyl	<0.3	<0.32	<0.34	<0.36	<0.31	<0.31	<0.31	<0.33	<0.32	<0.32	<0.33	<0.32
2,2',3,4,4',6,6'-Heptachlorobiphenyl	<0.24	<0.25	<0.26	<0.28	<0.24	<0.24	<0.24	<0.25	<0.25	<0.24	<0.25	<0.24
2,2',3,4,5'-Pentachlorobiphenyl	<0.36	<0.38	<0.4	<0.42	<0.37	<0.36	<0.36	<0.38	<0.38	<0.37	<0.38	<0.37
2,2',3,5'-Tetrachlorobiphenyl	<0.31	<0.33	<0.35	<0.37	<0.32	<0.32	<0.32	<0.34	<0.33	<0.33	<0.34	<0.33
2,2',4,4',5,5'-Hexachlorobiphenyl	<0.12	<0.13	<0.14	<0.15	<0.13	<0.13	<0.13	<0.13	<0.13	<0.13	<0.13	<0.13
2,2',4,5'-Tetrachlorobiphenyl	<0.35	<0.37	<0.39	<0.41	<0.36	<0.36	<0.36	<0.37	<0.36	<0.36	<0.37	<0.36
2,2',4,5,5'-Pentachlorobiphenyl	<0.17	<0.18	<0.19	<0.2	<0.17	<0.17	<0.17	<0.18	<0.18	<0.18	<0.18	<0.18
2,2',5,5'-Tetrachlorobiphenyl	<0.14	<0.14	<0.15	<0.16	<0.14	<0.14	<0.14	<0.15	<0.14	<0.14	<0.15	<0.14
2,2',5-Trichlorobiphenyl	<0.3	<0.32	<0.34	<0.35	<0.31	<0.3	<0.3	<0.32	<0.31	<0.31	<0.32	<0.31
2,3',4,4',5-Pentachlorobiphenyl	<0.31	<0.32	<0.34	<0.36	<0.32	<0.31	<0.31	<0.33	<0.32	<0.32	<0.33	<0.32
2,3',4,4'-Tetrachlorobiphenyl	<0.3	<0.32	<0.34	<0.36	<0.31	<0.31	<0.31	<0.32	<0.32	<0.31	<0.32	<0.31
2,3,3',4,4',5-Hexachlorobiphenyl	<0.29	<0.31	<0.33	<0.34	<0.3	<0.3	<0.3	<0.31	<0.3	<0.3	<0.31	<0.3
2,3,3',4,4'-Pentachlorobiphenyl	<0.19	<0.2	<0.21	<0.22	<0.2	<0.19	<0.19	<0.2	<0.2	<0.2	<0.2	<0.2
2,4'-Dichlorobiphenyl	<0.2	<0.21	<0.22	<0.23	<0.2	<0.2	<0.2	<0.21	<0.21	<0.2	<0.21	<0.2
2,4,4'-Trichlorobiphenyl	<0.24	<0.26	<0.28	<0.29	<0.25	<0.25	<0.25	<0.26	<0.26	<0.26	<0.26	<0.26
3,3',4,4',5,5'-Hexachlorobiphenyl	<0.28	<0.29	<0.31	<0.33	<0.28	<0.28	<0.28	<0.3	<0.29	<0.29	<0.3	<0.29
3,3',4,4',5-Pentachlorobiphenyl	<0.31	<0.33	<0.35	<0.37	<0.32	<0.32	<0.32	<0.34	<0.33	<0.33	<0.34	<0.33
3,3',4,4'-Tetrachlorobiphenyl	<0.26	<0.28	<0.29	<0.31	<0.27	<0.27	<0.27	<0.28	<0.27	<0.27	<0.28	<0.27
Dichlorobiphenyl	<0.23	<0.25	<0.26	<0.28	<0.24	<0.24	<0.24	<0.25	<0.24	<0.24	<0.25	<0.24
Heptachlorobiphenyl	<0.23	<0.25	<0.26	<0.28	<0.24	<0.24	<0.24	<0.25	<0.24	<0.24	<0.25	<0.24
Hexachlorobiphenyl (Cl 6)	<0.23	<0.25	<0.26	<0.28	<0.24	<0.24	<0.24	<0.25	<0.24	<0.24	<0.25	<0.24
Monochlorobiphenyl	<0.23	<0.25	<0.26	<0.28	<0.24	<0.24	<0.24	<0.25	<0.24	<0.24	<0.25	<0.24
Nonachlorobiphenyl	<0.23	<0.25	<0.26	<0.28	<0.24	<0.24	<0.24	<0.25	<0.24	<0.24	<0.25	<0.24
Octachlorobiphenyl	<0.23	<0.25	<0.26	<0.28	<0.24	<0.24	<0.24	<0.25	<0.24	<0.24	<0.25	<0.24
Pentachlorobiphenyl	<0.23	<0.25	<0.26	<0.28	<0.24	<0.24	<0.24	<0.25	<0.24	<0.24	<0.25	<0.24
Tetrachlorobiphenyl	<0.23	<0.25	<0.26	<0.28	<0.24	<0.24	<0.24	<0.25	<0.24	<0.24	<0.25	<0.24
Trichlorobiphenyl	<0.23	<0.25	<0.26	<0.28	<0.24	<0.24	<0.24	<0.25	<0.24	<0.24	<0.25	<0.24

Appendix F. Sediment PCB And Aroclor Data

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-7	1-8	4-6	4-6
Year	1998	1998	1998	1998	1998	1998	1998	1998	1998	1999
PCBs (µg/kg)										
8 - 2,4'-Dichlorobiphenyl	<0.14	<0.09	<0.15	<0.11	<0.12	<0.16	<0.11	<0.15	<0.12	<0.06
18 - 2,2',5-Trichlorobiphenyl	<0.29	<0.2	<0.32	<0.23	<0.26	<0.33	<0.24	<0.31	<0.25	0.03
28 - 2,4,4'-Trichlorobiphenyl	<0.2	<0.13	<0.22	0.011	0.007	0.013	0.018	0.004	0.008	<0.08
44 - 2,2',3,5'-Tetrachlorobiphenyl	0.044	0.024	0.056	0.061	<0.09	0.072	0.87	0.027	0.026	<0.04
49 - 2,2',4,5'-Tetrachlorobiphenyl	<0.22	<0.15	0.38	<0.17	<0.19	0.44	0.4	0.38	0.36	0.032
52 - 2,2',5,5'-Tetrachlorobiphenyl	0.052	0.033	0.046	0.095	0.048	0.089	1.4	0.031	0.029	<0.04
66 - 2,3',4,4'-Tetrachlorobiphenyl	<0.1	0.016	<0.12	0.035	<0.09	<0.12	0.25	<0.11	0.017	0.041
77 - 3,3',4,4'-Tetrachlorobiphenyl	<0.16	<0.11	0.41	0.19	0.12	0.92	<0.13	0.99	0.78	<0.07
87 - 2,2',3,4,5'-Pentachlorobiphenyl	0.048	0.026	0.022	0.058	0.016	0.091	1.2	0.015	0.021	0.053
101 - 2,2',4,5,5'-Pentachlorobiphenyl	0.12	0.068	0.064	0.14	0.041	0.26	<0.07	0.025	0.046	0.081
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.064	0.04	0.036	0.062	0.025	0.15	0.57	<0.1	0.031	0.029
118 - 2,3',4,4',5-Pentachlorobiphenyl	<0.1	0.085	0.078	0.23	0.038	0.54	1.9	0.068	0.08	0.084
126 - 3,3',4,4',5-Pentachlorobiphenyl	0.053	0.059	<0.27	0.11	<0.22	<0.28	0.31	<0.26	<0.21	<0.1
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	<0.1	0.024	0.012	0.045	0.023	0.1	<0.09	<0.11	0.029	0.024
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	0.16	0.091	0.076	0.15	0.05	0.6	1.4	0.044	0.072	0.077
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	0.46	0.12	0.23	0.39	0.061	0.89	1.3	0.16	0.092	0.21
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	<0.22	0.09	<0.24	<0.17	<0.19	0.049	0.23	<0.24	0.01	0.012
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<0.22	<0.15	<0.24	<0.17	<0.19	<0.25	<0.18	0.013	<0.19	0.05
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	0.037	0.018	0.021	<0.33	0.005	0.18	<0.34	0.009	0.013	<0.17
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.14	0.14	0.15	0.48	0.037	0.79	<0.1	0.042	0.044	0.047
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	0.04	0.03	0.024	0.13	0.009	0.25	0.22	<0.24	0.006	<0.09
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	<0.22	0.1	0.12	0.13	0.1	0.15	1.7	0.13	0.22	0.051
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.077	0.11	0.14	0.41	0.017	0.72	<0.15	0.043	0.022	0.036
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	0.037	<0.06	<0.09	0.052	<0.08	0.11	0.068	0.014	<0.07	0.024
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	<0.07	0.072	0.028	0.32	<0.06	1.5	0.31	0.019	0.011	0.042
209 - Decachlorobiphenyl	0.41	0.058	0.076	0.13	0.022	2.6	1.4	<0.09	<0.07	0.018

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-6	1-6	1-7	1-7	1-7	1-8
Replicate	1	1	1	1	1	1	1	3	1	2	3	1
Year	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998
Aroclors (µg/kg)												
Total Aroclor 1221	<9	<9	<14	<10	<12	<15	<1.4	<27	<11	<0.92	<18	<14
Total Aroclor 1232	<9	<9	<14	<10	<12	<15	<1.4	<27	<11	<0.92	<18	<14
Total Aroclor 1242	<9	<9	<14	<10	<12	<15	<1.4	<27	<11	<0.92	<18	<14
Total Aroclor 1248	<9	<9	<14	<10	<12	<15	<1.4	<27	<11	<0.92	<18	<14
Total Aroclor 1254	<9	<9	<14	<10	<12	<15	1.9	<27	35	32	27	<14
Total Aroclor 1260	<9	<9	<14	<10	<12	<15	<1.4	<27	<11	25	15	<14
Total Aroclor 1262	<9	<9	<14	6.4	<12	<15	<1.4	<27	8.1	<0.92	<18	<14
Total Aroclor 1268	<9	<9	<14	<10	<12	15	<1.4	<27	<11	<0.92	<18	<14

Appendix F: Sediment PCB And Aroclor Data

Station	1-3.5	1-5/6/7	4-1	4-2	4-3	4-3-1	4-5	4-8	Cruise IV Control	Cruise V Control
Year	1999	1999	1999	1999	1999	1999	1999	1999	1999	1999
PCBs (µg/kg)										
8 - 2,4'-Dichlorobiphenyl	0.021	1.6	<0.06	<0.06	<0.06	<0.06	<0.06	<0.05	<0.04	<0.04
18 - 2,2',5-Trichlorobiphenyl	0.02	<0.27	0.032	0.077	0.047	0.034	0.026	<0.1	<0.09	<0.09
28 - 2,4,4'-Trichlorobiphenyl	<0.07	<0.18	<0.09	<0.09	<0.09	<0.09	<0.08	<0.07	<0.06	0.16
44 - 2,2',3,5'-Tetrachlorobiphenyl	0.29	<0.09	<0.04	<0.05	<0.04	<0.04	<0.04	<0.03	0.14	<0.03
49 - 2,2',4,5'-Tetrachlorobiphenyl	0.094	1.7	0.04	0.034	0.033	0.03	0.027	0.057	0.035	0.036
52 - 2,2',5,5'-Tetrachlorobiphenyl	0.4	0.043	<0.04	<0.04	<0.04	0.05	<0.04	<0.03	0.021	0.011
66 - 2,3',4,4'-Tetrachlorobiphenyl	0.6	0.44	0.1	0.085	0.12	0.15	0.02	0.036	0.019	0.029
77 - 3,3',4,4'-Tetrachlorobiphenyl	<0.06	<0.15	<0.07	<0.07	<0.07	0.021	0.072	<0.05	0.025	0.028
87 - 2,2',3,4,5'-Pentachlorobiphenyl	0.49	0.078	0.062	0.05	0.09	0.072	0.032	0.058	0.024	0.024
101 - 2,2',4,5,5'-Pentachlorobiphenyl	0.8	0.18	0.065	0.098	0.082	0.075	0.026	0.012	0.016	0.07
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.23	0.12	0.039	0.026	0.048	0.039	0.017	0.012	0.009	0.006
118 - 2,3',4,4',5-Pentachlorobiphenyl	0.71	0.35	0.12	0.13	0.11	0.11	0.036	0.042	0.013	0.024
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.09	<0.23	<0.11	<0.11	<0.11	<0.11	<0.1	<0.08	<0.08	<0.08
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	0.21	<0.1	0.048	0.044	0.038	0.049	0.007	0.007	0.007	<0.03
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	0.9	0.19	0.11	0.081	0.1	0.091	0.029	0.043	0.011	0.007
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	0.95	0.21	0.18	0.12	0.17	0.16	0.1	0.066	0.025	0.02
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	0.11	0.085	<0.1	0.012	0.016	0.013	<0.09	0.01	<0.07	<0.07
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	0.072	<0.2	0.075	0.095	0.083	0.072	0.12	0.04	0.009	0.009
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	0.069	<0.39	<0.19	<0.19	<0.19	0.018	<0.17	<0.14	<0.13	<0.13
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.34	0.18	0.25	0.07	0.059	0.16	0.031	0.019	0.012	0.018
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	0.11	0.21	0.067	0.04	0.033	0.054	<0.09	<0.07	<0.07	<0.07
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	0.045	0.2	0.033	0.053	0.037	0.041	0.041	0.018	0.029	0.04
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.35	0.18	0.49	0.055	0.066	0.26	0.024	0.048	0.015	0.026
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	<0.03	<0.08	<0.04	<0.04	<0.04	<0.04	<0.04	<0.03	0.006	0.009
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	0.7	0.15	2.3	0.096	0.22	1	0.056	0.032	0.037	0.038
209 - Decachlorobiphenyl	0.36	0.13	0.47	0.042	0.12	0.2	0.014	0.055	0.018	0.049

Station	4-6	4-6	4-6	1-3.5	1-5/6/7	4-1	4-2	4-3	4-3-1	4-5	4-6	4-8
Replicate	1	2	3	1	1	1	1	1	1	1	1	1
Year	1998	1998	1998	1999	1999	1999	1999	1999	1999	1999	1999	1999
Aroclors (µg/kg)												
Total Aroclor 1221	<11	<1	<20	<1.4	<1.8	<1.8	<1.8	<1.8	<1.8	<1.6	<1.6	<1.3
Total Aroclor 1232	<11	<1	<20	<1.4	<1.8	<1.8	<1.8	<1.8	<1.8	<1.6	<1.6	<1.3
Total Aroclor 1242	<11	<1	<20	<1.4	<1.8	<1.8	<1.8	<1.8	<1.8	<1.6	<1.6	<1.3
Total Aroclor 1248	<11	<1	<20	<1.4	<1.8	<1.8	<1.8	<1.8	<1.8	<1.6	<1.6	<1.3
Total Aroclor 1254	<11	<1	<20	<1.4	<1.8	<1.8	<1.8	<1.8	<1.8	<1.6	<1.6	<1.3
Total Aroclor 1260	<11	<1	<20	<1.4	<1.8	<1.8	<1.8	<1.8	<1.8	<1.6	<1.6	<1.3
Total Aroclor 1262	<11	<1	<20	<1.4	<1.8	<1.8	<1.8	<1.8	<1.8	<1.6	<1.6	<1.3
Total Aroclor 1268	<11	<1	<20	3.3	1.7	8.2	<1.8	<1.8	3.8	<1.6	<1.6	<1.3

Appendix G. Sediment Core PCB Data

Station	1-2	1-2	1-2	1-2	1-2	1-2	1-2	1-2	1-2	1-2
Core Depth (cm)	0-1	1-2	2-3	3-4	4-5	5-6	6-7	7-8	8-9	9-10
Year	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998
PCBs (µg/kg)										
8 - 2,4'-Dichlorobiphenyl	<0.041	<0.041	<0.033	<0.046	<0.041	<0.044	<0.045	<0.036	<0.045	<0.044
18 - 2,2',5-Trichlorobiphenyl	<0.086	<0.087	<0.07	<0.096	<0.086	<0.093	<0.095	<0.077	<0.095	<0.093
28 - 2,4,4'-Trichlorobiphenyl	0.0048	0.0083	0.0024	<0.066	0.0088	<0.063	<0.065	<0.052	<0.065	<0.063
44 - 2,2',3,5'-Tetrachlorobiphenyl	0.019	0.0089	<0.024	<0.033	0.018	<0.032	<0.032	<0.026	<0.032	<0.032
49 - 2,2',4,5'-Tetrachlorobiphenyl	<0.064	<0.065	<0.052	<0.072	<0.064	<0.07	<0.071	<0.057	<0.071	<0.07
52 - 2,2',5,5'-Tetrachlorobiphenyl	0.052	0.016	0.021	0.01	0.028	<0.027	<0.028	0.0096	0.014	0.02
66 - 2,3',4,4'-Tetrachlorobiphenyl	<0.031	<0.031	<0.025	<0.035	<0.031	<0.034	<0.035	<0.028	<0.034	<0.034
77 - 3,3',4,4'-Tetrachlorobiphenyl	<0.047	<0.047	<0.038	<0.053	<0.047	<0.051	<0.052	<0.042	<0.052	<0.051
87 - 2,2',3,4,5'-Pentachlorobiphenyl	0.04	0.011	0.012	0.0074	0.019	0.014	0.016	0.0073	0.0086	0.021
101 - 2,2',4,5,5'-Pentachlorobiphenyl	0.12	0.024	0.029	0.012	0.03	0.051	0.058	0.019	0.024	0.041
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.055	<0.028	<0.022	<0.031	<0.027	<0.03	<0.03	<0.024	<0.03	0.028
118 - 2,3',4,4',5-Pentachlorobiphenyl	0.11	0.038	0.033	0.016	0.044	0.016	0.026	0.024	<0.034	0.05
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.072	<0.073	<0.058	<0.081	<0.072	<0.078	<0.08	<0.064	<0.08	<0.078
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	0.023	0.0095	<0.025	<0.035	<0.031	<0.034	<0.035	<0.028	<0.034	0.014
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	0.11	0.042	0.029	0.017	0.048	0.034	0.042	0.018	0.02	0.058
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	0.13	0.048	0.036	0.017	0.035	0.045	0.052	0.024	0.02	0.046
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	0.0062	0.006	0.0059	<0.072	<0.064	<0.07	<0.071	<0.057	<0.071	<0.07
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<0.064	<0.065	<0.052	<0.072	<0.064	<0.07	<0.071	<0.057	<0.071	<0.07
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	<0.12	<0.12	0.0034	<0.14	<0.12	<0.13	<0.14	<0.11	<0.14	<0.13
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.087	0.037	0.02	0.0085	0.0083	0.014	0.011	0.019	0.01	0.017
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	0.032	0.01	0.0065	0.0036	0.0016	0.0047	0.0055	0.004	0.0034	0.0028
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	0.036	0.044	0.032	0.04	0.032	0.032	0.028	0.027	0.036	0.032
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.088	0.045	0.037	0.033	0.035	0.045	0.047	0.047	0.041	0.049
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	<0.025	<0.026	<0.02	<0.028	<0.025	<0.027	<0.028	<0.023	<0.028	<0.027
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	0.034	0.017	0.014	<0.024	<0.021	0.0029	0.0028	<0.019	<0.024	<0.023
209 - Decachlorobiphenyl	0.061	0.022	0.018	0.0058	<0.025	0.011	0.035	<0.023	0.0096	0.017

Appendix G: Sediment Core PCB Data

Station	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7	1-5/6/7
Core Depth (cm)	**0-0.5**	**0.5-1**	**1-1.5**	**1.5-2**	**2-2.5**	**2.5-3**	**3-4**	**4-5**	**5-6**	**6-7**	**7-8**	**8-9**	**9-10**
Year	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**
PCBs (µg/kg)													
8 - 2,4'-Dichlorobiphenyl	0.7	0.59	1.3	1.3	1.2	1.2	1.1	0.87	1	0.52	1.2	1.1	0.89
18 - 2,2',5-Trichlorobiphenyl	<0.3	<0.2	<0.3	<0.2	<0.2	0.7	<0.2	<0.2	<0.2	0.26	<0.2	<0.2	<0.2
28 - 2,4,4'-Trichlorobiphenyl	<0.2	<0.2	<0.2	<0.2	<0.2	1.6	<0.1	<0.1	<0.2	0.56	<0.2	<0.2	<0.1
44 - 2,2',3,5'-Tetrachlorobiphenyl	<0.1	<0.1	0.31	<0.1	<0.1	22	<0.1	<0.1	<0.1	1.3	<0.1	<0.1	<0.1
49 - 2,2',4,5'-Tetrachlorobiphenyl	0.2	0.13	0.25	0.16	0.16	8.9	0.14	0.13	0.17	0.19	0.18	0.15	0.1
52 - 2,2',5,5'-Tetrachlorobiphenyl	<0.1	0.14	0.7	<0.1	0.15	32	<0.1	<0.1	<0.1	0.14	<0.1	<0.1	0
66 - 2,3',4,4'-Tetrachlorobiphenyl	0.67	<0.1	<0.1	<0.1	<0.1	120	0.46	<0.1	0.68	0.27	<0.1	<0.1	<0.1
77 - 3,3',4,4'-Tetrachlorobiphenyl	<0.2	<0.1	<0.1	<0.1	<0.1	<3	<0.1	<0.1	<0.1	<0.7	<0.1	<0.1	<0.1
87 - 2,2',3,4,5'-Pentachlorobiphenyl	<0.2	0.22	3.2	0.16	0.27	140	0.16	<0.1	<0.2	0.13	<0.2	<0.2	<0.1
101 - 2,2',4,5,5'-Pentachlorobiphenyl	0.58	0.38	4.4	0.36	0.52	210	0.38	0.2	0.26	0.16	0.17	0.17	0.11
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.46	0.25	3.6	0.25	0.34	160	0.25	0.13	0.17	0.04	0.08	0.1	0.04
118 - 2,3',4,4',5-Pentachlorobiphenyl	0.88	0.6	7.2	0.57	0.94	320	0.66	<0.1	<0.1	0.06	<0.1	0.47	0.37
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<1.1	<0.2	<0.2	<0.1
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	0.17	0.1	2.7	0.1	0.11	100	0.09	0.04	0.04	<0.5	0.02	0.02	0.02
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	0.67	0.34	10	0.4	0.54	350	0.42	0.18	0.2	0.08	0.16	0.11	0.1
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	0.68	0.33	5.8	0.37	0.44	220	0.36	0.2	0.25	0.2	0.17	0.15	0.12
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	0.08	0.1	1.9	0.14	0.16	67	0.12	<0.1	<0.2	0.19	<0.2	<0.2	<0.1
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.1	<0.1	<0.2	<1	<0.2	<0.2	<0.1
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	0.3	<0.4	1.5	<0.3	<0.3	44	<0.3	0.09	<0.3	0.12	<0.3	<0.3	<0.2
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.52	0.26	1.9	0.31	0.32	45	0.3	0.12	0.48	0.12	0.06	0.05	0.05
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	0.25	0.08	0.49	0.1	0.14	14	<0.1	<0.1	0.12	0.31	<0.2	<0.2	<0.1
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	0.23	0.15	<0.2	0.21	0.19	<0.2	0.16	0.21	0.23	0.25	0.24	0.24	0.16
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.45	0.15	0.71	0.2	0.2	<0.2	0.28	0.08	0.79	<0.9	0.09	0.07	0.05
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	0.96	<0.1	0.49	<0.1	<0.1	1.7	<0.1	<0.1	<0.1	<0.4	<0.1	<0.1	0
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	2.4	0.32	0.28	1.8	0.45	1.4	3.8	0.62	5.1	0.28	0.39	0.36	0.11
209 - Decachlorobiphenyl	1.2	0.42	0.27	0.83	0.62	0.78	1.4	0.44	1.8	0.14	0.34	0.4	0.12

Appendix G: Sediment Core PCB Data

Station	1-6	1-6	1-6	1-6	1-6	1-6	1-6	1-6	1-6	1-6	1-6
Core Depth (cm)	**0-0.5**	**0.5-1**	**1-1.5**	**1.5-2**	**2-2.5**	**2.5-3**	**3-4**	**4-5**	**5-6**	**6-7**	**7-8**
Year	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**
PCBs (µg/kg)											
8 - 2,4'-Dichlorobiphenyl	<0.16	<0.25	<0.23	<0.24	<0.2	<0.2	<0.16	<0.16	<0.16	<0.14	<0.14
18 - 2,2',5-Trichlorobiphenyl	<0.33	<0.52	<0.49	<0.52	<0.42	<0.42	<0.34	<0.33	<0.33	<0.28	<0.3
28 - 2,4,4'-Trichlorobiphenyl	0.26	<0.35	<0.33	<0.35	<0.29	<0.28	<0.23	<0.23	<0.23	<0.19	<0.2
44 - 2,2',3,5'-Tetrachlorobiphenyl	<0.11	<0.18	<0.16	<0.18	<0.14	<0.14	<0.12	<0.11	<0.11	<0.096	<0.1
49 - 2,2',4,5'-Tetrachlorobiphenyl	0.12	<0.39	<0.36	<0.39	<0.32	<0.31	<0.25	<0.25	<0.25	<0.21	0.089
52 - 2,2',5,5'-Tetrachlorobiphenyl	<0.097	0.27	0.12	<0.15	<0.12	<0.12	<0.1	<0.099	<0.098	<0.084	0.15
66 - 2,3',4,4'-Tetrachlorobiphenyl	<0.12	<0.19	<0.18	<0.19	<0.15	<0.15	<0.12	<0.12	<0.12	<0.1	<0.11
77 - 3,3',4,4'-Tetrachlorobiphenyl	<0.18	<0.28	<0.26	<0.28	<0.23	<0.23	<0.18	<0.18	<0.18	<0.15	<0.16
87 - 2,2',3,4,5'-Pentachlorobiphenyl	0.21	0.23	0.18	<0.39	<0.32	<0.31	<0.25	<0.25	<0.25	<0.21	0.25
101 - 2,2',4,5,5'-Pentachlorobiphenyl	0.3	0.62	0.38	<0.14	<0.12	<0.11	<0.092	<0.091	<0.091	<0.077	0.33
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.18	<0.16	0.27	<0.16	<0.13	<0.13	<0.11	<0.11	<0.1	<0.09	0.23
118 - 2,3',4,4',5-Pentachlorobiphenyl	0.46	0.49	0.43	<0.19	<0.15	<0.15	0.2	<0.12	<0.12	<0.1	0.48
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.28	<0.44	<0.41	<0.43	<0.36	<0.35	<0.28	<0.28	0.15	<0.24	<0.25
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	<0.12	<0.19	<0.18	<0.19	<0.15	<0.15	<0.12	<0.12	<0.12	<0.1	0.25
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	0.6	0.62	0.44	<0.21	0.35	<0.17	<0.14	<0.14	<0.14	<0.12	0.77
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	0.57	0.98	0.65	<0.19	0.38	<0.15	<0.12	<0.12	<0.12	<0.1	0.47
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	<0.25	<0.39	<0.36	<0.39	<0.32	<0.31	<0.25	<0.25	<0.25	<0.21	0.15
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<0.25	<0.39	<0.36	<0.39	<0.32	<0.31	<0.25	<0.25	<0.25	<0.21	<0.23
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	0.34	<0.74	<0.7	<0.74	<0.6	<0.6	<0.48	<0.48	<0.48	<0.4	<0.43
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.57	<0.21	<0.2	<0.21	<0.17	<0.17	<0.14	<0.14	<0.14	<0.12	<0.12
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	<0.25	<0.39	<0.36	<0.39	<0.32	<0.31	<0.25	<0.25	<0.25	<0.21	<0.23
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	0.11	<0.39	0.18	<0.39	<0.32	<0.31	<0.25	<0.25	<0.25	<0.21	<0.23
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.3	0.56	0.29	<0.33	<0.27	<0.26	<0.22	<0.21	<0.21	<0.18	0.29
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	<0.097	<0.15	<0.14	<0.15	<0.12	<0.12	<0.1	<0.099	<0.098	<0.084	<0.09
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	<0.082	<0.13	<0.12	<0.13	<0.1	<0.1	<0.085	<0.084	<0.083	<0.071	<0.08
209 - Decachlorobiphenyl	0.5	<0.15	<0.14	<0.15	<0.12	<0.12	<0.1	<0.099	<0.098	<0.084	0.35

Appendix G: Sediment Core PCB Data

Station	1-7	1-7	1-7	1-7	1-7	1-7	1-7	1-7	1-7	1-7	1-7	1-7	1-7	1-7
Core Depth (cm)	**0-0.5**	**0.5-1**	**1-1.5**	**1.5-2**	**2-2.5**	**2.5-3**	**3-4**	**4-5**	**5-6**	**6-7**	**7-8**	**8-9**	**9-10**	**10-11**
Year	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**
PCBs (µg/kg)														
8 - 2,4'-Dichlorobiphenyl	<0.13	<0.14	<0.14	<0.11	<0.11	0.23	<0.1	<0.093	<0.1	<0.11	0.26	<0.095	<0.1	<0.1
18 - 2,2',5-Trichlorobiphenyl	<0.28	<0.29	<0.29	<0.23	<0.23	0.25	<0.21	<0.19	<0.21	<0.24	0.18	<0.2	<0.22	<0.21
28 - 2,4,4'-Trichlorobiphenyl	<0.19	<0.2	0.089	<0.16	0.095	1.8	<0.14	0.15	0.26	<0.16	1.2	0.093	<0.15	0.13
44 - 2,2',3,5'-Tetrachlorobiphenyl	<0.096	<0.099	<0.098	0.62	0.16	0.16	<0.071	0.1	<0.072	<0.08	<0.067	<0.068	<0.076	<0.073
49 - 2,2',4,5'-Tetrachlorobiphenyl	<0.21	<0.22	<0.22	0.18	<0.17	0.55	<0.16	<0.15	0.12	<0.18	0.42	<0.15	0.11	<0.16
52 - 2,2',5,5'-Tetrachlorobiphenyl	<0.083	0.2	0.14	0.84	0.22	0.52	0.13	0.14	0.12	<0.07	0.42	<0.059	<0.066	0.088
66 - 2,3',4,4'-Tetrachlorobiphenyl	<0.1	0.36	<0.1	0.6	0.12	0.23	<0.076	<0.071	<0.077	<0.086	<0.071	<0.073	<0.081	<0.078
77 - 3,3',4,4'-Tetrachlorobiphenyl	<0.15	<0.16	<0.16	<0.13	<0.13	<0.16	<0.11	<0.11	<0.12	<0.13	<0.11	<0.11	<0.12	<0.12
87 - 2,2',3,4,5'-Pentachlorobiphenyl	<0.21	0.75	<0.22	2.8	0.45	<0.21	<0.16	0.14	<0.16	<0.18	<0.15	<0.15	<0.17	<0.16
101 - 2,2',4,5,5'-Pentachlorobiphenyl	<0.077	1.1	0.2	4.4	0.58	0.34	0.2	0.3	0.25	0.2	0.14	0.11	0.11	0.19
105 - 2,3,3',4,4'-Pentachlorobiphenyl	<0.09	0.65	<0.092	4.6	0.51	0.15	0.094	0.21	0.13	<0.075	0.17	<0.064	0.14	<0.068
118 - 2,3',4,4',5-Pentachlorobiphenyl	0.25	1.8	0.19	9.8	0.98	0.56	0.17	0.34	0.32	0.19	0.26	<0.073	0.17	0.18
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.24	<0.24	<0.24	<0.2	<0.2	<0.24	<0.18	<0.16	<0.18	<0.2	<0.16	<0.17	<0.19	<0.18
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	<0.1	0.36	<0.1	2.9	0.76	<0.1	<0.076	<0.071	<0.077	<0.086	<0.071	<0.073	<0.081	<0.078
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	<0.12	1.9	0.19	12	1.9	0.21	0.26	0.38	0.27	0.18	<0.08	<0.082	<0.091	0.22
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	<0.1	2	0.27	8.5	1.2	0.4	0.61	0.31	0.34	0.31	1.5	0.3	0.16	0.32
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	<0.21	0.17	<0.22	2.3	0.64	<0.21	<0.16	<0.15	<0.16	<0.18	<0.15	<0.15	<0.17	<0.16
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<0.21	<0.22	<0.22	<0.18	<0.17	<0.21	<0.16	<0.15	<0.16	<0.18	<0.15	<0.15	<0.17	<0.16
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	<0.4	<0.42	<0.41	2.7	1.2	<0.41	0.28	<0.28	<0.3	<0.34	<0.28	<0.29	<0.32	0.19
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	<0.12	0.43	<0.12	6.1	1.7	<0.12	1.1	0.31	0.22	0.24	<0.08	<0.082	0.15	0.26
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	<0.21	0.41	<0.22	1.3	0.33	<0.21	0.22	<0.15	<0.16	<0.18	<0.15	<0.15	<0.17	<0.16
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	<0.21	<0.22	<0.22	0.12	<0.17	<0.21	0.067	<0.15	0.1	<0.18	0.08	<0.15	0.088	0.11
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	<0.18	0.5	<0.18	2.8	0.44	<0.18	0.6	0.16	0.16	<0.15	<0.12	<0.13	<0.14	0.14
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	<0.083	<0.086	<0.085	0.63	<0.068	<0.084	<0.062	<0.058	<0.062	<0.07	<0.058	<0.059	<0.066	<0.063
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	<0.07	<0.073	<0.072	1	1.4	<0.071	0.54	<0.049	<0.053	<0.059	<0.049	<0.05	<0.055	<0.053
209 - Decachlorobiphenyl	0.96	0.51	0.36	2.8	63	0.28	0.32	0.4	<0.062	0.22	0.25	0.11	0.23	<0.063

Appendix G: Sediment Core PCB Data

Station	1-8	1-8	1-8	1-8	1-8	1-8	1-8	1-8	1-8	1-8
Core Depth (cm)	**0-1**	**1-2**	**2-3**	**3-4**	**4-5**	**5-6**	**6-7**	**7-8**	**8-9**	**9-10**
Year	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**
PCBs (µg/kg)										
8 - 2,4'-Dichlorobiphenyl	<0.075	<0.062	<0.073	<0.068	<0.088	<0.098	<0.096	<0.11	<0.11	<0.091
18 - 2,2',5-Trichlorobiphenyl	<0.16	<0.13	<0.15	<0.14	<0.18	<0.2	<0.2	<0.23	<0.24	<0.19
28 - 2,4,4'-Trichlorobiphenyl	0.0062	<0.089	0.0074	0.0018	0.013	<0.14	<0.14	<0.16	0.0062	<0.13
44 - 2,2',3,5'-Tetrachlorobiphenyl	0.022	<0.045	0.055	0.022	0.013	0.013	0.038	0.026	<0.082	<0.065
49 - 2,2',4,5'-Tetrachlorobiphenyl	<0.12	<0.098	<0.11	<0.11	<0.14	<0.15	<0.15	<0.17	<0.18	<0.14
52 - 2,2',5,5'-Tetrachlorobiphenyl	0.045	0.037	0.096	0.055	0.036	0.024	0.089	0.052	0.037	0.024
66 - 2,3',4,4'-Tetrachlorobiphenyl	<0.057	<0.048	<0.056	<0.052	<0.067	<0.075	<0.073	<0.083	<0.087	<0.07
77 - 3,3',4,4'-Tetrachlorobiphenyl	0.19	0.2	<0.083	0.32	0.24	0.15	0.19	0.18	0.17	0.12
87 - 2,2',3,4,5'-Pentachlorobiphenyl	0.032	0.027	0.16	0.043	0.041	0.019	0.029	<0.17	0.023	0.015
101 - 2,2',4,5,5'-Pentachlorobiphenyl	0.075	0.058	0.31	0.091	0.07	0.036	0.059	0.068	0.042	0.022
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.053	0.037	0.19	0.079	0.048	<0.065	0.034	0.079	0.074	0.068
118 - 2,3',4,4',5-Pentachlorobiphenyl	0.1	0.087	0.37	0.13	0.15	0.078	0.081	0.14	0.13	0.097
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.13	<0.11	<0.13	<0.12	<0.15	<0.17	<0.17	<0.19	<0.2	<0.16
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	<0.057	0.015	0.14	0.019	<0.067	<0.075	<0.073	<0.083	<0.087	<0.07
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	0.088	0.084	0.5	0.14	0.12	0.063	0.038	0.076	0.028	<0.078
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	0.12	0.11	0.55	0.17	0.13	0.12	0.12	0.17	0.16	0.15
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	0.0028	0.0035	0.048	0.007	0.0088	<0.15	<0.15	<0.17	<0.18	<0.14
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<0.12	<0.098	<0.11	<0.11	<0.14	<0.15	<0.15	<0.17	<0.18	<0.14
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	<0.22	<0.19	0.17	0.017	<0.26	<0.29	<0.29	<0.33	<0.34	<0.27
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.071	0.077	0.55	0.067	0.031	0.04	0.048	0.05	0.037	0.028
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	0.013	0.025	0.18	0.034	0.015	0.012	<0.15	<0.17	<0.18	<0.14
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	0.06	0.051	0.12	0.06	0.073	0.085	0.081	0.1	0.11	0.087
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.059	0.089	0.43	0.068	0.029	0.071	0.042	0.066	<0.15	<0.12
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	<0.046	<0.039	0.066	<0.042	<0.054	<0.061	0.02	<0.067	<0.071	<0.056
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	<0.039	0.033	0.35	0.0091	0.0034	0.017	0.019	0.025	<0.06	<0.048
209 - Decachlorobiphenyl	0.049	0.17	2.3	0.067	0.063	0.095	<0.06	0.033	<0.071	<0.056

Appendix G: Sediment Core PCB Data

Station	4-2	4-2	4-2	4-2	4-2	4-2	4-2	4-2	4-2	4-2	4-2	4-2	4-2
Core Depth (cm)	**0-0.5**	**0.5-1**	**1-1.5**	**1.5-2**	**2-2.5**	**2.5-3**	**3-4**	**4-5**	**5-6**	**6-7**	**7-8**	**8-9**	**9-10**
Year	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**
PCBs (µg/kg)													
8 - 2,4'-Dichlorobiphenyl	<0.2	0.2	0.13	0.26	0.23	0.2	0.09	0.09	0.1	0.1	0.09	0.09	0.14
18 - 2,2',5-Trichlorobiphenyl	<0.4	<0.4	<0.4	<0.5	<0.5	<0.6	<0.3	<0.3	<0.3	<0.4	<0.3	<0.3	<0.4
28 - 2,4,4'-Trichlorobiphenyl	0.27	0.22	0.2	0.74	0.33	0.26	0.45	0.11	0.23	0.55	<0.2	<0.2	<0.3
44 - 2,2',3,5'-Tetrachlorobiphenyl	<0.1	<0.1	<0.1	<0.2	<0.2	<0.2	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
49 - 2,2',4,5'-Tetrachlorobiphenyl	0.18	0.14	0.11	0.15	0.2	0.22	0.1	0.11	0.1	0.09	0.08	0.13	0.15
52 - 2,2',5,5'-Tetrachlorobiphenyl	<0.1	<0.1	<0.1	<0.2	<0.1	<0.2	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
66 - 2,3',4,4'-Tetrachlorobiphenyl	0.09	0.14	0.17	<0.2	0.17	<0.2	0.06	0.04	<0.1	<0.1	<0.1	<0.1	<0.1
77 - 3,3',4,4'-Tetrachlorobiphenyl	0.11	0.11	<0.2	<0.3	<0.3	<0.3	<0.1	<0.1	<0.2	<0.2	<0.1	<0.2	<0.2
87 - 2,2',3,4,5'-Pentachlorobiphenyl	0.05	<0.3	<0.3	<0.4	<0.3	<0.4	<0.2	0.02	0.02	<0.3	0.02	0.03	0.05
101 - 2,2',4,5,5'-Pentachlorobiphenyl	0.16	0.1	0.09	<0.1	0.07	<0.2	<0.1	0.06	0.05	0.03	0.04	<0.1	0.12
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.12	0.08	0.09	<0.2	<0.2	<0.2	<0.1	0.03	0.04	0.03	0.05	0.05	0.12
118 - 2,3',4,4',5-Pentachlorobiphenyl	0.2	0.13	0.15	0.09	0.14	0.1	0.05	0.07	0.08	0.06	0.08	0.08	0.18
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.4	<0.4	<0.4	<0.4	<0.4	<0.5	<0.2	<0.2	<0.3	<0.3	<0.2	<0.3	<0.3
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	0.06	0.04	0.04	<0.2	<0.2	<0.2	<0.1	<0.1	0.02	<0.1	0.02	<0.1	0.06
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	0.19	0.1	0.14	0.08	0.1	0.07	0.03	0.05	0.06	0.06	0.08	0.05	0.16
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	0.22	0.12	0.16	0.1	0.14	0.07	0.04	0.07	0.06	0.07	0.08	0.06	0.16
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	<0.3	<0.3	<0.3	<0.4	<0.3	<0.4	<0.2	<0.2	<0.2	<0.3	<0.2	<0.2	<0.3
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<0.3	<0.3	<0.3	<0.4	<0.3	<0.4	<0.2	<0.2	<0.2	0.05	<0.2	<0.2	<0.3
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	0.06	0.06	0.38	0.03	0.05	0.05	0.02	0.04	0.02	0.02	0.03	0.03	0.04
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.17	0.1	0.12	0.12	0.16	0.12	0.05	0.06	0.04	0.19	0.08	0.03	0.2
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	<0.3	<0.3	<0.3	<0.4	<0.3	<0.4	0.02	<0.2	<0.2	0.03	<0.2	<0.2	0.03
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	0.16	0.16	0.16	0.24	0.19	0.24	0.11	0.11	0.14	0.16	0.1	0.13	0.14
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.2	0.15	0.18	0.18	0.2	0.21	0.1	0.09	0.06	0.37	0.13	0.08	0.36
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	<0.1	<0.1	<0.1	<0.2	<0.1	<0.2	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	0.75	0.55	0.65	0.62	0.68	0.85	0.37	0.32	0.28	2.2	0.81	0.3	1.9
209 - Decachlorobiphenyl	0.08	0.08	0.09	0.07	0.09	0.12	0.05	0.05	0.05	0.33	0.12	0.04	0.32

Appendix G: Sediment Core PCB Data

Station	4-3	4-3	4-3	4-3	4-3	4-3	4-3	4-3	4-3	4-3	4-3	4-3	4-3
Core Depth (cm)	**0-0.5**	**0.5-1**	**1-1.5**	**1.5-2**	**2-2.5**	**2.5-3**	**3-4**	**4-5**	**5-6**	**6-7**	**7-8**	**8-9**	**9-10**
Year	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**
PCBs (µg/kg)													
8 - 2,4'-Dichlorobiphenyl	0.09	0.06	0.21	0.13	0.07	0.07	0.07	0.05	0.08	0.05	0.11	0.09	0.06
18 - 2,2',5-Trichlorobiphenyl	0.08	0.07	<0.3	0.2	0.05	0.09	0.09	0.05	<0.1	0.04	0.03	0.05	0.03
28 - 2,4,4'-Trichlorobiphenyl	0.14	0.18	<0.2	0.55	0.19	0.19	0.18	0.18	0.17	0.17	0.15	0.18	0.14
44 - 2,2',3,5'-Tetrachlorobiphenyl	0.16	0	<0.1	0.12	0.07	0	0	0	0	0	0	0	0
49 - 2,2',4,5'-Tetrachlorobiphenyl	0.15	0.08	<0.2	0.16	0.08	0.08	0.07	0.08	0.08	0.06	0.06	0.05	0.07
52 - 2,2',5,5'-Tetrachlorobiphenyl	0	0	<0.1	0	0	0	0	0	0	0	0	0	0
66 - 2,3',4,4'-Tetrachlorobiphenyl	0.13	0.05	<0.1	0.31	0.14	0.05	0.06	0.06	0.08	0.05	0.02	0.03	0.02
77 - 3,3',4,4'-Tetrachlorobiphenyl	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
87 - 2,2',3,4,5'-Pentachlorobiphenyl	0.14	0.05	<0.2	0.11	0.08	0.05	0.05	0.04	0.04	0.04	0.02	0.02	0.02
101 - 2,2',4,5,5'-Pentachlorobiphenyl	0.2	0.11	<0.1	0.24	0.12	0.08	0.05	0.05	0.06	0.04	0.02	0.01	0.01
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.13	0.09	0.11	0.14	0.09	0.08	0.06	0.04	0.04	0.03	0.02	0.02	0.02
118 - 2,3',4,4',5-Pentachlorobiphenyl	0.21	0.11	0.1	0.2	0.14	0.1	0.09	0.07	0.09	0.06	0.03	0.03	0.03
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.1	<0.1	<0.2	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	0.07	0.03	<0.1	0.06	0.04	0.04	0.02	0.01	0.02	0.01	0.01	0	0
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	0.23	0.11	0.12	0.2	0.15	0.1	0.09	0.07	0.08	0.05	0.04	0.03	0.02
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	0.21	0.11	0.4	0.22	0.13	0.1	0.08	0.09	0.08	0.05	0.03	0.03	0.02
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	<0.1	<0.1	<0.2	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	0.12	0.06	<0.2	0.08	0.08	0.08	0.08	0.07	0.06	0.05	0.06	0.04	<0.1
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	0.04	0.04	<0.4	0.05	0.04	0.03	0.01	0.02	0.02	0.02	0.01	0.03	0.02
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.1	0.05	<0.1	0.08	0.05	0.05	0.04	0.01	0.03	0.02	0.01	0	0.01
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	0.04	0.03	0.09	0.05	0.04	0.03	0.03	0.01	0.02	0.01	0.02	<0.1	0.01
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	0.1	0.08	<0.2	0.09	0.09	0.1	0.09	0.09	0.09	0.09	0.1	0.1	0.07
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.1	0.06	<0.2	0.1	0.07	0.09	0.05	0.06	0.05	0.04	0.04	0.06	0.04
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	0.11	0.05	<0.1	0.08	0.06	0	0	0	0	0	0	0	0
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	0.21	0.07	0.07	0.1	0.11	0.29	0.07	0.09	0.06	0.1	0.08	0.07	0.18
209 - Decachlorobiphenyl	0.04	0.02	0.02	0.03	0.03	0.06	0.01	0.02	0.03	0.02	0.02	0.01	0.03

Appendix G: Sediment Core PCB Data

Station	4-3.1	4-3.1	4-3.1	4-3.1	4-3.1	4-3.1	4-3.1	4-3.1	4-3.1	4-3.1	4-3.1	4-3.1	4-3.1
Core Depth (cm)	**0-0.5**	**0.5-1**	**1-1.5**	**1.5-2**	**2-2.5**	**2.5-3**	**3-4**	**4-5**	**5-6**	**6-7**	**7-8**	**8-9**	**9-10**
Year	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**
PCBs (µg/kg)													
8 - 2,4'-Dichlorobiphenyl	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
18 - 2,2',5-Trichlorobiphenyl	0.09	<0.1	0.03	<0.1	0.07	0.06	<0.1	0.06	0.04	0.04	0.02	<0.1	<0.1
28 - 2,4,4'-Trichlorobiphenyl	0.36	0.05	0.03	0.07	0.25	0.16	0.1	0.09	0.07	0.07	0.09	0.17	0.09
44 - 2,2',3,5'-Tetrachlorobiphenyl	0	0	0	0	0.49	0.58	0.47	0	0	0	0	0	0
49 - 2,2',4,5'-Tetrachlorobiphenyl	0.11	0.09	0.11	0.09	0.09	0.08	0.07	0.07	0.06	0.08	0.1	0.08	0.07
52 - 2,2',5,5'-Tetrachlorobiphenyl	0.1	0.06	0.06	0.08	0.1	0.09	0.07	0.08	0.08	0.04	0.11	0.1	0
66 - 2,3',4,4'-Tetrachlorobiphenyl	0.43	0.12	0.1	0.1	0.4	0.29	0.08	0.06	0.05	0.14	0.21	0.33	0.05
77 - 3,3',4,4'-Tetrachlorobiphenyl	<0.1	<0.1	<0.1	<0.1	0.06	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
87 - 2,2',3,4,5'-Pentachlorobiphenyl	0.09	0.06	0.05	0.14	0.05	0.09	0.04	0.03	0.08	0.03	0.05	0.04	<0.1
101 - 2,2',4,5,5'-Pentachlorobiphenyl	0.19	0.12	0.04	0.07	0.1	0.19	0.05	0.05	0.02	0.04	0.1	0.04	0.01
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.06	0.05	0.05	0.05	0.06	0.07	0.03	0.02	0.02	0.02	0.04	0.04	0
118 - 2,3',4,4',5-Pentachlorobiphenyl	0.18	0.17	0.14	0.12	0.14	0.22	0.09	0.08	0.07	0.07	0.12	0.08	0.03
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	0.05	0.07	0.05	0.04	0.05	0.06	0.03	0.02	0.02	0.02	0.03	0.03	0
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	0.16	0.56	0.1	0.11	0.11	0.53	0.07	0.07	0.05	0.12	0.08	0.14	<0.1
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	0.25	0.7	0.17	0.2	0.21	0.88	0.09	0.09	0.06	0.24	0.16	0.26	0.04
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	0.04	0.23	0.07	0.03	0.03	0.15	0.03	0.03	0.03	0.09	0.05	0.08	0.03
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.08	0.63	0.06	0.05	0.07	0.51	0.04	0.03	0.03	0.17	0.03	0.23	0.01
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	0.04	0.1	0.03	0.03	0.04	0.19	0.03	0.01	0.02	0.05	<0.1	0.06	<0.1
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	0.11	0.07	0.07	0.07	0.08	0.07	0.08	0.09	0.09	0.07	0.09	0.07	0.1
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.06	0.21	0.08	0.06	0.06	0.32	0.05	0.03	0.05	0.09	0.03	0.08	0.02
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	0	0.18	0.1	0.06	0.06	0.07	0	0.04	0	0.07	0	0.05	0
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	0.09	0.2	0.11	0.1	0.09	0.12	0.1	0.07	0.23	0.08	0.05	0.08	0.1
209 - Decachlorobiphenyl	0.1	0.06	0.07	0.08	0.08	0.11	0.09	0.05	0.07	0.03	0.02	0.03	0.07

Appendix H. Sediment PAH Data

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-7	1-8	4-6	4-6
Year	1998	1998	1998	1998	1998	1998	1998	1998	1998	1999
PAHs (µg/kg)										
Naphthalene	1.9	1.2	2.1	2.2	1.6	1.9	2.2	1.7	1.7	0.98
C1-Naphthalenes	1	0.61	1.3	0.93	1.2	1.2	1.7	1	0.95	1
C2-Naphthalenes	2.4	<0.59	<0.95	2.2	<0.77	<0.98	4.8	<0.94	<0.75	2.6
C3-Naphthalenes	1.9	<0.59	<0.95	<0.7	<0.77	<0.98	<0.72	<0.94	<0.75	3.3
C4-Naphthalenes	<0.87	<0.59	<0.95	<0.7	<0.77	<0.98	38	<0.94	<0.75	<1.2
Acenaphthylene	0.23	<0.19	0.12	0.15	<0.24	<0.31	<0.23	<0.3	0.14	0.23
Acenaphthene	0.24	0.21	0.23	0.23	0.18	<0.33	2.3	<0.32	0.18	<0.4
Biphenyl	0.29	0.25	0.4	0.41	0.29	0.44	0.75	0.33	0.39	0.25
Fluorene	0.44	0.27	0.36	0.36	0.4	<0.4	3	0.31	0.23	0.26
C1-Fluorenes	<0.36	<0.24	<0.39	<0.28	<0.32	<0.4	17	<0.38	<0.31	<0.49
C2-Fluorenes	<0.36	<0.24	<0.39	<0.28	<0.32	<0.4	32	<0.38	<0.31	<0.49
C3-Fluorenes	<0.36	<0.24	<0.39	<0.28	<0.32	<0.4	34	<0.38	<0.31	<0.49
Anthracene	0.42	0.1	0.19	0.22	0.26	0.28	2	0.17	0.16	0.13
Phenanthrene	1.5	0.94	1.7	1.4	1.7	1.8	5.7	1.5	1.1	1
C1-Phenanthrenes/anthracenes	1.8	0.78	<0.52	1.6	1.2	1.3	6	1.1	0.74	0.97
C2-Phenanthrenes/anthracenes	2	<0.32	<0.52	<0.38	<0.42	<0.54	14	<0.51	<0.41	1.9
C3-Phenanthrenes/anthracenes	1.8	<0.32	<0.52	<0.38	<0.42	<0.54	15	<0.51	<0.41	1.4
C4-Phenanthrenes/anthracenes	<0.48	<0.32	<0.52	<0.38	<0.42	<0.54	8.8	<0.51	<0.41	<0.65
Dibenzothiophene	0.31	0.29	0.3	0.34	0.22	0.39	2	0.26	0.14	<0.26
C1-Dibenzothiophenes	0.74	<0.13	<0.21	0.67	<0.17	<0.22	11	<0.21	<0.17	<0.26
C2-Dibenzothiophenes	1.6	<0.13	<0.21	<0.16	<0.17	<0.22	38	<0.21	<0.17	<0.26
C3-Dibenzothiophenes	2	<0.13	<0.21	<0.16	<0.17	<0.22	47	<0.21	<0.17	<0.26
Fluoranthene	2.2	0.42	1.1	1.1	0.51	1.1	2.9	0.98	0.72	0.69
Pyrene	1.8	0.47	0.94	0.96	0.71	0.62	2.8	0.57	1.1	0.95
C1-Fluoranthenes/pyrenes	2.2	<0.17	<0.28	1.3	<0.22	3	3.4	1.9	1.1	1
C2-Fluoranthenes/pyrenes	2.2	<0.17	<0.28	<0.2	<0.22	<0.29	4	<0.27	<0.22	1.2
C3-Fluoranthenes/pyrenes	1.7	<0.17	<0.28	<0.2	<0.22	<0.29	<0.21	<0.27	<0.22	<0.34
Benzo[a]anthracene	1.3	<0.23	<0.36	<0.27	<0.29	<0.38	<0.28	<0.36	<0.29	<0.45
Chrysene	1.5	<0.18	<0.3	<0.22	<0.24	<0.31	<0.23	<0.3	<0.24	0.57
C1-Chrysenes	1.4	<0.18	<0.3	0.6	<0.24	<0.31	<0.23	<0.3	0.8	0.59
C2-Chrysenes	1.2	<0.18	<0.3	<0.22	<0.24	<0.31	<0.23	<0.3	<0.24	<0.37
C3-Chrysenes	<0.27	<0.18	<0.3	<0.22	<0.24	<0.31	<0.23	<0.3	<0.24	<0.37
C4-Chrysenes	<0.27	<0.18	<0.3	<0.22	<0.24	<0.31	<0.23	<0.3	<0.24	<0.37
Benzo[b]fluoranthene	2.7	0.41	0.65	0.71	0.55	1.2	0.94	0.46	0.8	0.7
Benzo[k]fluoranthene	0.83	0.12	0.16	0.16	0.11	0.25	0.26	0.11	0.22	0.31
Benzo[e]pyrene	1.7	0.41	0.62	0.9	0.51	0.8	0.91	0.44	0.85	0.56
Benzo[a]pyrene	1.3	0.21	0.44	0.36	0.21	0.46	0.66	0.25	0.49	0.3
Perylene	1.7	0.97	1.5	1.2	1.2	2.6	1.4	1.2	0.97	0.6
Indeno[1,2,3,-c,d]pyrene	1.2	0.14	0.37	0.54	0.24	0.64	0.38	0.33	0.52	0.41
Dibenzo[a,h]anthracene	0.46	<0.23	0.86	0.19	<0.3	0.52	<0.28	0.23	0.077	0.23
Benzo[g,h,i]perylene	1.6	1	1.3	0.7	0.65	1.2	0.97	0.74	0.72	0.55

Appendix H: Sediment PAH Data

Station	1-3.5	1-5/6/7	4-1	4-2	4-3	4-3.1	4-5	4-8	Cruise IV Control	Cruise V Control
Year	1999	1999	1999	1999	1999	1999	1999	1999	1999	1999
PAHs (µg/kg)										
Naphthalene	1.1	1.1	1	1.1	1	1.6	0.81	0.83	2.3	2.6
C1-Naphthalenes	0.94	1.4	1.3	1.2	1.2	1.4	0.98	0.93	4.4	5.4
C2-Naphthalenes	2.6	<1.4	3.9	3.8	2.9	3	2.1	2.5	7.2	9.1
C3-Naphthalenes	2.3	3.5	3.5	3.6	2.2	2.4	<1.2	2.2	7.3	7.8
C4-Naphthalenes	<1	<1.4	<1.3	<1.3	<1.3	<1.3	<1.2	<0.95	5.5	7.5
Acenaphthylene	0.17	<0.43	0.24	<0.42	0.2	0.27	0.16	<0.3	<0.28	0.21
Acenaphthene	0.14	<0.46	0.14	<0.44	0.17	0.15	<0.41	<0.32	<0.3	0.26
Biphenyl	0.24	0.26	0.29	0.32	0.27	0.3	0.16	0.18	1	1.5
Fluorene	0.24	<0.56	0.22	0.35	0.2	0.31	0.21	0.35	1	1.1
C1-Fluorenes	<0.43	<0.56	<0.53	<0.54	<0.53	<0.53	<0.5	<0.39	2	2.9
C2-Fluorenes	<0.43	<0.56	<0.53	<0.54	<0.53	<0.53	<0.5	<0.39	6	7
C3-Fluorenes	<0.43	<0.56	<0.53	<0.54	<0.53	<0.53	<0.5	<0.39	5.8	6.2
Anthracene	0.11	0.098	0.18	0.2	0.14	0.19	0.12	0.31	0.22	0.25
Phenanthrene	1	0.74	1	0.92	0.94	1.2	0.59	0.79	4.2	5.5
C1-Phenanthrenes/anthracenes	1.2	1.3	1.3	1.3	1.4	1.7	0.74	1.1	7.3	9.1
C2-Phenanthrenes/anthracenes	2.8	2.6	4.2	3.3	2.5	3.6	1.7	2.1	8	9.1
C3-Phenanthrenes/anthracenes	1.8	2.3	1.8	1.7	1.9	2.4	1.4	<0.52	3.7	4.2
C4-Phenanthrenes/anthracenes	<0.57	<0.74	<0.71	<0.71	<0.71	<0.71	<0.66	<0.52	<0.49	2.5
Dibenzothiophene	0.17	<0.3	<0.29	<0.29	<0.29	<0.29	<0.27	0.19	0.44	0.62
C1-Dibenzothiophenes	0.51	<0.3	<0.29	<0.29	<0.29	<0.29	<0.27	<0.21	0.92	1.1
C2-Dibenzothiophenes	1.3	<0.3	<0.29	<0.29	<0.29	<0.29	<0.27	<0.21	1.2	1.3
C3-Dibenzothiophenes	1.6	<0.3	<0.29	<0.29	<0.29	<0.29	<0.27	<0.21	<0.2	<0.2
Fluoranthene	0.94	0.61	0.96	0.98	0.94	1.5	0.53	0.62	1.2	1.2
Pyrene	0.94	0.44	1.8	1.5	1.6	2.1	0.72	0.98	1.4	1.6
C1-Fluoranthenes/pyrenes	0.97	0.73	1.8	1.3	1.6	2.1	0.89	1.6	3.5	4
C2-Fluoranthenes/pyrenes	1.5	<0.4	2	1.4	1.7	2.4	1.1	2.4	4.7	5.3
C3-Fluoranthenes/pyrenes	<0.3	<0.4	<0.38	<0.38	<0.38	<0.38	<0.35	3.5	3.5	4.2
Benzo[a]anthracene	<0.4	<0.52	<0.5	<0.5	<0.5	<0.5	<0.46	<0.36	<0.34	<0.34
Chrysene	0.69	0.52	0.93	0.82	0.75	1	0.35	<0.3	1.2	1.5
C1-Chrysenes	0.68	<0.42	1	0.96	0.94	1	0.52	0.85	1.6	2
C2-Chrysenes	<0.33	<0.42	2.5	<0.41	2.8	<0.41	<0.38	<0.3	1.9	3
C3-Chrysenes	<0.33	<0.42	<0.41	<0.41	<0.41	<0.41	<0.38	4.4	<0.28	1.6
C4-Chrysenes	<0.33	<0.42	<0.41	<0.41	<0.41	<0.41	<0.38	6.6	<0.28	<0.28
Benzo[b]fluoranthene	0.72	0.48	1.4	1.1	1.4	1.6	0.51	1.5	0.73	1.2
Benzo[k]fluoranthene	0.28	0.45	0.52	0.43	0.48	0.64	0.28	0.55	0.19	0.4
Benzo[e]pyrene	0.43	0.39	0.9	0.78	0.86	1	0.36	0.68	0.55	0.88
Benzo[a]pyrene	0.3	0.35	0.6	0.59	0.52	0.78	0.29	0.49	0.23	0.32
Perylene	0.73	1.3	6	1.6	1.9	2	0.6	2.5	0.43	0.82
Indeno[1,2,3,-c,d]pyrene	0.33	0.23	0.62	0.6	0.72	0.99	0.27	0.32	0.19	0.26
Dibenzo[a,h]anthracene	0.15	0.23	0.16	0.19	0.16	0.21	0.18	0.27	0.17	0.15
Benzo[g,h,i]perylene	0.42	0.5	0.97	0.75	0.9	1.1	0.36	0.64	0.43	0.52

Appendix I. Sediment Metals Data

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-7	1-8	4-6	4-6
Year	1998	1998	1998	1998	1998	1998	1998	1998	1998	1999
Metals (µg/g, except where noted)										
Aluminum (%)	0.94	2.09	1.49	1.42	2	1.26	1.89	1.16	1.26	1.5
Cadmium	2.34	1.3	3.96	1.16	2.15	2.25	7	2.95	0.28	0.31
Chromium	87.9	187	161	115	191	103	223	110	108	147
Copper	61.5	37	109	32.4	52.4	84.4	137	50.8	28.7	28
Iron (%)	1.53	4.87	1.94	3.84	3.76	1.57	3.77	1.65	2.93	3.39
Lead	4.5	9.2	3.1	16.9	4.5	5.8	13.3	4.5	5.3	0.028
Mercury	0.05	0.052	0.107	0.06	0.055	0.063	0.062	0.052	0.033	4.9
Nickel	95.4	40.5	215	57.2	57.4	96.2	64.1	91.8	35.4	24.1
Silver	0.53	0.26	1.58	0.29	0.31	0.62	0.43	0.54	0.09	0.16
Zinc	126	77.2	248	256	90	143	141	126	85.2	72.3

Station	1-3.5	1-5/6/7	4-1	4-2	4-3	4-3.1	4-5	4-8	Cruise IV Control
Year	1999	1999	1999	1999	1999	1999	1999	1999	1999
Metals (µg/g, except where noted)									
Aluminum (%)	2.03	0.4	2.46	2	1.88	1.93	1.8	1.78	4.96
Cadmium	0.54	4.37	0.44	0.33	0.31	0.31	0.29	0.47	0.12
Chromium	227	114	145	125	123	106	197	231	71.8
Copper	78.2	67.6	33.9	26.5	29.8	28.5	29.9	23.9	4.2
Iron (%)	5.89	1.23	4.18	3.99	3.97	3.46	4.61	5.63	1.19
Lead	0.043	0.049	0.054	0.043	0.041	0.052	0.032	0.054	0.034
Mercury	24.3	1.8	5	4.2	4.2	4.1	4.7	5.3	4.1
Nickel	25	88.9	33.4	27.5	27.6	28.3	23.9	25.8	28.4
Silver	0.21	0.31	0.18	0.15	0.14	0.15	0.15	0.19	0.11
Zinc	78.4	108	83.2	71.2	71.5	70	74.9	61.4	26.9

Appendix J. Tissue PCB Concentration Data

Macoma nasuta

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-7	1-8	4-6	Cruise II Control	Zero Time II
Year	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998
PCBs (µg/kg)											
8 - 2,4'-Dichlorobiphenyl	<2.5	<1.1	<2.5	<2.8	<2.3	<1.7	<2.7	<2.4	<2.7	<2.6	<2.4
18 - 2,2',5-Trichlorobiphenyl	<3.3	0.24	<3.3	<3.7	<3	<2.2	<3.5	<3.2	<3.6	<3.4	<3.2
28 - 2,4,4'-Trichlorobiphenyl	<3.1	0.22	0.31	0.42	0.31	<2.1	<3.3	0.32	0.54	1.1	0.47
44 - 2,2',3,5'-Tetrachlorobiphenyl	<2.6	1.1	<2.7	0.43	1.4	0.74	0.44	0.4	<2.9	<2.8	<2.6
49 - 2,2',4,5'-Tetrachlorobiphenyl	1.2	1.2	1.1	1.5	1.7	0.99	1.6	1.9	1.9	4.6	3.1
52 - 2,2',5,5'-Tetrachlorobiphenyl	1.1	2	1.5	1.1	2.6	1	1.4	2.8	1.8	2.6	1
66 - 2,3',4,4'-Tetrachlorobiphenyl	0.39	0.57	0.67	0.9	0.9	0.47	1.1	0.73	1.7	2.8	0.73
77 - 3,3',4,4'-Tetrachlorobiphenyl	<5.8	<2.6	<5.9	<6.6	<5.4	<3.9	<6.3	<5.7	<6.4	<6.1	<5.7
87 - 2,2',3,4,5'-Pentachlorobiphenyl	0.68	0.96	0.45	0.32	1.1	0.46	0.56	1.4	0.5	0.75	0.95
101 - 2,2',4,5,5'-Pentachlorobiphenyl	1.5	3.4	1.4	2	3.5	1.8	2.4	5.2	2.6	3.9	2.4
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.58	0.68	0.47	0.59	0.78	0.36	0.91	1.2	1	1.5	1.3
118 - 2,3',4,4',5-Pentachlorobiphenyl	1.5	1.8	1.7	1.5	2.4	1.2	1.9	4.3	2.6	4.7	3.1
126 - 3,3',4,4',5-Pentachlorobiphenyl	<4.8	<2.1	<4.9	<5.4	<4.5	<3.2	<5.2	<4.7	<5.2	<5	<4.7
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	<7.7	<3.4	<7.8	<8.7	<7.1	<5.1	<8.3	<7.5	<8.4	0.69	0.77
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	1.8	1.7	2	1	2.6	1.6	2.2	4.2	2.8	4.3	4.4
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	2	3.2	2.6	2	3.4	2.1	3.1	6.2	3.2	4.5	4.1
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	<3.9	0.18	<4	<4.4	<3.6	<2.6	<4.2	0.16	<4.2	<4.1	0.81
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<3.9	<1.7	<4	<4.4	<3.6	<2.6	<4.2	<3.8	<4.2	<4.1	<3.8
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	<3.8	0.21	<3.9	<4.3	<3.5	0.26	<4.1	0.64	<4.2	0.3	<3.7
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.45	0.74	0.36	<6.9	0.63	0.76	<6.6	2.6	0.39	0.73	0.84
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	<3.9	0.35	0.34	<4.4	<3.6	0.22	<4.2	1.2	0.36	0.22	0.26
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	1.5	0.69	1.3	1.7	1.9	1.3	1.4	1.9	1.6	1.8	3.2
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.93	1.2	0.63	0.35	1.3	0.83	0.7	3.5	0.46	0.74	1.2
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	<5.4	<2.4	<5.5	<6.1	<5	<3.6	<5.8	0.59	<5.9	<5.7	<5.3
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	<4.5	0.14	<4.6	<5.1	<4.2	<3	<4.8	1.1	<4.9	<4.7	0.18
209 - Decachlorobiphenyl	<2.5	0.078	<2.5	<2.8	<2.3	<1.7	<2.7	5.6	<2.7	<2.6	0.16

Macoma nasuta

Station	1-5/6/7	4-1	4-2	4-3	4-5	Cruise IV Control	Cruise V Control	Zero Time IV	Zero Time V
Year	1999	1999	1999	1999	1999	1999	1999	1999	1999
PCBs (µg/kg)									
8 - 2,4'-Dichlorobiphenyl	<1.3	<1.4	<1.3	<1.4	<1.4	<2.5	<3.6	<1.3	<2.8
18 - 2,2',5-Trichlorobiphenyl	<0.82	<0.93	<0.87	<0.88	<0.89	<1.6	<2.3	<0.83	<1.8
28 - 2,4,4'-Trichlorobiphenyl	0.18	1.3	2.1	1.1	1.4	2.3	1.1	1.4	0.82
44 - 2,2',3,5'-Tetrachlorobiphenyl	0.79	<3.2	<3	<3	<3.1	<5.6	<7.9	<2.9	<6.3
49 - 2,2',4,5'-Tetrachlorobiphenyl	0.48	0.78	1.2	0.84	0.84	2.2	1.1	0.7	0.92
52 - 2,2',5,5'-Tetrachlorobiphenyl	<2.2	<2.5	1.6	<2.4	<2.4	<4.4	<6.3	<2.3	0.98
66 - 2,3',4,4'-Tetrachlorobiphenyl	1.6	1.4	1.9	1.3	1.7	4.3	2.4	2	0.96
77 - 3,3',4,4'-Tetrachlorobiphenyl	<1.8	5.1	6.1	4.5	5	14	6.3	6.3	<4.1
87 - 2,2',3,4,5'-Pentachlorobiphenyl	<1.2	<1.3	<1.2	<1.2	<1.2	<2.3	<3.2	<1.2	<2.6
101 - 2,2',4,5,5'-Pentachlorobiphenyl	2.2	3.6	4.4	3	3.2	8.4	4.8	4.2	4.5
105 - 2,3,3',4,4'-Pentachlorobiphenyl	1.2	1.6	1.5	1.8	1.8	3.9	2.4	1.8	1.6
118 - 2,3',4,4',5-Pentachlorobiphenyl	1.8	4.4	5.6	3.8	4.2	10	5.8	5	5.3
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.58	<0.66	<0.61	<0.62	<0.63	<1.1	<1.6	<0.58	<1.3
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	0.58	0.72	0.72	0.54	0.48	1.5	0.96	0.92	1.3
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	3.4	4.4	5	4.8	3.4	10	6.1	<1.5	6.9
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	3.6	6	6	4.7	5.5	13	6.5	6.5	6.8
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	0.47	<0.75	0.73	<0.71	0.37	1.4	0.91	0.56	1.1
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<0.69	<0.79	<0.74	<0.74	<0.75	<1.4	<1.9	<0.7	<1.6
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	<0.82	<0.94	<0.87	<0.88	<0.9	<1.6	<2.3	0.58	1.5
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.61	1.3	0.99	0.86	0.77	1.5	1.2	1.1	2
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	<0.51	<0.58	0.35	<0.55	<0.56	0.18	<1.4	0.16	<1.2
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	<0.53	<0.6	<0.56	<0.57	<0.57	<1	<1.5	<0.54	<1.2
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.46	0.57	0.95	0.34	0.46	1.2	0.75	0.74	1.4
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	<0.78	<0.89	<0.83	0.16	<0.85	0.32	<2.2	0.21	<1.8
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	0.43	0.25	0.29	0.26	0.3	0.6	0.24	0.18	0.4
209 - Decachlorobiphenyl	0.51	0.21	0.16	<1.1	0.14	0.53	0.15	0.24	0.19

Nephtys caecoides

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-7	1-8	4-6	Cruise II Control	Zero Time II
Year	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998
PCBs (µg/kg)											
8 - 2,4'-Dichlorobiphenyl	<1.5	<2.5	<1.4	<1.7	<2.5	<2.5	<1.6	<1.6	<1.7	<1.7	<1.4
18 - 2,2',5-Trichlorobiphenyl	<2	<3.2	0.36	<2.2	<3.2	<3.3	<2.1	1.8	<2.2	<2.2	<1.8
28 - 2,4,4'-Trichlorobiphenyl	0.19	0.65	0.17	<2.1	0.66	<3.1	0.18	0.41	0.24	0.68	<1.8
44 - 2,2',3,5'-Tetrachlorobiphenyl	1	0.34	1.1	0.82	<2.6	0.34	1.3	6.9	1.3	2.4	<1.5
49 - 2,2',4,5'-Tetrachlorobiphenyl	1.1	2.1	0.91	1	1.6	1.3	1.3	5.4	1.6	4.1	1.3
52 - 2,2',5,5'-Tetrachlorobiphenyl	1.5	2.4	1.5	1.1	2	1.2	2.4	15	2.2	4	0.29
66 - 2,3',4,4'-Tetrachlorobiphenyl	0.61	1.5	0.61	0.56	1.6	0.64	0.69	2	1	2.3	<2.5
77 - 3,3',4,4'-Tetrachlorobiphenyl	<3.5	<5.8	<3.3	<3.9	<5.8	<5.8	<3.8	<3.8	<4	<4	<3.3
87 - 2,2',3,4,5'-Pentachlorobiphenyl	0.49	1.4	0.55	0.32	0.88	0.42	1.2	14	0.61	1.1	0.18
101 - 2,2',4,5,5'-Pentachlorobiphenyl	2.5	4.2	2.5	1.8	4.1	1.4	3.8	29	3	4.8	0.65
105 - 2,3,3',4,4'-Pentachlorobiphenyl	0.43	1.2	0.37	<2.2	1	<3.3	0.62	11	0.51	1.2	0.26
118 - 2,3',4,4',5-Pentachlorobiphenyl	1.4	4.6	1.2	1	3.2	1.2	2	26	1.6	3.7	0.58
126 - 3,3',4,4',5-Pentachlorobiphenyl	<2.9	<4.8	<2.7	<3.2	<4.8	<4.8	<3.1	<3.2	<3.3	<3.3	<2.7
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	0.41	<7.6	<4.3	<5.2	<7.6	<7.7	<5	<5.1	<5.3	0.52	<4.4
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	1.8	4.6	1.9	0.94	2.7	2	2	22	2	3	0.93
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	2.6	5.1	2.4	2.1	4	2.3	3.4	9	3	4.6	1.7
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	<2.4	<3.8	<2.2	<2.6	<3.8	<3.9	<2.5	3.1	<2.7	0.29	<2.2
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<2.4	<3.8	<2.2	<2.6	<3.8	<3.9	<2.5	<2.6	<2.7	<2.6	<2.2
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	0.28	0.33	0.19	<2.6	<3.8	<3.8	<2.5	2	<2.6	0.43	0.098
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.88	0.94	0.57	0.52	0.72	0.43	0.96	4.8	0.74	1	0.42
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	0.17	0.31	0.3	0.22	0.34	<3.9	0.25	1.6	0.29	0.36	0.14
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	1.1	1.6	1.2	1.1	1.6	1.8	1.4	<2.6	1.6	1.4	0.93
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.76	1.6	0.69	0.67	1.1	0.39	0.8	4.5	0.92	1	0.33
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	<3.3	<5.4	<3	<3.7	<5.4	<5.4	<3.5	0.38	<3.7	<3.7	<3.1
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	<2.7	0.32	<2.5	<3	0.22	<4.5	<2.9	0.79	<3.1	<3.1	0.2
209 - Decachlorobiphenyl	<1.5	0.46	<1.4	<1.7	0.57	<2.5	<1.6	1.1	<1.7	<1.7	<1.4

Nephtys caecoides

Station	1-5/6/7	4-1	4-2	4-3	4-5	Cruise IV Control	Cruise V Control	Zero Time IV	Zero Time V
Year	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**
PCBs (µg/kg)									
8 - 2,4'-Dichlorobiphenyl	<0.88	<0.84	<0.47	<0.44	<0.82	<0.46	<0.84	<0.91	<2.6
18 - 2,2',5-Trichlorobiphenyl	<0.57	<0.54	<0.3	<0.28	0.19	1.6	<0.54	<0.59	<1.7
28 - 2,4,4'-Trichlorobiphenyl	0.82	0.51	0.58	1.6	0.9	1.2	0.72	0.97	0.38
44 - 2,2',3,5'-Tetrachlorobiphenyl	1.7	1.5	2.2	3.2	1.5	2.8	1.6	<2	<5.9
49 - 2,2',4,5'-Tetrachlorobiphenyl	<0.3	0.43	0.74	1	0.78	1.5	0.7	<0.31	<0.9
52 - 2,2',5,5'-Tetrachlorobiphenyl	<1.6	1	<0.83	4.3	<1.5	2.5	<1.5	<1.6	0.82
66 - 2,3',4,4'-Tetrachlorobiphenyl	1.7	0.96	1.6	2.6	1.5	2.9	1.3	0.96	0.46
77 - 3,3',4,4'-Tetrachlorobiphenyl	<1.3	1.4	1.5	<0.63	<1.2	2.9	<1.2	<1.3	<3.8
87 - 2,2',3,4,5'-Pentachlorobiphenyl	<0.81	0.6	<0.43	0.69	0.69	0.86	<0.76	<0.83	0.66
101 - 2,2',4,5,5'-Pentachlorobiphenyl	2	3	3.9	4.3	3.9	3.3	2.1	<0.61	1.3
105 - 2,3,3',4,4'-Pentachlorobiphenyl	1.3	0.59	0.9	0.87	1.1	1	0.92	0.73	0.36
118 - 2,3',4,4',5-Pentachlorobiphenyl	1.2	1.4	1.8	2.3	1.9	2.4	1.9	1	1.1
126 - 3,3',4,4',5-Pentachlorobiphenyl	<0.4	<0.38	<0.21	<0.2	<0.38	<0.21	<0.38	<0.42	<1.2
128 - 2,2',3,3',4,4'-Hexachlorobiphenyl	0.36	0.21	0.13	0.37	0.25	0.26	0.31	0.18	0.31
138 - 2,2',3,4,4',5'-Hexachlorobiphenyl	1.3	1.4	1.8	2	1.8	2.1	1.8	1.4	2
153 - 2,2',4,4',5,5'-Hexachlorobiphenyl	0.91	2	2.1	2.6	3.2	3.6	2.9	2	2
156 - 2,3,3',4,4',5-Hexachlorobiphenyl	<0.46	<0.44	0.27	0.24	<0.43	<0.24	<0.44	0.22	<1.4
169 - 3,3',4,4',5,5'-Hexachlorobiphenyl	<0.48	<0.46	<0.26	<0.24	<0.45	<0.25	<0.46	<0.5	<1.4
170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl	<0.57	<0.55	<0.3	<0.28	<0.54	<0.3	<0.54	<0.59	<1.7
180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.72	<1	<0.58	<0.54	0.44	<0.56	0.93	0.58	0.67
183 - 2,2',3,4,4',5',6-Heptachlorobiphenyl	<0.36	0.16	0.14	0.2	0.21	0.16	0.12	0.095	0.18
184 - 2,2',3,4,4',6,6'-Heptachlorobiphenyl	<0.37	<0.35	<0.19	<0.18	<0.34	<0.19	<0.35	<0.38	<1.1
187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl	0.26	0.32	0.4	0.55	0.35	0.53	0.47	0.35	0.51
195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl	0.053	<0.52	0.095	0.15	0.06	0.14	<0.52	<0.56	0.27
206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl	0.13	0.077	0.066	0.052	0.071	0.13	<6.1	<6.6	0.3
209 - Decachlorobiphenyl	0.13	0.057	0.18	0.032	0.21	<0.38	<0.69	0.068	<2.2

Appendix K. Tissue PAH Data

Macoma nasuta

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-7	1-8	4-6	Cruise II Control	Zero Time II
Year	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998
PAHs (µg/kg)											
Naphthalene	14	6.6	13	15	13	19	13	13	17	9.7	10
C1-Naphthalenes	9.4	4.8	9.6	9.1	9.5	6.5	9.2	9.5	8.9	7.5	7.4
C2-Naphthalenes	17	9.3	18	<14	13	6.7	15	15	21	16	12
C3-Naphthalenes	<8.8	<3.9	<9	<10	<8.2	<5.9	<9.5	<8.7	<9.7	<9.3	<8.6
C4-Naphthalenes	0	0	0	0	0	0	0	0	0	0	0
Acenaphthylene	4.2	1.5	4.7	4.8	1.6	0.57	5.2	6.3	2.2	2	14
Acenaphthene	1.2	1.7	1.2	2.2	1.8	0.79	1.8	1.8	12	1.3	3.3
Biphenyl	6.1	4	6.5	7.8	6.8	4.1	7	5.4	5.8	2.7	3
Fluorene	3.8	2	3.9	4.2	5.1	1.7	4	4.6	3.5	2.7	5
C1-Fluorenes	7.9	3.1	9.6	9.4	7.1	<3.4	11	11	<5.5	<5.3	8.9
C2-Fluorenes	0	0	26	20	0	0	25	34	0	0	0
C3-Fluorenes	0	0	0	0	0	0	0	26	0	0	0
Anthracene	21	2.4	25	24	3.1	1.2	24	34	12	5.7	55
Phenanthrene	13	7.2	12	12	17	6.5	16	11	13	11	58
C1-Phenanthrenes/anthracenes	12	5.1	12	12	8.8	3.7	12	15	12	12	84
C2-Phenanthrenes/anthracenes	18	0	19	20	0	0	25	28	17	34	150
C3-Phenanthrenes/anthracenes	0	0	24	0	0	0	0	32	16	38	120
C4-Phenanthrenes/anthracenes	0	0	0	0	0	0	0	34	0	0	240
Dibenzothiophene	1.8	0.79	1.2	1.2	1.5	0.56	1.3	1.4	<7.8	1.1	3.3
C1-Dibenzothiophenes	<1.8	2.1	4.7	<2	4.6	2.2	4.5	6.4	<1.9	3.6	14
C2-Dibenzothiophenes	<3.3	<1.5	<3.4	<3.8	<3.1	<2.2	<3.6	<3.3	<3.6	14	47
C3-Dibenzothiophenes	<5.8	<2.6	<5.9	<6.6	<5.4	<3.9	<6.3	<5.7	<6.4	18	52
Fluoranthene	12	3.6	12	11	6.5	3	12	14	14	44	770
Pyrene	9	5.7	11	7.9	7	4.8	11	12	11	40	700
C1-Fluoranthenes/pyrenes	160	0	350	330	10	5	410	440	84	100	630
C2-Fluoranthenes/pyrenes	66	0	140	110	16	0	150	190	47	67	190
C3-Fluoranthenes/pyrenes	23	0	34	0	0	0	57	54	19	24	78
Benzo[a]anthracene	27	<2.8	46	25	<5.8	<4.2	29	48	42	140	610
Chrysene	32	3	33	18	12	3.8	27	53	52	190	830
C1-Chrysenes	41	0	50	38	4.8	0	46	77	59	110	260
C2-Chrysenes	0	0	0	0	0	0	0	0	27	0	100
C3-Chrysenes	0	0	0	0	0	0	0	0	0	0	25
C4-Chrysenes	0	0	0	0	0	0	0	0	0	0	0
Benzo[b]fluoranthene	48	4	72	43	22	5.6	55	97	56	150	600
Benzo[k]fluoranthene	37	1.8	57	37	14	3.2	52	76	44	90	420
Benzo[e]pyrene	33	2.7	48	29	14	4.8	38	68	39	100	400
Benzo[a]pyrene	50	3.2	56	48	13	3	49	77	55	94	390
Perylene	14	<1.9	13	14	6.3	2.1	9.6	12	14	19	77
Indeno[1,2,3,-c,d]pyrene	9.3	1.7	17	13	12	4	15	17	8.6	12	110
Dibenzo[a,h]anthracene	3.2	0.44	4.2	3.1	2.4	<2.8	3.9	3.8	3	3.2	26
Benzo[g,h,i]perylene	13	1.8	22	16	12	4.1	19	23	12	16	130

Appendix K: Tissue PAH Data

Macoma nasuta

Station	1-5/6/7	4-1	4-2	4-3	4-5	Cruise IV Control	Cruise V Control	Zero Time IV	Zero Time V
Year	1999	1999	1999	1999	1999	1999	1999	1999	1999
PAHs (µg/kg)									
Naphthalene	21	17	22	18	18	20	18	20	8
C1-Naphthalenes	14	14	17	14	11	13	12	<8	<3
C2-Naphthalenes	<9.8	<12	<11	<11	<10	<9.6	<10	<11	<4
C3-Naphthalenes	<6.9	<8.3	<7.6	<8	<7.1	<6.7	<7.3	<7.4	<2.8
C4-Naphthalenes	<9.1	<11	<10	<10	<9.4	<8.8	<9.6	<9.8	<3.7
Acenaphthylene	<4.5	<5.5	<5	<5.3	<4.7	5	<4.8	<4.9	<1.9
Acenaphthene	3.4	<5.5	<5.1	<5.3	<4.8	5.5	<4.9	<5	<1.9
Biphenyl	4.7	<5.8	<5.3	<5.5	<5	7.1	<5.1	<5.1	<2
Fluorene	5.1	<6.9	<6.4	<6.7	<6	6.5	5.1	<6.2	<2.4
C1-Fluorenes	<3.9	<4.7	<4.4	<4.6	<4.1	<3.8	<4.2	<4.2	<1.6
C2-Fluorenes	<9.1	<11	<10	<10	<9.4	<8.8	<9.6	<9.8	<3.7
C3-Fluorenes	<9.1	<11	<10	<10	<9.4	<8.8	<9.6	<9.8	<3.7
Anthracene	2.5	<7.4	<6.8	<7.1	4.7	7.8	4.4	6.6	7.5
Phenanthrene	16	14	20	14	21	28	18	29	14
C1-Phenanthrenes/anthracenes	7.9	<11	<10	<10	<9.4	17	12	20	15
C2-Phenanthrenes/anthracenes	<9.1	<11	<10	<10	<9.4	29	<9.6	<9.8	<3.7
C3-Phenanthrenes/anthracenes	<9.1	<11	<10	<10	<9.4	29	<9.6	<9.8	<3.7
C4-Phenanthrenes/anthracenes	<9.1	<11	<10	<10	<9.4	<8.8	<9.6	<9.8	<3.7
Dibenzothiophene	<5.6	<6.7	<6.2	<6.4	<5.8	<5.4	<5.9	<6	<2.3
C1-Dibenzothiophenes	<1.4	<1.7	<1.5	<1.6	<1.4	<1.4	<1.5	<1.5	<0.57
C2-Dibenzothiophenes	<2.6	<3.1	<2.9	<3	<2.7	<2.5	<2.8	<2.8	<1.1
C3-Dibenzothiophenes	<4.5	<5.5	<5	<5.3	<4.7	<4.4	<4.8	<4.9	<1.9
Fluoranthene	11	7.8	12	9.8	12	28	16	58	51
Pyrene	7.6	7.5	13	11	12	26	16	40	38
C1-Fluoranthenes/pyrenes	<9.1	<11	<10	<10	<9.4	28	23	38	31
C2-Fluoranthenes/pyrenes	<9.1	<11	<10	<10	<9.4	<8.8	<9.6	<9.8	<3.7
C3-Fluoranthenes/pyrenes	<9.1	<11	<10	<10	<9.4	<8.8	<9.6	<9.8	<3.7
Benzo[a]anthracene	<4.9	<5.9	<5.4	<5.7	<5.1	<4.8	<5.2	<5.3	<2
Chrysene	<4.5	<5.4	<5	<5.2	<4.6	<4.3	<4.8	25	22
C1-Chrysenes	<9.1	<11	<10	<10	<9.4	<8.8	<9.6	<9.8	11
C2-Chrysenes	<9.1	<11	<10	<10	<9.4	<8.8	<9.6	<9.8	<3.7
C3-Chrysenes	<9.1	<11	<10	<10	<9.4	<8.8	<9.6	<9.8	<3.7
C4-Chrysenes	<9.1	<11	<10	<10	<9.4	<8.8	<9.6	<9.8	<3.7
Benzo[b]fluoranthene	2.4	<3.7	4.2	<3.6	3.1	11	7.2	19	13
Benzo[k]fluoranthene	4.7	<7.9	2.7	<7.6	2.4	6.7	3.7	9.6	6.5
Benzo[e]pyrene	2.3	<4.9	<4.5	<4.7	<4.2	8.2	6	12	10
Benzo[a]pyrene	1.7	<4.2	<3.8	<4	<3.6	3.3	2.6	7.7	4.6
Perylene	1.6	<4.2	<3.8	<4	<3.6	3	2.2	4.7	3.9
Indeno[1,2,3,-c,d]pyrene	<3.4	<4.2	<3.8	<4	<3.6	3.5	2.8	7.8	<1.4
Dibenzo[a,h]anthracene	<3.3	<3.9	<3.6	<3.8	<3.4	2.4	<3.5	<3.5	<1.3
Benzo[g,h,i]perylene	1.7	<4.1	<3.8	18	<3.5	5.6	2.9	7.3	4.3

Appendix K: Tissue PAH Data

Nephtys caecoides

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-7	1-8	4-6	Cruise II Control	Zero Time II
Year	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998	1998
PAHs (µg/kg)											
Naphthalene	9.1	13	8.4	11	14	14	8.8	10	17	8.4	7.4
C1-Naphthalenes	7.4	9.7	5.9	8.8	8.9	9.4	7.3	8	10	5.1	8.9
C2-Naphthalenes	15	20	7.8	16	13	18	16	12	19	9.2	13
C3-Naphthalenes	<5.4	<8.8	<5	<6	<8.8	<8.8	<5.8	<5.8	<6.1	<6	<5
C4-Naphthalenes	0	0	0	0	0	0	0	0	0	0	0
Acenaphthylene	1.1	3.2	0.64	1	4.2	7.1	1.1	1.3	2.6	1.7	1.1
Acenaphthene	2	1.6	0.92	1.8	1.5	1.9	1.1	2	1.6	1.5	1.5
Biphenyl	5.4	6.6	4.4	5	5.9	8.6	4.3	9.4	5.7	2.5	2.8
Fluorene	2.9	3.8	1.7	3.7	4.1	4.1	2.5	3.1	4.3	2.1	3.3
C1-Fluorenes	5.2	8.8	4.3	4.7	9.3	9	5.1	3.3	5.6	4	5
C2-Fluorenes	0	20	0	0	25	23	0	12	0	0	0
C3-Fluorenes	0	0	0	0	0	0	0	0	0	0	0
Anthracene	3	21	2	2.6	26	27	3.1	3	3.8	3.6	1.5
Phenanthrene	12	15	8	11	18	20	9.3	11	18	12	13
C1-Phenanthrenes/anthracenes	8.8	14	5.2	8	14	12	9.6	6.7	9	9.2	7.9
C2-Phenanthrenes/anthracenes	0	25	0	0	21	23	0	0	0	0	0
C3-Phenanthrenes/anthracenes	0	28	0	0	0	27	0	0	0	0	0
C4-Phenanthrenes/anthracenes	0	36	0	0	0	0	0	0	0	0	0
Dibenzothiophene	1.2	1.7	0.64	1	1.4	1.7	0.75	0.98	1.1	0.98	1.3
C1-Dibenzothiophenes	3.4	5	2.2	2.8	3.7	4	4.5	3.7	2.9	2.6	2.7
C2-Dibenzothiophenes	<2	<3.3	<1.9	<2.2	<3.3	<3.3	16	<2.2	<2.3	<2.3	<1.9
C3-Dibenzothiophenes	<3.5	<5.8	<3.3	<3.9	<5.8	<5.8	14	<3.8	<4	<4	<3.3
Fluoranthene	5.8	15	4.1	3.8	19	13	4.3	5.7	10	9	4.5
Pyrene	6.8	11	6	5.2	12	11	8.4	8.1	12	13	5.2
C1-Fluoranthenes/pyrenes	0	180	5	6.1	310	410	10	14	6.6	5.1	4.4
C2-Fluoranthenes/pyrenes	0	89	0	0	110	160	0	19	0	0	0
C3-Fluoranthenes/pyrenes	0	29	0	0	36	41	0	24	0	0	0
Benzo[a]anthracene	<3.8	37	<3.5	<4.2	40	27	<4.1	<4.2	<4.3	<4.3	<3.6
Chrysene	2.8	54	1.7	2.1	46	24	2.9	<3.8	5.1	8.4	2.6
C1-Chrysenes	0	59	0	0	66	42	0	12	3.1	0	2.4
C2-Chrysenes	0	0	0	0	52	0	0	0	0	0	0
C3-Chrysenes	0	0	0	0	0	0	0	0	0	0	0
C4-Chrysenes	0	0	0	0	0	0	0	0	0	0	0
Benzo[b]fluoranthene	22	73	1.4	<2.7	71	42	6.8	6	3.3	2.5	1.4
Benzo[k]fluoranthene	<5.1	53	1.1	4	58	35	<5.5	1.4	2.5	1.3	0.9
Benzo[e]pyrene	<3.2	46	7.2	3.7	46	29	3.2	3.9	2.9	8.5	6.9
Benzo[a]pyrene	<2.7	58	1.2	7.8	60	43	6.2	4.4	4.1	3.4	4.7
Perylene	<2.7	9	<2.5	<3	11	6.8	2.4	<2.9	<3	0.83	2.1
Indeno[1,2,3,-c,d]pyrene	<2.7	17	<2.5	<3	15	10	<2.9	1.2	1.4	0.85	0.99
Dibenzo[a,h]anthracene	<2.5	4	<2.3	<2.8	3.9	3.3	<2.7	<2.8	<2.9	<2.9	0.67
Benzo[g,h,i]perylene	<2.6	20	0.59	<2.9	19	14	2.9	7.5	2	1.2	1.1

Nephtys caecoides

Station	1-5/6/7	4-1	4-2	4-3	4-5	Cruise IV Control	Cruise V Control	Zero Time IV	Zero Time V
Year	1999	1999	1999	1999	1999	1999	1999	1999	1999
PAHs (µg/kg)									
Naphthalene	14	10	11	12	11	9.4	11	22	5.5
C1-Naphthalenes	14	8.1	8.5	12	9.5	7.4	9.6	9.7	9.6
C2-Naphthalenes	<7.3	<7.2	<6.8	<6.9	<6.5	<5.6	<6.9	<7.8	<3.8
C3-Naphthalenes	<5.1	<5	<4.8	<4.8	<4.6	<3.9	<4.8	<5.4	<2.6
C4-Naphthalenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
Acenaphthylene	<3.3	<3.3	<3.1	<3.2	<3	<2.6	<3.2	<3.6	<1.7
Acenaphthene	<3.4	<3.3	<3.2	<3.2	<3	<2.6	<3.2	<3.6	<1.8
Biphenyl	3.8	<3.5	<3.3	<3.3	<3.2	<2.7	<3.4	<3.8	<1.8
Fluorene	3.8	<4.2	<4	4	<3.8	<3.3	3.9	<4.5	<2.2
C1-Fluorenes	<2.9	<2.8	<2.7	<2.7	<2.6	<2.2	<2.8	<3.1	<1.5
C2-Fluorenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
C3-Fluorenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
Anthracene	2.7	4.7	2.6	3.9	3.7	3.1	3.5	<4.8	3.4
Phenanthrene	16	7.9	9	12	9.6	8.9	15	11	9
C1-Phenanthrenes/anthracenes	9.4	<6.6	<6.3	7.1	<6	<5.2	10	<7.2	9
C2-Phenanthrenes/anthracenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
C3-Phenanthrenes/anthracenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
C4-Phenanthrenes/anthracenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
Dibenzothiophene	<4.1	<4	<3.8	<3.9	<3.7	<3.2	<3.9	<4.4	<2.1
C1-Dibenzothiophenes	<1	<1	<0.96	<0.97	<0.92	<0.79	<0.98	<1.1	<0.53
C2-Dibenzothiophenes	<1.9	<1.9	<1.8	<1.8	<1.7	<1.5	<1.8	<2	<0.99
C3-Dibenzothiophenes	<3.3	<3.3	<3.1	<3.2	<3	<2.6	<3.2	<3.6	<1.7
Fluoranthene	8.5	4.8	4.9	6	5.4	5.6	8	5.8	5.1
Pyrene	8.5	8	7.6	10	7.1	9.4	10	10	8
C1-Fluoranthenes/pyrenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
C2-Fluoranthenes/pyrenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
C3-Fluoranthenes/pyrenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
Benzo[a]anthracene	<3.6	<3.6	<3.4	13	<3.2	<2.8	<3.4	<3.9	<1.9
Chrysene	<3.3	<3.2	<3.1	<3.1	<3	<2.6	<3.1	<3.5	<1.7
C1-Chrysenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
C2-Chrysenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
C3-Chrysenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
C4-Chrysenes	<6.7	<6.6	<6.3	<6.3	<6	<5.2	<6.4	<7.2	<3.5
Benzo[b]fluoranthene	<2.3	<2.2	<2.1	<2.1	<2	2.2	<2.2	<2.4	<1.2
Benzo[k]fluoranthene	<4.8	<4.7	<4.5	<4.5	<4.3	1.3	<4.6	<5.2	<2.5
Benzo[e]pyrene	<3	<2.9	<2.8	<2.8	<2.7	1.6	<2.8	<3.2	<1.5
Benzo[a]pyrene	<2.5	<2.5	<2.4	<2.4	<2.3	<2	<2.4	<2.7	<1.3
Perylene	<2.5	<2.5	<2.4	<2.4	<2.3	<2	<2.4	<2.7	<1.3
Indeno[1,2,3,-c,d]pyrene	<2.5	<2.5	<2.4	<2.4	<2.3	<2	<2.4	<2.7	<1.3
Dibenzo[a,h]anthracene	<2.4	<2.4	<2.3	<2.3	<2.2	<1.9	<2.3	<2.6	<1.2
Benzo[g,h,i]perylene	<2.5	<2.5	<2.3	<2.4	<2.2	<1.9	<2.4	<2.7	<1.3

Appendix L. Tissue Metals Data

Macoma nasuta

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-7	1-8	4-6	Cruise II Control	Zero Time II
Year	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**
Metals (µg/g, except where noted)											
Cadmium	0.84	0.64	0.7	0.66	0.66	0.84	0.37	0.34	0.65	0.22	0.3
Chromium	4.84	12.5	7.98	4.21	5.4	6.86	22.1	7.64	4.54	21.7	28.9
Copper	101	75.5	85.5	96.8	105	102	54	92	95.9	143	80.8
Iron	1040	710	637	514	896	595	864	880	571	721	1710
Mercury	0.902	0.718	0.824	0.928	1.01	0.772	0.726	0.978	0.843	0.479	0.486
Lead	3.99	4.68	5.13	4.81	5.12	4.1	3.34	4.84	5.29	2.66	2.68
Nickel	9.97	14.1	13.3	11	11.2	11.8	23.7	20.5	11.6	15.7	23.1
Silver	0.495	0.445	0.464	0.447	0.438	0.535	0.242	0.378	0.5	0.541	0.389
Zinc	257	281	231	322	285	244	235	313	245	227	233
Dry Weight (%)	9.775	16.95	10	9.56	11.7	12.9	9.72	9.88	9.595	9.87	10.75
Lipid (%)	6.99	8.6	5.9	5.02	10.7	8.96	5.16	5.76	5.76	3.59	4.89

Station	1-5/6/7	4-1	4-2	4-3	4-5	Cruise IV Control	Cruise V Control	Zero Time IV	Zero Time V
Year	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**
Metals (µg/g, except where noted)									
Cadmium	1.76	0.48	0.48	0.46	0.46	0.64	0.9	0.42	
Chromium	3.6	3.1	2.6	2.7	3.1	2.5	5	6.4	
Copper	172	58.2	46.8	56.9	44.4	41.8	160	68.4	
Iron	518	568	484	495	525	543	1010	1150	
Mercury	0.283	0.228	0.212	0.281	0.227	0.186	0.305	0.244	
Lead	3.27	2.92	2.48	2.6	2.12	2.95	4.6	3.48	
Nickel	10.3	9.4	8.5	8.6	7.9	6.4	10.5	9.5	
Silver	0.39	0.32	0.27	0.27	0.31	0.21	0.59	0.31	
Zinc	263	258	247	243	288	298	474	200	
Dry Weight (%)	12	10.6	11.3	11.2	11	13.6	11.7	11.8	10.6
Lipid (%)	4.555	5.025	5.095	5.005	4.505	6.845	4.465	6.385	6.07

Appendix L: Tissue Metals Data

Nephtys caecoides

Station	1-1	1-2	1-3	1-4	1-5	1-6	1-7	1-8	4-6	Cruise II Control	Zero Time II
Year	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**	**1998**
Metals (µg/g, except where noted)											
Cadmium	1.5	1.23	1.19	1.06	0.96	1.98	0.68	1.34	0.79	0.67	0.84
Chromium	1.42	7.03	5.39	15.9	4.1	1.61	1.16	1.38	6.92	0.82	32.4
Copper	7.95	12.2	5.31	5.32	7.06	13.7	8.67	9.06	8.05	10.7	5.19
Iron	499	514	439	413	418	439	363	408	395	537	1190
Mercury	0.179	0.101	0.121	0.133	0.15	0.082	0.158	0.112	0.119	0.076	0.057
Lead	0.31	0.65	0.68	1.19	2.27	0.87	1.01	1.32	0.21	0.3	1.51
Nickel	3.68	8.68	5.43	9.98	4.1	5.28	2.49	2.97	6.12	2.16	22.6
Silver	0.08	0.054	0.067	0.054	0.034	0.095	0.07	0.06	0.043	0.097	0.019
Zinc	220	207	216	207	228	258	193	244	189	198	138
Dry Weight (%)	17	14.55	17.7	14.6	12.6	13.6	16.25	16.7	15.15	15.95	18.2
Lipid (%)	12.8	4.92	11.5	11.2	5.6	5.32	11.3	12.8	11.1	6.9	9.67

Station	1-5/6/7	4-1	4-2	4-3	4-5	Cruise IV Control	Cruise V Control	Zero Time IV	Zero Time V
Year	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**	**1999**
Metals (µg/g, except where noted)									
Aluminum	33.8	37	27.5	45.5	19.4	33.6	75	12.5	
Cadmium	3.18	1.22	1.09	1.3	1.26	1.3	0.99	0.94	
Chromium	0.5	0.4	0.3	0.3	0.3	0.5	0.7	0.4	
Copper	78.5	8.5	9	9.3	8.1	23.5	21.9	6.7	
Iron	318	378	354	377	297	417	415	338	
Mercury	0.035	0.069	0.057	0.068	0.056	0.046	0.04	0.045	
Lead	0.27	0.35	0.33	0.38	0.3	0.67	0.26	0.3	
Nickel	2	2.2	1.8	2.1	2	3.4	1.5	1.7	
Silver	0.29	0.09	0.08	0.09	0.08	0.18	0.17	0.05	
Zinc	360	223	213	227	214	235	199	156	
Dry Weight (%)	17	18.1	16.2	17.4	18.3	16.7	18.2	16.5	14.8
Lipid (%)	9.955	10.23	9.52	11.86	11.35	8.885	8.84	9.205	9.83

Appendix M. Literature Survey

The scientific literature was examined for useful references prior to completion of the 1994 SINKEX draft report. The principal literature reference source for this earlier survey was the Aquatic Sciences and Fisheries Abstracts database accessed at the Scripps Institute of Oceanography Library. Since the completion of the initial literature search, additional literature searches were performed. The literature was examined for useful references to include the period from the late 1970s to the first quarter of 2002. The literature databases examined were: 1) the Aquatic Sciences and Fisheries Abstracts; 2) Current Contents; 3) Sea Grant Publications; and 4) the Marine, Oceanographic, and Freshwater Resources database . The keywords used initially were:

- (PCB or PCBs)
- polychlorinated
- (organochlorine or organochlorines)
- chlorinated hydrocarbons
- (biphenyl or biphenyls)

The keyword search was also combined to use the compound form:

- (PCB or PCBs) and (marine or seawater) or (ocean or estuarine)

This combined search term was applied in order to better focus on the oceanic-marine-estuarine literature since the general terms were returning many off-topic references such as laboratory mammalian studies and terrestrial data. Searches included the keywords in all record fields where abstracts, summaries, and other data fields were provided as opposed to simply the title field.

The literature survey also focused on physical PCB subjects such as felt material, coatings, and insulation. A list of databases searched where information was found are listed below.

- INSPEC 1969-1997/Jan W4; (c) 1997 Institution of Electrical Engineers
- Ei Compendex(R) 1970-1997/Mar W1; (c) 1997 Engineering Info. Inc.
- World Surface Coatings Abs 1976-1996/Nov; (c) 1996 Paint Research Assn.
- METADEX(R) 1966-1997/Feb B2; (c) 1997 Cambridge Scientific Abs
- Aluminium Ind Abs 1968-1997/Feb; (c) 1997 Cambridge Scientific Abs
- Inside Conferences 1993-1996; (c) 1996 BLDSC all rts. reserv.
- JICST-EPlus 1985-1997/Dec W5; (c) 1997 Japan Science and Tech Corp(JST)
- Wilson Appl. Sci & Tech Abs 1983-1997/Dec; (c) 1997 The HW Wilson Co.
- Pascal 1973-1997/Jan; (c) 1997 INIST/CNRS
- Analytical Abstracts 1980-1997/Feb; (c) 1997 Royal Soc Chemistry
- ChemEng & Biotec Abs 1970-1997/Jan; (c)1997 RoySocChm,DECHEMA,FizChemie
- Chemical Safety NewsBase 1981-1997/Feb; (c) 1997 Royal Soc Chemistry
- Polymer Online; (c) 1990 John Wiley & Sons Inc.

- RAPRA Rubber & Plastics 1972-1997/Jan B3; (c) 1997 RAPRA Technol Ltd
- CLAIMS(R)/US PATENTS ABS 1950-1996/NOV; (c) 1997 IFI/Plenum Data Corp.
- Current BioTech Abs 1983-1997/Feb; Royal Soc Chem & DECHEMA
- Scisearch(R) Cited Ref Sci 1974-1997/Jan W3; (c) 1997 Inst for Sci Info
- CA SEARCH(R) 1967-1996/UD=12605; (c) 1997 American Chemical Society
- NTIS 64-1997/Jan W1; Comp&distr 1997 NTIS, Intl Copyright All Rights Re
- Energy SciTec 1974-1997/Nov B2; (c) 1997 Contains copyrighted material
- Aerospace Database 1962-1997/Dec; (c) 1997 AIAA
- PIRA 1975-1997Feb W2

The processing of the literature search results was accomplished in the following manner:

1. Access data from databases and save as text files.
2. Create and run scripts written with data conversion software (Data Junction Cambio V.6.0-7.0) to convert text files to suitable import format (i.e., ASCII delimited files).
3. Merge data into an import table for MS Access and remove duplicates.
4. Merge import table with existing MS Access PCB database and again remove duplicates. The current SINKEX bibliographic database exists in MicroSoft Office 97 Access format. It is fully searchable and output files can be created by use of appropriate queries and report formats.

Confidence in the bibliographic database content increased as the degree of returned duplicate records increased, indicating that the sources and search terms used were effective. Additional records were added when found referenced in documents accessed, or while not directly concerned with PCBs, their content was relevant to SINKEX. The total record count is currently at 2757 references, of which 603 are held as hardcopy documents.

Appendix N. Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX ADCP I	2SX_ADCP	None	Seawater	Regular	1998.09.06	32.758092	-119.591412
SINKEX ADCP I	2SX_ADCP	None	Seawater	Regular	1998.09.06	32.758092	-119.591412
SINKEX 4-1	2SX0001	1	Sediment	Regular	1998.09.07	32.765967	-119.592994
SINKEX 4-1	2SX0001	1	Sediment	Regular	1998.09.07	32.765967	-119.592994
SINKEX 1-1	2SX0002	None	Sediment	Composite	1998.09.08	32.756996	-119.593274
SINKEX 1-1	2SX0002	None	Sediment	Composite	1998.09.08	32.756996	-119.593274
SINKEX 1-4	2SX0003	None	Sediment	Composite	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0003	None	Sediment	Composite	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0003.11	None	Sediment	Regular	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0003.11	None	Sediment	Regular	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0004	None	Sediment	Regular	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0004	None	Sediment	Regular	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0005	1	Sediment	Regular	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0005	1	Sediment	Regular	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0006	None	Sediment	Regular	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0006	None	Sediment	Regular	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0007	None	Sediment	Regular	1998.09.08	32.756839	-119.592758
SINKEX 1-4	2SX0007	None	Sediment	Regular	1998.09.08	32.756839	-119.592758
SINKEX 1-7	2SX0008	None	Sediment	Composite	1998.09.08	32.756813	-119.593564
SINKEX 1-7	2SX0008	None	Sediment	Composite	1998.09.08	32.756813	-119.593564
SINKEX 1-7	2SX0008.11	None	Sediment	Regular	1998.09.08	32.756813	-119.593564
SINKEX 1-7	2SX0008.11	None	Sediment	Regular	1998.09.08	32.756813	-119.593564
SINKEX 1-1	2SX0102	1	Sediment	Regular	1998.09.08	32.756996	-119.593274
SINKEX 1-1	2SX0102	1	Sediment	Regular	1998.09.08	32.756996	-119.593274
SINKEX 1-1	2SX0103	1	Sediment	Regular	1998.09.08	32.756996	-119.593274
SINKEX 1-1	2SX0103	1	Sediment	Regular	1998.09.08	32.756996	-119.593274
SINKEX II MISC	T1	None	Sediment	Regular	1998.09.08	32.756901	-119.593028
SINKEX II MISC	T1	None	Sediment	Regular	1998.09.08	32.756901	-119.593028
SINKEX 1-3	2SX0009	1	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0009	1	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0010	1	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0010	1	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0011	None	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0011	None	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0012	1	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0012	1	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0013	None	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0013	None	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0014	None	Sediment	Split	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0014	None	Sediment	Split	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0014DUP	None	Sediment	Split	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0014DUP	None	Sediment	Split	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0015	None	Sediment	Composite	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0015	None	Sediment	Composite	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0015.b	1	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-3	2SX0015.b	1	Sediment	Regular	1998.09.09	32.756951	-119.59276
SINKEX 1-8	2SX0016	None	Sediment	Composite	1998.09.09	32.757009	-119.593544
SINKEX 1-8	2SX0016	None	Sediment	Composite	1998.09.09	32.757009	-119.593544
SINKEX 1-5	2SX0017	1	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0017	1	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0018	None	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0018	None	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0019	None	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0019	None	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0020	None	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0020	None	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0021	1	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0021	1	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0022	1	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0022	1	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0023	None	Sediment	Composite	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0023	None	Sediment	Composite	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0023.11	None	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-5	2SX0023.11	None	Sediment	Regular	1998.09.11	32.756833	-119.593051
SINKEX 1-2	2SX0024	None	Sediment	Composite	1998.09.11	32.756988	-119.593058
SINKEX 1-2	2SX0024	None	Sediment	Composite	1998.09.11	32.756988	-119.593058

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 1-2	2SX0024.11	None	Sediment	Regular	1998.09.11	32.756988	-119.593058
SINKEX 1-2	2SX0024.11	None	Sediment	Regular	1998.09.11	32.756988	-119.593058
SINKEX 1-6	2SX0025	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0025	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0025Re-Analysis	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0025Re-Analysis	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0025Re-Analysis2	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0025Re-Analysis2	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0026	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0026	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0027	1	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0027	1	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0028	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0028	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0029	1	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0029	1	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030	1	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030	1	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 .5-1cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 .5-1cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 0-.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 0-.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 1.5-2cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 1.5-2cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 1-1.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 1-1.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 2.5-3cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 2.5-3cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 2-2.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 2-2.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 3-4cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 3-4cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 4-5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 4-5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 5-6cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 5-6cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 6-7cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 6-7cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 7-8cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0030 7-8cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0031	None	Sediment	Composite	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0031	None	Sediment	Composite	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0031.11	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0031.11	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0031.b	1	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0031.b	1	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 4-6	2SX0036	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0036	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0037	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0037	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0037Re-Analysis	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0037Re-Analysis	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0037Re-Analysis2	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0037Re-Analysis2	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0038	1	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0038	1	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0039	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0039	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0041	None	Sediment	Composite	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0041	None	Sediment	Composite	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0041.11	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 4-6	2SX0041.11	None	Sediment	Regular	1998.09.11	32.750609	-119.600484
SINKEX 1-6	2SX0052 .5-1cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 .5-1cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 0-.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 0-.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 1.5-2cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 1.5-2cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 10-11cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 10-11cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 1-1.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 1-6	2SX0052 1-1.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 2.5-3cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 2.5-3cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 2-2.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 2-2.5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 3-4cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 3-4cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 4-5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 4-5cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 5-6cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 5-6cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 6-7cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 6-7cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 7-8cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 7-8cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 8-9cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 8-9cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 9-10cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX 1-6	2SX0052 9-10cm	None	Sediment	Regular	1998.09.11	32.756827	-119.593272
SINKEX STN I	2SX0104	1	Sediment	Regular	1998.09.12	32.758813	-119.592448
SINKEX STN I	2SX0104	2	Sediment	Regular	1998.09.12	32.758813	-119.592448
SINKEX STN I	2SX0104	3	Sediment	Regular	1998.09.12	32.758813	-119.592448
SINKEX STN I	2SX0104	4	Sediment	Regular	1998.09.12	32.758813	-119.592448
SINKEX STN I	2SX0104	1	Sediment	Regular	1998.09.12	32.758813	-119.592448
SINKEX STN I	2SX0104	2	Sediment	Regular	1998.09.12	32.758813	-119.592448
SINKEX STN I	2SX0104	3	Sediment	Regular	1998.09.12	32.758813	-119.592448
SINKEX STN I	2SX0104	4	Sediment	Regular	1998.09.12	32.758813	-119.592448
SINKEX STS I	2SX0106	1	Sediment	Regular	1998.09.12	32.755485	-119.592778
SINKEX STS I	2SX0106	2	Sediment	Regular	1998.09.12	32.755485	-119.592778
SINKEX STS I	2SX0106	3	Sediment	Regular	1998.09.12	32.755485	-119.592778
SINKEX STS I	2SX0106	4	Sediment	Regular	1998.09.12	32.755485	-119.592778
SINKEX STS I	2SX0106	1	Sediment	Regular	1998.09.12	32.755485	-119.592778
SINKEX STS I	2SX0106	2	Sediment	Regular	1998.09.12	32.755485	-119.592778
SINKEX STS I	2SX0106	3	Sediment	Regular	1998.09.12	32.755485	-119.592778
SINKEX STS I	2SX0106	4	Sediment	Regular	1998.09.12	32.755485	-119.592778
SINKEX 1-8	2SX0042	None	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0042	None	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0043	None	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0043	None	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0044	None	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0044	None	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0045	1	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0045	1	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0046	1	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0046	1	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0047	1	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-8	2SX0047	1	Sediment	Regular	1998.09.13	32.757009	-119.593544
SINKEX 1-7	2SX0048	None	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0048	None	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0049	None	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0049	None	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0050	None	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0050	None	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0050Re-Analysis	None	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0050Re-Analysis	None	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0050Re-Analysis2	None	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0050Re-Analysis2	None	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0051	1	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0051	1	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0052	1	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0052	1	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0053	1	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-7	2SX0053	1	Sediment	Regular	1998.09.13	32.756813	-119.593564
SINKEX 1-2	2SX0054	None	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-2	2SX0054	None	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-2	2SX0055	None	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-2	2SX0055	None	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-2	2SX0056	None	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-2	2SX0056	None	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-2	2SX0057	1	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-2	2SX0057	1	Sediment	Regular	1998.09.13	32.756988	-119.593058

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 1-2	2SX0058	1	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-2	2SX0058	1	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-2	2SX0059	1	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-2	2SX0059	1	Sediment	Regular	1998.09.13	32.756988	-119.593058
SINKEX 1-1	2SX0060	None	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0060	None	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0061	None	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0061	None	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0062	None	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0062	None	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0063	1	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0063	1	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0064	1	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0064	1	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0065	1	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX 1-1	2SX0065	1	Sediment	Regular	1998.09.13	32.756996	-119.593274
SINKEX II MISC	2SX9999-MISC	None	Organism	Assorted	1998.09.13	32.756901	-119.593028
SINKEX II MISC	2SX9999-MISC	None	Organism	Assorted	1998.09.13	32.756901	-119.593028
SINKEX Fish Trap 3	2SX0066	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0066	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0066.600	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0066.600	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0066Re-Analysis	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0066Re-Analysis	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0066Re-Analysis2	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0066Re-Analysis2	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0067	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0067	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0067.600	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 3	2SX0067.600	None	Anoplopoma fimbria	Regular	1998.09.14	32.757057	-119.593525
SINKEX Fish Trap 4	2SX0068	None	Lithodes couesi	Regular	1998.09.14	32.756797	-119.593488
SINKEX Fish Trap 4	2SX0068	None	Lithodes couesi	Regular	1998.09.14	32.756797	-119.593488
SINKEX Fish Trap 4	2SX0068.600	None	Lithodes couesi	Regular	1998.09.14	32.756797	-119.593488
SINKEX Fish Trap 4	2SX0068.600	None	Lithodes couesi	Regular	1998.09.14	32.756797	-119.593488
SINKEX Fish Trap 4	2SX0068RE	None	Lithodes couesi	Re-Analysis	1998.09.14	32.756797	-119.593488
SINKEX Fish Trap 4	2SX0069	None	Lithodes couesi	Regular	1998.09.14	32.756797	-119.593488
SINKEX Fish Trap 4	2SX0069	None	Lithodes couesi	Regular	1998.09.14	32.756797	-119.593488
SINKEX Fish Trap 4	2SX0069.600	None	Lithodes couesi	Regular	1998.09.14	32.756797	-119.593488
SINKEX Fish Trap 4	2SX0069.600	None	Lithodes couesi	Regular	1998.09.14	32.756797	-119.593488
SINKEX Fish Trap 5	2SX0070	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0070	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0070.600	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0070.600	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0070Re-Analysis	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0070Re-Analysis	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0070Re-Analysis2	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0070Re-Analysis2	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0071	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0071	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0071.600	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX Fish Trap 5	2SX0071.600	None	Anoplopoma fimbria	Regular	1998.09.14	32.80828333	-119.6264
SINKEX STE I	2SX0105	1	Sediment	Regular	1998.09.14	32.757793	-119.589982
SINKEX STE I	2SX0105	2	Sediment	Regular	1998.09.14	32.757793	-119.589982
SINKEX STE I	2SX0105	3	Sediment	Regular	1998.09.14	32.757793	-119.589982
SINKEX STE I	2SX0105	4	Sediment	Regular	1998.09.14	32.757793	-119.589982
SINKEX STE I	2SX0105	1	Sediment	Regular	1998.09.14	32.757793	-119.589982
SINKEX STE I	2SX0105	2	Sediment	Regular	1998.09.14	32.757793	-119.589982
SINKEX STE I	2SX0105	3	Sediment	Regular	1998.09.14	32.757793	-119.589982
SINKEX STE I	2SX0105	4	Sediment	Regular	1998.09.14	32.757793	-119.589982
SINKEX STW I	2SX0107	1	Sediment	Regular	1998.09.14	32.75709	-119.595008
SINKEX STW I	2SX0107	2	Sediment	Regular	1998.09.14	32.75709	-119.595008
SINKEX STW I	2SX0107	3	Sediment	Regular	1998.09.14	32.75709	-119.595008
SINKEX STW I	2SX0107	4	Sediment	Regular	1998.09.14	32.75709	-119.595008
SINKEX STW I	2SX0107	1	Sediment	Regular	1998.09.14	32.75709	-119.595008
SINKEX STW I	2SX0107	2	Sediment	Regular	1998.09.14	32.75709	-119.595008
SINKEX STW I	2SX0107	3	Sediment	Regular	1998.09.14	32.75709	-119.595008
SINKEX STW I	2SX0107	4	Sediment	Regular	1998.09.14	32.75709	-119.595008
SINKEX 1-4	2SX0003.b	1	Sediment	Regular	1998.09.15	32.756839	-119.592758
SINKEX 1-4	2SX0003.b	1	Sediment	Regular	1998.09.15	32.756839	-119.592758
SINKEX 4-6	2SX0041.b	1	Sediment	Regular	1998.09.15	32.750609	-119.600484
SINKEX 4-6	2SX0041.b	1	Sediment	Regular	1998.09.15	32.750609	-119.600484

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 1-4	2SX0072	1	Sediment	Regular	1998.09.15	32.756839	-119.592758
SINKEX 1-4	2SX0072	1	Sediment	Regular	1998.09.15	32.756839	-119.592758
SINKEX 1-4	2SX0073	1	Sediment	Regular	1998.09.15	32.756839	-119.592758
SINKEX 1-4	2SX0073	1	Sediment	Regular	1998.09.15	32.756839	-119.592758
SINKEX 1-4NB	2SX0074	None	Anoplopoma fimbria	Regular	1998.09.15	32.756848	-119.592755
SINKEX 1-4NB	2SX0074	None	Anoplopoma fimbria	Regular	1998.09.15	32.756848	-119.592755
SINKEX 1-4NB	2SX0074.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.756848	-119.592755
SINKEX 1-4NB	2SX0074.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.756848	-119.592755
SINKEX 1-5NB	2SX0075	None	Lithodes couesi	Regular	1998.09.15	32.756841	-119.593048
SINKEX 1-5NB	2SX0075	None	Lithodes couesi	Regular	1998.09.15	32.756841	-119.593048
SINKEX 1-5NB	2SX0075.600	None	Lithodes couesi	Regular	1998.09.15	32.756841	-119.593048
SINKEX 1-5NB	2SX0075.600	None	Lithodes couesi	Regular	1998.09.15	32.756841	-119.593048
SINKEX 1-5NB	2SX0075RE	None	Lithodes couesi	Re-Analysis	1998.09.15	32.756841	-119.593048
SINKEX PB	2SX0076	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0076	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0076.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0076.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0077	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0077	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0077.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0077.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0077Re-Analysis	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0077Re-Analysis	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0077Re-Analysis2	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0077Re-Analysis2	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0078	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0078	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0078.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0078.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0079	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0079	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0079.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0079.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0080	1	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0080	1	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0080.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0080.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0081	1	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0081	1	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0081.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX PB	2SX0081.600	None	Anoplopoma fimbria	Regular	1998.09.15	32.75701	-119.592944
SINKEX 1-1	2SX0002.b	1	Sediment	Regular	1998.09.16	32.756996	-119.593274
SINKEX 1-1	2SX0002.b	1	Sediment	Regular	1998.09.16	32.756996	-119.593274
SINKEX 1-8	2SX0016.b	1	Sediment	Regular	1998.09.16	32.757009	-119.593544
SINKEX 1-8	2SX0016.b	1	Sediment	Regular	1998.09.16	32.757009	-119.593544
SINKEX 1-5	2SX0023.b	1	Sediment	Regular	1998.09.16	32.756833	-119.593051
SINKEX 1-5	2SX0023.b	1	Sediment	Regular	1998.09.16	32.756833	-119.593051
SINKEX 1-2	2SX0024.b	1	Sediment	Regular	1998.09.16	32.756988	-119.593058
SINKEX 1-2	2SX0024.b	1	Sediment	Regular	1998.09.16	32.756988	-119.593058
SINKEX 1-8	2SX0082	1	Sediment	Regular	1998.09.16	32.757009	-119.593544
SINKEX 1-8	2SX0082	1	Sediment	Regular	1998.09.16	32.757009	-119.593544
SINKEX 1-8	2SX0083	1	Sediment	Regular	1998.09.16	32.757009	-119.593544
SINKEX 1-8	2SX0083	1	Sediment	Regular	1998.09.16	32.757009	-119.593544
SINKEX 1-2	2SX0084	1	Sediment	Regular	1998.09.16	32.756988	-119.593058
SINKEX 1-2	2SX0084	1	Sediment	Regular	1998.09.16	32.756988	-119.593058
SINKEX 1-2	2SX0085	1	Sediment	Regular	1998.09.16	32.756988	-119.593058
SINKEX 1-2	2SX0085	1	Sediment	Regular	1998.09.16	32.756988	-119.593058
SINKEX 1-3	2SX0086	1	Sediment	Regular	1998.09.16	32.756951	-119.59276
SINKEX 1-3	2SX0086	1	Sediment	Regular	1998.09.16	32.756951	-119.59276
FIT	2SX0087	1	Sediment	Regular	1998.09.16		
FIT	2SX0087	1	Sediment	Regular	1998.09.16		
FIT	2SX0088	1	Sediment	Regular	1998.09.16		
FIT	2SX0088	1	Sediment	Regular	1998.09.16		
SINKEX 4-6	2SX0089	1	Sediment	Regular	1998.09.16	32.750609	-119.600484
SINKEX 4-6	2SX0089	1	Sediment	Regular	1998.09.16	32.750609	-119.600484
SINKEX 1-7	2SX0008.b	1	Sediment	Regular	1998.09.17	32.756813	-119.593564
SINKEX 1-7	2SX0008.b	1	Sediment	Regular	1998.09.17	32.756813	-119.593564
SINKEX 1-6	2SX0090	1	Sediment	Regular	1998.09.17	32.756827	-119.593272
SINKEX 1-6	2SX0090	1	Sediment	Regular	1998.09.17	32.756827	-119.593272
SINKEX 1-6	2SX0091	1	Sediment	Regular	1998.09.17	32.756827	-119.593272
SINKEX 1-6	2SX0091	1	Sediment	Regular	1998.09.17	32.756827	-119.593272

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 1-7	2SX0092	1	Sediment	Regular	1998.09.17	32.756813	-119.593564
SINKEX 1-7	2SX0092	1	Sediment	Regular	1998.09.17	32.756813	-119.593564
SINKEX 1-7	2SX0093	1	Sediment	Regular	1998.09.17	32.756813	-119.593564
SINKEX 1-7	2SX0093	1	Sediment	Regular	1998.09.17	32.756813	-119.593564
SINKEX 1-7	2SX0094	1	Sediment	Regular	1998.09.17	32.756813	-119.593564
SINKEX 1-7	2SX0094	1	Sediment	Regular	1998.09.17	32.756813	-119.593564
SINKEX 1-7	2SX0095	1	Sediment	Regular	1998.09.17	32.756813	-119.593564
SINKEX 1-7	2SX0095	1	Sediment	Regular	1998.09.17	32.756813	-119.593564
SINKEX 1-1	2SX0096	1	Sediment	Regular	1998.09.17	32.756996	-119.593274
SINKEX 1-1	2SX0096	1	Sediment	Regular	1998.09.17	32.756996	-119.593274
SINKEX 1-1	2SX0097	1	Sediment	Regular	1998.09.17	32.756996	-119.593274
SINKEX 1-1	2SX0097	1	Sediment	Regular	1998.09.17	32.756996	-119.593274
SINKEX 1-4	2SX0098	1	Sediment	Regular	1998.09.17	32.756839	-119.592758
SINKEX 1-4	2SX0098	1	Sediment	Regular	1998.09.17	32.756839	-119.592758
SINKEX 1-4	2SX0099	1	Sediment	Regular	1998.09.17	32.756839	-119.592758
SINKEX 1-4	2SX0099	1	Sediment	Regular	1998.09.17	32.756839	-119.592758
SINKEX 1-5	2SX0100	1	Sediment	Regular	1998.09.17	32.756833	-119.593051
SINKEX 1-5	2SX0100	1	Sediment	Regular	1998.09.17	32.756833	-119.593051
SINKEX 1-5	2SX0101	1	Sediment	Regular	1998.09.17	32.756833	-119.593051
SINKEX 1-5	2SX0101	1	Sediment	Regular	1998.09.17	32.756833	-119.593051
MECASI	2SX0002Rhepox	1	Sediment	Composite	1998.10.23		
MECASI	2SX0002Rhepox	2	Sediment	Composite	1998.10.23		
MECASI	2SX0002Rhepox	3	Sediment	Composite	1998.10.23		
MECASI	2SX0002Rhepox	4	Sediment	Composite	1998.10.23		
MECASI	2SX0002Rhepox	5	Sediment	Composite	1998.10.23		
MECASI	2SX0003Rhepox	1	Sediment	Composite	1998.10.23		
MECASI	2SX0003Rhepox	2	Sediment	Composite	1998.10.23		
MECASI	2SX0003Rhepox	3	Sediment	Composite	1998.10.23		
MECASI	2SX0003Rhepox	4	Sediment	Composite	1998.10.23		
MECASI	2SX0003Rhepox	5	Sediment	Composite	1998.10.23		
MECASI	2SX0008Rhepox	1	Sediment	Composite	1998.10.23		
MECASI	2SX0008Rhepox	2	Sediment	Composite	1998.10.23		
MECASI	2SX0008Rhepox	3	Sediment	Composite	1998.10.23		
MECASI	2SX0008Rhepox	4	Sediment	Composite	1998.10.23		
MECASI	2SX0008Rhepox	5	Sediment	Composite	1998.10.23		
MECASI	2SX0015Rhepox	1	Sediment	Composite	1998.10.23		
MECASI	2SX0015Rhepox	2	Sediment	Composite	1998.10.23		
MECASI	2SX0015Rhepox	3	Sediment	Composite	1998.10.23		
MECASI	2SX0015Rhepox	4	Sediment	Composite	1998.10.23		
MECASI	2SX0015Rhepox	5	Sediment	Composite	1998.10.23		
MECASI	2SX0016Rhepox	1	Sediment	Composite	1998.10.23		
MECASI	2SX0016Rhepox	2	Sediment	Composite	1998.10.23		
MECASI	2SX0016Rhepox	3	Sediment	Composite	1998.10.23		
MECASI	2SX0016Rhepox	4	Sediment	Composite	1998.10.23		
MECASI	2SX0016Rhepox	5	Sediment	Composite	1998.10.23		
MECASI	2SX0023Rhepox	1	Sediment	Composite	1998.10.23		
MECASI	2SX0023Rhepox	2	Sediment	Composite	1998.10.23		
MECASI	2SX0023Rhepox	3	Sediment	Composite	1998.10.23		
MECASI	2SX0023Rhepox	4	Sediment	Composite	1998.10.23		
MECASI	2SX0023Rhepox	5	Sediment	Composite	1998.10.23		
MECASI	2SX0024Rhepox	1	Sediment	Composite	1998.10.23		
MECASI	2SX0024Rhepox	2	Sediment	Composite	1998.10.23		
MECASI	2SX0024Rhepox	3	Sediment	Composite	1998.10.23		
MECASI	2SX0024Rhepox	4	Sediment	Composite	1998.10.23		
MECASI	2SX0024Rhepox	5	Sediment	Composite	1998.10.23		
MECASI	2SX0031Rhepox	1	Sediment	Composite	1998.10.23		
MECASI	2SX0031Rhepox	2	Sediment	Composite	1998.10.23		
MECASI	2SX0031Rhepox	3	Sediment	Composite	1998.10.23		
MECASI	2SX0031Rhepox	4	Sediment	Composite	1998.10.23		
MECASI	2SX0031Rhepox	5	Sediment	Composite	1998.10.23		
MECASI	2SX0041Rhepox	1	Sediment	Composite	1998.10.23		
MECASI	2SX0041Rhepox	2	Sediment	Composite	1998.10.23		
MECASI	2SX0041Rhepox	3	Sediment	Composite	1998.10.23		
MECASI	2SX0041Rhepox	4	Sediment	Composite	1998.10.23		
MECASI	2SX0041Rhepox	5	Sediment	Composite	1998.10.23		
MECASI	2SX0002Nean	1	Sediment	Composite	1998.10.27		
MECASI	2SX0002Nean	10	Sediment	Composite	1998.10.27		
MECASI	2SX0002Nean	2	Sediment	Composite	1998.10.27		
MECASI	2SX0002Nean	3	Sediment	Composite	1998.10.27		
MECASI	2SX0002Nean	4	Sediment	Composite	1998.10.27		
MECASI	2SX0002Nean	5	Sediment	Composite	1998.10.27		

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
MECASI	2SX0002Nean	6	Sediment	Composite	1998.10.27		
MECASI	2SX0002Nean	7	Sediment	Composite	1998.10.27		
MECASI	2SX0002Nean	8	Sediment	Composite	1998.10.27		
MECASI	2SX0002Nean	9	Sediment	Composite	1998.10.27		
MECASI	2SX0003Nean	1	Sediment	Composite	1998.10.27		
MECASI	2SX0003Nean	10	Sediment	Composite	1998.10.27		
MECASI	2SX0003Nean	2	Sediment	Composite	1998.10.27		
MECASI	2SX0003Nean	3	Sediment	Composite	1998.10.27		
MECASI	2SX0003Nean	4	Sediment	Composite	1998.10.27		
MECASI	2SX0003Nean	5	Sediment	Composite	1998.10.27		
MECASI	2SX0003Nean	6	Sediment	Composite	1998.10.27		
MECASI	2SX0003Nean	7	Sediment	Composite	1998.10.27		
MECASI	2SX0003Nean	8	Sediment	Composite	1998.10.27		
MECASI	2SX0003Nean	9	Sediment	Composite	1998.10.27		
MECASI	2SX0008Nean	1	Sediment	Composite	1998.10.27		
MECASI	2SX0008Nean	10	Sediment	Composite	1998.10.27		
MECASI	2SX0008Nean	2	Sediment	Composite	1998.10.27		
MECASI	2SX0008Nean	3	Sediment	Composite	1998.10.27		
MECASI	2SX0008Nean	4	Sediment	Composite	1998.10.27		
MECASI	2SX0008Nean	5	Sediment	Composite	1998.10.27		
MECASI	2SX0008Nean	6	Sediment	Composite	1998.10.27		
MECASI	2SX0008Nean	7	Sediment	Composite	1998.10.27		
MECASI	2SX0008Nean	8	Sediment	Composite	1998.10.27		
MECASI	2SX0008Nean	9	Sediment	Composite	1998.10.27		
MECASI	2SX0015Nean	1	Sediment	Composite	1998.10.27		
MECASI	2SX0015Nean	10	Sediment	Composite	1998.10.27		
MECASI	2SX0015Nean	2	Sediment	Composite	1998.10.27		
MECASI	2SX0015Nean	3	Sediment	Composite	1998.10.27		
MECASI	2SX0015Nean	4	Sediment	Composite	1998.10.27		
MECASI	2SX0015Nean	5	Sediment	Composite	1998.10.27		
MECASI	2SX0015Nean	6	Sediment	Composite	1998.10.27		
MECASI	2SX0015Nean	7	Sediment	Composite	1998.10.27		
MECASI	2SX0015Nean	8	Sediment	Composite	1998.10.27		
MECASI	2SX0015Nean	9	Sediment	Composite	1998.10.27		
MECASI	2SX0016Nean	1	Sediment	Composite	1998.10.27		
MECASI	2SX0016Nean	10	Sediment	Composite	1998.10.27		
MECASI	2SX0016Nean	2	Sediment	Composite	1998.10.27		
MECASI	2SX0016Nean	3	Sediment	Composite	1998.10.27		
MECASI	2SX0016Nean	4	Sediment	Composite	1998.10.27		
MECASI	2SX0016Nean	5	Sediment	Composite	1998.10.27		
MECASI	2SX0016Nean	6	Sediment	Composite	1998.10.27		
MECASI	2SX0016Nean	7	Sediment	Composite	1998.10.27		
MECASI	2SX0016Nean	8	Sediment	Composite	1998.10.27		
MECASI	2SX0016Nean	9	Sediment	Composite	1998.10.27		
MECASI	2SX0023Nean	1	Sediment	Composite	1998.10.27		
MECASI	2SX0023Nean	10	Sediment	Composite	1998.10.27		
MECASI	2SX0023Nean	2	Sediment	Composite	1998.10.27		
MECASI	2SX0023Nean	3	Sediment	Composite	1998.10.27		
MECASI	2SX0023Nean	4	Sediment	Composite	1998.10.27		
MECASI	2SX0023Nean	5	Sediment	Composite	1998.10.27		
MECASI	2SX0023Nean	6	Sediment	Composite	1998.10.27		
MECASI	2SX0023Nean	7	Sediment	Composite	1998.10.27		
MECASI	2SX0023Nean	8	Sediment	Composite	1998.10.27		
MECASI	2SX0023Nean	9	Sediment	Composite	1998.10.27		
MECASI	2SX0024Nean	1	Sediment	Composite	1998.10.27		
MECASI	2SX0024Nean	10	Sediment	Composite	1998.10.27		
MECASI	2SX0024Nean	2	Sediment	Composite	1998.10.27		
MECASI	2SX0024Nean	3	Sediment	Composite	1998.10.27		
MECASI	2SX0024Nean	4	Sediment	Composite	1998.10.27		
MECASI	2SX0024Nean	5	Sediment	Composite	1998.10.27		
MECASI	2SX0024Nean	6	Sediment	Composite	1998.10.27		
MECASI	2SX0024Nean	7	Sediment	Composite	1998.10.27		
MECASI	2SX0024Nean	8	Sediment	Composite	1998.10.27		
MECASI	2SX0024Nean	9	Sediment	Composite	1998.10.27		
MECASI	2SX0031Nean	1	Sediment	Composite	1998.10.27		
MECASI	2SX0031Nean	10	Sediment	Composite	1998.10.27		
MECASI	2SX0031Nean	2	Sediment	Composite	1998.10.27		
MECASI	2SX0031Nean	3	Sediment	Composite	1998.10.27		
MECASI	2SX0031Nean	4	Sediment	Composite	1998.10.27		
MECASI	2SX0031Nean	5	Sediment	Composite	1998.10.27		
MECASI	2SX0031Nean	6	Sediment	Composite	1998.10.27		

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
MECASI	2SX0031Nean	7	Sediment	Composite	1998.10.27		
MECASI	2SX0031Nean	8	Sediment	Composite	1998.10.27		
MECASI	2SX0031Nean	9	Sediment	Composite	1998.10.27		
MECASI	2SX0041Nean	9	Sediment	Composite	1998.10.27		
MECASI	2SX0041Nean	1	Sediment	Composite	1998.10.27		
MECASI	2SX0041Nean	10	Sediment	Composite	1998.10.27		
MECASI	2SX0041Nean	2	Sediment	Composite	1998.10.27		
MECASI	2SX0041Nean	3	Sediment	Composite	1998.10.27		
MECASI	2SX0041Nean	4	Sediment	Composite	1998.10.27		
MECASI	2SX0041Nean	5	Sediment	Composite	1998.10.27		
MECASI	2SX0041Nean	6	Sediment	Composite	1998.10.27		
MECASI	2SX0041Nean	7	Sediment	Composite	1998.10.27		
MECASI	2SX0041Nean	8	Sediment	Composite	1998.10.27		
SINKEX 1-8	2SX0046D0-1	None	Sediment	Regular	1998.10.31	32.757009	-119.593544
SINKEX 1-8	2SX0046D1-2	None	Sediment	Regular	1998.10.31	32.757009	-119.593544
SINKEX 1-8	2SX0046D2-3	None	Sediment	Regular	1998.10.31	32.757009	-119.593544
SINKEX 1-8	2SX0046D3-4	None	Sediment	Regular	1998.10.31	32.757009	-119.593544
SINKEX 1-8	2SX0046D4-5	None	Sediment	Regular	1998.10.31	32.757009	-119.593544
SINKEX 1-8	2SX0046D5-6	None	Sediment	Regular	1998.10.31	32.757009	-119.593544
SINKEX 1-8	2SX0046D6-7	None	Sediment	Regular	1998.10.31	32.757009	-119.593544
SINKEX 1-8	2SX0046D7-8	None	Sediment	Regular	1998.10.31	32.757009	-119.593544
SINKEX 1-8	2SX0046D8-9	None	Sediment	Regular	1998.10.31	32.757009	-119.593544
SINKEX 1-8	2SX0046D9-10	None	Sediment	Regular	1998.10.31	32.757009	-119.593544
SINKEX 1-2	2SX0058D0-1	None	Sediment	Regular	1998.10.31	32.756988	-119.593058
SINKEX 1-2	2SX0058D1-2	None	Sediment	Regular	1998.10.31	32.756988	-119.593058
SINKEX 1-2	2SX0058D2-3	None	Sediment	Regular	1998.10.31	32.756988	-119.593058
SINKEX 1-2	2SX0058D3-4	None	Sediment	Regular	1998.10.31	32.756988	-119.593058
SINKEX 1-2	2SX0058D4-5	None	Sediment	Regular	1998.10.31	32.756988	-119.593058
SINKEX 1-2	2SX0058D5-6	None	Sediment	Regular	1998.10.31	32.756988	-119.593058
SINKEX 1-2	2SX0058D6-7	None	Sediment	Regular	1998.10.31	32.756988	-119.593058
SINKEX 1-2	2SX0058D7-8	None	Sediment	Regular	1998.10.31	32.756988	-119.593058
SINKEX 1-2	2SX0058D8-9	None	Sediment	Regular	1998.10.31	32.756988	-119.593058
SINKEX 1-2	2SX0058D9-10	None	Sediment	Regular	1998.10.31	32.756988	-119.593058
MECASI	2SX00015-Nephtys	1	Sediment	Composite	1998.11.06		
MECASI	2SX0002Mac	1	Sediment	Composite	1998.11.06		
MECASI	2SX0002-Nephtys	1	Sediment	Composite	1998.11.06		
MECASI	2SX0003Mac	1	Sediment	Composite	1998.11.06		
MECASI	2SX0003-Nephtys	1	Sediment	Composite	1998.11.06		
MECASI	2SX0008Mac	1	Sediment	Composite	1998.11.06		
MECASI	2SX0008-Nephtys	1	Sediment	Composite	1998.11.06		
MECASI	2SX0015Mac	1	Sediment	Composite	1998.11.06		
MECASI	2SX0016Mac	1	Sediment	Composite	1998.11.06		
MECASI	2SX0016-Nephtys	1	Sediment	Composite	1998.11.06		
MECASI	2SX0023Mac	1	Sediment	Composite	1998.11.06		
MECASI	2SX0023-Nephtys	1	Sediment	Composite	1998.11.06		
MECASI	2SX0024Mac	1	Sediment	Composite	1998.11.06		
MECASI	2SX0024-Nephtys	1	Sediment	Composite	1998.11.06		
MECASI	2SX0031Mac	1	Sediment	Composite	1998.11.06		
MECASI	2SX0031-Nephtys	1	Sediment	Composite	1998.11.06		
MECASI	2SX0041Mac	1	Sediment	Composite	1998.11.06		
MECASI	2SX0041-Nephtys	1	Sediment	Composite	1998.11.06		
MECASI	2SX0002-Macoma	None	Macoma nasuta	Regular	1998.11.07		
MECASI	2SX0002-Macoma-Re-Analysis	None	Macoma nasuta	Regular	1998.11.07		
MECASI	2SX0002-Nephtys	None	Nephtys caecoides	Regular	1998.11.07		
MECASI	2SX0003-Macoma	None	Macoma nasuta	Regular	1998.11.07		
MECASI	2SX0003-Nephtys	None	Nephtys caecoides	Regular	1998.11.07		
MECASI	2SX0008-Macoma	None	Macoma nasuta	Regular	1998.11.07		
MECASI	2SX0008-Nephtys	None	Nephtys caecoides	Regular	1998.11.07		
MECASI	2SX0015-Macoma	None	Macoma nasuta	Regular	1998.11.07		
MECASI	2SX0015-Nephtys	None	Nephtys caecoides	Regular	1998.11.07		
MECASI	2SX0016-Macoma	None	Macoma nasuta	Regular	1998.11.07		
MECASI	2SX0016-Macoma-Re-Analysis	None	Macoma nasuta	Regular	1998.11.07		
SINKEX 1-8	2SX0016-Macoma-Re-Analysis2	None	Macoma nasuta	Regular	1998.11.07	32.757009	-119.593544
MECASI	2SX0016-Nephtys	None	Nephtys caecoides	Regular	1998.11.07		
MECASI	2SX0023-Macoma	None	Macoma nasuta	Regular	1998.11.07		
MECASI	2SX0023-Nephtys	None	Nephtys caecoides	Regular	1998.11.07		
MECASI	2SX0024-Macoma	None	Macoma nasuta	Regular	1998.11.07		
MECASI	2SX0024-Nephtys	None	Nephtys caecoides	Regular	1998.11.07		
MECASI	2SX0031-Macoma	None	Macoma nasuta	Regular	1998.11.07		
MECASI	2SX0031-Nephtys	None	Nephtys caecoides	Regular	1998.11.07		
MECASI	2SX0041-Macoma	None	Macoma nasuta	Regular	1998.11.07		

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
MECASI	2SX0041-Nephtys	None	Nephtys caecoides	Regular	1998.11.07		
SINKEX 4-5	4SX001.01	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.01	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.02	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.02	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.03	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.03	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.04	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.04	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.05	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.05	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.06	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.06	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10-1	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10-1	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10-2	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10-2	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10-3	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10-3	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10-4	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10-4	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10-5	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10-5	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.20	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.20	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.21	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.21	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.22	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.22	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.23	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.23	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.24	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.24	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.25	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.25	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.26	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.26	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.27	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.27	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.28	None	Sediment	Split	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.28	None	Sediment	Split	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.28DUP	None	Sediment	Split	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.28DUP	None	Sediment	Split	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.29	None	Sediment	Split	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.29	None	Sediment	Split	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.29DUP	None	Sediment	Split	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.29DUP	None	Sediment	Split	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.30	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.30	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.31	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.31	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.40	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.40	None	Sediment	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.70	None	Water	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.70	None	Water	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.71	None	Water	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.71	None	Water	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.72	None	Water	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.72	None	Water	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.73	None	Water	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.73	None	Water	Regular	1999.09.21	32.747821	-119.593007
SINKEX 4-5	4SX001.10	None	Sediment	Composite	1999.09.22	32.747821	-119.593007
SINKEX 4-5	4SX001.10	None	Sediment	Composite	1999.09.22	32.747821	-119.593007
SINKEX 4-7	4SX_MEC_4-7_Infauna	1	Sediment	Regular	1999.09.23	32.757183	-119.603678
SINKEX 4-7	4SX_MEC_4-7_Infauna	1	Sediment	Regular	1999.09.23	32.757183	-119.603678
SINKEX Fish Trap 10	4SX_MEC_MiscInvert1_Infauna	1	Organism	Regular	1999.09.23	32.756816	-119.51395
SINKEX Fish Trap 10	4SX_MEC_MiscInvert1_Infauna	1	Organism	Regular	1999.09.23	32.756816	-119.51395
SINKEX Fish Trap 8	4SX024.602	None	Anoplopoma fimbria	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.602	None	Anoplopoma fimbria	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.603	None	Anoplopoma fimbria	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.603	None	Anoplopoma fimbria	Regular	1999.09.23	32.71183	-119.646

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX Fish Trap 8	4SX024.604	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.604	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.605	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.605	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.61	None	Anoplopoma fimbria	Split	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.61	None	Anoplopoma fimbria	Split	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.61DUP	None	Anoplopoma fimbria	Split	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.61DUP	None	Anoplopoma fimbria	Split	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.62	None	Anoplopoma fimbria	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.62	None	Anoplopoma fimbria	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.63	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.63	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.64	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.64	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.65	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.65	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.66	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX Fish Trap 8	4SX024.66	None	Paralomis multispina	Regular	1999.09.23	32.71183	-119.646
SINKEX 4-5	4SX_MEC_4-5_Infauna	1	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX_MEC_4-5_Infauna	1	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.07	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.07	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.08	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.08	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.10-10	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.10-10	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.10-6	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.10-6	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.10-7	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.10-7	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.10-8	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.10-8	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.10-9	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-5	4SX001.10-9	None	Sediment	Regular	1999.09.24	32.747821	-119.593007
SINKEX 4-3	4SX002.01	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.01	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.02	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.02	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.03	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.03	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.04	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.04	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.10	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.10	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.20	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.20	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.21	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.21	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.22	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.22	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.23	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.23	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.24	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.24	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.25	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.25	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.26	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.26	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.27	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.27	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.28	None	Sediment	Split	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.28	None	Sediment	Split	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.28DUP	None	Sediment	Split	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.28DUP	None	Sediment	Split	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.29	None	Sediment	Split	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.29	None	Sediment	Split	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.29DUP	None	Sediment	Split	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.29DUP	None	Sediment	Split	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.30	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.30	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.31	None	Sediment	Regular	1999.09.24	32.757043	-119.582293

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 4-3	4SX002.31	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.40	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.40	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.41	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.41	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.42	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.42	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.43	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.43	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.44	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.44	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.45	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.45	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.46	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.46	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.47	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.47	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.48	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.48	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.49	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.49	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.50	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.50	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.51	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.51	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.52	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.52	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.53	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-3	4SX002.53	None	Sediment	Regular	1999.09.24	32.757043	-119.582293
SINKEX 4-2	4SX003.10-1	None	Sediment	Regular	1999.09.24	32.762117	-119.585617
SINKEX 4-2	4SX003.10-1	None	Sediment	Regular	1999.09.24	32.762117	-119.585617
SINKEX 4-2	4SX003.10-2	None	Sediment	Regular	1999.09.24	32.762117	-119.585617
SINKEX 4-2	4SX003.10-2	None	Sediment	Regular	1999.09.24	32.762117	-119.585617
SINKEX 4-2	4SX_MEC_4-2_Infauna	1	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX_MEC_4-2_Infauna	1	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.01	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.01	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.02	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.02	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.03	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.03	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.04	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.04	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.05	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.05	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.06	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.06	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.07	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.07	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.08	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.08	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10	None	Sediment	Composite	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10	None	Sediment	Composite	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-3	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-3	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-4	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-4	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-5	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-5	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-6	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-6	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-7	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-7	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-8	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-8	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-9	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.10-9	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.20	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.20	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.40	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.40	None	Sediment	Regular	1999.09.25	32.762117	-119.585617

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 4-2	4SX003.41	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.41	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.42	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.42	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.43	None	Sediment	Split	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.43	None	Sediment	Split	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.43DUP	None	Sediment	Split	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.43DUP	None	Sediment	Split	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.44	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.44	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.45	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.45	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.46	None	Sediment	Split	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.46	None	Sediment	Split	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.46DUP	None	Sediment	Split	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.46DUP	None	Sediment	Split	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.47	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.47	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.48	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.48	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.49	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.49	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.50	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.50	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.51	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.51	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.52	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.52	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.53	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.53	None	Sediment	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.70	None	Water	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.70	None	Water	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.71	None	Water	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.71	None	Water	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.72	None	Water	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.72	None	Water	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.73	None	Water	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-2	4SX003.73	None	Water	Regular	1999.09.25	32.762117	-119.585617
SINKEX 4-3.1	4SX_MEC_4.3.1_Infauna	1	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX_MEC_4.3.1_Infauna	1	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-4	4SX_MEC_4-4_Infauna	1	Sediment	Regular	1999.09.26	32.750433	-119.585417
SINKEX 4-4	4SX_MEC_4-4_Infauna	1	Sediment	Regular	1999.09.26	32.750433	-119.585417
SINKEX 4-3.1	4SX004.01	None	Sediment	Split	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.01	None	Sediment	Split	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.02	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.02	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.03	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.03	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.04	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.04	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.05	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.05	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.06	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.06	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10	None	Sediment	Composite	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10	None	Sediment	Composite	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-1	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-1	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-2	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-2	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-3	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-3	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-4	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-4	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-5	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-5	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-6	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-6	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.20	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.20	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.21	None	Sediment	Regular	1999.09.26	32.757495	-119.580312

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 4-3.1	4SX004.21	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.22	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.22	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.23	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.23	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.24	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.24	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.25	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.25	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.26	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.26	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.27	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.27	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.28	None	Sediment	Split	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.28	None	Sediment	Split	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.28DUP	None	Sediment	Split	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.28DUP	None	Sediment	Split	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.29	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.29	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.30	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.30	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.31	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.31	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.32	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.32	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.40	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.40	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.41	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.41	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.42	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.42	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.43	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.43	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.44	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.44	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.45	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.45	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.46	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.46	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.47	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.47	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.48	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.48	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.49	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.49	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.50	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.50	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.51	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.51	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.52	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.52	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.53	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-3.1	4SX004.53	None	Sediment	Regular	1999.09.26	32.757495	-119.580312
SINKEX 4-1	4SX_MEC_4-1_Infauna	1	Sediment	Regular	1999.09.27	32.765967	-119.592994
SINKEX 4-1	4SX_MEC_4-1_Infauna	1	Sediment	Regular	1999.09.27	32.765967	-119.592994
SINKEX Fish Trap 5	4SX_MEC_NWFT_Infauna	1	Organism	Regular	1999.09.27	32.80828333	-119.6264
SINKEX Fish Trap 5	4SX_MEC_NWFT_Infauna	1	Organism	Regular	1999.09.27	32.80828333	-119.6264
SINKEX 4-3.1	4SX004.07	None	Sediment	Regular	1999.09.27	32.757495	-119.580312
SINKEX 4-3.1	4SX004.07	None	Sediment	Regular	1999.09.27	32.757495	-119.580312
SINKEX 4-3.1	4SX004.08	None	Sediment	Regular	1999.09.27	32.757495	-119.580312
SINKEX 4-3.1	4SX004.08	None	Sediment	Regular	1999.09.27	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-7	None	Sediment	Regular	1999.09.27	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-7	None	Sediment	Regular	1999.09.27	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-8	None	Sediment	Regular	1999.09.27	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-8	None	Sediment	Regular	1999.09.27	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-9	None	Sediment	Regular	1999.09.27	32.757495	-119.580312
SINKEX 4-3.1	4SX004.10-9	None	Sediment	Regular	1999.09.27	32.757495	-119.580312
SINKEX 4-1	4SX005.10-1	None	Sediment	Regular	1999.09.27	32.765967	-119.592994
SINKEX 4-1	4SX005.10-1	None	Sediment	Regular	1999.09.27	32.765967	-119.592994
SINKEX 4-1	4SX005.10-2	None	Sediment	Regular	1999.09.27	32.765967	-119.592994
SINKEX 4-1	4SX005.10-2	None	Sediment	Regular	1999.09.27	32.765967	-119.592994

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 4-1	4SX005.10-3	None	Sediment	Regular	1999.09.27	32.765967	-119.592994
SINKEX 4-1	4SX005.10-3	None	Sediment	Regular	1999.09.27	32.765967	-119.592994
SINKEX 4-1	4SX005.10-4	None	Sediment	Regular	1999.09.27	32.765967	-119.592994
SINKEX 4-1	4SX005.10-4	None	Sediment	Regular	1999.09.27	32.765967	-119.592994
SINKEX Fish Trap 5	4SX023.601	None	Paralomis multispina	Regular	1999.09.27	32.80828333	-119.6264
SINKEX Fish Trap 5	4SX023.601	None	Paralomis multispina	Regular	1999.09.27	32.80828333	-119.6264
SINKEX Fish Trap 5	4SX023.62	None	Paralomis multispina	Regular	1999.09.27	32.80828333	-119.6264
SINKEX Fish Trap 5	4SX023.62	None	Paralomis multispina	Regular	1999.09.27	32.80828333	-119.6264
SINKEX Fish Trap 5	4SX023.63	None	Paralomis multispina	Regular	1999.09.27	32.80828333	-119.6264
SINKEX Fish Trap 5	4SX023.63	None	Paralomis multispina	Regular	1999.09.27	32.80828333	-119.6264
SINKEX 4-8	4SX_MEC_4-8_Infauna	1	Sediment	Regular	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX_MEC_4-8_Infauna	1	Sediment	Regular	1999.09.28	32.76321	-119.600542
SINKEX Fish Trap 10	4SX_MEC_MiscInvert2_Infauna	1	Organism	Regular	1999.09.28	32.756816	-119.51395
SINKEX Fish Trap 10	4SX_MEC_MiscInvert2_Infauna	1	Organism	Regular	1999.09.28	32.756816	-119.51395
SINKEX 4-1	4SX005.01	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.01	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.02	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.02	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.03	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.03	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.04	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.04	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.05	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.05	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.06	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.06	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.07	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.07	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.08	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.08	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10	None	Sediment	Composite	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10	None	Sediment	Composite	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-10	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-10	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-5	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-5	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-6	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-6	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-7	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-7	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-8	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-8	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-9	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.10-9	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.20	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.20	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.21	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.21	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.22	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.22	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.23	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.23	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.24	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.24	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.25	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.25	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.26	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.26	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.27	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.27	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.28	None	Sediment	Split	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.28	None	Sediment	Split	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.28DUP	None	Sediment	Split	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.28DUP	None	Sediment	Split	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.29	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.29	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.30	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.30	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.31	None	Sediment	Split	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.31	None	Sediment	Split	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.31DUP	None	Sediment	Split	1999.09.28	32.765967	-119.592994

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 4-1	4SX005.31DUP	None	Sediment	Split	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.32	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.32	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.40	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-1	4SX005.40	None	Sediment	Regular	1999.09.28	32.765967	-119.592994
SINKEX 4-8	4SX006.01	None	Sediment	Regular	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.01	None	Sediment	Regular	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.03	None	Sediment	Regular	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.03	None	Sediment	Regular	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.04	None	Sediment	Split	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.04	None	Sediment	Split	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.04DUP	None	Sediment	Split	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.04DUP	None	Sediment	Split	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.10	None	Sediment	Regular	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.10	None	Sediment	Regular	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.88	None	Sediment	Regular	1999.09.28	32.76321	-119.600542
SINKEX 4-8	4SX006.88	None	Sediment	Regular	1999.09.28	32.76321	-119.600542
SINKEX Fish Trap 10	4SX_MEC_EFT_Infauna	1	Organism	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX_MEC_EFT_Infauna	1	Organism	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX021.99	None	Organism	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX021.99	None	Organism	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX021.99-1	None	Organism	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX021.99-1	None	Organism	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX021.99-2	None	Organism	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX021.99-2	None	Organism	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX021.99-3	None	Organism	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX021.99-3	None	Organism	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX022.600	None	Anoplopoma fimbria	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX022.600	None	Anoplopoma fimbria	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX022.61	None	Anoplopoma fimbria	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX022.61	None	Anoplopoma fimbria	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX022.64	None	Anoplopoma fimbria	Regular	1999.09.29	32.756816	-119.51395
SINKEX Fish Trap 10	4SX022.64	None	Anoplopoma fimbria	Regular	1999.09.29	32.756816	-119.51395
MECASI	4SX001.90	None	Sediment	Composite	1999.10.26		
MECASI	4SX001.91	None	Sediment	Composite	1999.10.26		
MECASI	4SX003.90	None	Sediment	Composite	1999.10.26		
MECASI	4SX003.91	None	Sediment	Split	1999.10.26		
MECASI	4SX004.90	None	Sediment	Composite	1999.10.26		
MECASI	4SX004.91	None	Sediment	Composite	1999.10.26		
MECASI	4SX005.90	None	Sediment	Composite	1999.10.26		
MECASI	4SX005.91	None	Sediment	Split	1999.10.26		
MECASI	4SX001.10Mac	1	Sediment	Composite	1999.10.27		
MECASI	4SX001.10Mac	2	Sediment	Regular	1999.10.27		
MECASI	4SX001.10Mac	3	Sediment	Regular	1999.10.27		
MECASI	4SX001.10Mac	4	Sediment	Regular	1999.10.27		
MECASI	4SX001.10Mac	5	Sediment	Regular	1999.10.27		
MECASI	4SX003.10Mac	1	Sediment	Composite	1999.10.27		
MECASI	4SX003.10Mac	2	Sediment	Regular	1999.10.27		
MECASI	4SX003.10Mac	3	Sediment	Regular	1999.10.27		
MECASI	4SX003.10Mac	4	Sediment	Regular	1999.10.27		
MECASI	4SX003.10Mac	5	Sediment	Regular	1999.10.27		
MECASI	4SX004.10Mac	1	Sediment	Composite	1999.10.27		
MECASI	4SX004.10Mac	2	Sediment	Regular	1999.10.27		
MECASI	4SX004.10Mac	3	Sediment	Regular	1999.10.27		
MECASI	4SX004.10Mac	4	Sediment	Regular	1999.10.27		
MECASI	4SX004.10Mac	5	Sediment	Regular	1999.10.27		
MECASI	4SX005.10Mac	1	Sediment	Composite	1999.10.27		
MECASI	4SX005.10Mac	2	Sediment	Regular	1999.10.27		
MECASI	4SX005.10Mac	3	Sediment	Regular	1999.10.27		
MECASI	4SX005.10Mac	4	Sediment	Regular	1999.10.27		
MECASI	4SX005.10Mac	5	Sediment	Regular	1999.10.27		
MECASI	4SX001.10Nean	1	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Nean	10	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Nean	2	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Nean	3	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Nean	4	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Nean	5	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Nean	6	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Nean	7	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Nean	8	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Nean	9	Sediment	Composite	1999.10.29		

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
MECASI	4SX001.10Rhepox	1	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Rhepox	2	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Rhepox	3	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Rhepox	4	Sediment	Composite	1999.10.29		
MECASI	4SX001.10Rhepox	5	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Nean	1	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Nean	10	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Nean	2	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Nean	3	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Nean	4	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Nean	5	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Nean	6	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Nean	7	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Nean	8	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Nean	9	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Rhepox	1	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Rhepox	2	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Rhepox	3	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Rhepox	4	Sediment	Composite	1999.10.29		
MECASI	4SX003.10Rhepox	5	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Nean	1	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Nean	10	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Nean	2	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Nean	3	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Nean	4	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Nean	5	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Nean	6	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Nean	7	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Nean	8	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Nean	9	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Rhepox	1	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Rhepox	2	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Rhepox	3	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Rhepox	4	Sediment	Composite	1999.10.29		
MECASI	4SX004.10Rhepox	5	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Nean	1	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Nean	10	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Nean	2	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Nean	3	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Nean	4	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Nean	5	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Nean	6	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Nean	7	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Nean	8	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Nean	9	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Rhepox	1	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Rhepox	2	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Rhepox	3	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Rhepox	4	Sediment	Composite	1999.10.29		
MECASI	4SX005.10Rhepox	5	Sediment	Composite	1999.10.29		
ADCP Second	5SX_ADCP	None	Seawater	Regular	1999.11.15	32.755788	-119.594292
ADCP Second	5SX_ADCP	None	Seawater	Regular	1999.11.15	32.755788	-119.594292
ADCP Second	5SX_MEC_ADCP_Infauna	1	Organism	Regular	1999.11.15	32.755788	-119.594292
ADCP Second	5SX_MEC_ADCP_Infauna	1	Organism	Regular	1999.11.15	32.755788	-119.594292
S4	5SX_S4	None	Seawater	Regular	1999.11.15	32.75673	-119.592668
S4	5SX_S4	None	Seawater	Regular	1999.11.15	32.75673	-119.592668
SINKEX 1-5/6/7	5SX001.01	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.01	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.02	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.02	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.03	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.03	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.04	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.04	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.05	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.05	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.06	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.06	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.07	2	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.07	2	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.08	2	Sediment	Regular	1999.11.16	32.756791	-119.5933

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 1-5/6/7	5SX001.08	2	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.10	1	Sediment	Composite	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.10	1	Sediment	Composite	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.19	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.19	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.40	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.40	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.41	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.41	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.42	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.42	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.43	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.43	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.44	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.44	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.45	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.45	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.46	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.46	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.47	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.47	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.48	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.48	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.49	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.49	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.50	1	Sediment	Split	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.50	1	Sediment	Split	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.50DUP	1	Sediment	Split	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.50DUP	1	Sediment	Split	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.51	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.51	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.52	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.52	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.53	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.53	1	Sediment	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.70	1	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.70	1	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.71	2	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.71	2	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.72	3	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.72	3	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.73	4	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.73	4	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.74	1	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.74	1	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.75	2	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.75	2	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.76	3	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.76	3	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.77	4	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-5/6/7	5SX001.77	4	Water	Regular	1999.11.16	32.756791	-119.5933
SINKEX 1-3.5	5SX002.01	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.01	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.02	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.02	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.03	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.03	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.04	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.04	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.05	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.05	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.06	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.06	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.40	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.40	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.41	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.41	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.42	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.42	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.43	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.43	1	Sediment	Regular	1999.11.16	32.756937	-119.592729

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX 1-3.5	5SX002.44	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.44	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.45	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.45	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.46	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.46	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.47	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.47	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.48	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.48	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.49	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.49	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.50	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.50	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.51	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.51	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.52	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.52	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.53	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX 1-3.5	5SX002.53	1	Sediment	Regular	1999.11.16	32.756937	-119.592729
SINKEX WFT	5SX_MEC_WFT_Crab_Infauna	1	Organism	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX_MEC_WFT_Crab_Infauna	1	Organism	Regular	1999.11.17	32.75718833	-119.671845
SINKEX 4-6	5SX003.01	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.01	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.02	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.02	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.03	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.03	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.04	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.04	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.05	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.05	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.06	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.06	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.40	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.40	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.41	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.41	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.42	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.42	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.43	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.43	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.44	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.44	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.45	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.45	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.46	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.46	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.47	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.47	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.48	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.48	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.49	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.49	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.50	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.50	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.51	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.51	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.52	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.52	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.53	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX 4-6	5SX003.53	1	Sediment	Regular	1999.11.17	32.750609	-119.600484
SINKEX WFT	5SX011.60	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.60	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.600	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.600	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.601	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.601	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.602	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.602	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.603	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX WFT	5SX011.603	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.604	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.604	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.61	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.61	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.62	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.62	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.63	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.63	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.64	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.64	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.65	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.65	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.66	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.66	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.67	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.67	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.68	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.68	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.69	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX011.69	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.60	1	Anoplopoma fimbria	Split	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.60	1	Anoplopoma fimbria	Split	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.600	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.600	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.601	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.601	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.602	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.602	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.603	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.603	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.604	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.604	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
ADLittle	5SX012.60DUP	1	Anoplopoma fimbria	Split	1999.11.17		
ADLittle	5SX012.60DUP	1	Anoplopoma fimbria	Split	1999.11.17		
SINKEX WFT	5SX012.61	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.61	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.62	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.62	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.63	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.63	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.64	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.64	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.65	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.65	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.66	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.66	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.67	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.67	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.68	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.68	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.69	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX012.69	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.60	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.60	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.600	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.600	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.601	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.601	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.603	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.603	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.604	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.604	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.605	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.605	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.61	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.61	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.62	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.62	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.63	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.63	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
SINKEX WFT	5SX013.64	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.64	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.65	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.65	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.66	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.66	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.67	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.67	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.68	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.68	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.69	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX WFT	5SX013.69	1	Anoplopoma fimbria	Regular	1999.11.17	32.75718833	-119.671845
SINKEX RFT	5SX014.60	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.60	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.600	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.600	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.601	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.601	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.602	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.602	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.603	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.603	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.61	1	Anoplopoma fimbria	Split	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.61	1	Anoplopoma fimbria	Split	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.61DUP	1	Anoplopoma fimbria	Split	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.61DUP	1	Anoplopoma fimbria	Split	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.62	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.62	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.63	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.63	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.64	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.64	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.65	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.65	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.66	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.66	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.67	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
SINKEX RFT	5SX014.67	1	Anoplopoma fimbria	Regular	1999.11.17	32.756879	-119.59354
MECASI	5SX001.12	1	Sediment	Composite	1999.11.23		
MECASI	5SX001.12	10	Sediment	Composite	1999.11.23		
MECASI	5SX001.12	2	Sediment	Composite	1999.11.23		
MECASI	5SX001.12	3	Sediment	Composite	1999.11.23		
MECASI	5SX001.12	4	Sediment	Composite	1999.11.23		
MECASI	5SX001.12	5	Sediment	Composite	1999.11.23		
MECASI	5SX001.12	6	Sediment	Composite	1999.11.23		
MECASI	5SX001.12	7	Sediment	Composite	1999.11.23		
MECASI	5SX001.12	8	Sediment	Composite	1999.11.23		
MECASI	5SX001.12	9	Sediment	Composite	1999.11.23		
MECASI	5SX001.13	1	Sediment	Composite	1999.11.24		
MECASI	5SX001.13	2	Sediment	Regular	1999.11.24		
MECASI	5SX001.13	3	Sediment	Regular	1999.11.24		
MECASI	5SX001.13	4	Sediment	Regular	1999.11.24		
MECASI	5SX001.13	5	Sediment	Regular	1999.11.24		
MECASI	5SX001.16	1	Sediment	Composite	1999.11.24		
MECASI	4SX001.14T	None	Macoma nasuta	Regular	1999.11.25		
MECASI	4SX001.15	None	Macoma nasuta	Regular	1999.11.25		
MECASI	4SX001.17T	None	Nephtys caecoides	Regular	1999.11.25		
MECASI	4SX001.18	None	Nephtys caecoides	Regular	1999.11.25		
MECASI	4SX003.14T	None	Macoma nasuta	Regular	1999.11.25		
MECASI	4SX003.15	None	Macoma nasuta	Regular	1999.11.25		
MECASI	4SX003.17T	None	Nephtys caecoides	Regular	1999.11.25		
MECASI	4SX003.18	None	Nephtys caecoides	Regular	1999.11.25		
MECASI	4SX004.14T	None	Macoma nasuta	Split	1999.11.25		
MECASI	4SX004.15	None	Macoma nasuta	Regular	1999.11.25		
MECASI	4SX004.17T	None	Nephtys caecoides	Regular	1999.11.25		
MECASI	4SX004.18	None	Nephtys caecoides	Regular	1999.11.25		
MECASI	4SX005.14T	None	Macoma nasuta	Split	1999.11.25		
ADLittle	4SX005.14TDUP	None	Macoma nasuta	Split	1999.11.25		
MECASI	4SX005.15	None	Macoma nasuta	Regular	1999.11.25		
MECASI	4SX005.17T	None	Nephtys caecoides	Regular	1999.11.25		
MECASI	4SX005.18	None	Nephtys caecoides	Regular	1999.11.25		

Appendix N: Sample Collection Date And Location Data

SITE LABEL	SAMPLE LABEL	DUPLICATE LABEL	SAMPLE MEDIUM	TYPE NAME	SAMPLE DATE	LATITUDE	LONGITUDE
MECASI	5SX001.15	1	Macoma nasuta	Regular	1999.11.25		
MECASI	5SX001.18	1	Nephtys caecoides	Regular	1999.11.25		
MECASI	5SX001.14T	1	Macoma nasuta	Regular	2000.01.11		
MECASI	5SX001.17T	1	Nephtys caecoides	Regular	2000.01.11		
MECASI	5SX001.11	1	Sediment	Composite	2000.04.24		
MECASI	5SX001.11	2	Sediment	Composite	2000.04.24		
MECASI	5SX001.11	3	Sediment	Composite	2000.04.24		
MECASI	5SX001.11	4	Sediment	Composite	2000.04.24		
MECASI	5SX001.11	5	Sediment	Composite	2000.04.24		
MECASI	5SX001.90	1	Sediment	Composite	2000.05.15		
MECASI	5SX001.91	1	Sediment	Composite	2000.05.15		
FIT	4SX005.91DUP	None	Sediment	Split	2000.06.09		
SINKEX 4-3.1	4SX004.01DUP	None	Sediment	Split	2000.06.12	32.757495	-119.580312
MECASI	4SX004.14TDUP	None	Macoma nasuta	Split	2000.06.27		
SINKEX 1-6	2SX0026DUP	None	Sediment	Split	2002.04.24	32.756827	-119.593272
SINKEX 1-7	2SX0049DUP	None	Sediment	Split	2002.04.24	32.756813	-119.593564
SINKEX 4-8	4SX006.03DUP	None	Sediment	Split	2002.04.24	32.76321	-119.600542