HEAVY METALS, PCBs AND PAHs IN MARINE ORGANISMS FROM FOUR HARBOR LOCATIONS ON GUAM

A PILOT STUDY

Gary R.W. Denton
Lucrina P. Concepcion
H. Rick Wood
Vance S. Elfin
Greg T. Pangelinan

WERI
WATER AND ENVIRONMENTAL RESEARCH INSTITUTE
OF THE WESTERN PACIFIC
UNIVERSITY OF GUAM

Technical Report No. 87
June 30, 1999

HEAVY METALS, PCBs AND PAHs IN MARINE ORGANISM FROM FOUR HARBOR LOCATIONS ON GUAM

A PILOT STUDY

Gary R.W. Denton
Lucrina P. Concepcion
H. Rick Wood

Water & Environmental Research Institute
of the Western Pacific
University of Guam

&

Vance S. Eflin
Greg T. Pangelinan

Guam Environmental Protection Agency

June 30, 1999

Intrepid explorer and master boatman, Greg Pangelinan, maneuvers the GEPA boat to a mooring site in Apra Harbor

ACKNOWLEDGMENTS

Our thanks to Mike Ham, Coastal Zone Administrator, Guam Bureau of Planning, for providing us with the opportunity to do this work and for his continued interest in the project. We would also like to thank Mr. Jesus Salas, Administrator of Guam Environmental Protection Agency (GEPA), and Dr. Galt Siegrist, Director of WERI at the University of Guam, for their constant support and encouragement. A debt of gratitude is also owed to Carmen Sian-Denton for proofreading the text, and to Norma Blas for organizing the photocopying and binding of the final document. This work was funded, in part, by the National Oceanographic and Atmospheric Administration, Office of Ocean and Coastal Resource Management, and the Guam Coastal Management Program, Bureau of Planning, Government of Guam, through NOAA Grant Award # NA77OZ0184.

TABLE OF CONTENTS

Page

ACKNOWLEDGEMENTS v

LIST OF FIGURES xi

LIST OF TABLES xi

ABSTRACT 1

INTRODUCTION 3

MATERIALS & METHODS

1. HARBOUR SITES 7
   - 1.1 Agana Boat Basin 7
   - 1.2 Apra Harbor 7
   - 1.3 Agat Marina 10
   - 1.4 Merizo Pier 10
2. SAMPLE COLLECTION AND PREPARATION 10
3. HEAVY METAL ANALYSIS 15
   - 3.1 Mercury 15
   - 3.2 All Other Metals 16
4. PCB AND PAH ANALYSIS 16
   - 4.1 Solvent Extraction 16
   - 4.2 Silica/Alumina Column Cleanup 20
   - 4.3 Chromatographic Parameters for PCB Analysis 20
   - 4.4 Chromatographic Parameters for PAH Analysis 21
5. PRESENTATION OF DATA 24

RESULTS & DISCUSSION

1. HEAVY METALS IN HARBOR BIOTA 35
   - 1.1 Silver (Ag) 35
     - *1.1.1 Ag in Algae* 35
     - *1.1.2 Ag in Sponges* 35
     - *1.1.3 Ag in Corals* 35
     - *1.1.4 Ag in Sea Cucumbers* 35
     - *1.1.5 Ag in Mollusks* 36
     - *1.1.6 Ag in Crustaceans* 36
     - *1.1.7 Ag in Ascidians* 36
     - *1.1.8 Ag in Fish* 37
     - *1.1.9 Concluding Remarks* 37

Page

1.2 ARSENIC (As) 37
1.2.1 As in Algae 37
1.2.2 As in Sponges 37
1.2.3 As in Corals 38
1.2.4 As in Sea Cucumbers 38
1.2.5 As in Mollusks 38
1.2.6 As in Crustaceans 39
1.2.7 As in Ascidians 39
1.2.8 As in Fish 39
1.2.9 Concluding Remarks 39
1.3 CADMIUM (Cd) 39
1.3.1 Cd in Algae 40
1.3.2 Cd in Sponges 40
1.3.3 Cd in Corals 40
1.3.4 Cd in Sea Cucumbers 41
1.3.5 Cd in Mollusks 41
1.3.6 Cd in Crustaceans 41
1.3.7 Cd in Ascidians 42
1.3.8 Cd in Fish 42
1.3.9 Concluding Remarks 42
1.4 CHROMIUM (Cr) 42
1.4.1 Cr in Algae 43
1.4.2 Cr in Sponges 43
1.4.3 Cr in Corals 43
1.4.4 Cr in Sea Cucumbers 43
1.4.5 Cr in Mollusks 44
1.4.6 Cr in Crustaceans 44
1.4.7 Cr in Ascidians 44
1.4.8 Cr in Fish 44
1.4.9 Concluding Remarks 44
1.5 COPPER (Cu) 44
1.5.1 Cu in Algae 45
1.5.2 Cu in Sponges 45
1.5.3 Cu in Corals 46
1.5.4 Cu in Sea Cucumbers 46
1.5.5 Cu in Mollusks 46
1.5.6 Cu in Crustaceans 47
1.5.7 Cu in Ascidians 48
1.5.8 Cu in Fish 48
1.5.9 Concluding Remarks 48

Page

1.6 MERCURY (Hg) 49
1.6.1 Hg in Algae 49
1.6.2 Hg in Sponges 50
1.6.3 Hg in Corals 50
1.6.4 Hg in Sea Cucumbers 50
1.6.5 Hg in Mollusks 50
1.6.6 Hg in Crustaceans 51
1.6.7 Hg in Ascidians 51
1.6.8 Hg in Fish 52
1.6.9 Concluding Remarks 52
1.7 NICKEL (Ni) 53
1.7.1 Ni in Algae 53
1.7.2 Ni in Sponges 53
1.7.3 Ni in Corals 53
1.7.4 Ni in Sea Cucumbers 54
1.7.5 Ni in Mollusks 54
1.7.6 Ni in Crustaceans 55
1.7.7 Ni in Ascidians 55
1.7.8 Ni in Fish 55
1.7.9 Concluding Remarks 55
1.8 LEAD (Pb) 55
1.8.1 Pb in Algae 56
1.8.2 Pb in Sponges 56
1.8.3 Pb in Corals 56
1.8.4 Pb in Sea Cucumbers 57
1.8.5 Pb in Mollusks 57
1.8.6 Pb in Crustaceans 58
1.8.7 Pb in Ascidians 58
1.8.8 Pb in Fish 58
1.8.9 Concluding Remarks 59
1.9 TIN (Sn) 59
1.9.1 Sn in Algae 59
1.9.2 Sn in Sponges 60
1.9.3 Sn in Corals 60
1.9.4 Sn in Sea Cucumbers 60
1.9.5 Sn in Mollusks 61
1.9.6 Sn in Crustaceans 61
1.9.7 Sn in Ascidians 61
1.9.8 Sn in Fish 62
1.9.9 Concluding Remarks 62

Page

1.10 Zinc (Zn) 62
1.10.1 Zn in Algae 63
1.10.2 Zn in Sponges 63
1.10.3 Zn in Corals 64
1.10.4 Zn in Sea Cucumbers 65
1.10.5 Zn in Mollusks 65
1.10.6 Zn in Crustaceans 66
1.10.7 Zn in Ascidians 67
1.10.8 Zn in Fish 67
1.10.9 Concluding Remarks 67
2. PCBs in Harbor Biota 81
2.1 PCBs in Algae 82
2.2 PCBs in Sponges 82
2.3 PCBs in Corals 83
2.4 PCBs in Sea Cucumbers 83
2.5 PCBs in Mollusks 87
2.6 PCBs in Crustaceans 88
2.7 PCBs in Ascidians 88
2.8 PCBs in Fish 88
2.9 Concluding Remarks 89
3. PAHs in Harbor Biota 103
3.1 PAHs in Algae 104
3.2 PAHs in Sponges 105
3.3 PAHs in Corals 106
3.4 PAHs in Sea Cucumbers 106
3.5 PAHs in Mollusks 106
3.6 PAHs in Crustaceans 108
3.7 PAHs in Ascidians 109
3.8 PAHs in Fish 109
3.9 Concluding Remarks 109

GENERAL CONCLUSIONS

1. The Implementation of a Marine Monitoring Program Using Biological Indicators: Some Preliminary Considerations 121
1.1 Species Selection 121
1.2 Sample Variability 126
1.3 Program Design 126
1.4 Site Selection 127
2. Evaluation of Data in Relation to Current Food Standards 127
3. Recommendations For Future Work 129

BIBLIOGRAPHY 131

LIST OF FIGURES

Page

Figure 1: Guam Harbors Visited 5
Figure 2: Agana Boat Basin 8
Figure 3: Apra Harbor 9
Figure 4: Agat Marine 11
Figure 5: Merizo Pier 12
Figure 6: Polychlorinated Biphenyls in Sponges from Apra Harbor 84
Figure 7: Polychlorinated Biphenyls in Soft Corals from Apra Harbor 85
Figure 8: Polychlorinated Biphenyls in Sea Cucumber from Apra Harbor 86
Figure 9: Polychlorinated Biphenyls in Fish from Apra Harbor (*Caranx malempugus:* Dry Dock Island) 90
Figure 10: Polychlorinated Biphenyls in Fish from Apra Harbor (*Monodactylus argenteus:* Commercial Port) 91
Figure 11: Polychlorinated Biphenyls in Fish from Apra Harbor (*Caranx malempugus:* Hotel Wharf and Dry Dock Island) 92

LIST OF TABLES

Page

Table 1: Flora and Fauna Sampled During the Present Survey 13
Table 2: Analysis of Standard Reference Materials 17
Table 3: PCBs Congeners in Calibration Standard used to Quantify PCB Homologues in Biota Samples from Harbor Sites on Guam 22
Table 4: Unsubstituted PAHs in Calibration Standard Used to Quantify PAH Levels in Biota Samples from Harbor Sites on Guam 23
Table 5: Heavy Metal Concentrations (µg/g dry wt.) in Marine Organisms from Other Regions of the World 25
Table 6: PCB Concentrations in Marine Organisms from Other Regions of the World 29
Table 7: PAH Concentrations in Marine Organisms from Other Regions of the World 31
Table 8: Heavy Metals in Seaweed from Guam Harbor Waters 68
Table 9: Heavy Metals in Sponges from Guam Harbor Waters 69
Table 10: Heavy Metals in Corals from Guam Harbor Waters 70
Table 11: Heavy Metals in Sea Cucumbers from Guam Harbor Waters 71
Table 12: Heavy Metals in Bivalve Mollusks from Guam Harbor Waters (Oysters) 72
Table 13: Heavy Metals in Bivalve Mollusks from Guam Harbor Waters (Chamids and Spondylids) 73
Table 14: Heavy Metals in Octopus, Mantis Shrimp and Ascidians from Guam Harbor Waters 74
Table 15: Heavy Metals in Tissues of Fish from Guam Harbor Waters 75
Table 16: PCB Homologues in Seaweed from Guam Harbor Waters 93

LIST OF TABLES (cont.)

		Page
Table 17:	PCB Homologues in Sponges and Soft Corals from Guam Harbor Waters	94
Table 18:	PCB Homologues in Sea Cucumbers from Guam Harbor Waters	95
Table 19:	PCB Homologues in Bivalve Mollusks from Guam Harbor Waters (Oysters)	96
Table 20:	PCB Homologues in Bivalve Mollusks from Guam Harbor Waters (Chamids and Spondylids)	97
Table 21:	PCB Homologues in Octopus, Mantis Shrimp and Ascidians from Guam Harbor Waters	98
Table 22:	PCB Homologues in Tissues of Fish from Guam Harbor Waters	99
Table 23:	PAHs in Seaweed from Guam Harbor Waters	110
Table 24:	PAHs in Sponges and Soft Corals from Guam Harbor Waters	111
Table 25:	PAHs in Sea Cucumbers from Guam Harbor Waters	112
Table 26:	PAHs in Bivalve Mollusks from Guam Harbor Waters (Oysters)	113
Table 27:	PAHs in Bivalve Mollusks from Guam Harbor Waters (Chamids and Spondylids)	114
Table 28:	PAHs in Octopus, Mantis Shrimp and Ascidians from Guam Harbor Waters	115
Table 29:	PAHs in Tissues of Fish from Guam Harbor Waters	116
Table 30:	Evaluation of Bioindicator Potential of the Various Organisms Analyzed During this Study	125
Table 31:	Compilation of Legal Limits for Hazardous Metals and PCBs in Fish and Fisheries Products	128

ABSTRACT

The data presented herein represents Phase II of a two part program to evaluate levels of heavy metals, polychlorinated biphenyls (PCBs) and polycyclic aromatics (PAHs) in abiotic and biotic components of four harbor environments on Guam. Phase I focused on sediments and clearly identified areas of enrichment for all three contaminant groups in Agana Boat Basin, the outer portion of Apra Harbor, and the Merizo Pier area. The data from this investigation were presented and discussed at length in an earlier report (Denton *et al.* 1997). In the present study, all four harbors were revisited and dominant biotic representatives were collected in order to evaluate contaminant movement into marine food chains. The sampling sites ranged from relatively enriched through to relatively clean and were identified during Phase I of the study. The dominant biotic groups considered were algae, sponges, soft and hard corals, sea cucumbers, bivalves, and fish. Representatives of each were collected from all four harbor locations. In addition, a limited number of ascidians, an octopus, and a stomatopod crustacean were collected from Apra Harbor.

The findings of the survey were evaluated, following a detailed comparative analysis with published findings, for similar and related species from elsewhere. It was concluded that Guam's harbor environments are generally clean by world standards, although mild to moderate enrichment of the biota with arsenic, copper, lead, mercury, tin and PCBs was evident at certain sites.

Oysters from Agana Boat Basin and Apra Harbor were heavily contaminated with copper and zinc. Sponges, soft corals and sea cucumbers from Apra Harbor also contained relatively high concentrations of arsenic, presumably reflecting releases of this element from fuel combustion as well as from past uses in biocides and wood preservatives. All three biotic groups from this location were also relatively enriched with PCBs, a feature they had in common with the majority of fish captured here. Sea cucumbers and fish from Apra Harbor also contained higher mercury concentrations than specimens from the other harbor sites.

The data for tin contrasted sharply with the findings described above. For this element, levels were appreciably higher in sponges, soft corals and sea cucumbers from within the smaller boat harbors compared with those from Apra Harbor. These findings are in line with reports from elsewhere, that marinas and small boat harbors are generally more prone to tin (TBT) problems than larger ports and harbors; a factor attributed to the higher density of boating traffic and permanently moored water-craft. However, they are not supported by our previous sediment data for tin at each of these locations.

None of the fish or shellfish contained levels of any contaminant that exceeded current U.S. FDA food standards or guidance limits. The absence of an FDA food standard for copper and zinc was duly noted in light of the high levels of these metals in oyster from Agana Boat Basin and Apra Harbor. Levels found in these bivalves frequently exceeded the Australian food standards for both elements. There was no evidence to support an increase in the biological availability of silver, chromium, nickel or PAHs at any of the harbor sites examined.

INTRODUCTION

Historically, the sea has been a major source of protein to the people of Guam and, notwithstanding the variety of imported foods, fishing is still an important occupational and recreational activity today. The fringing reefs, lagoons and offshore waters provide habitats for a great diversity of edible marine life, including a variety of algae, mollusks, crustaceans, sea cucumbers (bêche-de-mer), and many different kinds of fish. Local inhabitants commonly harvest representatives from each of these groups for sale or home consumption (Amesbury *et al.* 1986).

By virtue of Guam's geographic location, these resources have been relatively isolated from the adverse effects of pollution generated by the industrialized nations of the world. However, Guam has undergone tremendous commercial growth and development over the last 10-15 years, particularly in areas related to the tourism and hospitality industry. In addition, the local population has grown appreciably in the wake of improved living standards and a generally healthier job market. Such expansions, although economically desirable on one hand, have greatly contributed to Guam's waste disposal, pollution, and environmental management problems on the other.

Up until a few years ago, much of the marine environment surrounding the island was considered to be pristine. Today, coastal waters along much of central Guam's western shoreline are now utilized for a variety of water sports including recreational and commercial boating and jet skiing activities. Moreover, a number of bays on this side of the island are inundated with storm water runoff from hotel car parks and adjacent highways during the wet season, while others receive wastewater discharges from several of the island's primary sewage treatment plants.

Further anthropogenic expansion into Guam's coastal waters seems almost inevitable given the long-term growth and development predicted for the island. Therefore, it is imperative that the ecological impact of such progress and its effects on the delicate balance of the environment be carefully monitored, in order that a harmonious and viable ecosystem can be developed and maintained.

The precise impact of man's current level of intrusion into Guam's coastal waters is largely unknown. We also know very little about the degree of chemical contamination derived from the activities and events outlined above, and the accompanying water quality changes they bring about. Clearly, such information is vital if the ecological, recreational, and commercial potential of our nearshore waters is to be preserved.

Recognizing this important need, the Guam Bureau of Planning established the Guam Coastal Management Program (GCMP) to develop management strategies for the sustainable development of resources within this environmentally sensitive area. This included the identification and evaluation of major coastal point and non-point pollution sources, the identification of potential health risks to consumers of contaminated fisheries, and the establishment of a sensibly planned and readily implemented pollution monitoring program.

As a first step in this direction GCMP approached the Water & Environmental Research Institute (WERI), at the University of Guam, to undertake a preliminary baseline survey of heavy metals, PCBs, and PAHs in abiotic and biotic components from four harbors on the western side of the island (Fig. 1). The rationale behind the study was that harbor environments are often enriched in various organic and inorganic pollutants derived primarily from watercraft of one sort or another. Other important contaminant sources in these areas are wind-blown dust and surface runoff from a multitude of contributing harbor activities. Thus, marine harbors usually represent "worst case" nearshore conditions within any particular area.

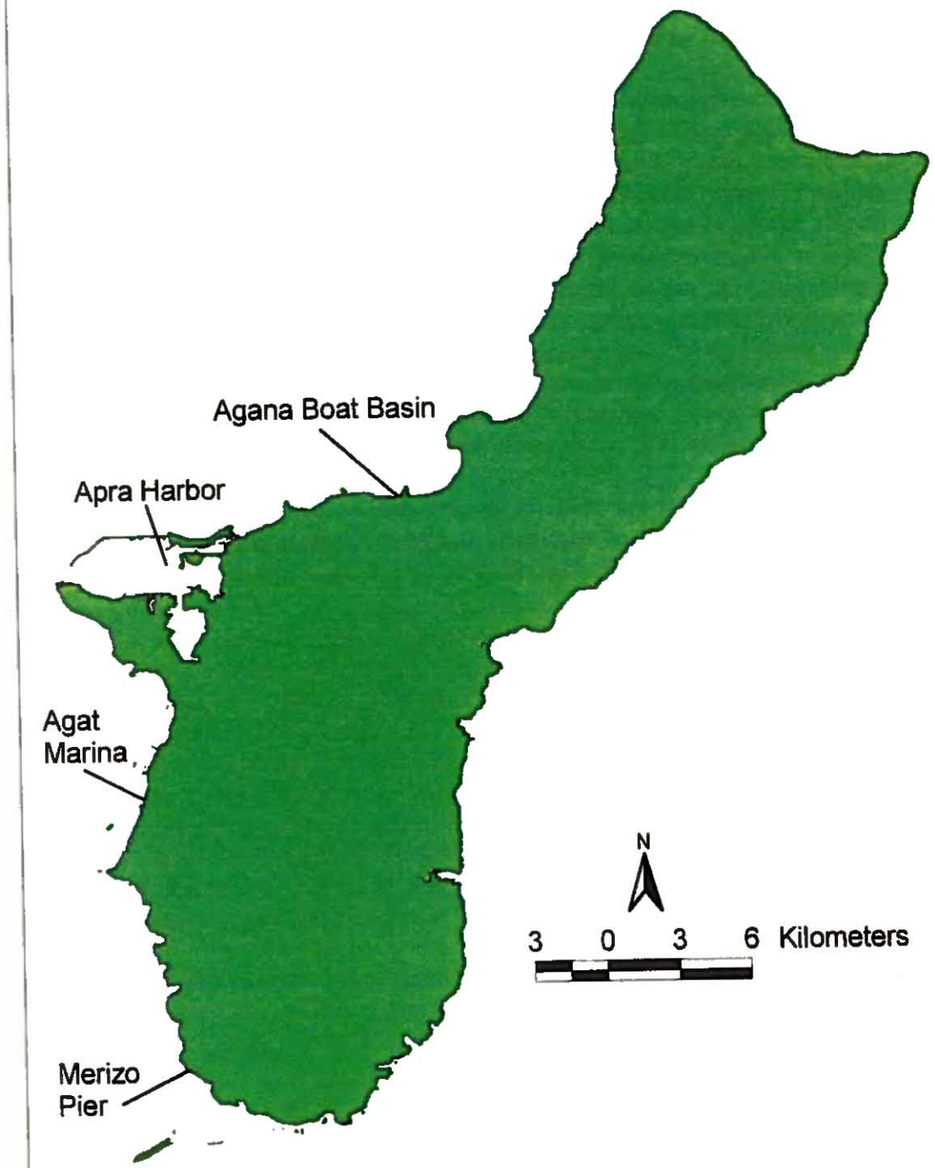
The contaminant groups mentioned above are important both from an ecotoxicological and public health standpoint and included representatives that are prevalent and persistent in the environment, have a high bioaccumulation potential, and exert harmful effects on biological systems at relatively low concentrations.

The major objectives of the study were as follows:

- Determine the presence and abundance of a range of heavy metals and several PCB and PAH congeners in sediments and biota from strategic sites within Agana Boat Basin, Apra Harbor, Agat Marina and Merizo Pier.
- Highlight localized 'hot-spots' and specific point sources of contamination.
- Develop numerical sediment quality guidelines to assist in the decision making process related to any future disposal of locally dredged sedimentary materials at sea.
- Evaluate the bioaccumulation potential of sediment bound contaminants within identified areas of enrichment, identify vulnerable foci within local marine food chains and indicate which organisms exhibit the highest bioaccumulation factors.
- Initiate the provision of a sound database with which future levels may be compared and evaluated.
- Provide data of immediate public health importance for those species frequently consumed by man.
- Assess the degree of background contamination at each location by reference to levels reported in clean and polluted environment elsewhere and with special reference to other tropical regions of the world.
- Provide a bank of data upon which GCMP and others may draw when evaluating environmental problems relating to the management and maintenance of water quality and the protection of marine resources within Guam's coastal waters.

The study was conducted in two distinct phases. Phase 1 focused on the chemical analysis of sediments taken immediately adjacent to suspected sources of chemical contamination (piers, jetties, docksides, refueling stations, navigational channels, etc.) as well as along fixed transects that followed presumed chemical concentration gradients. Overall, a total of 46 sub-tidal sites were examined. The survey clearly demonstrated enrichment of all contaminant groups in Agana Boat Basin, Outer Apra Harbor and Merizo Pier, although by world standards, the majority of sites within each location were considered to be relatively clean.

Fig. 1: Guam Harbors Visited



The highest levels of all three chemical groups were found at Apra Harbor, the largest and oldest port on Guam. Here, moderate to heavy enrichment of various heavy metals, PCBs and PAHs were identified in sediments collected in the vicinity of Hotel Wharf, Commercial Port, and Dry Dock Island. The lowest contaminant levels were almost always encountered at Agat Marina, a recently constructed small boat harbor to the south of Agana. Full details of the study are presented in an earlier WERI technical report (Denton *et al.* 1997). Copies of this report are available upon written request from the Director of the Institute.

The study reported herein comprises Phase II of the program, designed specifically to monitor heavy metals, PAHs and PCBs in marine organisms from within each of the four harbor locations mentioned above. Emphasis has been given to dominant flora and fauna from clean and contaminated harbor sites identified during Phase 1. These have included organisms from various trophic levels, in addition to those frequently harvested for human consumption. The primary focus of the investigation was on biotic groups popularly used as bioindicators of chemical pollution, e.g., macro algae, bivalve mollusks and certain fish. These organisms generally possess little to no regulatory capacity for some or all of the above contaminants and hence, tissue levels mirror biologically available amounts derived from their immediate surroundings. In addition to these so called 'sentinel' species, some attention was directed towards the collection and analysis of other leading ecosystem representatives, including sponges, ascidians (sea squirts), corals and holothurians (sea cucumbers).

This program is the first of its kind for Guam and, indeed, for Micronesia, and should therefore command the interest of regulators and policy makers involved with the protection and management of coastal waters within the tropics and neo-tropical zones of the world.

MATERIALS AND METHODS

1. HARBOR SITES

General information relating to each harbor studied is given below. Biota collection sites were based upon sediment contamination profiles identified during Phase 1 of the program.

1.1 Agana Boat Basin:

Agana Boat Basin was the most northerly of the four harbors examined during the present study. It is located in the western shores of Agana, the capital and business center of the island, and has been used for small pleasure and commercial craft for over 40 years. The facility is divided into two discrete areas by a breakwater that separates the inner permanent moorings and floating walkways from an outer lagoonal area. It is protected from the ocean swell by a larger outer breakwater and connects with the open sea via a deep-water channel along its eastern edge (Fig. 2). The collection of biota focused on the inner boat basin, a relatively contaminated area with restricted water circulation. Sediments from this section contained high levels of copper, lead and zinc, and moderate levels chromium, mercury, tin, PCBs and PAHs (Denton *et al.* 1997). Primary pollution sources in this area, apart from the high intensity of watercraft, included a storm drain outlet, a refueling station and a nearby wastewater treatment plant. Biota of interest that were absent from the inner boat basin were collected from the outer lagoon (see Fig. 2)

1.2 Apra Harbor:

Apra Harbor is the largest harbor on Guam, and has been used by small pleasure and commercial craft and larger commercial and military shipping for more than a century. Geographically it is divided into an inner and outer area. The US navy has used the inner harbor as a ship repair and maintenance facility for the last 55 years. Sediments from this area and the immediately adjacent portion of the outer harbor are known to be high in copper, mercury, nickel, lead, tin and zinc (Belt Collins 1993). Sedimentary levels of PCBs and PAHs in this area are currently unknown. The outer harbor includes Sasa Bay, a safe refuge and permanent mooring site for a number of privately owned sailing craft; Dry Dock Island, a US navy dry dock facility that is now obsolete; and a series of wharves along the northern perimeter for the unloading of large container ships. Primary pollution sources in this area, aside from the major shipping and harbor activities, included several fuel piers and fuel storage depots (tank farms), electrical substations and transformers, and stormwater runoff from wharves, piers and adjacent buildings.

Sites selected for biota analysis were Hotel Wharf, Shell Fox-1 Fuel Pier, the western end of Commercial Port, Dry Dock Island, and Echo Wharf (Fig. 3). The Echo Wharf area was selected as a control site based on low sedimentary levels of all contaminants examined earlier. Sediments from the remaining sites were found to be moderately to highly enriched with the following contaminants:

- Hotel Wharf (copper, lead, mercury, tin, zinc, PCBs, PAHs)
- Shell Fox-1 Fuel Pier (copper, lead, mercury, zinc, PCBs, PAHs)
- Western Commercial Port (copper, lead, mercury, zinc, PCBs)
- Dry Dock Island (copper, lead, mercury, zinc, PCBs, PAHs)

Fig. 2: Agana Boat Basin
(Biota Sampling Sites)

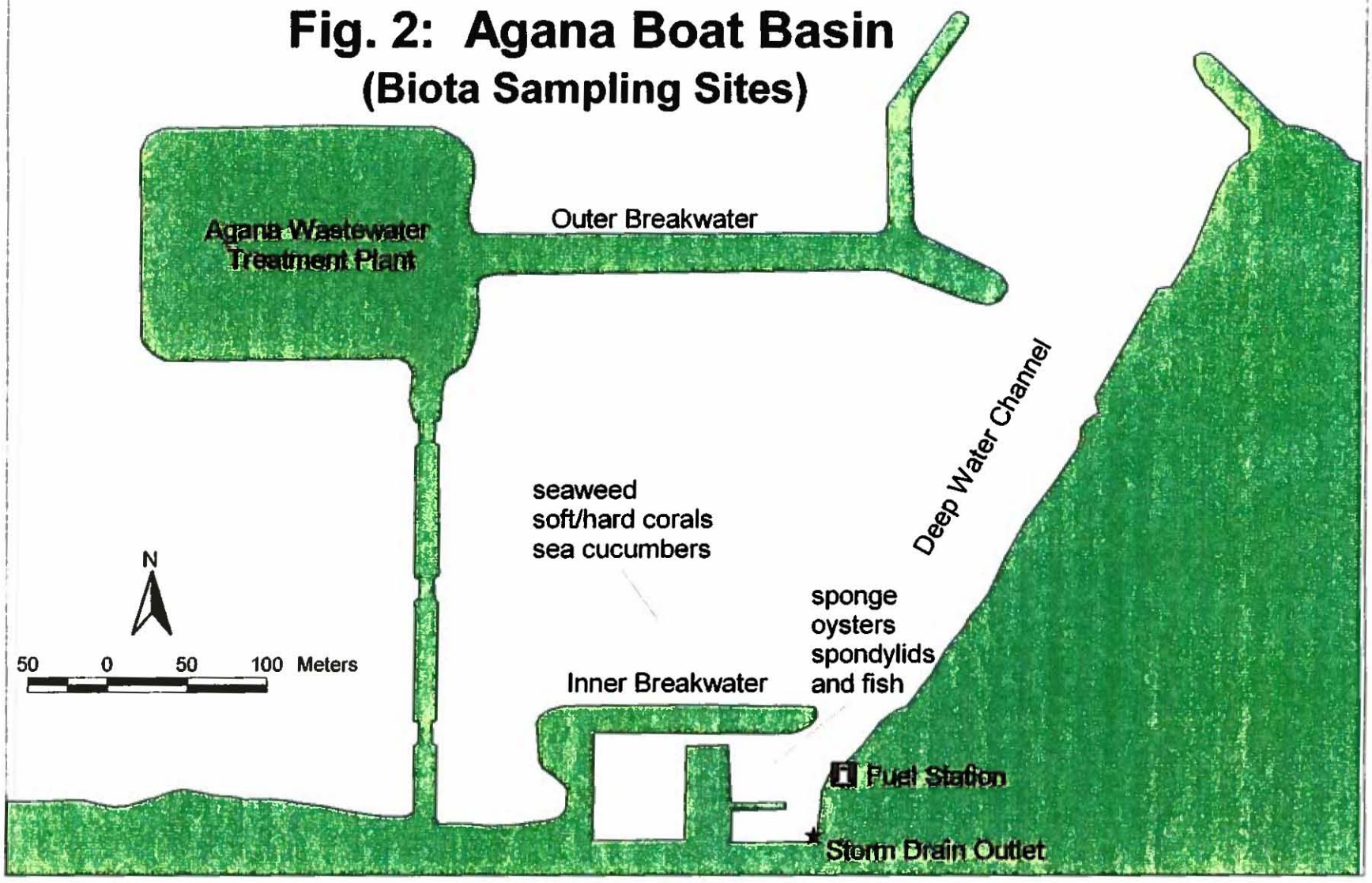
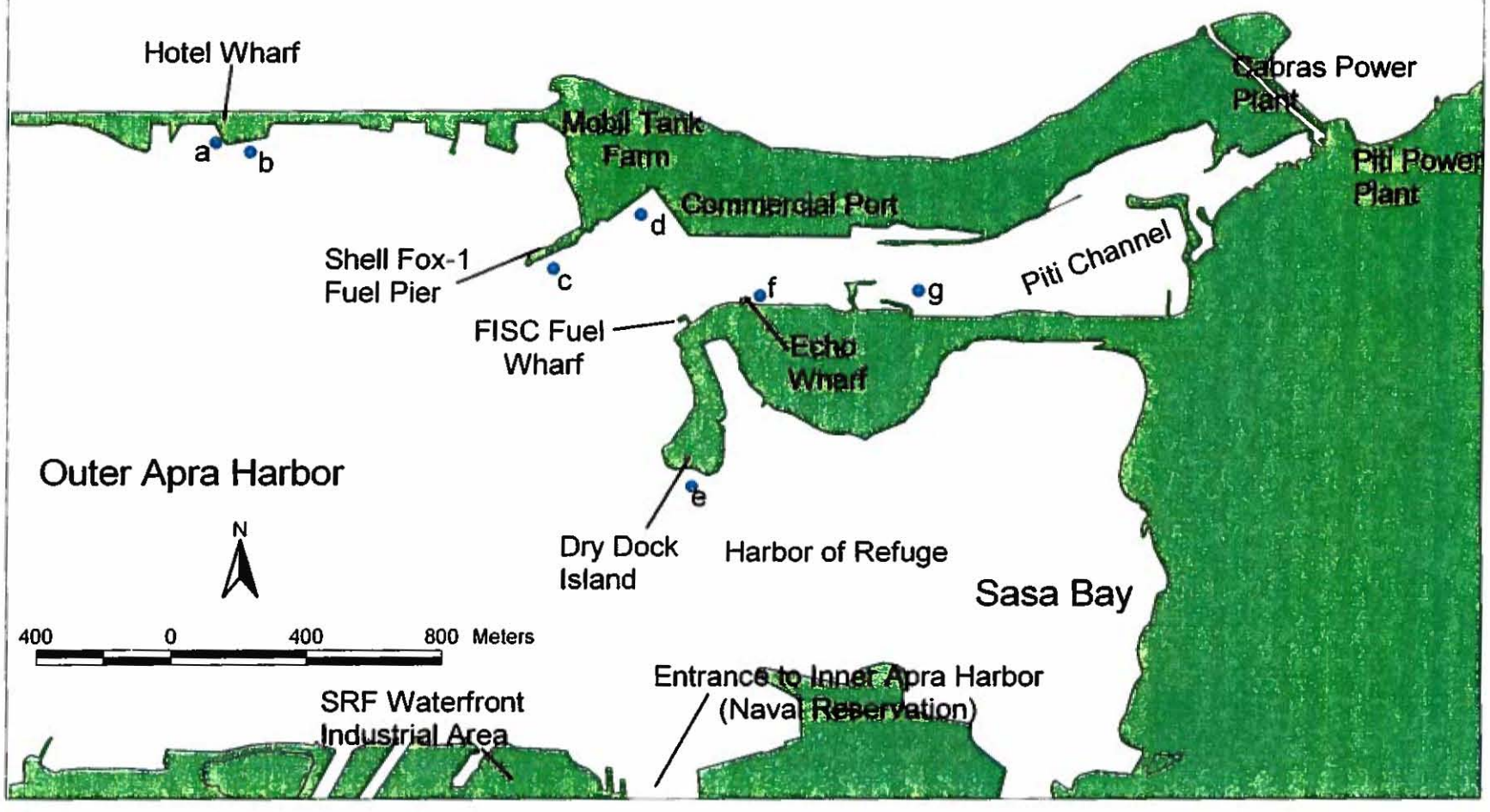


Fig. 3: Apra Harbor
(Biota Sampling Sites a-g)



1.3 Agat Marina:
Agat Marina is a relatively new, small boat harbor that has been in existence since 1990. It is located approximately 8 km south of Apra Harbor in the semi rural setting of Agat village. Permanent mooring sites are available for about 50 vessels. Although sediments from this harbor were lightly contaminated with chromium, they were classified as clean for all other contaminants examined (Denton *et al.* 1997). Potential sources of pollution in this area are limited to contributions from watercraft, stormwater runoff from the adjacent car park area, and a refueling pier at the southern entrance. There may also be some impact from the Agat sewage treatment plant that discharges primary treated effluent nearshore, in about 2 m of water, approximately 3 km to the north. Biota samples were collected from various points throughout the harbor (Fig. 4)

1.4 Merizo Pier:
The Merizo Pier area is located within Guam's largest barrier reef and is the southernmost harbor facility on island. This small boat launch site has been in existence for about 35 years and is a popular area for recreational boating and related water sports activities. It is also the gateway to Cocos Island, a popular tourist spot located about 3 km off shore and accessed by ferry. The Cocos Island ferry pier, more or less, marks the southern limit of the impacted coastline, which extends northwest for about 200 m to a large public pier and popular fishing spot. A deep-water navigation channel running parallel to the beach is situated about 25 m offshore. The general layout of the area suggests that the waters are well mixed by the prevailing winds, tides, and ocean currents.

Sediments from the deep-water channel were previously classified as clean for all contaminants of interest (Denton *et al.* 1997). However, those collected in shallower waters closer to shore demonstrated moderate to heavy enrichment with copper, lead, tin and zinc, especially in the vicinity of the Cocos Island Ferry terminal. PCB and PAH contamination of these sediments, on the other hand, was generally light. Potential sources of pollution are largely restricted to the ongoing boating activities, a couple of derelict and partially submerged barges and a shoreline refueling station that services the Cocos Island ferries. Biota samples were collected along the entire length of the impacted shoreline (Fig. 5)

2. Sample Collection and Preparation

A listing of all the organisms collected for analysis is shown in Table 1. While not exhaustive, it includes representatives of several major phyla in addition to a number of organisms of direct and potential economic importance. It also readily demonstrates the species that are most widely distributed and, therefore, of the greatest use for future pollution monitoring programs. We point out that not all species were available at all sites visited.

Biota samples were collected between June 3, 1998 and January 30, 1999. In most cases the organisms were collected by scuba diver and were simply handpicked off the ocean floor, coral reef, or side of a submerged structure. However, the bivalves did not readily facilitate this method of collection and were usually removed from their point of attachment with the aid of a hammer and chisel. Fish taken during the study were captured using spear gun and hook and line.

Fig. 4: Agat Marina
(Biota Sampling Sites)

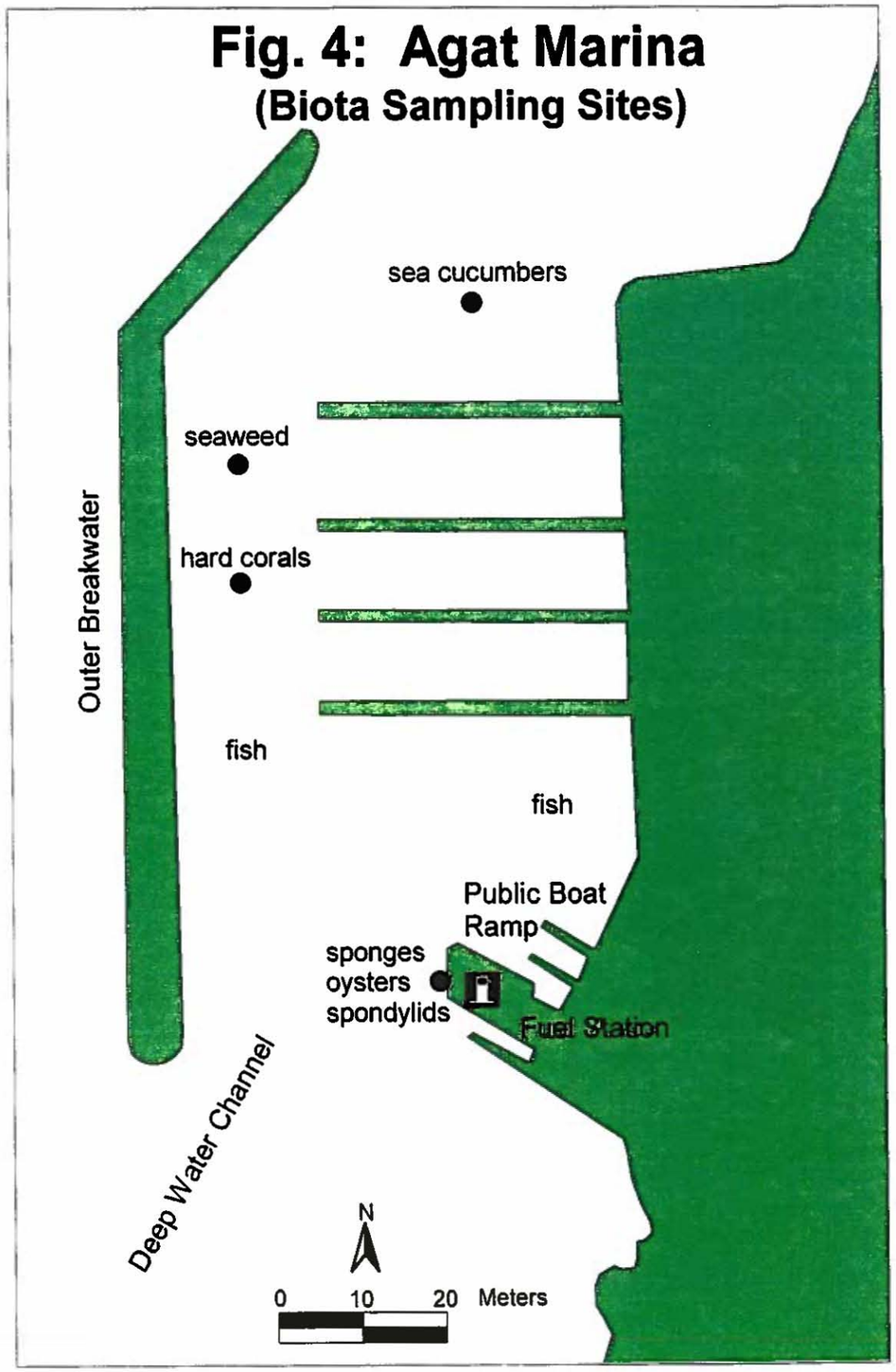


Table 1

Flora and Fauna Sampled During the Present Survey

Species Collected for Analysis	Agana Boat Basin	Apra Harbor (site a)	Apra Harbor (site b)	Apra Harbor (site c)	Apra Harbor (site d)	Apra Harbor (site e)	Apra Harbor (site f)	Apra Harbor (site g)	Agat Marina	Merizo Pier
BROWN ALGA										
Padina sp.	x	x		x	x	x	x		x	x
SPONGES										
Callyspongia diffusa									x	
Cinachyra sp.	x									x
Clathria vulpina ?									x	x
Dysidea sp.				x	x		x		x	
Liosina cf. granularis						x				
Stylotella aurantium			x			x				x
Yellow bread sponge									x	
Yellow sponge (red outside)				x						
Brown wart sponge			x			x	x			
Orange brown wart sponge						x				
HARD CORALS										
Acropora formosa						x				
Fungia concinna				x						
Fungia echidata						x				
Herpolitha limax				x		x				
Pocilopora damicornis	x			x	x		x		x	x
SOFT CORALS										
Sinularia sp.	x			x		x				x
SEA CUCUMBERS										
Bohadschia argus	x		x	x		x			x	x
Holothuria atra	x					x		x	x	x
BIVALVE MOLLUSKS										
Chama lazarus			x	x	x	x	x			x
Chama brassica					x					
Saccostrea cucullata				x						x
Spondylus ? multimuricatus	x								x	
Striostrea cf. mytiloides	x	x				x	x		x	x
CEPHALOPOD MOLLUSK										
Octopus cyanea				x						
STOMATOPOD CRUSTACEAN										
Gonodactylus sp. (mantis shrimp)						x				
TUNICATES										
Ascidia sp.			x			x				
Rhopalaea				x	x	x				

Table 1 (cont.)

Flora and Fauna Sampled During the Present Survey

Species Collected for Analysis	Agana Boat Basin	Apra Harbor (site a)	Apra Harbor (site b)	Apra Harbor (site c)	Apra Harbor (site d)	Apra Harbor (site e)	Apra Harbor (site f)	Apra Harbor (site g)	Agat Marina	Merizo Pier
FISH										
Acanthurus xanthopterus	x			x			x			
Balistoides viridescens										x
Bolbometopon muricatum				x						
Caranx ignobilis	x									
Caranx melampygus			x			x				
Caranx sexfasciatus	x			x	x					
Cephalopholis sonnerati										x
Cheilinus chlorourus									x	
Cheilinus fasciatus				x						
Cheilinus trilobatus										x
Ctenochaetus binotatus					x					
Ctenochaetus striatus						x	x		x	
Epibulus insidiator				x		x				
Epinephelus merra										x
Gerres argyreus	x				x					
Gymnothorax javanicus			x							
Leiognathus equulus									x	
Lethrinus rubrioperculatus									x	x
Lutjanus kasmira										x
Monodactylus argenteus	x				x					
Naso annulatus						x				
Naso unicornis		x	x							
Odenus niger									x	
Parupeneus barberinus										x
Parupeneus cyclostomus										x
Parupeneus multifasciatus										x
Saurida gracilis	x								x	
Saurida nebulosa			x							x
Scarus sordidus						x				
Siganus spinus	x									
Sufflamen chrysoptera						x				
Valamugil engeli				x						

Key to Apra Harbor Sites:

Apra Harbor (site a) = Western end of Hotel Warf
Apra Harbor (site b) = Central Hotel Warf
Apra Harbor (site c) = Shell Fox-1 Fuel Pier
Apra Harbor (site d) = Northwestern end of Commercial Port
Apra Harbor (site e) = Southern end of Dry Dock Island
Apra Harbor (site f) = Eastern end of Echo Warf
Apra Harbor (site g) = Off Port Authority Beach

Upon collection, all samples except the bivalves were immediately wrapped in aluminum foil and placed on ice. The bivalves were held in seawater for approximately 6 h to allow them to purge their gut contents. In the laboratory, all organism were thoroughly cleaned of epiphytic growth and/or adhering particulate material before subsampling for analysis. With algae, the holdfasts and older, more encrusted portions of the plant were discarded and only the fronds were taken for analysis. With the sponges, it was also necessary to carefully pare away sediment laden portions of the exterior and interior surfaces prior to subsampling. The sponges and ascidians were analyzed whole. Likewise, the entire soft parts of the bivalves were taken for analysis. In contrast, specific tissues were removed from the sea cucumbers (dorsal body wall and hemal system), octopus (tentacle and liver), mantis shrimp (tail muscle and gonad) and fish (axial muscle and liver). With fish, muscle samples were taken immediately below and parallel to the dorsal fin (left side of the body for heavy metals and right side for PCBs and PAHs).

Samples for heavy metal analysis were stored in acid-cleaned, polypropylene vials while those for PCB and PAH analyses were wrapped in aluminum foil and placed in precleaned glass jars. All tissue samples were held at -20°C until required for analysis.

Samples for the analysis of all metals, except mercury, were performed on tissues dried to constant weight, in an oven, at 60°C. Owing to the relatively high volatility of mercury, analysis was conducted on wet rather than dry tissues.

Appropriate analytical methods for the above contaminants were adapted from the current SW-846 protocols developed by U.S. EPA (USEPA 1995) for the physical and chemical evaluation of solid waste, in addition to those recommended by the NOAA National Status and Trends Program for Marine Environmental Quality (NOAA 1993a-d). Appropriate quality control and quality assurance procedures including full procedural blanks, matrix spikes, and certified reference materials were built into the analytical protocols.

3. HEAVY METAL ANALYSIS

All tissue samples were analyzed for heavy metals following conventional wet oxidation procedures in hot mineral acids. The digestion procedures were essentially similar to EPA method 3050A, SW-846 (USEPA 1995) with minor modifications as outlined below.

3.1 Mercury:

Approximately 1 g of wet tissue was accurately weighed into a 125 ml Erlenmeyer glass flask and allowed to stand overnight in 10 ml of a 2:1 mixture of concentrated nitric and sulfuric acids. Several bivalve samples that were too big to analyze individually were split into two or more portions and digested separately. The following day the cold digests were heated to 100°C in a boiling water bath for 3-hours. Each flask was loosely capped with a Teflon stopper to facilitate good refluxing and exclude extraneous contaminants. After cooling, the digests were made up to volume with deionized water (75-ml), and analyzed by flameless atomic absorption spectroscopy (AAS) using the syringe technique described by Stainton (1971). Calibration standards (5-20 ng/l) were made up in 10% nitric acid containing 0.05% potassium dichromate as a preservative (Feldman 1974).

3.2 All Other Metals

Between 1-3 g of dried tissue were accurately weighed into the digestion flasks described above. Approximately 10 ml of concentrated nitric acid was added to each flask and they were allowed to stand overnight. The following day the digests were heated to 100°C ± 5°C and allowed to reflux for 2-3 days. The solutions were then evaporated to dryness and further additions of acid were made as necessary to completed digestion. Finally, digests were made up to volume with 10% nitric acid (10 ml/g tissue weight) and analyzed by AAS within 5 working days. Blanks (two per batch of 40 digests) were treated similarly. Corrections for non-atomic absorption were made simultaneously by the instrument.

Arsenic and tin were analyzed by cold vapor AAS using the hydride generation technique. For arsenic, between 50-1000 µl of sample were accurately dispensed into a polypropylene reaction vessel containing 4 ml of 1.5% HCl. The total volume was adjusted to 5 ml with 10% nitric acid. Arsine gas was generated by reduction of the sample with 1% sodium borohydride in 3% sodium hydroxide. All calibration standards (1-10 µg/l) and sample dilutions were made up in 10% nitric acid.

For tin, 1 ml of sample was added to 5 ml of saturated boric acid (50g/l). For smaller sample volumes, adjustments to a 6-ml total volume were made using 10% nitric acid in order to minimize changes in pH. Stannane gas was generated with 0.5% sodium borohydride in 3% sodium hydroxide. Calibration standards (5-20 µg/l) were made up in saturated boric acid solution on a daily basis. Levels of both metals in each sample were calculated by standard addition to compensate for matrix interference.

All other metals were analyzed directly by conventional flame Atomic Absorption Spectroscopy (AAS). All methods were validated using standard reference materials and or spiked tissue composites as shown in Table 2.

4. PCB and PAH Analysis

All samples were analyzed for these contaminants with the exception of the hard corals. All solvents used were pesticide grade and were checked for interfering contaminants following a 500-fold volume reduction before use (50 ml to 100 µl). Surrogates and internal standards used to determine PCB recoveries were PCB 103 (100 pg/µl) and petachloronitrobenzene (250 pg/µl) respectively. The equivalent compounds used for PAH analysis were deuterated acenaphthene and benzo[a]pyrene as the surrogates (50 ng/µl), and deuterated naphthalene as the internal standard (50 ng/µl). The extraction and cleanup procedures outlined below were customarily performed on sets of five wet tissue samples with an accompanying method blank.

4.1 Solvent Extraction:

The samples were removed from the freezer and allowed to thaw. Using stainless steel scissors and forceps, approximately 3 ± 0.1 g of tissue sample were accurately weighed to the nearest 0.01 g into a 50-ml Teflon centrifuge tube. All bivalve specimens were macerated and thoroughly mixed in their glass storage jars beforehand using a Tekmar Tissumizer probe. A sub-sample was then transferred into a centrifuge tube using a Teflon coated spatula.

Table 2

Recovery of Heavy Metals from Standard Reference Materials
(Data are Mean ± 95% Confidence Limits)

Metal	Apple Leaves (SRM 1515)		Bovine Liver (SRM 1577b)	
	This Study	Certified Value	This Study	Certified Value
	µg/g dry wt		µg/g dry wt	
Arsenic	0.032 ± 0.026	0.038 ± 0.007	0.060 ± 0.026	0.05*
Cadmium	<0.04 - 0.07	0.013 ± 0.002	0.58 ± 0.17	0.50 ± 0.03
Copper	5.02 ± 0.18	5.64 ± 0.24	152 ± 31	160 ± 8
Chromium	0.82 ± 0.57	0.3*	1.05 ± 1.04	-
Mercury	0.057 ± 0.012	0.044 ± 0.004	0.005 ± 0.011	0.003*
Nickel	0.66 ± 0.20	0.91 ± 0.12	<0.18 - 0.23	-
Lead	0.47 ± 0.32	0.470 ± 0.024	<0.30 - <0.38	0.129 ± 0.004
Silver	<0.09 - <0.11	-	<0.10 - <0.13	0.039 ± 0.007
Tin	0.003 - 0.03	<0.2*	<0.004 - 0.07	-
Zinc	11.2 ± 3.28	12.5 ± 0.3	110 ± 16.9	127 ± 16

* Certified Value not available. Dashes indicate no data

Table 2 (cont.)

Recovery of PCBs from Standard Reference Material and Spiked Oyster Composite

PCB Congener	Certified Mean plus/minus (95% Confidence Limits)	This Study: Mean and (Range)	Spike Added (ng)	Recovered Amount (ng) Mean and (Range)
SRM 2974: Marine Mussel			Oyster Composite	
PCB 8	no value	no value	10	11 (9.8 - 12.1)
PCB 18	26.8 (23.5 - 30.1)[a]	14.9 (11.6 - 18.7)	10	9.8 (8.9 - 10.6)
PCB 28	79 (64 - 94)[a]	59.2 (41.5 - 77)	10	13.4 (11.9 - 15)
PCB 52	115 (103 - 127)	76.5 (57.1 - 93.9)	10	5.0 (3.1 - 6.9)
PCB 44	72.7 (65 - 80.4)	50.6 (41.1 - 60.1)	10	12.2 (10.9 - 13.6)
PCB 66	101.4 (96 - 106.8)	77.1 (62.1 - 86.3)	10	12.2 (10.6 - 13.7)
PCB 101	128 (118 - 138)	102.9 (75.8 - 119.1)	10	8.9 (6 - 11.8)
PCB 77	no value	no value	10	15.8 (13.7 - 18)
PCB 118	130.8 (125.5 - 136.1)	125.5 (101.7 - 144.4)	10	11.1 (9.5 - 12.7)
PCB 153	145.2 (136.4 - 154)	92.5 (86.3 - 103.3)	10	7.5 (6.9 - 8)
PCB 105	53 (49.2 - 56.8)	41.6 (36.1 - 47.6)	10	11.7 (9.9 - 13.6)
PCB 138	134 (124 - 144)	65.5 (56.4 - 77.8)	10	7.2 (6.3 - 8.3)
PCB 126	no value	no value	10	13.8 (11.2 - 16.3)
PCB 187	34 (31.5 - 36.5)	21.1 (17.9 - 23.3)	10	6.4 (5.1 - 7.8)
PCB 128	22 (18.5 - 25.5)	13.1 (10.3 - 15.1)	10	8.8 (7.5 - 10.2)
PCB 180	17.1 (13.3 - 20.9)	7.7 (5.1 - 9.3)	10	5.5 (4.6 - 6.5)
PCB 170	5.5 (4.4 - 6.6)	2.1 (1.2 - 2.8)	10	6.9 (5.8 - 8.1)
PCB 195	no value	no value	10	5.6 (4.7 - 6.5)
PCB 206	no value	no value	10	3.2 (2.5 - 3.9)
PCB 209	no value	no value	10	1.8 (1.3 - 2.3)

a = unconfirmed reference value only

Table 2 (cont.)

Recovery of PAHs from Spiked Oyster Composite

PAH Congener	Spike Added (µg)	Recovered Amount (µg) Mean and (Range)
Naphthalene	1	0.16 (0.15 - 0.17)
Acenaphthylene	1	0.23 (0.11 - 0.33)
Acenaphthene	1	0.26 (0.11 - 0.36)
Fluorene	1	0.22 (0.19 - 0.25)
Phenanthrene	1	0.40 (0.22 - 0.54)
Anthracene	1	0.34 (0.18 - 0.48)
Fluoranthene	1	0.41 (0.24 - 0.55)
Pyrene	1	0.42 (0.23 - 0.56)
Benzo(a)anthracene	1	0.33 (0.22 - 0.43)
Chrysene	1	0.40 (0.24 - 0.53)
Benzo(b)fluoranthene	1	0.39 (0.22 - 0.53)
Benzo(k)fluoranthene	1	0.39 (0.21 - 0.53)
Benzo(a)pyrene	1	0.34 (0.19 - 0.48)
Dibenzo(a,h)anthracene	1	0.39 0.22 - 0.53)
benzo(g,h,i)perylene	1	0.38 (0.20 - 0.54)
indenol(1,2,3-cd)pyrene	1	0.39 (0.22 - 0.52)

Following the addition of 10 g of anhydrous, granular sodium sulfate (heated to 600°C overnight), 20 ml of methylene dichloride, and 100 µl of the PCB and PAH surrogates, each tissue sample was homogenized using the Tissumizer (setting 50 for approximately two minutes). After rinsing down the probe into the centrifuge tube with clean solvent, the extract was centrifuged at 2000 rpm for 5 minutes before decanting into a Turbo-Vap™ evaporator tube (Zymark). The extraction was repeated once more and added to the contents of the evaporator tube. After volume reduction to approximately 0.5 ml, the extract was quantitatively transferred to a 10-ml graduated, glass centrifuge tube with two 0.5-ml rinses of methylene chloride. The tube was placed in a warm water bath and the extract volume reduced to ~0.25 ml under a gentle stream of nitrogen. Solvent exchange into hexane (~1.0 ml) and further reduction in volume (~0.2 ml) was necessary before cleanup.

4.2 Silica/Alumina Column Cleanup:

Cleanup was accomplished with small columns of silica gel (grade 923, 100-200 mesh) and neutral alumina (F-20, 80-200 mesh). Both adsorbents were activated and cleaned by heating to 600°C overnight. The adsorbents were supported in glass, chromatographic columns, 280 mm in length and 7 mm internal diameter (i.d.). These were obtained commercially obtained from *Supelco*. The upper 80-mm of each column was expanded to form a 50-ml solvent reservoir. Just prior to use, the columns were plugged at their lower end with cotton wool, rinsed with clean solvent and allowed to drain. Upon packing, each column was filled with methylene chloride. The solvent was prevented from draining by a Teflon cap fitted over the lower end of the column. Slurries of alumina (1 g) and silica gel (2 g) were sequentially washed into the column reservoir with methylene chloride taking care to allow for the displacement of trapped air bubbles. After settling (facilitated by gently tapping the column), the individual alumina and silica gel portions of the column were approximately 3.2 cm and 9 cm in length respectively. Packed columns were washed with a further 20-ml of methylene chloride followed by 2 x 20-ml volumes of pentane in final preparation. The laboratory temperature was kept lower than 27°C at all times to avoid vapor pockets from forming in the columns.

The concentrated tissue extract was transferred to the cleanup column after draining the pentane wash to the packing top. Two rinses of ~0.25 ml of hexane were used to complete the transfer. The column was eluted with 5 ml of pentane (discarded) followed by 10 ml of 50% methylene chloride in pentane. The latter fraction containing the PCBs and PAHs was collected in a 10-ml graduated, glass centrifuge tube, evaporated to 5 ml and split into two 2.5-ml fractions. The first fraction was solvent exchanged with hexane for PCB analysis while the second fraction was solvent exchanged with acetonitrile for PAH determination. Both fractions were reduced to a final volume of 0.1 ml before transfer to clean, glass auto-sampler vials with small volume inserts (250 µl). Finally, 10 µl of the appropriate internal standard was added to each vial before chromatographic analysis.

4.3 Chromatographic Parameters for PCB Analysis:

PCB analysis was carried out by Gas Chromatography (Varian 3400CX) using an electron capture detector and a 60 m x 0.25 mm i.d. fused silica MDN-5S, polymethyl-5% phenyl-siloxane (0.25µm film thickness) capillary column (Supelco). Gas flows (nitrogen), through the column and the detector, were 1 ml/min and 30 ml/min respectively. The initial column

temperature was maintained at 50°C for the first minute of each run. It was then ramped to 150°C at 30°C/min, then to 280°C at 25°C/min, where it was held for 20 min to give a total run time of 76 min. Both the injector and detector temperatures were held constant at 280°C and 310°C respectively.

PCB quantification was accomplished using a 20-congener calibration standard representing PCB homologues Cl_2 to Cl_{10} (NOAA 1993a). The congeners, listed in Table 3, were selected on the basis of their potential toxicity, bioaccumulation and/or frequency of occurrence in environmental samples. Complete chromatographic separation of all congeners was achieved although several of them are known to co-elute with other PCB congeners present in commercial PCB mixtures (Table 3).

PCB homologue concentrations were estimated from the data by summing values obtained for congeners of similar chlorine content. The "total" PCB content of the sample was calculated from the sum of the individual congener data (Σ_{20}PCB). PCB congener recoveries from the certified standard reference material (SRM 1974) and a spiked oyster composite were generally within acceptable limits (Table 2). Method detection limits for individual chlorobiphenyls in the standard mix ranged from 0.02-0.15 ng/g.

4.4 Chromatographic Parameters for PAH Analysis:

PAH analysis was achieved by High Performance Liquid Chromatography (HPLC) using a fluorescence/UV (diode array) detector system and a 10 cm x 4.6 cm i.d., stainless steel, LC-PAH column (Supelco) containing a porous silica stationary phase (3 µm particle size). Following sample injection, isocratic elution with acetonitrile/water (4:6, v/v) occurred for the first 0.3 min, followed by a linear gradient to 100% acetonitrile over the next 10 min. Elution with 100% acetonitrile continued for a further 5 min before the run was terminated. The solvent flow rate through the column was held constant at 2 ml/min.

Quantification with the more sensitive fluorescence detector was achieved with excitation at 280 nm and emission at 380 nm. The diode array provided a synchronous absorption scan from 190-357 nm, with a wavelength difference of 4 nm, and was used primarily for confirmatory analysis at the higher levels of detection.

The calibration standards were made up containing the 16 PAHs recommended as priority pollutants by the Wold Health Organization (WHO), the European Economic Community (EEC) and the U.S. EPA. These priority pollutants are all parental compounds (i.e., they contain no alkyl substituents) and are major constituents of pyrolytic sources of PAHs. They are listed in Table 4 together with their molecular weights and structural identities. Method detection limits with the fluorescence detector were as follows: naphthalene (34 ng/g), acenaphthene (4 ng/g), fluorene (8 ng/g), phenanthrene (3 ng/g), anthracene (2 ng/g), fluoranthene (5 ng/g), pyrene (3 ng/g), benzo(a)anthracene (1 ng/g), chrysene (1 ng/g), benzo(b)fluoranthene (5 ng/g), benzo(k)fluoranthene (4 ng/g), benzo(a)pyrene (3 ng/g), dibenzo(a,h) anthracene (8 ng/g), and benzo(g,h,i)perylene (13 ng/g). Detection limits for the non-fluorescing PAHs, acenaphthylene and indenol(1,2,3-cd)pyrene, were 3 ng/g and 6 ng/g respectively, using the UV diode array detector.

Table 3

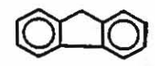
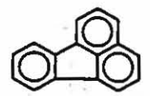
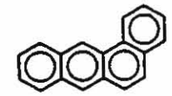
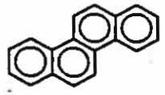
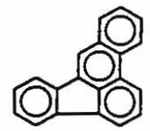
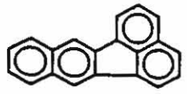
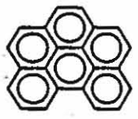
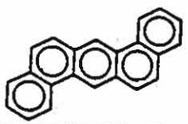
PCB Congeners in Calibration Standard used to Quantify PCB Homologues in Sediment Samples from Harbor Sites on Guam

PCB Congeners in Calibration Standard			Co-eluting PCB Congeners		
IUPAC[1] Number	Chlorine Atoms/mol.	Structural Arrangement	IUPAC Number	Chlorine Atoms/mol.	Structural Arrangement
8[a] (A1221/1242)	2	2,4'	5[a]	2	2,3
18[b] (A1016/1242)	3	2,2',5	15[a] (A1221/1242)	2	4,4'
28[b] (A1016/1242)	3	2,4,4'	31[a] (A1242)	3	2,4',5
44[b] (A1242/1254)	4	2,2',3,5'	none		
52[b] (A1242/1254)	4	2,2',5,5'	43[a]	4	2,2',3,5
66[b] (A1254)	4	2,3',4,4'	80[a]	4	3,3',5,5'
			95	5	2,2',3,5',6
77[a,c]	4	3,3',4,4'	154[a]	6	2,2',4,4'5,6
101[b] (A1254/1260)	5	2,2',4,5,5'	79[a]	4	3,3',4,5'
105[b]	5	2,3,3',4,4'	none		
118[b] (A1254/1260)	5	2,3',4,4',5	106[a]	5	2, 3,3',4,5
126[a,c]	5	3,3',4,4',5	129	6	2,2',3,3',4,5'
128[b]	6	2,2',3,3',4,4'	none		
138[b] (A1254/1260)	6	2,2',3,4,4',5'	158	6	2,3,3',4,4',6
153[b] (A1254/1260)	6	2,2',4,4',5,5'	none		
170[b] (A1260)	7	2,2',3,3',4,4',5	none		
180[b] (A1260)	7	2,2',3,4,4',5,5'	none		
187[b]	7	2,2',3,4',5,5',6	159[a]	6	2,3,3',4,5,5'
			182[a]	7	2,2',3,4,4',5,6'
195[a]	8	2,2',3,3',4,4',5,6	none		
206[a]	9	2,2',3,3',4,4',5,5',6	none		
209[a]	10	2,2',3,3',4,4',5,5',6,6'	none		

[a] not common (<10% occurrence) in environmental samples (from McFarland and Clarke 1989).
[b] major component of environmental mixtures (from NOAA 1993a); [c] highly toxic planar PCB. [1] International Union of Pure & Applied Chemistry.
Labels in parentheses indicate dominant components (≥ 2% by wt.) of the commercial PCB mixtures: Aroclors 1016, 1221, 1242, 1254 & 1260 (from De Voogt *et al.* 1990)
Compilation of chromatographic data from Ballschmiter and Zell (1980); Holden (1986); Ballschmiter *et al.* (1987); De Voogt *et al.* (1990); Rebbert *et al.* (1992); Wise *et al.* (1993); Schantz *et al.* (1993); Bright *et al.* (1995), using 60 m DB-5 (or equivalent) high resolution GC columns.

Table 4

Unsubstituted PAHs in Calibration Standard used to Quantify PAH Levels in Biota Samples from Harbor Sites on Guam

IUPAC[1] Nomenclature	Molecular Wt.	Structural Identity
Naphthalene	128.19	
Acenaphthylene	152.21	
Acenaphthene	154.21	
Fluorene	166.23	
Phenanthrene	178.24	
Anthracene	178.24	
Fluoranthene	202.26	
Pyrene*	202.26	
Benzo(a)anthracene*	228.30	
Chrysene*	228.30	
Benzo(b)fluoranthene*	252.32	
Benzo(k)fluoranthene*	252.32	
Benzo(a)pyrene*	252.32	
Benzo(ghi)perylene	276.34	
Indeno(1,2,3-cd)pyrene*	276.34	
Dibenzo(a,h)anthracene*	278.36	

[1] International Union of Pure and Applied Chemistry; * = known carcinogen

All calculations were based on peak area comparisons of components sharing identical retention times in both sample and standard. From these data, the "total" PAH (Σ_{16}PAH) content of the sample was calculated. PAH recoveries from spiked oyster tissue composites were disappointingly low (Table 2) reflecting perhaps the inadequacies of the cleanup procedure. Nevertheless, they were considered sufficient for the preliminary screening purposes of this project.

5. PRESENTATION OF DATA

All the chemical data accumulated hitherto has been tabulated separately for each contaminant group and is presented in ascending order of organism complexity starting with algae and culminating with fish. It is organized in a way that facilitates quick reference to the concentration and distribution of contaminant levels between sites for any particular species. No adjustments have been made for percentage recoveries from tissue spikes and standard reference materials.

Notes on the significance of the findings precede the tabulated data for each contaminant group. Levels normally encountered in seawater and sediments from clean and contaminated areas are included to facilitate a better understanding of environmental distribution patterns. Comparisons are also made with levels reported in the literature for marine organisms from elsewhere with emphasis, where possible, on those from tropical waters. A selection of published data has been tabulated for easy reference and appears in Tables 5-7 at the end of the current section. From such comparisons, a preliminary appraisal of the degree of contamination, exhibited by biotic resources from within Guam harbors, has been made.

A detailed comparative analysis of sedimentary concentrations with data from other parts of the world, together with likely contaminant sources and suggested sediment quality guidelines for Guam, are presented in a companion report prepared earlier (Denton *et al.* 1997).

Table 5

Heavy Metals in Marine Organisms (μg/g dry weight) from Other Regions of the World

Species	Location	Ag	As	Cd	Cr	Cu	Hg[a]	Ni	Pb	Sn	Zn	Reference
BROWN ALGAE												
Padina australis	Gt. Barrier Reef, Australia	nd	nd	0.4-0.6	nd	2.0-3.0	0.001-0.004	1.0-1.4	<0.9-5.0	nd	3.8-9.5	Denton & Burdon-Jones 1986a
Padina commersonni	Singapore coastal waters	nd	nd	0.4-0.6	2.9-6.5	3.8-7.3	<0.01[b]	4.0-6.5	4.3-7.9	nd	20.7-50.1	Bok & Keong 1976
Padina gymnospora	Puerto Rico	nd	nd	nd	nd	nd	nd	23.0-32.0	nd	nd	nd	Stevenson & Ufret 1966
Padina tenuis	Penang Island, Malaysia	nd	nd	7.1	25.6	5.7	1.025[b]	nd	17.1	nd	45.5	Sivalingam, 1978, 1980
Padina tenuis	Townsville coastal waters, Australia	<0.1-0.4	nd	0.2-1.4	1.4-10.0	1.4-5.1	nd	0.7-8.4	<0.3-6.2	nd	3.7-30	Burdon-Jones *et al.* 1982
Padina tetrastromatica	Goa coastal waters, India	nd	nd	nd	nd	3.2-7.9	nd	8.0-18.3	3.0-28.3	nd	4.5-11.7	Agadi *et al.* 1978
Padina tetrastromatica	Goa coastal waters, India	nd	4.8-12.6	nd	nd	8.7-20.1	nd	nd	nd	nd	20.2-31.5	Zingde *et al.* 1976
Padina tetrastromatica	Townsville coastal waters, Australia	<0.1-0.4	nd	0.2-1.2	1.6-8.3	2.0-9.7	nd	0.9-4.0	1.1-4.9	nd	5.5-25.7	Burdon-Jones *et al.* 1982
Padina tetrastromatica	Townsville Harbor (lower reaches)	<0.1-0.4	nd	0.2-0.6	2.1-9.9	4.4-11.1	nd	0.7-5.6	2.0-10.2	nd	67.2-166	Burdon-Jones *et al.* 1982
Padina tetrastromatica	Townsville Harbor (upper reaches)	<0.1	nd	<0.4	31.5	58.9	nd	13.1	108	nd	440	Burdon-Jones et al.1975
Padina sp.	Israeli coast	nd	nd	nd	nd	nd	0.065[b]	nd	nd	nd	nd	Hornung *et al.* 1981
Padina sp.	Penang Island, Malaysia	nd	nd	nd	nd	nd	0.100[b]	nd	nd	nd	nd	Sivalingam 1980
Padina sp.	Lizard Island, Great Barrier Reef	nd	nd	0.2	nd	2.2	0.002	1.1	<0.74	nd	5.9	Denton & Burdon-Jones 1986a
Padina sp.	Agana Boat Basin, Guam	0.89	32.2	0.3	0.68	1.53	<0.002	1.18	0.46	<0.01	11	This Study
Padina sp.	Apra Harbor, Guam	<0.1-<0.1	5.8-38.1	0.2-0.5	1.3-3.0	2.6-36.6	0.007-0.026	1.1-3.2	2.6-6.5	<0.01	45.1-192	This Study
Padina sp.	Agat Marina, Guam	<0.1	20.5	<0.1	2.7	4.1	<0.002	2.9	<0.25	<0.01	18.7	This Study
Padina sp.	Merizo Pier, Guam	<0.1	17.4	<0.1	14.1	27.2	0.003	2.28	8.07	<0.01	78.3	This Study
SOFT CORALS												
Alcyoniumdigitatum	Irish Sea, UK	nd	nd	4.1	nd	9.7	nd	17	24	nd	46	Riley and Segar 1970
Gorgonian sp.	Gt. Barrier Reef, Australia	nd	nd	0.8-3.0	nd	2.8-4.3	<0.001-<0.003	<0.3-<0.3	<0.6-<0.7	nd	2.9-12.2	Denton & Burdon-Jones 1986b
Litophyton sp.	Gt. Barrier Reef, Australia	nd	nd	2.6	nd	1.9	<0.002	70	<0.6	nd	4.7	Denton & Burdon-Jones 1986b
Sarcophyton acutangulum	Townsville coastal waters, Australia	<0.1	nd	1.6-9.7	nd	1.8-3.2	<0.06	0.13	0.8-1.5	nd	12.6-19.3	Burdon Jones and Klumpp 1979
Sarcophyton acutangulum	Lizard Island, Gt. Barrier Reef, Australia	nd	nd	0.2-1.3	nd	1.9-5.7	nd	<0.2-0.9	<0.5-<0.8	nd	13.0-29.9	Burdon-Jones & Denton 1984a
Sarcophyton glaucum	Heron Island, Gt,. Barrier Reef, Australia	nd	nd	0.5-2.5	nd	2.2-6.3	nd	<0.2-0.9	<0.4-<0.9	nd	4.2-15.8	Burdon-Jones & Denton 1984a
Sarcophyton trocheliophorum	Orpeus Island, Gt. Barrier Reef, Australia	nd	nd	0.5-3.7	nd	1.3-4.2	nd	<0.2-0.8	<0.4-<0.9	nd	9.9-26.9	Burdon-Jones & Denton 1984a
Sarcophyton sp.	Gt. Barrier Reef, Australia	nd	nd	0.4-2.1	nd	2.5-4.5	all <0.002	<0.4-<0.9	<0.5-<0.9	nd	8.6-29.0	Denton & Burdon-Jones 1986b
Sinularia eracta	Townsville coastal waters, Australia	<0.1	nd	0.1-0.2	nd	0.5-0.8	<0.06	<0.5	0.4-0.4	nd	0.4-0.8	Burdon Jones and Klumpp 1979
Sinularia sp.	Gt. Barrier Reef, Australia	nd	nd	0.5-1.1	nd	2.3-3.2	all <0.002	<0.3-<0.4	<0.6-<0.8	nd	1.5-9.7	Denton & Burdon-Jones 1986b
Sinularia sp.	Agana Boat Basin, Guam	2.7	0.01	0.1	<0.15	1.0	0.004	0.8	<0.3	10.5	74.5	This Study
Sinularia sp.	Apra Harbor, Guam	all <0.1	1.6-2.3	0.1-0.2	0.3-0.3	0.4-0.9	0.007-0.013	0.5-0.7	<0.3-<0.4	0.13-0.24	76.3-143	This Study
Sinularia sp.	Merizo Pier, Guam	<0.1	<0.01	<0.1	<0.2	0.6	0.022	0.2	<0.3	7.1	38.9	This Study
HARD CORALS												
Acropora formosa	Gt. Barrier Reef, Australia	nd	nd	0.02-0.2	nd	0.1-0.5	nd	0.1-0.8	<0.1-<0.4	nd	0.4-1.2	Denton & Burdon-Jones 1986b
Acropora formosa	Apra Harbor, Guam	<0.1	0.14	0.1	0.3	<0.1	0.017	2.12	<0.3	<0.01	1.7	This Study
Fungia concinna	Gt. Barrier Reef, Australia	nd	nd	0.02-0.03	nd	0.3-0.5	nd	<0.1-0.3	<0.1-<0.3	nd	0.8-1.5	Denton & Burdon-Jones 1986b
Fungia concinna	Apra Harbor, Guam	0.2	0.25	0.1	0.3	1.1	<0.011	<0.2	<0.3	0.06	3.1	This Study
Fungia fungites	Gt. Barrier Reef, Australia	nd	nd	0.02-0.1	nd	0.2-0.4	nd	<0.1-0.2	<0.1-0.7	nd	0.6-1.1	Denton & Burdon-Jones 1986b
Fungia echidata	Apra Harbor, Guam	0.1	0.19	0.1	0.2	0.5	0.007	0.3	<0.3	<0.01	1.8	This Study
Herpolitha limax	Apra Harbor, Guam	<0.1-1.2	0.17-0.20	0.1- 0.1	0.3-0.3	0.9-1.5	<0.005-0.015	all <0.2	<0.3-<0.4	all <0.01	2.2-4.1	This Study
Pocilopora damicornis	Agana Boat Basin, Guam	<0.1	<0.01	0.1	<0.1	0.1	<0.006	<0.2	<0.3	0.16	1.29	This Study
Pocilopora damicornis	Apra Harbor, Guam	<0.1-0.3	0.41-67	0.1-0.2	<0.1-0.3	<0.1-0.2	<0.005-<0.007	0.2-0.3	all <0.3	all <0.01	7.0-7.7	This Study
Pocilopora damicornis	Agat Marina, Guam	<0.1	<0.01	<0.1	<0.1	0.2	0.005	<0.1	<0.2	0.63	3.3	This Study
Pocilopora damicornis	Merizo Pier, Guam	<0.1	<0.01	<0.1	<0.2	<0.1	0.004	<0.2	<0.4	0.37	3.8	This Study

a = Hg determined as ug/g wet weight; b = Hg determined as ug/g dry weight; nd = no data.

Table 5 (cont.)

Heavy Metals in Marine Organisms (μg/g dry weight) from Other Regions of the World

Species	Location	Ag	As	Cd	Cr	Cu	Hg[a]	Ni	Pb	Sn	Zn	Reference
SEA CUCUMBERS												
Holothuria sp. (whole)	Japanese waters	nd	nd	nd	nd	1.9[c]	nd	nd	14.4[c]	nd	8.7[c]	Matsumoto 1964
Holothuria sp. (whole)	Townsville coastal waters, Australia	all <0.2	nd	<0.2	<0.3-6.3	<0.3-3.5	nd	all <0.5	<0.4-3.8	nd	13.9-39.4	Denton, unpublished data
Holothuria atra. (muscle)	Agana Boat Basin, Guam	0.2	<0.01	0.1	<0.1	1.4	0.008	<0.2	<0.4	10.6	12.6	This Study
Holothuria atra. (muscle)	Apra Harbor, Guam	all <0.1	13.6-23.2	<0.1-0.1	<0.1-0.3	0.7-1.2	0.007-0.008	<0.2	all <0.3	0.11-0.16	15.5-17.9	This Study
Holothuria atra. (muscle)	Agat Marina, Guam	<0.2	all <0.01	<0.1-0.1	<0.1-<0.2	1.3-1.7	0.022-0.014	<0.2-<0.3	<0.4-<0.6	9.76-21.5	15.4-17.0	This Study
Holothuria atra. (muscle)	Merizo Pier, Guam	<0.1	<0.01	0.1	<0.2	2.5	0.008	<0.2	<0.4	10.7	21.2	This Study
Molpadia intermedia (muscle)	Georgia Strait, Vancouver (dump site)	nd	nd	1.7	2.2	26	nd	1.7	1.4	nd	171	Thompson & Paton 1978
Stichopus tremulus (unknown)	Not stated	nd	nd	2.6	nd	57.0	nd	38.0	21.0	nd	140	Noddack & Noddack 1939
Stichopus variagatus (muscle)	Lizard Island, Gt. Barrier Reef, Australia	nd	nd	all <0.1	nd	1.5-2.1	<0.001-0.003	<0.3-<0.4	<0.6-<0.9	nd	1.9-13.9	Burdon-Jones & Denton 1984a
Stichopus variagatus (muscle)	Orpheus Island, Gt. Barrier Reef, Australia	nd	nd	<0.1-<0.2	nd	1.7-2.1	<0.001-0.002	<0.5-<0.8	<0.8-<0.9	nd	5.7-10.3	Burdon-Jones & Denton 1984a
Stichopus variagatus (muscle)	Heron Island, Gt. Barrier Reef, Australia	nd	nd	all <0.1	nd	1.5-1.8	<0.001-0.001	<0.3-<0.5	<0.8-<0.9	nd	3.3-9.5	Burdon-Jones & Denton 1984a
Bohadschia argus (muscle)	Agana Boat Basin, Guam	<0.1	<0.01	0.1	0.1	0.9	0.007	0.3	<0.4	14.5	12.5	This Study
Bohadschia argus (muscle)	Apra Harbor, Guam	<0.1	7.8-17.7	0.1-0.1	<0.2-0.4	0.6-2.3	0.005-0.005	1.0-1.4	<0.3-0.6	0.11-0.26	13.8-18.0	This Study
Bohadschia argus (muscle)	Agat Marina, Guam	<0.1	all <0.01	0.1-0.1	all <0.1	0.7-0.7	0.001-0.003	0.7-1.0	all <0.4	7.25-19.3	8.3-16.6	This Study
Bohadschia argus (muscle)	Merizo Pier, Guam	<0.1	<0.01	0.1	<0.1	0.6	0.003	1.1	<0.4	14.8	11	This Study
BIVALVES (Oysters)												
Crassostrea gigas	Saldanha Bay, S. Africa	nd	nd	3.7-9.0	nd	32-33	nd	1.0-1.6	1	nd	424	Watling & Watling 1976a and b
Crassostrea gigas	Hong Kong Waters	nd	nd	1.2[c]	nd	16.7[c]	0.06	nd	0.3[c]	nd	80.5[c]	Phillips *et al.* 1982
Saccostrea amasa	Townsville coastal waters, Australia	0.3-7.5	nd	0.7-5.9	<0.2-1.2	219-518	nd	<0.3-1.6	<0.3-1.8	nd	1163-2443	Burdon-Jones *et al.* 1979
Saccostrea amasa	Townsville Harbor, Australia	<0.2-5.1	nd	0.9-3.9	<0.3-8.6	417-1775	nd	<0.2-1.8	<0.2-1.3	nd	1916-9073	Burdon-Jones *et al.* 1979
Saccostrea amasa	Wistari Reef, Gt. Barrier Reef, Australia	nd	nd	2.6-5.5	nd	33.1-189	0.015-0.019	0.5-1.7	<0.5-<0.6	nd	54.4-130	Burdon-Jones & Denton 1984a
Saccostrea cucullata	Hong Kong Waters	nd	nd	1.5-14.7	nd	219-1413	nd	nd	nd	nd	998-8629	Phillips 1979
Saccostrea cucullata	Townsville Harbor, Australia	0.6-8.5	nd	1.0-3.5	0.2-1.7	450-1423	nd	<0.2-2.8	<0.2-2.3	nd	2577-8840	Burdon-Jones *et al.* 1979
Saccostrea cucullata	Townsville coastal waters, Australia	nd	nd	nd	nd	nd	0.06	nd	nd	nd	nd	Denton & Breck 1981
Saccostrea cucullata	Townsville coastal waters, Australia	0.4-8.3	nd	1.4-3.5	<0.2-1.2	280-720	nd	<0.2-2.3	<0.2-1.4	nd	1012-2752	Burdon-Jones *et al.* 1979
Saccostrea cucullata	Apra Harbor, Guam	<0.1-0.6	8.3-21.8	0.5-0.7	<0.3-<0.5	661-1911	0.043-0.078	<0.4-1.2	<0.3-1.1	0.24-0.70	2262-4722	This Study
Saccostrea cucullata	Merizo Pier, Guam	4.1-4.9	21.3-32.9	0.6-0.8	1.0-1.2	598-715	0.020	1.3-1.5	<0.3-0.4	all <0.01	1086-1225	This Study
Striostrea cf. mytiloides	Agana Boat Basin, Guam	0.1-3.0	16.5-35.5	0.4-0.8	0.8-9.0	500-3047	0.080-0.149	0.4-3.6	0.7-12.2	<0.01-0.09	2002-8375	This Study
Striostrea cf. mytiloides	Apra Harbor, Guam	<0.1-1.3	9.5-25.1	0.2-1.0	<0.2-0.9	496-2971	0.031-0.048	0.7-1.4	<0.2-0.6	0.11-0.27	2800-6280	This Study
Striostrea cf. mytiloides	Agat Marina, Guam	<0.1-0.2	28.7-38.4	0.6-1.0	1.5-2.0	689-962	0.016-0.022	1.6-2.7	<0.3-<0.7	0.01-0.05	2492-5393	This Study
Striostrea cf. mytiloides	Merizo Pier, Guam	<0.1	27.2	0.6	2.2	815	nd	2.7	6.5	<0.02	3571	This Study
BIVALVES (Chamids)												
Chama brassica	Apra Harbor, Guam	<0.1-0.6	23.6-51.6	0.2-0.7	4.0-6.2	6.8-11.2	0.033-0.312	14.9-25.1	<0.3-2.0	0.03-0.23	79.4-387	This Study
Chama iostoma	Townsville coastal waters, Australia	0.6-11.8	nd	2.3-12.1	nd	5.0-20.3	0.073-0.093	4.0-20.5	<0.5-10	nd	55.7-180	Burdon-Jones & Klumpp 1979
Chama iostoma	Orpheus Island, Gt. Barrier Reef, Australia	nd	nd	2.1-28.5	nd	3.1-31.9	0.018-0.326	9.5-54.5	all <0.9	nd	41.0-164	Burdon-Jones & Denton 1984b
Chama iostoma	Several sites, Gt. Barrier Reef, Australia	nd	nd	1.0-23.3	nd	3.5-6.7	0.006-0.083	6.2-38.9	<0.4-1.5	nd	56.0-319	Burdon-Jones & Denton 1984a
Chama lazarus	Apra Harbor, Guam	<0.1-0.2	21.6-331	0.1-0.8	0.6-2.9	4.4-129	0.020-1.041	1.3-7.8	<0.3-0.9	<0.01-0.37	46.2-202	This Study
Chama lazarus	Merizo Pier, Guam	<0.1-0.2	103-225	0.2-0.2	0.5-0.7	5.4-9.7	0.018	1.9-3.5	<0.4-<0.7	<0.02-0.05	127-227	This Study
Chama pacifica	Several sites, Gt. Barrier Reef, Australia	nd	nd	5.9-9.9	nd	3.7-4.3	nd	14.3-20.5	all <0.9	nd	48.8-102	Burdon-Jones & Denton 1984a
Chama plinthota	Aureed Island, Torres Strait, N. Australia	nd	46.0-1150	3.3-130	1.9-12.0	3.3-110	0.01-0.34[b]	6.0-190	<0.1-4.6	nd	24.0-220	Dight & Gladstone 1993
Chama plinthota	Kokope Island, Torres Strait, N. Australia	nd	59.0-1400	4.7-78.0	4.0-20.0	2.1-109	0.03-0.21[b]	5.9-80.0	all <0.1	nd	52.9-132	Dight & Gladstone 1993

a = Hg determined as ug/g wet weight; b = Hg determined as ug/g dry weight; c = metal determined on a wet weight basis; nd = no data

Table 5 (cont.)

Heavy Metals in Marine Organisms (μg/g dry weight) from Other Regions of the World

Species	Location	Ag	As	Cd	Cr	Cu	Hg[a]	Ni	Pb	Sn	Zn	Reference
BIVALVES (Spondylids)												
Spondylus ducalis	Orpheus Island, Gt. Barrier Reef, Australia	nd	nd	21.1-48.2	nd	15.0-42.9	nd	30.9-61.4	3.7-7.5	nd	175-518	Burdon-Jones & Denton 1984a
Spondylus ducalis	Heron Island, Gt. Barrier Reef, Australia	nd	nd	16.1-34.2	nd	5.5-12.2	0.055-0.084	41.8-96.1	1.4-6.0	nd	45.5-472	Burdon-Jones & Denton 1984a
Spondylus ducalis	Wistari Reef, Gt. Barrier Reef, Australia	nd	nd	14.9-26.9	nd	6.2-14.0	0.015-0.033	39.0-54.0	1.0-1.9	nd	44.3-191	Burdon-Jones & Denton 1984a
Spondylus ducalis	Gt. Barrier Reef, Australia	nd	nd	14.5-40.7	nd	8.3-17.0	0.036-0.039	72.3-116	2.5-5.5	nd	82.6-159	Burdon-Jones & Denton 1984a
Spondylus varians	Gt. Barrier Reef, Australia	nd	nd	7.5-9.2	nd	13.7-22.5	0.017-0.017	15.8-39.2	3.0-4.6	nd	34.7-72.6	Burdon-Jones & Denton 1984a
Spondylus? multimuricatus	Agana Boat Basin , Guam	0.4-1.7	33.0-52.3	5.3-6.9	2.9-9.6	271-432	0.001-0.001	13.7-18.0	72.8-88.6	0.28-0.33	404-730	This Study
Spondylus? multimuricatus	Agat Marina, Guam	<0.1-0.3	46.7-195	3.9-6.8	0.6-6.8	52.5-328	0.002-0.004	23.0-65.2	1.8-6.3	0.07-0.19	213-858	This Study
CEPHALOPODS (Cuttlefish)												
Sepia sp. (whole)	Japanese Waters										50.0[c]	Matsumoto *et al.* 1964
Sepia sp. (mantle)	Townsville Coastal Waters, Australia	<0.1	nd	0.1	0.9	0.7	0.15-0.25	<0.2	1.7	nd	4.1	Denton, unpublished data
Sepia sp. (liver)	Townsville Coastal Waters, Australia	1.8	nd	132	<0.3	660	0.19-0.39	3.8	<0.3	nd	331	Denton, unpublished data
CEPHALOPODS (Squid)												
Loligo formosana (whole)	Townsville Coastal Waters, Australia	<0.3-<0.4	nd	0.4-1.0	<0.4-<0.8	25.9-26.2	nd	<0.8-<0.9	<1.0-<1.1	nd	47.8-51.5	Burdon-Jones *et al.* 1975
Loligo formosana (tentacle)	Townsville Coastal Waters, Australia	0.1	nd	0.3	0.2	35.4	nd	<0.3	<0.2	nd	59.5	Denton, unpublished data
Loligo formosana (mantle)	Townsville Coastal Waters, Australia	<0.1	nd	0.1	<0.2	12.5	nd	<0.2	<0.2	nd	40.4	Denton, unpublished data
Loligo formosana (mantle)	Townsville Coastal Waters, Australia	nd	nd	nd	nd	nd	0.10-0.09	nd	nd	nd	nd	Denton & Breck 1981
Loligo formosana (liver)	Townsville Coastal Waters, Australia	2.6	nd	8.5-27.5	<0.3	140-361	nd	<0.5	<0.4	nd	94.1-234	Denton, unpublished data
Loligo formosana (liver)	Townsville Coastal Waters, Australia	nd	nd	nd	nd	nd	0.05	nd	nd	nd	nd	Denton & Breck 1981
Loligo opalescens (liver)	California Coast	25-45	nd	85-121	nd	5350-8370	nd	nd	nd	nd	247-449	Martin and Flegal 1975
Loligo vulgaris (whole)	USA - Gulf of Mexico	nd	nd	1.0-2.6	nd	28.8-70.7	nd	2.6-5.3	nd	nd	71.8-86.5	Forster *et al.* 1972
Loligo vulgaris (whole)	USA - North Pacific Coastal Waters	nd	nd	0.6-5.8	nd	52.0-85.9	nd	nd	3.1-5.0	nd	64.4-105	Cutshall & Holton 1972
CEPHALOPODS (Octopus)												
Octopus vulgaris (whole)	USA - North Atlantic Coastal Waters	nd	nd	0.5	nd	5	nd	nd	0.5	nd	43.0	Windom 1972
Octopus sp. (whole)	Japanese Waters	nd	nd	nd	nd	nd	nd	nd	1.0[c]	nd	106[c]	Matsumoto *et al.* 1964
Octopus sp. (tentacle)	Apra Harbor, Guam	< 0.12	96.4	0.06	< 0.16	12.1	0.047	< 0.2	< 0.3	0.17	69.5	This Study
Octopus sp. (liver)	Apra Harbor, Guam	4.40	44.3	7.8	1.9	5680	0.242	4.7	24.8	0.77	573	This Study
CRUSTACEANS (Shrimp)												
Pink shrimp (muscle)	USA - Pacific Coastal Waters	0.1-0.4	nd	0.5-1.0	<0.5	4.1-6.5	nd	nd	nd	nd	37-59	Robertson *et al.* 1972
Pistol shrimp (whole)	Townsville Coastal Waters, Australia	0.6	nd	<0.2	0.5	139	nd	<0.2	<0.4	nd	88.3	Denton, unpublished data
Callianasa sp. (whole)	Townsville Coastal Waters, Australia	<0.1	nd	<0.2	1.7	115	nd	1.5	<0.4	nd	87.5	Denton, unpublished data
Penaeus esculentus (whole)	Townsville Coastal Waters, Australia	0.6-0.8	nd	0.5-0.9	0.4-0.4	84.7-90.3	nd	<0.7-<0.7	<0.9-3.0	nd	67.2-164	Burdon-Jones *et al.* 1975
Penaeus merguiensis (whole)	Townsville Coastal Waters, Australia	0.9	nd	<0.1	<0.5	54.6	nd	<0.6	4.6	nd	59.1	Burdon-Jones *et al.* 1975
Penaeus merguiensis (muscle)	Townsville Coastal Waters, Australia	<0.1-<0.4	nd	<0.1-<0.1	<0.1-<0.5	12.9-40.8	nd	<0.1-<0.4	<0.3-<0.6	nd	20.2-55.2	Denton, unpublished data
Penaeus merguiensis (hepato)	Townsville Coastal Waters, Australia	3.9	nd	4.3	0.6	346	nd	11.7	<0.3	nd	138	Denton, unpublished data
Penaeus merguiensis (gonad)	Townsville Coastal Waters, Australia	0.5	nd	<0.2	<0.3	49.9	nd	0.7	2.5	nd	199	Denton, unpublished data
Mantis shrimp (muscle)	Apra Harbor, Guam	0.27	5.06	0.36	0.57	11.0	0.075	< 0.23	< 0.39	0.090	125	This study
Mantis shrimp (gonad)	Apra Harbor, Guam	1.43	4.58	9.11	0.91	3195	0.085	< 0.81	< 1.38	0.251	148	This study

a = Hg determined as ug/g wet weight; b = Hg determined as ug/g dry weight; c = metal determined on a wet weight basis; nd = no data

Table 5 (cont.)

Heavy Metals in Marine Organisms (μg/g dry weight) from Other Regions of the World

Species	Location	Ag	As	Cd	Cr	Cu	Hg[a]	Ni	Pb	Sn	Zn	Reference
FISH (Muscle)												
Several spp.	Caribbean waters	nd	nd	0.3-16.6	nd	1-94.2	nd	1.3-7.9	10.9-36.5	nd	10.8-117	Forster *et al.* 1972
25 spp.	north Atlantic	nd	6.4	<0.1-1.6	nd	<0.3-23.0		nd	nd	nd		Windom *et al.* 1973
4 spp.	San Antonio Bay, USA	nd	nd	0.1-1.8	nd	1.3-2.3	nd	nd	0.2-0.9	nd	19-37	Sims & Presley, 1976
8 spp.	Spain & Portugal - Atlantic coast	nd	nd	<0.1-0.8	nd	0.6-8.0	0.05-0.43	nd	1.2-2.0	nd	9-120	Stenner & Nickless 1975
Several spp.	Tuscan coasts, Italy	nd	nd	nd	nd	nd	0.10-0.64	nd	0.1-0.5[c]	nd	nd	Buggiani & Vannuchi 1980
20 spp.	E. Mediterranean	nd	nd	nd	nd	nd	0.02-0.88	nd	nd	nd	nd	Yanni & Sachs 1978
11 spp.	Israeli coast	nd	nd	0.1-0.7	0.5-4.9	0.7-23.5	nd	<0.1-10.8	<0.1-4.8	nd	0.5-84.3	Roth & Hornung 1977
4 spp.	Persian Gulf	nd	nd	nd	nd	nd	0.04-0.56	nd	nd	nd	nd	Parvaneh 1979
9 spp.	Goa, India	nd	nd	nd	nd	2.3-32.5	nd	nd	nd	nd	7.5-76.5	Zingde *et al.* 1976
4 spp.	Bombay coast, India	nd	nd	nd	nd	nd	0.062-0.470	nd	nd	nd	nd	Somayajula & Rama 1972
6 spp.	W. Malaysia	nd	nd	<0.1-0.1[c]	nd	nd	0.003-0.15	nd	<0.1-0.5[c]	nd	0.73-10.0[c]	Babji *et al.* 1979
10 spp.	Upper coast of Thailand	nd	nd	nd	nd	nd	0.006-0.150	nd	nd	nd	nd	Choovaparinapivat & Marasveta 1979
8 spp.	Japan coastal waters	nd	nd	0.02-0.13	nd	nd	0.02-0.74[b]	nd	<0.1-0.6	nd	19.3-87.5	Einaga 1977
16 spp.	New Guinea.	nd	nd	nd	nd	nd	0.02-5.70	nd	nd	nd	nd	Sorentino 1979
	Cockburn Sound, W. Australia	nd	nd	0.1-0.6	0.2-0.8	0.2-5.8	nd	0.1-3.9	0.6-4.4	nd	nd	Plaskett & Potter 1979
15 spp.	Townsville Coastal Waters, Australia	<0.1-0.2	nd	all <0.1	0.1-0.6	0.7-3.8	nd	<0.1-1.2	<0.2-1.0	nd	8.3-126	Burdon-Jones *et al.* 1975
48 spp.	Townsville Coastal Waters, Australia	nd	nd	nd	nd	nd	0.03-1.30	nd	nd	nd	nd	Denton & Breck 1981
50 spp.	Great Barrier Reef, Australia	nd	nd	all <0.1	nd	0.47-2.4	<0.002-1.9	all <0.5	all <0.7	nd	4.3-41.8	Denton & Burdon-Jones 1986c
8 spp.	Agana Boat Basin, Guam	all <0.2	1.4-10.8	all <0.1	<0.1-0.6	0.3-0.8	0.009-0.165	all <0.4	all <0.9	<0.01-0.02	8.4-48.9	This Study
17 spp.	Apra Harbor, Guam	<0.1-0.2	0.63-24.1	all <0.1	all <0.5	0.5-7.8	0.012-0.660	all <0.4	all <0.8	0.02-0.41	8.3-34.2	This Study
6 spp.	Agat Marine, Guam	all <0.2	1.3-47.3	all <0.1	all <0.3	0.3-0.9	0.003-0.214	all <0.4	all <0.8	<0.01-0.07	11.5-24.3	This Study
10 spp.	Merizo Pier, Guam	<0.1-0.3	1.7-77.6	all <0.1	<0.1-0.5	0.3-0.8	0.011-0.066	<0.2-<0.7	<0.4-<1.3	all <0.01	9.6-24.3	This Study
FISH (Liver)												
Several spp.	Caribbean waters	nd	nd	3.8-4.2	nd	10.7-718	nd	2.6-5.3	14.1-50.7	nd	14.3-1558	Forster *et al.* 1972
15 species	Townsville Coastal Waters, Australia	<0.2-3.0	nd	0.1-6.7	<0.6-2.8	5.7-540	nd	<0.2-7.4	<0.3-4.6	nd	49.6-588	Burdon-Jones *et al.* 1975
44 spp.	Townsville Coastal Waters, Australia	nd	nd	nd	nd	nd	0.01-3.53	nd	nd	nd	nd	Denton & Breck 1981
50 spp.	Great Barrier Reef, Australia	nd	nd	0.8-209	nd	1.1-1593	0.007-10.09	all <0.5	all <0.7	nd	62.9-2335	Denton & Burdon-Jones 1986c
7 spp.	Agana Boat Basin, Guam	<0.1-1.7	0.4-7.2	0.2-1.8	<0.2<1.0	5.4-188	0.010-1.028	<0.2-<1.0	<0.4-10.8	0.01-0.29	52.8-485	This Study
17 spp.	Apra Harbor, Guam	<0.1-5.1	1.3-9.5	0.1-4.8	<0.13-4.8	2.6-1920	0.020-2.197	<0.2-<0.8	<0.3-2.1	0.18-9.67	22.0-540	This Study
2 spp.	Agat Marine, Guam	all <0.4	1.4-7.5	0.3-1.9	all <0.6	9.1-90.0	0.018-0.637	all <0.6	<0.8-<1.1	0.02-0.55	52.6-212	This Study
6 spp.	Merizo Pier, Guam	<0.2-2.3	1.9-18.2	0.7-2.9	<0.3-<1.6	3.4-71.7	0.010-0.761	<0.3-<1.7	<0.5-3.9	<0.01-0.11	31.8-375	This Study

a = Hg determined as ug/g wet weight; b = Hg determined as ug/g dry weight; c = metal determined on a wet weight basis; nd = no data

Table 6

PCB Concentrations in Marine Organisms from Other Regions of the World

Species	Location	Total PCB (ng/g)	Reference
BIVALVES			
Oysters	Dominican Republic	19.5 - 51.2*	Sbriz *et al.* 1998
Mussels & Oysters	Puerto Rico	18.3 - 55.1*	GREG 1993
Mussels & Oysters	Puerto Rico	3.80 - 36.1*	GREG 1994
Mussels & Oysters	Puerto Rico	32.3 - 83.0*	GREG 1995
Mussels & Oysters	Cuba	15.3*	IMW Program, Sericano unpublished results
Mussels & Oysters	Jamaica	14.9 - 25.4*	IMW Program, Sericano unpublished results
Mussels & Oysters	Trinidad and Tobago	8.54 - 15.6*	IMW Program, Sericano unpublished results
Perna viridis (Mussel)	Junk Bay, Hong Kong	1904*	Phillips 1985
Perna viridis (Mussel)	Hong Kong	245 - 1667*	Phillips 1986
Mytillus galloprovincialis (Mussel)	Catalan Mediterranean Coast	2.19 - 51.1	Porte & Albaiges 1994
CRUSTACEANS			
Macropipus tuberculat (Crab)	Catalan Mediterranean Coast	10.2 - 90.5	Porte & Albaiges 1994

nd = not detected; * = data expressed in ng/g dry weight

Table 6 (cont.)

PCB Concentrations in Marine Organisms from Other Regions of the World

Species	Location	Total PCB (ng/g)	Reference
FISH (muscle)			
5 spp.	Egypt, Abu Qir Bay	55.6 - 89.5	El Nabawi *et al.* 1987
Tilapia nilotica	Egypt, Iduku Lake	16.0 - 17.0	El Nabawi *et al.* 1987
Tilapia zilli	Egypt, Maryut Lake	21.9	El Nabawi *et al.* 1987
Mullus barbatus (Red mullet)	Greece, Alexandroupolis	7.9 - 14.6	Giouranovitis-Psyllidou *et al.* 1994
3 spp.	Egypt, Ebro delta	1.8 - 20.3	Pastor *et al.* 1996
25 spp.	England and Wales	nd - 2100	Murray and Portmann 1984
5 spp.	Australia, Brisbane River	nd - 940	Shaw and Connell 1980
Myoxocephalus quadricornus (Sculpin)	Canada, Victoria Island	nd - 220	Bright *et al.* 1995
6 spp.	England	4.0 - 69.0	Franklin 1987
5 spp.	Australia, Brisbane River	100 - 2100	Shaw and Connell 1982
Platycephalus bassensis (Flathead)	Australia, Port Philip Bay	2.7 - 42.5	Nicholson *et al.* 1994
4 spp.	Australia, Richomond River	nd - 97.1	Williams and Krogh 1993
4 spp.	Australia, Wallis Lake	nd - 93.5	Williams and Krogh 1993
4 spp.	Australia, Parramatta River	nd - 217.6	Williams and Krogh 1993
4 spp.	Australia, Georges River	nd - 136.3	Williams and Krogh 1993
4 spp.	Australia, St. Georges Basin	nd - 222.4	Williams and Krogh 1993
3 spp.	Australia, Botany Bay	60.0 - 410	Scribner *et al.* 1987
6 spp.	Australia, Georges and Cooks River	141 - 10140	Roach and Runcie 1998
5 spp.	Australia, Sydney Harbour	10.0 - 3782	Roach and Runcie 1998
3 spp.	Catalan Mediterranean Coast	3.10 - 482	Porte & Albaiges 1994
FISH (liver)			
Thunnus thynnus	Catalan Mediterranean Coast	112 - 275	Porte & Albaiges 1994

nd = not detected; * = data expressed in ng/g dry weight

Table 7

PAH Concentrations in Marine Organisms from Other Regions of the World

Species	Location	Total PAH (ng/g)	n[1]	Reference
BIVALVES				
Mytillus galloprovincialis (Mussel)	Mediterranean Sea	25.1 - 337*	14	Baumard *et al.* 1998
Spondylus sp. (Rock Scallop)	Askar, Bahrain	124*	5	Fowler *et al.* 1993
Saccostrea cucullata (Rock Oyster)	Oman	92 -496*	5	Fowler *et al.* 1993
5 spp.	Gulf of Naples	185 - 295	16	Cocchieri *et al.* 1990
Mytillus galloprovincialis (Mussel)	Catalan Mediterranean Coast	190 - 5490	total[2]	Porte & Albaiges 1993
Crassostrea virginica (Oyster)	Palmetto Bay, South Carolina	269 - 520	total	Marcus & Stokes 1985
Crassostrea virginica (Oyster)	Outdoor resorts, South Carolina	134 - 247	total	Marcus & Stokes 1985
Crassostrea virginica (Oyster)	Fripp Island, South Carolina	21 - 55	total	Marcus & Stokes 1985
Mytilus edulis (Bay mussel)	Oregon	27 - 986	total	Mix & Schaffer 1983
Placopecten magellanicus (Sea scallop)	New York Bight	127	total	Humason & Gadbois 1982
CRUSTACEANS				
Polybius henslowi (Crab)	Mediterranean Sea	82.8 - 102*	14	Baumard *et al.* 1998
Mysids (Shrimp)	Mediterranean Sea	220*	14	Baumard *et al.* 1998
Euphausiids (Shrimp)	Mediterranean Sea	509*	14	Baumard *et al.* 1998
Shrimp	Mediterranean Sea	6200 - 6400*	total[2]	Yilmaz *et al.* 1998
Macropipus tuberculat (Crab)	Catalan Mediterranean Coast	60 - 930	total[2]	Porte & Albaiges 1993
Cancer irroratus (Rock Crab)	New York Bight	1600	total	Humason & Gadbois 1982
Cancer irroratus (Rock Crab)	Long Island Sound	1290	total	Humason & Gadbois 1982
Homarus americanus (Lobster)	New York Bight	367	total	Humason & Gadbois 1982
Homarus americanus (Lobster)	Long Island Sound	328	total	Humason & Gadbois 1982

nd = not detected; * = data expressed in ng/g dry weight

[1] = number of individual PAHs analyzed; [2] = quantified as chrysene

Table 7 (cont.)

PAH Concentrations in Marine Organisms from Other Regions of the World

Species	Location	Total PAH (ng/g)	n[1]	Reference
FISH (Muscle)				
2 spp.	Georges Bank	5000 - 38000*	total	Boehm and Hirtzer 1982
2 spp.	Georges Bank	14 - 18*	> 13	Boehm and Hirtzer 1982
Parophrys vetulus	Puget Sound, Washington	nd*	23	Malins *et al.* 1984, 1985
2 spp.	Finnish Archipelago	85 - 150*	14	Rainio *et al.* 1986
3 spp.	Turkey	1000 - 8000*	total[2]	Salihoglu *et al.* 1987
4 spp.	Donano Natural Park, Spain	nd - 11000*	total	Albaiges *et al.* 1987a, 1987b
2 spp.	Arabian Gulf	66000 -689000*	total[2]	El Deeb and El Ebiary 1988
Mullus barbatus	Adriatic Sea	020 - 170*	total[2]	Dujmov and Sucevic 1989
Tilefish	Middle Atlantic Bight	1.96 - 3.95*	24	Steimle *et al.* 1990
Gailus morhua	NW Atlantic	nd*	27	Hellou *et al.* 1993
2 spp.	Mediterranean Sea	14.7 - 139*	14	Baumard *et al.* 1998
Arius thalassinus (Sea catfish)	Ras Al Jousah, Kuwait	139*	5	Fowler *et al.* 1993
Lethrinus nebulosus (Pigface bream)	Safaniya, Saudi Arabia	39.1 - 322.2*	5	Fowler *et al.* 1993
6 spp.	Bahrain	1.9 - 135*	5	Fowler *et al.* 1993
Epinephelus suillus	Dubai, UAE	18.4*	5	Fowler *et al.* 1993
5 spp.	Oman	10.5 - 38.2*	5	Fowler *et al.* 1993
14 spp.	Gulf of Naples	94 - 1930	16	Cocchieri *et al.* 1990
8 spp.	Mediterranean Sea	1100 - 10700*	total[2]	Yilmaz *et al.* 1998
Mugil sp.	Mersin Harbour, Mediterranean Sea	10000 - 14500*	total[2]	Yilmaz *et al.* 1998

nd = not detected; * = data expressed in ng/g dry weight
[1] = number of individual PAHs analyzed; [2] = quantified as chrysene

Table 7 (cont.)

PAH Concentrations in Marine Organisms from Other Regions of the World

Species	Location	Total PAH (ng/g)	n[1]	Reference
FISH (Muscle)				
3 spp.	Catalan Mediterranean Coast	40 - 190	total[2]	Porte & Albaiges 1993
3 spp.	New York Bight	315 - 536	total	Humason & Gadbois 1982
3 spp.	Long Island Sound	86 - 124	total	Humason & Gadbois 1982
FISH (Liver)				
2 spp.	Georges Bank	127000 - 885000*	total	Boehm and Hirtzer 1982
2 spp.	Georges Bank	204 - 902*	> 13	Boehm and Hirtzer 1982
Parophrys vetulus	Puget Sound, Washington	72 - 989*	23	Malins *et al.* 1994, 1995
2 spp.	Finnish Archipelago	590 - 2225*	14	Rainio *et al.* 1986
3 spp.	Turkey	5000 - 75000*	total[2]	Salihoglu *et al.* 1987
2 spp.	Donano Natural Park, Spain	8000 - 602000*	total	Albaiges *et al.* 1987a, 1987b
2 spp.	Arabian Gulf	76000 - 677000*	total[2]	El Deeb and El Ebiary 1988
Tilefish	Middle Atlantic Bight	21.96 - 12.8*	24	Steimle *et al.* 1990
Gailus morhua	NW Atlantic	nd - 585*	27	Hellou *et al.* 1993
Lethrinus nebulosus (Pigface bream)	Safaniya, Saudi Arabia	457 - 2920*	5	Fowler *et al.* 1993
Epinephelus suillus	Dubai, UAE	117*	5	Fowler *et al.* 1993
3 spp.	Oman	12 - 32*	5	Fowler *et al.* 1993
Thunnus thynnus	Catalan Mediterranean Coast	80 - 270	total[2]	Porte & Albaiges 1993

nd = not detected; * = data expressed in ng/g dry weight
[1] = number of individual PAHs analyzed; [2] = quantified as chrysene

RESULTS & DISCUSSION

1. HEAVY METALS IN HARBOR BIOTA

The heavy metal data obtained during the present study are summarized in Tables 8-15 at the end of this section. The following text is organized on a metal by metal basis and the data are discussed with reference to levels found by other workers in similar and related species from elsewhere in the world. The bioindicator potential of each group of organisms is also discussed where appropriate. All referenced data are expressed on a dry weight basis unless stated otherwise. The Guam data can be conveniently expressed on a wet weight basis if so desired using the water content data recorded in each table.

1.1 Silver (Ag):

Silver ranks among the most toxic of heavy metals to marine organisms (Moore 1991). Levels in abiotic components of the marine environment are usually low. Dissolved levels in seawater are generally less than 0.001 µg/l (Shafer 1995) while levels in uncontaminated sediments are in the order of 0.1 µg/g (Bryan and Langston 1992). Sedimentary silver concentrations in highly polluted environments can exceed 100 µg/g (Skei *et al.* 1972). Levels previously reported by us for Guam harbor sediments were consistently below an analytical detection limit of ~0.2 µg/g indicating that silver is not an element of environmental concern locally (Denton *et al.* 1997). Levels found in biota during the present investigation are discussed below.

1.1.1 Ag in Algae:

In the present study, silver levels in the brown alga, *Padina* sp., were below the limits of analytical detection except at Agana Boat Basin where the pooled tissue composite yielded a value of 0.89 µg/g (Table 8). Burdon-Jones *et al.* (1982) reported silver concentrations of <0.1-0.4 µg/g for this genus taken from Townsville Harbor, Australia (Table 5). Levels recorded in other phaeophyceae generally do not exceed 0.4 µg/g (Preston *et al.* 1972, Bryan and Uysal 1978, Burdon-Jones *et al.* 1975) although Bryan and Hummerstone (1977) gave a maximum value of 2.42 µg/g for *Fucus* spp. collected from the metal enriched Looe estuary in Cornwall, UK.

1.1.2 Ag in Sponges:

Silver levels found in sponges during the current study were low and ranged from <0.11-0.47 µg/g. The highest concentrations occurred in specimens from Apra Harbor and Agana Boat Basin (Table 9). We were unable to locate any comparative silver data for sponges from elsewhere.

1.1.3 Ag in Corals:

Silver does not concentrate up the food chain and so residues are typically low in invertebrates from most surface waters (Moore 1991). Reported levels for soft and hard corals rarely exceed 0.1 µg/g (Veek and Turekian 1968, Riley and Segar 1970, Burdon-Jones and Klumpp 1979). The relatively high level of 2.7 µg/g recorded in *Sinularia* sp. from the Agana Boat Basin during the present study (Table 10) is of interest because it supports the mild enrichment demonstrated by *Padina* sp. collected from this area.

1.1.4 Ag in Sea Cucumbers:

Silver levels in almost all species of echinoderms examined by others are either low, non-detectable, or near the limits of analytical detection (Eisler 1981). The results of the present study are in line with these findings apart from one relatively high value of 4.9 µg/g determined in the hemal system of a specimen of *Holothuria atra* from the Port Authority Beach area in Apra Harbor (Table 11). Papadopoulu *et al.* (1976) reported whole body silver concentrations of 0.05 µg/g for the sea cucumber, *Holothuria tubulosa.*

1.1.5 Ag in Mollusks:

Mollusks show considerable inter- and intra-specific variations in silver concentrations. In most cases, the highest reported levels coincide with samples taken from polluted environments (Alexander and Young 1976, Fowler and Oregioni 1976, Greig 1979). Oysters appear to have a greater affinity for this element than either mussels or scallops (Brooks and Rumsby 1965). Levels reported for this group commonly fall between 0.1 and 10 µg/g (Thurberg *et al.* 1974,Watling 1976, Goldberg *et al.* 1978, Greig and Wenzloff 1978) as seen during the present study (Table 12). However, Windom and Smith (1972) found high levels ranging from 28.0-82.0 µg/g in oysters from the Georgia coast, USA.

Comparative data for silver in the other bivalve species collected during the present study is almost nonexistent (Table 13). Burdon-Jones and Klumpp reported 0.6-11.8 µg/g for *Chama iostoma* from Townsville coastal waters, Australia, and is somewhat higher than reported here for *C. brassica.* These authors also looked at silver in the separated tissues of *Spondylus ducalis* and found maximum levels of 11.3 and 13.7 µg/g in the digestive gland and kidney respectively. Levels in both tissues seemed to decrease with increased distance offshore, a trend presumably related to the proximity of contamination sources.

While the digestive gland and kidney are the sites of silver deposition in bivalve mollusks, it is the liver that usually accumulates the highest concentration of this element in cephalopod mollusks. This was evident for octopus collected from Apra Harbor during the current study (Table 14) and has previously been demonstrated with squid (Denton, unpublished data). Interestingly, the highest recorded silver levels in squid liver are 25.0 µg/g and 45 µg/g found in *Loligo opalescens* from the central and southern California coasts respectively (Martin and Flagal 1975).

1.1.6 Ag in Crustaceans:

Crustaceans generally contain low tissue levels of silver ranging from 0.5 µg/g or less, in muscle and gonad, to 1-10 µg/g in the hepatopancrease (Bertine and Goldberg 1972, Greig *et al.* 1977, Hall *et al.* 1978). Thus, levels found in mantis shrimp tissues during the present study were not considered unusual (Table 14).

1.1.7 Ag in Ascidians:

Few studies have focused on heavy metal in tunicates. Papadopoulu and Kanias (1977) looked at silver in whole *Ciona intestinalis* and *Microcosmus sulcatus* and found very low levels of 0.021 and 0.031 µg/g respectively. Tunicates from Apra Harbor generally showed similarly low levels of this element in their tissues (Table 14).

1.1.8 Ag in Fish:

In contrast to the situation with tunicates, there is a wealth of data describing heavy metal levels in fish. Public health interests in species commonly consumed by man have largely driven this research. According to Eisler (1991), biomagnification of silver rarely occurs in fish, even under the most polluted conditions. Consequently, silver levels in fish muscle never exceed 0.2 µg/g wet weight and are almost always <0.1 µg/g wet weight. The findings of the present study confirm this statement (Table 15). Like most other metals, silver tends to be more concentrated in the liver of fish although levels rarely exceed 1 µg/g wet weight. During the present work, higher levels were found in less than 3% of liver samples analyzed.

1.1.9 Concluding Remarks:

Clearly, none of the organisms examined were excessively enriched in silver, confirming earlier conclusions regarding this element's low level abundance in our local harbor environments (Denton *et al.* 1997).

1.2 Arsenic (As):

Although arsenic has several oxidation states, the chemical form normally encountered in the environment is not particularly toxic to aquatic organisms (Moore 1991). Soluble arsenic levels in seawater are normally around 2-4 µg/l (Riley and Chester 1971, Bowen 1979) while levels in uncontaminated sediments are in the order of 5 µg/g (Bryan and Langston 1992). Levels previously reported by us for local harbor sediments ranged from <1.0-17.0 µg/g with the highest levels occurring in samples from Hotel Wharf in Apra Harbor. Values of 1-3 µg/g were considered to be fairly typical of clean carbonate sediments on Guam (Denton *et al.* 1997). In highly contaminated environments, arsenic levels in sediments can exceed 1000 µg/g (Langston 1984, 1985).

1.2.1 As in Algae:

Appreciable amounts of arsenic are present in most marine species and most of this is in the organic form. In algae for example, lipid soluble dimethyl arsenate usually accounts for well over 90% of the total arsenic present (Klumpp and Peterson 1979). It should be emphasized that most of the organic arsenic in algae is the result of metabolic transformations within the plants themselves and not direct uptake from water (Moore 1991). Average arsenic levels in algae (all types) are around 20 µg/g according to Bryan (1976) with normal ranges between 2 and 60 µg/g (Eisler 1981). Levels determined in *Padina* sp. during the present study fell within these limits (Table 8).

1.2.2 As in Sponges:

Data on arsenic levels in sponges are limited. Leatherland and Burton (1974) recorded 2.8 µg/g in the bread sponge, *Halichondrea panicea,* from Southampton waters in the UK. In our study, we determined relatively high levels of arsenic (5.96-47.7 µg/g) in the majority of sponges collected from Apra Harbor. In contrast, levels were either at or below detection in specimens taken from all other harbor sites (Table 9).

1.2.3 As in Corals:

Corals from Apra Harbor generally contained the highest arsenic concentrations determined during the present investigation (Table 10). However, levels were generally lower than found in algae and sponges, with the notable exception of *Pocilopora damicornis* from beneath the Shell Fox-1 Fuel Pier (site c). The arsenic level measured in this particular specimen was 67.1 µg/g. We were unable to locate any previous studies of arsenic abundance in corals although some data exists for other coelenterates. For example, Leatherland and Burton found 72.0 µg/g in the sea anemone, *Telia felina*, from the Solent estuary, a major shipping highway in the south of England. In his review paper, Bryan (1976) estimates average arsenic levels for coelenterates to be around 20 µg/g.

1.2.4 As in Sea Cucumbers:

The echinoderms are another group that has received little attention in terms of their trace metal content. Bryan (1976) reports an average arsenic value of 5 µg/g for the group as a whole, but draws attention to the fact that his estimate is derived from very few data. Based on our findings for Guam harbors, it would seem that arsenic levels are appreciably lower than this, at least in sea cucumbers. For example, both *Holothuria atra* and *Bohadschia argus* from Agana Boat Basin, Agat Marina and Merizo Pier contained less than 0.01 µg/g in their body wall muscle (Table11). Levels were slightly higher in the hemal system but did not exceed 0.2 µg/g in any of the samples analyzed. Levels in both tissues were considerably higher in all specimens collected from within Apra Harbor. These findings once again point towards the increased biological availability of arsenic in this area.

1.2.5 As in Mollusks:

Mollusks are known to be unusually rich in arsenic compounds. For example, the whole soft parts of the file shell, *Pinna nobilis*, from the Mediterranean were reported to contain up to 670 µg/g (Papadopoulou 1973). Closer to home, the chamid, *Chama plinthota*, from the Torres Strait was found to contain a maximum of 1400 µg/g (Dight and Gladstone 1993). Fortunately, such compounds consist primarily of organic pentavalent species, non-toxic forms with little implications from a human health perspective. Most other bivalves generally contain much lower arsenic levels than the two examples cited above. Oysters, for example, normally contain around 10 µg/g (Förstner 1980) although the natural range can extend from 1-15 µg/g (Eisler 1981). Arsenic levels measured in oysters during the present study frequently exceeded 20 µg/g and peaked at 38.4 µg/g in one specimen from Agat Marina (Table 12). Oysters from Apra Harbor generally contained the lowest concentrations of arsenic in contrast to the other animal groups described above. The utility of bivalves as indicators of arsenic pollution has yet to be unequivocally established.

The bivalve kidney is the primary deposition site for arsenic. In most bivalves these paired organs are anatomically inconspicuous but in spondylids and chamids they are enlarged. This could account for the relatively high arsenic levels observed in representatives from both groups analyzed during the present study (Table 13). The tridacnid clams are another group with enlarged kidneys. In fact, the kidneys of these bivalves account for up to 10% of the total flesh wet weight (Reid *et al.* 1984). Interestingly, one representative of this group, *Tridacna maxima*, was found to contain renal arsenic levels in excess of 1000 µg/g (Benson 1983).

Cephalopod mollusks show a similar affinity for arsenic as their bivalve relatives, and according to Bryan (1976), contain average concentrations of around 40 µg/g. The relatively high arsenic levels, determined in the liver (44.3 µg/g) and tentacles (96 µg/g) of the octopus captured in Apra Harbor during the present study are, therefore, unremarkable (Table 14). For comparative purpose, we note here that Leatherland and Burton (1974) reported arsenic levels of 73 µg/g in the mantle of the cuttlefish, *Sepia officianalis*, from north temperate waters.

1.2.6 As in Crustaceans:

Arsenic concentrations in decapod crustaceans range from 1-100 µg/g (Fowler and Unlu 1978) although average concentrations for the group are around 30 µg/g (Bryan 1976). Levels determined in the stomatopod from Apra Harbor tended towards the lower end of this range (Table 14).

1.2.7 As in Ascidians:

Tunicates are not exceptional accumulators of arsenic and average levels for the group, based on limited data, are in the order of 5 µg/g (Bryan 1976). Levels determined in two genera of ascidians from Apra Harbor during the present study ranged from 2.31-3.92 µg/g (Table 14). Whether these values are influenced by the mild enrichment of biologically available arsenic in this area remains to be determined.

1.2.8 As in Fish:

Arsenic concentrations in edible fish tissues are generally lower than those reported for edible portions of algae, crustaceans, and bivalve mollusks (Lunde 1977). Eisler (1971) conducted an extensive review of arsenic in fish tissue and concluded that while levels in muscle and liver tissues varied widely, most fell between 2.0 and 5.0 µg/g wet weight. The results of our study confirm this (Table 15). However, Eisler also noted that hepatic arsenic levels were usually higher than those found in muscle tissue, which is contrary to what we observed.

There is some evidence that fish are useful indicators of arsenic contamination. For example, Grimanis *et al.* (1978) found maximum levels of 18.0 and 142 µg/g in the flesh of *Gobius niger* from non-polluted and polluted areas of the Aegean Sea respectively. Likewise, Papadopoulu *et al.* (1973) recorded average concentrations of 18.0 and 39.0 µg/g in the flesh of *Pagellus erythrimus* from clean and contaminated areas of the Mediterranean.

1.2.9 Concluding Remarks:

The data generally point toward mild enrichment of biologically available forms of arsenic in the outer Apra Harbor area. Discrepancies between the various groups in this regard presumably reflect inter-specific differences in affinity and metabolic control over this element, in addition to variations in uptake from different fractions of the total available load (i.e., soluble, particulate, food-associated, or sediment-bound arsenic).

1.3 Cadmium (Cd):

Cadmium, particularly as the free cadmium ion, is highly toxic to most plant and animal species (Moore 1991). Cadmium concentrations in remote open ocean waters may be as low as 0.002 µg/l and rarely exceeds 0.5 µg/l in nearshore waters, even in heavily industrialized

areas (Yeates and Bewers 1987). Non-polluted sediments normally contain 0.2 µg/g or less but levels may exceed 100 µg/g at heavily contaminated sites (Naidu and Morrison 1994). Previously reported cadmium concentrations in Guam harbor sediments ranged from less than 0.1µg/g, in the great majority of samples analyzed, to 2.18 µg/g at Hotel Wharf in Apra Harbor. It should be mentioned, however, that two other surface sediment samples taken from Hotel Wharf at the same time yielded values of 0.27 and 0.35 µg/g indicating cadmium enrichment heterogeneity in this area.

1.3.1 Cd in Algae:

The ability of algae to accumulate cadmium from seawater is well documented and levels as high as 220 µg/g have been recorded in brown algae (*Fucus vesiculosus*) from the metal enriched Severn Estuary in the UK (Butterworth *et al* 1972). Levels recorded in *Padina* sp. during the present study ranged from <0.1 µg/g, in samples from Agat Marina and Merizo Pier area, to 0.5 µg/g in algae from Apra Harbor (Table 8). These values compare well with levels found in related species from Singapore coastal waters (Bok and Keong 1976) and the Australian Great Barrier Reef (Denton and Burdon-Jones 1986a). However, they are a little lower than those found in *Padina* sp. from elsewhere (Table 5). For example, Burdon-Jones *et al.* (1982) determined a maximum mean value of 1.4 µg/g in *Padina tenuis* from the coastal waters off Townsville, Australia, while Sivalingam (1978) reported a high of 7.1 µg/g for the same species from Penang, Malaysia.

While algae are generally considered to be useful biological indicators of dissolved cadmium, the presence of elevated levels of iron and/or manganese in the water can significantly reduce cadmium uptake (Moore 1991). This is thought to occur as a result of competition between the metals for cellular binding sites. Since harbors are typically enriched with both metals, some caution is required in interpreting cadmium contamination profiles in such areas from the analysis of algae alone. The work of Burdon-Jones *et al.* (1982) clearly demonstrated this problem. These researchers collected *Padina tetrostromatica* from Townsville Harbor, an area enriched with all three metals. Cadmium levels in algae, collected monthly for one year from this location, ranged from 0.2-0.6 µg/g compared with 0.2-1.2 µg/g at a control site.

1.3.2 Cd in Sponges:

Low levels of cadmium were found in all sponge samples collected during the present study. Values ranged from 0.11-0.86 µg/g with no obvious inter-site differences. Comparable data are rare and confined here to reports by Leatherland and Burton (1974), who found 0.85 µg/g in the bread sponge, *Halichondria panicea*, and Bernhard and Zattera (1975), who reported a range of 1.2-4.5 µg/g for several species of porifera from Puerto Rico.

1.3.3 Cd in Corals:

Cadmium concentrations in representative species of coelenterates, reviewed by Eisler (1971), ranged from 0.07-5.3 µg/g in whole organisms. A more recent survey of hard and soft corals, from unpolluted waters of the Great Barrier Reef, revealed levels of 0.02-0.2 µg/g and 0.1-9.7 µg/g in representatives of each group respectively (Burdon-Jones and Klumpp 1979, Burdon-Jones and Denton 1984a, Denton and Burdon Jones 1986b). These values encompass the range of cadmium concentrations determined in hard and soft corals during the present study.

1.3.4 Cd in Sea Cucumbers:

Echinoderms generally seem to contain cadmium levels of less than 1.0 µg/g. However, there are exceptions. For example, Riley and Segar (1970) found 4.5-5.3 µg/g in the starfish, *Solaster papposus*, from UK coastal waters, whilst Noddack and Noddack (1939) reported a high of 2.6 µg/g in the sea cucumber, *Stichopus tremulus*, from an unspecified location. Thompson and Paton (1978) determined a slightly lower maximum of 1.7 µg/g in body wall muscle of the sea cucumber, *Molpadia intermediai*, from a sediment disposal site in the Georgia Strait, Vancouver. In contrast, Burdon-Jones and Denton (1984a) failed to find cadmium above a detection limit of ~0.1 µg/g in the same tissue of *Stichopus variagatus* from unpolluted sections of the Great Barrier Reef. These studies strongly suggest that sea cucumbers have some bioindicator capacity for cadmium. If such is the case, the findings of the current study (Table 11) infer that none of the harbor sites visited were appreciably enriched with this element.

1.3.5 Cd in Mollusks:

Bivalve mollusks have been widely used to monitor cadmium pollution in aquatic environments. The fact that they are sessile and have a high affinity for cadmium, and several other metals of environmental concern, make them ideal candidates for coastal monitoring purposes. However, this latter quality also places severe constraints on their usefulness as a food resource when harvested from heavy metal contaminated waters.

There is considerable data for cadmium and other heavy metals in oysters. In clean environments, cadmium levels in the whole soft parts of oysters usually lie somewhere between 1.0 and 10 µg/g (Table 5). In grossly contaminated environments they are very much higher. For example, Talbot *et al.* (1976) reported a high of 174 µg/g in the flesh of *Ostrea angasi* taken from the polluted Port Phillip Bay area in Australia. Similarly, Ratkosky *et al.* (1974) found 30.7 µg/g wet weight in *Crassostrea gigas* taken close to a zinc refinery in Tasmanian waters. This translates to ~150 µg/g when recalculated on a dry weight basis. Levels encountered during the current study ranged from 0.2-1.0 µg/g and are among the lowest ever recorded for this group (Table 12).

Not much in the way of comparable data exists for the other bivalves analyzed during the present investigation. What little data there is has been incorporated into Table 5 and largely reflects the extensive work of Burdon-Jones and coworkers. Suffice to say cadmium levels in chamids and spondylids from Guam harbors are appreciable lower than those found in related species from the Great Barrier Reef and the Torres Strait.

Cephalopod mollusks tend to accumulate naturally high concentrations of cadmium and other trace elements in their livers (Table 5). In contrast, levels found in edible tissues are usually very much lower. There is no evidence from the literature to suggest that these organisms have any usefulness as bioindicators of heavy metal pollution.

1.3.6 Cd in Crustaceans:

Crustaceans naturally contain reasonably high levels of cadmium in their digestive gland (hepatopancreas) and occasionally in their gills and gonads (Burdon-Jones *et al.* 1975). Levels in muscle, while generally lower, often vary between 1-10 µg/g. However, the great majority

of values reported in the literature are less than 1 µg/g (Eisler 1981) as was noted in the present study with mantis shrimp (Table 14). There is some evidence to suggest that levels of cadmium in crustacean tissues are influenced by, and therefore reflective of, environmental levels (White and Rainbow 1982, Rainbow and White 1989)

1.3.7 Cd in Ascidians:
The little work that has focused on cadmium in tunicates, including the results of the present study, indicates that levels normally encountered in this group range between 0.1-3.0 µg/g (Leatherland *et al.* 1973, Eustace 1974, Letherland and Burton 1974). It is noteworthy that cadmium levels in all ascidians from Guam are at the lower end of this range (Table 14).

1.3.8 Cd in Fish:
Cadmium levels in fish muscle are generally less than 0.1 µg/g although there are occasional reports of levels 1 to 2 orders of magnitude higher in fish from contaminated areas (Forster *et al.* 1972, Halcrow *et al.* 1973, Sims and Presley 1976, Bohn and Fallis 1978). Levels determined in fish muscle during the present study were either undetectable or below 0.1 µg/g (Table 15). Denton and Burdon-Jones (1986c) reported similarly low values in muscle of 50 species of Australian fish from remote areas of the Great Barrier Reef. These authors also noted that cadmium was usually more concentrated in the livers of fish examined. In fact, levels often exceeded 20 µg/g and occasionally topped 100 µg/g in this tissue. They concluded that dietary difference between and within species were responsible for the highly variable hepatic cadmium levels encountered. Interestingly, hepatic cadmium levels determined in fish during our study were considerably lower and ranged from 0.2-4.8 µg/g.

1.3.9 Concluding Remarks:
Based on the foregoing data and discussions, it seems reasonable to assume that cadmium does not pose a threat to the health of ecosystems, or integrity of potential food resources, within any of the harbor environments examined.

1.4 Chromium (Cr):
Chromium is only moderately toxic to aquatic organisms (Moore 1991). Total dissolved chromium levels in seawater show little variability and range from around 0.6 µg/l in offshore areas to 1-2 µg/l in highly polluted areas (Riley and Chester 1971, Beukema *et al.* 1986). Nakayama *et al.* (1981) showed that dissolve chromium in the Pacific Ocean and Sea of Japan existed as 10-20% inorganic-Cr^{3+}, 25-40% inorganic-Cr^{6+}, and 45-65% organic-Cr. Levels in particulate form were also found to outweigh dissolved concentrations by a factor of 6 and 5.25 in each location respectively. From this we infer that sedimentary chromium levels rapidly accumulate in waters receiving elevated concentrations of this metal.

Chromium levels in uncontaminated sediments vary according to their mineralogical characteristics and range between 10-100 µg/g (Turekian and Wedepole 1961). Calcareous sediments of biogenic origin, like those found on Guam, are typically lower and normally contain 3-5 µg/g. In severely contaminated areas, sedimentary chromium concentrations have exceeded 2000 µg/g (Young and Means 1987). Chromium levels previously determined by us

in Guam harbor sediments ranged from 3.09-52.7 µg/g, and were indicative of fairly clean conditions overall with light to moderate enrichment in places (Denton *et al.* 1997).

1.4.1 Cr in Algae:
Surprisingly, the Merizo Pier area in the vicinity of the Cocos Island ferry terminal contained the highest levels of sedimentary chromium level given above. This moderate enrichment was also reflected in algae from this site with 14 µg/g being recorded in *Padina* sp. during the present study (Table 8). At the other harbor sites, levels ranged from 0.57-2.98 µg/g.

Burdon-Jones *et al.* (1975, 1982) reported chromium levels of 1.4-10 µg/g in *Padina* sp. from relatively clean coastal waters near Townsville, Australia, and a high of 31.5 µg/g in samples from the polluted upper reaches of Townsville Harbor. These values pale in comparison to the high of 140 µg/g recorded by Gryzhanková *et al.* (1973) for 19 species of algae from polluted Japanese coastal waters.

1.4.2 Cr in Sponges:
Chromium levels found in sponges from Guam harbors were not excessively high and ranged from 0.45-24.9 µg/g (Table 9). No enrichment was apparent in the Merizo Pier area. In fact, inter-specific differences in chromium levels outweighed any obvious inter-site differences. No comparative data were found in the literature to effectively evaluate levels of this element in local sponges.

1.4.3 Cr in Corals:
Coelenterates are another little worked group in terms of their elemental composition. This is especially true for chromium. One reference to a cold water, soft coral species (*Alcyonium digitatum*) recorded a chromium level of <0.4 µg/g (Riley and Segar 1970). Similarly low values of <0.15-0.31 µg/g were found in the soft coral, *Sinularia* sp. during the present study (Table 10).

Comparative data for chromium in hard corals is confined here to the work of Livingston and Thompson (1971). These authors measured several trace elements in 34 species of coral from the Caribbean. Deep-water species contained chromium levels ranging from 0.8-3.0 µg/g, whereas shallow water species, taken from chromium-rich, mineral areas, contained up to 35 µg/g. Levels determined in hard corals during the present study were 0.3 µg/g, or less, clearly an indication of a low ambient availability of this element in the surrounding waters.

1.4.4 Cr in Sea Cucumbers:
Chromium in sea cucumbers collected during the current investigation was largely confined to the hemal system. Levels in this tissue ranged from 6.27-31.9 µg/g in *Bohadschia argus*, and 0.88-8.58 µg/g in *Holothuria atra* (Table 11). Chromium concentrations in the muscle tissue of both species were mostly below a detection limit of ~0.2 µg/g. Fukai (1965) recorded a similar value of 0.28 µg/g in muscle tissue of the sea cucumber, *Holothuria forksalli*. In contrast, Thompson and Paton (1978) noted a relatively high chromium concentration of 2.2 µg/g in the body wall of *Molpadia intermedia*, collected from a sediment disposal site in Georgia Strait. These data imply that sea cucumbers are effective bioindicators of chromium

contamination, and that Guam harbor sediments are comparatively free of pollution by this element.

1.4.5 Cr in Mollusks:
Chromium levels in the edible tissues of uncontaminated marine mollusks usually lie between 0 5 and 3.0 µg/g (Eisler 1981). Levels recorded here, for oyster, chamids, spondylids, and octopus were mostly within this range (Tables 12-14).

1.4.6 Cr in Crustaceans:
In general, chromium seldom exceeds 2 µg/g in the edible tissues of crustaceans and is usually less than 1 µg/g (Burdon-Jones *et al.* 1975, Denton unpublished data, see Table 5). Results from the current study are in agreement with this (Table 14).

1.4.7 Cr in Ascidians:
Reported chromium levels in whole ascidian range from 5.5 µg/g in *Ciona intestinalis* (Papadopoulu and Kanias 1977) to 144 µg/g in *Eudistoma ritteri* (Levine 1961). Levels reported here for ascidians from Guam harbors were at the lower end of this scale and ranged from 1.03-5 08 µg/g in *Ascidia* sp., and from 1 82-9.65 µg/g in *Rhopalaea* sp. The utility of tunicates as indicators of heavy metal pollution is suggested by the work of Papadopoulu and Kanias (1977) but has yet to be substantiated.

1.4.8 Cr in Fish:
Chromium does not normally accumulate in fish tissues and levels in flesh are almost always less than 1 µg/g (Table 5). The work of Horowitz and Presley (1977) is a notable exception to this general rule. These authors determined chromium in the muscle tissue of 8 species of fish, from the outer continental shelf region of southern Texas, and reported levels of 2.0-7.7 µg/g. In our study, levels measured in fish muscle were predominantly below the limits of analytical detection and ranged from <0.1-0.6 µg/g (Table 15). Similarly low ranges have been reported for fish from Australian coastal waters (Burdon-Jones *et al.* 1975, Plaskett and Potter 1979).

1.4.9 Concluding Remarks:
Clearly, chromium is not an element of environmental concern in the areas investigated during this study

1.5 Copper (Cu):
Copper is highly toxic to plants and invertebrates (Brown and Ahsanulla 1971, Denton and Burdon Jones 1982), and ranks among the more toxic heavy metals to fish (Denton and Burdon-Jones 1986d, Moore 1991). Dissolved copper levels in open ocean surface waters are low, being generally in the order of 0.2 µg/l, or less. In uncontaminated nearshore surface waters, levels are significantly higher, often approaching 1 µg/l, while in highly polluted waters they may exceed 10 µg/l (Burdon-Jones and Denton 1984a). Copper levels in clean, non-geochemically enriched sediments are around 10 µg/g, or less. In contrast, severely polluted environments can yield sedimentary copper values in excess of 2000 µg/g (Legoburu and Canton 1991, Bryan and Langston 1992).

Copper levels previously determined by us in Guam harbor sediments ranged from 0.49-181 µg/g (Denton *et al.* 1997). The highest levels were encountered in samples from Hotel Wharf in Apra Harbor (85-181 µg/g), the western end of Commercial Port in Apra Harbor (72.7-127 µg/g), Dry Dock Island in Apra Harbor (35.7-75.4 µg/g), the inner harbor area of Agana Boat Basin (48.0-96.1 µg/g), and adjacent to the Cocos Island ferry terminal at Merizo Pier (83.1-168 µg/g). Biological samples were collected from these and other sites during the present investigation and the data are discussed below.

1.5.1 Cu in Algae:
According to Moore (1991), total copper levels in marine plants are typically less than 10 µg/g, except near polluting sources. This certainly appears to be true for algae. For example, Denton and Burdon-Jones (1986a) analyzed 47 species of algae from 20 sites, along the entire length of the Australian Great Barrier Reef, and reported values ranging from 0.74-7.2 µg/g. Most of the data fell between 1 and 4 µg/g. In an earlier investigation, these researchers analyzed *Padina tenuis* and *P. tetrostromatica* from Townsville coastal waters. Sampling was conducted at monthly intervals for one year. Copper levels were found to range from 2.0-9.7 µg/g and 1.4-5.1 µg/g in *P. tenuis* and *P. tetrostromatica* respectively. Only in the relatively polluted, upper reaches of Townsville Harbor did levels exceed 10 µg/g, and reached a high of 58.9 µg/g in *P. tetrostromatica* found growing there. Copper levels in the water from this particular site averaged 4.6 µg/l, at least an order of magnitude higher than average concentrations measured outside the harbor area (Burdon-Jones *et al.* 1982).

In the present study, copper levels in *Padina* sp. substantially surpassed 10 µg/g at the western end of Commercial Port (site d) and Dry Dock Island (site e) in Apra Harbor, and at the Cocos Island ferry terminal at Merizo Pier (Table 8). Clearly, areas of copper enrichment are indicated at each of these sites. Elsewhere in the study areas, copper levels in *Padina* sp. were low and ranged from 0.57-2.98 µg/g.

Algae have a relatively high accumulation capacity for copper and levels in excess of 100 µg/g are not unusual in species from highly polluted waters. For example, Bryan and Hummerstone (1973a) reported a maximum copper concentration of 301 µg/g in the thallus of the brown alga, *Fucus vesiculosus,* from a contaminated estuary in southwest England.

1.5.2 Cu in Sponges:
Most of the sponges analyzed during the current work contained reasonably high copper concentrations (Table 9). Whether this is a reflection of elevated ambient copper availability, or the group's natural affinity for this element, is not entirely clear. The copper concentration profiles depicted by *Dysidea* sp. certainly seem to parallel those of *Padina* sp. insofar as identifying the western end of Commercial Port as copper-enriched compared with Echo Wharf and Agat Marina. The elevated level of copper determined in an unidentified brown sponge from Hotel Wharf may well be reflective of the high sedimentary copper levels known to exist there. However, in the absence of adequate baseline data for local sponges, such claims are difficult to substantiate. An earlier study by Lowman *et al.* (1966) revealed copper levels in species of sponges from Puerto Rico of 8.5-31.0 µg/g. Most of the data gathered during the present study fall within, or just beyond, this range.

1.5.3 Cu in Corals:

Concentrations of copper in soft corals vary between species and between locations. For example, levels ranging from 6.2-9.7 µg/g were reported for the cold water species, *Alyconium digitata* (Culkin and Riley 1958, Riley and Segar 1970). In contrast, 10 tropical species from the Great Barrier Reef, including four unidentified species of *Simularia*, contained 1.9-4.5 µg/g (Denton and Burdon-Jones 1986b). Levels reported here for *Simularia* sp. are significantly lower again and range from 0.44-0.98 µg/g (Table 10). There is no evidence to suggest that soft corals have any bioindicator capability for this element.

Copper levels determined in hard corals during the current study, were as low and sometimes lower than those found in *Simularia* sp. Inter-specific differences in copper concentrations were clearly apparent, with *Acropora formosa* and *Pocilopora damicornis* demonstrating significantly lower affinities for this element than the other three species of hard coral analyzed. In all cases, however, levels were considerably below the range of 2.0-10 µg/g reported for shallow-water corals from the Caribbean (Livingston and Thompson 1971).

The use of hard corals to monitor environmental changes in trace metals has been implied by several workers (e.g., St. John 1973, Barnard *et al.* 1974, Buddemeier 1978). However, difficulties encountered in analyzing coral skeletons coupled with their low affinity for several elements of environmental concern, make them rather unattractive candidates for such purposes. Moreover, there is some evidence to suggest that members of this group can control both skeletal and soft tissue concentrations of certain essential metals like copper, zinc and iron (Brown and Holley 1982). If this is the case, then they are of little value as sentinel organisms for these particular elements.

1.5.4 Cu in Sea Cucumbers:

Whether the copper content of sea cucumbers is influenced by changes in this element's ambient availability is a matter of some debate. Burdon-Jones and Denton (1984a) found copper levels of 1.5-2.1 µg/g in the body wall of *Stichopus variagatus* from the Great Barrier Reef. After comparing their findings with the earlier copper data of Thompson and Paton (1978) for *Molpadia intermedia* (26 µg/g) from a sediment disposal site in the Georgia Strait, they concluded that sea cucumbers have bioindicator potential for this element. The findings of the present study do not support their claim, however (Table 11).

1.5.5 Cu in Mollusks:

Bivalve mollusks are widely used to monitor copper in the marine environment. Oysters, in particular, have an extraordinary capacity to accumulate copper and are highly sensitive to changes in this element's ambient availability. For this reason, they are one of the most popular bivalves used for heavy metal monitoring purposes. One of the highest copper levels recorded in oysters to date is 6,480 µg/g. This was measured in *Crassostrea gigas*, exposed to copper-enriched effluent water, from a power station in Wales, UK (Boyden and Romeril 1974).

Oysters from clean, non-geochemically enriched coastal areas contain copper levels of less than 100 µg/g when analyzed whole (Mackay *et al.* 1975a, Watling and Watling 1976a and b, Burdon-Jones and Denton 1984a). In mineralized areas, levels are typically higher. For

example, Australian oysters from clean, coastal waters near Townsville contained 200-500 µg/g (Burdon-Jones *et al.* 1977). Copper levels of up to 500 µg/g were found in Tasmanian oysters from areas of minimal metal pollution compared with up to 2,500 µg/g in specimens from contaminated areas (Thrower and Eustace 1974).

Oysters from harbor areas are typically high in copper, reflecting the increased environmental abundance of this element from sources such as algaecides and anti-fouling paints. Burdon-Jones *et al.* (1977) conducted monthly surveys of heavy metals in *Saccostrea amasa* and *S. cucullata*, from Townsville Harbor, over a one-year period. Mean monthly levels determine for each species ranged from 417-1,775 µg/g, and 661-1,911 µg/g for *S. amasa* and *S. cucullata* respectively. Phillips (1979) determined similarly high values in *S. glomerata* from Hong Kong waters (Table 5).

In the current study, the highest copper value recorded in an oyster was 3,047 µg/g. This was measured in a single specimen from the inner harbor area of Agana Boat Basin (Table 12). The geometric mean copper level for 13 oysters analyzed from this location was 1,968 µg/g and is comparable with the Australian study mentioned. It is also a clear indication of copper enrichment in this area. Oysters from Apra Harbor were also high in copper, with maximum values between 1,900 and 3,000 µg/g, at all sites examined. Maximum levels determined in the rather limited number of oysters available from Agat Marina and Merizo Pier area did not exceed 1,000 µg/g, possibly as a result of lower levels of available copper in these areas.

Not much is known about the bioindicator potential of the other bivalves examined during the present study, namely the chamids and spondylids. Some preliminary work carried out in Australia has shown that copper levels in *Chama iostoma* are linked to the reproductive cycle and are significantly higher in specimens with well-developed gonads (Burdon-Jones and Denton 1984b). Nevertheless, mean copper concentrations in this species, from unpolluted waters, rarely exceed 20 µg/g and usually drop below 10 µg/g after spawning. In the present study, mean levels determined in *C. lazurus* and *C. brassica* were mostly below 10 µg/g (Table 13). This strongly suggests that chamids can maintain levels of copper in their tissues to within certain limits irrespective of changes in this element's ambient availability. *Spondylus*, on the other hand, does not appear to possess the same regulatory capability. On the contrary, copper levels in local representatives of this group were generally much higher than those found in related species from clean waters of the Great Barrier Reef (Table 5).

Copper is naturally high in cephalopod mollusks and is largely related to the storage of copper in the liver and the presence of the copper-based respiratory pigment, haemocyanin, in the blood (Bryan 1976). It should be noted here, that while some bivalves also possess haemocyanin, oysters do not.

1.5.6 Cu in Crustaceans:

Copper levels in decapod crustaceans are also naturally high, particularly in the hepatopancreas and occasionally the gonad. Such high levels are associated with their metabolic requirements and the presence of haemocyanin in their blood in much the same way as for cephalopods. Since both groups are capable of metabolically regulating levels of copper in their tissue, they are of little value as bioindicators of copper pollution. In point of

interest, the copper concentration of 3,195 µg/g found in the gonad of the stomatopod crustacean from Apra Harbor, ranks among the highest values ever recorded for this tissue (Table14).

1.5.7 Cu in Ascidians:

Copper levels in ascidians analyzed during the present study ranged from 3.10-5.58 µg/g in *Ascidia* sp. to 6.46-9.87 µg/g in *Rhopalaea* sp., with no obvious differences between sites (Table 14). Comparable studies with other tunicates are rare. We note that Eustace (1974) reported a whole body copper value of 8.3 µg/g wet weight in *Ascidacea* sp. from the polluted Derwent estuary, in Tasmania. This translates to well over 100 µg/g, assuming 95 % water content, and is considerably higher than levels found in the two local species examined here. Bryan (1971) gives an overall average for the group of 30 µg/g but fails to disclose his sources of reference. In any event, the bioindicator potential of ascidians for copper does not look promising on the strength of the data gathered so far.

1.5.8 Cu in Fish:

Copper levels in fish flesh typically range between 0.5-2.0 µg/g in marine species (Denton and Burdon-Jones 1986c) although some extraordinarily high values have occasionally been reported in the literature (Table 5). This rather narrow range is thought to reflect the group's ability to metabolically regulate copper and other essential trace elements (Phillips 1980). Denton and Burdon-Jones (1986c) reviewed some of the more reliable trace element data-bases for fish from all over the world and concluded that mean copper levels in the axial muscle of fish from polluted waters frequently exceed 1.0 µg/g, whereas fish from uncontaminated waters almost always yield values of less than 1.0 µg/g. Their own work with fish from the Great Barrier Reef leant considerable weight to this assumption. It is significant to note, then, that 29 of the 38 fish taken from Apra Harbor (76%) contained copper levels in their muscle tissue greater than 1 µg/g. Levels ranged from 0.51-7.76 µg/g with an overall geometric mean of 1.64 µg/g. It is also noteworthy that copper concentrations in fish muscle from all other harbor sites were less than 1.0 µg/g.

In fish where both muscle and liver tissues were analyzed, hepatic copper levels were always higher and there was no obvious correlation between the two data sets. In fact, considerable inter- and intra-specific variation was apparent and there was no clear evidence for trophic level dependence. Several liver samples contained levels between 10-100 µg/g with one sample approaching 2000 µg/g. Similar observations were made by Denton and Burdon-Jones (1986c) with fish from the Great Barrier Reef (Table 5).

1.5.9 Concluding Remarks:

The data clearly identifies increased levels of biologically available copper in the inner section of Agana Boat Basin and in Apra Harbor, particularly in the Commercial Port area. A highly localized source of elevated copper availability is also evident at the Cocos Island ferry terminal, in Merizo. However, copper enrichment of the biota in these areas does not exceed that normally encountered in harbor environments in other parts of the world.

1.6 Mercury (Hg):

Mercury is highly toxic to aquatic organisms, particularly in the organic form (Moore 1991). Concentrations of dissolved mercury in the open ocean typically range from <0.010-0.003 µg/l (Miyake and Suzuki 1983). Slightly higher values of 0.003-0.02 µg/l are found closer to shore, and polluted estuarine waters may contain up to 0.06 µg/l (Baker 1977). Sediment concentrations of mercury in unpolluted, non-geochemically enriched areas, usually do not exceed 30 ng/g (Bryan and Langston 1992, Benoit *et al.* 1994), and may be as low as 4 ng/g (Knauer 1976). Estuarine sediments, adjacent to heavy industrialized areas or mercury mining activities, can be three to five orders of magnitude higher than this (Langston 1986, Benoit *et al.* 1994). Values in excess of 2000 µg/g were found in sediments from the grossly contaminated Minimata Bay area in Japan, following the mass mercury-poisoning episode of the late 1950's, and probably rank among the highest values ever reported for this element (Tokuomi 1969).

Mercury levels in Guam harbor sediments ranged from a low of 2.72 ng/g at Agat Marina, to a high of 741 ng/g at Hotel Wharf in Apra Harbor. Moderate enrichment was also noted at the Shell Fox-1 Fuel Pier (202-256 ng/g), the western end of Commercial Port (107-264 ng/g), and at Dry Dock Island (160-428 ng/g). All four sites were revisited during the present investigation.

The reader is reminded here, that all mercury data presented in Tables 8-15 are expressed on a wet weight rather than a dry weight basis. Where appropriate, these values have been recalculated on a dry weight basis during the following discussion (unless stated otherwise) to facilitate ease of comparison with levels recorded in some of the literature cited.

1.6.1 Hg in Algae:

Marine algae have a relatively high affinity for mercury. For example, a high of 20 µg/g was reported for the brown alga, *Ascophyllum nodosum*, from Hardangerfjord, in Norway (Haug *et al.* 1974). Apparently, wastewater discharged from a nearby metal smelter was the primary source of mercury pollution in this particular case (Myklestad *et al.* 1978). In an earlier study, Jones *et al.* (1972) measured mercury in 10 species of algae from the polluted Tay estuary in the UK and reported a maximum of 25.54 µg/g (6.26 µg/g wet weight) in the green alga, *Ulva lactuca*. This still stands as one of the highest values ever recorded for marine algae. Among the lowest values ever found, are those given by Denton and Burdon-Jones (1986a) for 48 species of algae from the Great Barrier Reef. In this instance, mercury concentrations ranged from <0.011-0.320 µg/g (<0.001-0.024 µg/g wet weight). These values are comparable with the values of 0.002-0.52 µg/g given by Kim (1972) for 17 species of algae from Korean waters. They are also within the range of values (not detectable to 1.03 µg/g) reported by Sivalingam (1980) for 26 tropical species from Malaysia.

Very low mercury concentrations were detected in *Padina* sp. during the current work. Levels ranged from <0.002-0.026 µg/g wet weight (Table 8), or <0.011-0.137 µg/g, when expressed on a dry weight basis. While these values are hardly indicative of polluted conditions, they do indicate a light enrichment of mercury in the Apra Harbor area.

1.6.2 Hg in Sponges:

Data for mercury levels in sponges are limited. Leatherland and Burton (1974) recorded 0.33 µg/g in the bread sponge, *Halichondria panicea,* from UK waters. This value is appreciably higher than those recorded for Guam sponges analyzed during the present study (Table 9).

1.6.3 Hg in Corals:

Concentrations of mercury in coelenterates are usually low, except for species collected from heavily contaminated areas. For example, Matida and Kumada (1969) reported a maximum value of 41 µg/g for a sea anemone from Minimata Bay. By way of contrast, Leatherland and Burton (1974) reported a very much lower value of 0.86 µg/g for the sea anemone, *Telia felina,* from the Solent estuary (UK). These data suggest that coelenterates may have bioindicator potential for mercury. However, no such quality was indicated from the present work with hard and soft corals. In fact, levels of mercury were extremely low in representatives of both groups (Table 10) and there was no correlation with levels determined earlier in sediments.

1.6.4 Hg in Sea Cucumbers:

Eisler (1981) reviewed the limited mercury data for echinoderms and concluded that values below 1.0 µg/g occur in specimens from non-polluted areas. This was confirmed by the work of Burdon-Jones and Denton (1984a) who found <0.019-0.056 µg/g in muscle of the sea cucumber, *Stichopus variagatus,* from the Great Barrier Reef. An almost identical range of 0.019-0.057 µg/g was determined in the body wall muscle of the sea cucumber *Bohadschia argus* during the present investigation (Table 11). A slightly higher range of 0.059-0.219 µg/g was noted for the same tissue of *Holothuria atra.* In neither case, however, did levels in muscle tissue mirror those determined earlier in sediment samples. In contrast, mercury concentrations in the hemal tissue generally did, and were highest in both species from the Apra Harbor area (Table 11). The utility of this tissue, as an indicator of mercury contamination, warrants further investigation.

1.6.5 Hg in Mollusks:

From the literature, it is clear that marine mollusks are excellent accumulators of mercury and there are numerous reports of various species being used as environmental indicators for this metal. Eisler (1981), in his comprehensive review of published information, conclude that levels above 1.0 µg/g wet weight in representatives of this group were always associated with mercury pollution. It is not surprising, therefore, that some of the highest concentrations ever recorded (10-100 µg/g) were found in mollusks from the mercury contaminated Minimata Bay area of Japan (Irukayama *et al.* 1961, 1967, Matida and Kumada 1969).

Reported mercury levels in tropical oysters, from clean reef waters in northern Australia, ranged from 0.015-0.019 µg/g wet weight (Burdon-Jones and Denton 1984a). These values are similar to those found in oysters from Agat Marina and Merizo Pier during the present study (Table 12). Harbor environments typically contain a greater abundance of heavy metals, including mercury, and a degree of elemental enrichment of the biota in such areas is to be expected. For example, a mean mercury concentration of 0.060 µg/g wet weight was reported for oysters from Townsville Harbor in north Queensland, Australia, (Denton and Breck 1981). Levels found here in oysters from Apra Harbor were of a similar order and

ranged from 0.022-0.078 µg/g wet weight. Specimens from Agana Boat Basin contained marginally higher concentrations of 0.080-0.149 µg/g wet weight. However, these values are well below the maximum of 10 µg/g (~2.0 µg/g wet weight) recorded in oysters from Minimata Bay during the late 1960's (Matida and Kumada 1969).

Burdon Jones and Denton (1984a) looked at mercury in the chamid, *Chama iostoma,* from pristine, offshore areas of the Great Barrier Reef, and reported levels that ranged from 0.006-0.032 µg/g wet weight. Nearer shore, the range widened from 0.018-0.326 µg/g wet weight. The authors concluded that chamids have potential as bioindicators of mercury pollution. Data from the current work tends to support their conclusion and infers enrichment in the Apra Harbor area when compared with previously reported data from elsewhere (Table 5).

Burdon-Jones and Klumpp (1979) conducted a similar study with the spondylid, *Spondylus ducalis,* but failed to establish a clear link between tissue levels of mercury and distance offshore. Likewise, Burdon-Jones and Denton (1984a) found identical mercury concentrations of 0.017 µg/g wet weight in *S. varians* collected from two locations, 10 km and 200 km offshore. In the present study, the mercury profiles depicted by *S. multimuricatus* were contrary to what was expected, based on our earlier sediment analysis. Moreover, levels were surprisingly low compared with levels found in related species from relatively clean Australian waters (Table 5). On the strength of these findings, we conclude that spondylids, hold little promise as bioindicators of mercury pollution.

Cephalopod mollusks appear to have relatively high affinities for mercury. For example, Renzoni *et al.* (1973) reported levels of 0.75-2.32 µg/g wet weight in the tentacles of *Octopus vulgaris* from a polluted section of the Tyrrhenian coast. Levels in the liver were appreciably higher and topped 200 µg/g wet weight in one individual. These values are far greater than those found in the same tissues of octopus from Apra Harbor during this study (Table 14).

1.6.6 Hg in Crustaceans:

Crustaceans tend to mirror environmental levels of mercury under certain conditions. The edible portions of two species from Minimata Bay, for example, yielded levels of 41 and 100 µg/g (~8 and 20 µg/g wet weight respectively) at the time the mercury pollution problem was discovered (Matida and Kumada 1969). Normally, however, mercury levels in crustacean tissues remain well below those considered hazardous for human consumption (Eisler 1981) and are of the same magnitude as those presented here for mantis shrimp from Apra Harbor (Table 14).

1.6.7 Hg in Ascidians:

Little published information exists for mercury in tunicates. Yannai and Sachs (1978) analyzed the ascidian, *Ciona intestinalis,* from the eastern Mediterranean area and found mercury levels of 0.03-0.12 µg/g wet weight, in whole organisms. Levels reported here for Apra Harbor ascidians were generally lower and ranged from 0.007-0.041 µg/g wet weight. Whether tunicates can adequately reflect changes in mercury's ambient availability remains to be unequivocally established although Matida and Kumada (1969) reported a high of 35 µg/g in one species from Minimata Bay.

1.6.8 Hg in Fish:

Mercury levels in fish are generally higher than found in most invertebrate species and tend to be age and trophic level dependant. Thus, highest natural levels are usually found in the larger, long-lived, predatory species like sharks, tuna, marlin and swordfish (Bligh and Armstrong 1971, Windom *et al.* 1972, Rivers *et al.* 1972, Nishigaki *et al.* 1973, Beckett and Freeman 1974, Mackay *et al.* 1975, Shultz and Crear 1976, Denton and Breck 1981). In some cases, levels of mercury in fish from remote areas have been known to exceed maximum values recommended for human consumption (Denton and Burdon Jones 1986c).

In non-polluted situations, mercury levels in fish are generally less than 0.2 µg/g wet weight (Holden 1973, Denton and Burdon-Jones 1986c). However, it is now generally agreed that fish possess little ability to regulate tissue levels of mercury in the same way as they can other essential elements like copper and zinc. Therefore, they serve as useful indicators of environmental contamination by this metal. Fish flesh analyzed from Minimata Bay, for example, contained up to 309.1 µg/g wet weight, way beyond levels considered safe for human consumption. It should be noted here that mercury has caused more problems to consumers of fish than any other inorganic compound.

In the present study, 11 out of 38 fish from Apra Harbor (29%) contained mercury in their axial muscle at concentrations above 0.2 µg/g wet weight (Table 15). The highest level (1.157 µg/g wet weight) occurred in one specimen of lizardfish, *Saurida nebulosa*, from the Hotel Wharf area. Other species analyzed from this site also contained relatively high concentrations of mercury in their muscle tissue, including the conger eel, *Gymnothorax javanicus*, (0.58 µg/g wet weight) and the snapper, *Caranx malampygus* (0.66 µg/g wet weight). It is noteworthy that all three fish are predatory species and the latter two were among the largest specimens captured during the study.

Only one fish from Agat Marina contained an axial muscle mercury concentration above 0.2 µg/g wet weight, but this was only a marginal exceedence (0.214 µg/g wet weight in the snapper, *Lethrinus rubrioperculatus*). Mercury levels in fish taken from Agana Boat Basin and Merizo Pier were consistently below 0.2 µg/g wet weight. This was due, in part, to the fact that specimens from both locations were relatively small-sized individuals.

Mercury was detected in all fish livers examined and was usually higher than levels found in flesh. A significant correlation ($P<0.05$) was found between the two tissues in the pooled data-sets for all specimens analyzed. A similar relationship has been noted for other tropical species (Denton and Breck 1981, Denton and Burdon-Jones 1986c).

1.6.9 Concluding Remarks:

The findings presented here confirm earlier suspicions of increased mercury availability to the biota in the outer Apra Harbor area. While not excessive, levels recorded indicate a need to expand the fish survey and focus more on larger representatives of some of the more popular table fish taken from this area. Emphasis should also be given to residential species within the inner Apra Harbor region where very high sediment levels of mercury have previously been reported (Belt Collins 1993).

1.7 Nickel (Ni):

Nickel is only moderately toxic to most species of aquatic plants and is one of the least toxic inorganic agents to invertebrates and fish (Denton and Burdon-Jones 1982, 1986d, Moore 1991). Open ocean concentrations of dissolved nickel normally lie between 0.1 and 0.3 µg/l (Boyle *et al.* 1981, Bruland 1979, Denton and Burdon-Jones 1986e). In polluted nearshore and estuarine waters, levels of between 5 and 30 µg/l have been reported (Halcrow *et al.* 1973, Abdulla and Royle 1974, Boyden 1975). Total nickel residues in clean coastal sediments typically range between 10 and 20 µg/g (Bryan and Langston 1992) but may fall below 1 µg/g in unpolluted coastal regions, away from nickel bearing geological formations (Moore 1991). In contaminated regions, concentrations may exceed 200 µg/g (Fowler 1993). Sedimentary nickel levels recently determined in Guam harbors ranged from <0.2-71.0 µg/g with areas of enrichment confined to Agat Marina and Merizo Pier. Baseline levels throughout the area were estimated at 1-3 µg/g.

1.7.1 Ni in Algae:

In general, algae from clean water areas contain relatively low concentrations of nickel although there are some notable exceptions, particularly among the Rhodophyta (Denton and Burdon-Jones 1986a). For example, the red algae, *Amansia glomerata* and *Ceratodyction spongiosm*, from remote sites along the Australian Great Barrier Reef, yielded highs of 17.0 and 36.9 µg/g respectively (Denton and Burdon-Jones 1986a). In contrast, levels found in the brown algae, *Padina* spp., from this area ranged from 1.0-1.5 µg/g. Much higher levels have been reported for this genus from relatively contaminated waters. For instance, Stevenson and Ufret (1966) reported levels of 23-32 µg/g in *P. gymnospora* from Puerto Rico, while Agadi *et al.* (1978) found 8.0-18.3 µg/g in *P. tetrostromatica* from Goa, in southern India. The same species from the upper reaches of Townsville Harbor contained a high of 13.1 µg/g (Burdon-Jones *et al.* 1975). In the present study, we determined nickel concentrations in *Padina* sp. ranging from ~1-3 µg/g (Table 8), indicative of low ambient levels of dissolved nickel in Guam harbor waters.

1.7.2 Ni in Sponges:

No previous reports of nickel levels in sponges were found in the literature. The data presented here, for Guam species, indicates that certain members of the group are capable of accumulating this element to respectable levels. However, there is no firm evidence to suggest that any of the species examined are useful bioindicators of nickel enrichment.

1.7.3 Ni in Corals:

From the limited available data it would appear that coelenterates normally do not concentrate nickel in their tissues. However, among the soft corals, there appears to be one or two exceptions. For example, *Lithophyton* sp. taken from Heron Island, on the Great Barrier Reef, contained 70 µg/g compared with levels of <0.5 µg/g in *Sarcophyton* and *Sinularia* spp. found growing beside it (Denton and Burdon-Jones 1986b). Likewise, the temperate soft coral, *Alcyonium digitatum*, from the Irish Sea, was found to contain 17.0 µg/g (Riley and Segar 1970). Soft corals analyzed during the course of the present work contained nickel levels of 0.2-0.8 µg/g (Table 10), in line with levels recorded earlier for these genera from Australian coastal waters (Burdon-Jones and Klumpp 1979, Burdon-Jones and Denton 1984b).

Hard corals may have indicator capabilities for nickel although this has yet to be substantiated. Nevertheless, species from geochemically enriched areas of the Caribbean were found to contain 2.0-23.0 µg/g (Livingston and Thompson 1971), whereas mean nickel levels in related species taken from the Great Barrier Reef ranged from 0.09-0.56 µg/g (Denton and Burdon-Jones 1986b). In the present study we noted 2.12 µg/g in *Acropora formosa* from Apra Harbor, slightly higher than found for the same species in Australian waters. Interestingly enough, nickel concentrations in *Pocilopora damicornis* from Apra Harbor were marginally higher than at the other harbor sites (Table 10).

1.7.4 Ni in Sea Cucumbers:

Few studies have considered nickel levels in echinoderms. Riley and Segar (1970) measured 1.5 µg/g in the whole starfish, *Asterias rubens*, while Stevenson and Ufret (1966) reported 49.0-52.0 µg/g in the skeleton of the sea urchin, *Echinometra lucunter*. Levels in sea cucumbers seem equally variable. For example, Noddack and Noddack (1939) reported a high value of 38 µg/g in the sea cucumber, Stichopus tremulus, while levels in the related species, *Stichopus variegatus*, from the Great Barrier Reef, were consistently below a detection limit of 0.8 µg/g (Burdon-Jones and Denton 1984a). In the present study, we noted distinct difference in nickel levels between the two species of sea cucumbers examined (Table 11). The higher levels were consistently encountered in *Bohadschia argus* and ranged from 0.28-1.38 µg/g and 0.39-0.96 µg/g in muscle and hemal system respectively. A similar value of 1.7 µg/g was reported for body wall muscle of the sea cucumber, *Molpadia intermedia*, from a dredge soil disposal site in the Georgia Strait (Thompson and Patton 1978).

1.7.5 Ni in Mollusks:

It is apparent from the literature, that there is a high degree of inter-specific variation in the ability of bivalves and other mollusks to accumulate nickel. For example, the soft parts of the slipper limpet, *Crepidula fornicata*, produced a value of 850 µg/g for Segar *et al.* (1971). However, the highest level reported to date is 5000 µg/g in the kidney of the giant clam, *Tridacna maxima*, from Flinder Reefs, 200 km off the north Queensland coast in northern Australia (Burdon-Jones and Denton 1984a).

Oysters are poor accumulators of nickel and have not been shown to be effective indicators of environmental quantities of this element. Levels reported by Burdon-Jones and coworkers for Australian species from Townsville Harbor and adjacent coastal waters, and Heron Island on the Great Barrier Reef, ranged from <0.2-2.8 µg/g (Table 5) and were not reflective of environmental differences in nickel availability. Nickel levels determined in oysters during the present study, were very similar and ranged from <0.4- 3.6 µg/g (Table 12).

The chamids and spondylids are far more affective accumulators of nickel than oysters, although their bioindicator capacity for this element remains in question. In spondylids, the kidney is the major site of nickel accumulation and levels in excess of 200 µg/g are commonplace (Burdon-Jones and Klumpp 1979). The tissue distribution of nickel and other trace elements in chamids awaits investigation. Nickel concentrations determined in both groups during the present study were generally lower than found in related species from Australian waters (Table 5).

Primary deposition sites for nickel in cephalopod mollusks seems to vary between subgroups. In octopus, the liver is the chief storage organ as shown here (Table 14). The same is true for cuttlefish (Table 5), whereas, in squid, levels are distributed fairly equally between tissues (Horowitz and Presley 1977).

1.7.6 Ni in Crustaceans:

Nickel levels in the edible tissues of crustaceans are typically low and rarely exceed 2.0 µg/g, according to data presented by Burdon-Jones *et al.* (1977) and Hall *et al.* (1978). Levels encountered in mantis shrimp during the present investigation are in agreement with these earlier findings. Interestingly, the exoskeleton has been found to have high nickel adsorbing properties in certain species (Yoshinari and Subramanian 1976, Fowler 1977).

1.7.7 Ni in Ascidians:

According to Bryan (1976), average nickel levels in ascidians are around 8 µg/g although he fails to pinpoint his data sources. We came across only one reference of any value and that was by Ikebe and Tanaka (1979). These authors reported a nickel concentration of 0.13 µg/g in the tunicate, *Halocynthia roretzi*, from an unspecified location. This translates to around 2.6 µg/g on a dry weight basis, assuming a water content of 95%, and lies within the range determined here for ascidians from Apra Harbor (Table 14).

1.7.8 Ni in Fish:

The flesh of most marine fish rarely contains nickel concentrations in excess of 1 µg/g, although levels of up to 10.8 µg/g have been reported in the literature (Roth and Hornung 1977). Plaskett and Potter (1979) gave values for nickel in fish muscle from Cockburn Sound, Australia, which ranged from 0.11-3.88 µg/g. Burdon-Jones *et al.* (1975) detected nickel in only one out of 18 fish from Townsville coastal waters. All the rest had levels below an analytical detection limit of 0.2-0.9 µg/g. Likewise, Denton and Burdon-Jones (1986c) failed to detect nickel in the axial muscle of 190 fish, representing 50 different species, from several different trophic levels along the length of the Great Barrier Reef. Hepatic nickel concentrations determined by these workers were also found to be below the limits of analytical detection. It comes as little surprise, then, that nickel residues were undetectable in muscle and liver tissues of every fish analyzed during the present study.

1.7.9 Concluding Remarks:

In light of the data presented, nickel does not appear to be a metal of environmental concern in any of the harbor environments investigated.

1.8 Lead (Pb):

Although inorganic lead is only moderately toxic to aquatic plants and animals, organolead compounds, particularly those used as antiknock agents in gasoline, are highly toxic to all forms of life (Denton and Burdon-Jones 1986d, Moore 1991). Inorganic lead is barely soluble in seawater and levels in open ocean waters typically range from 0.005-0.015 µg/l. Even in highly polluted waters, levels are unlikely to rise above 0.05 µg/l (Burnett *et al.* 1977). Thus, particulate lead accounts for >75% of total lead in most waters (Moore 1991).

Total lead levels in clean, non-geochemically enriched sediments are in the order of 25 µg/g or less, but may exceed 400 µg/g near wastewater outfalls (Schafer and Bascom 1976, UNEP 1985, Louma and Phillips 1988, Bryan and Langston 1992,). In severely polluted locations, near mining activities, or industrial processes that utilize lead, sedimentary lead concentrations may exceed 2000 µg/g (Jones 1986, Bryan and Langston 1992). The highest level reported to date is 266,000 µg/g in sediments adjacent to a battery factory in Suva Harbor, Fiji (Naidu and Morrison 1994).

Lead levels previously reported for Guam harbor sediments ranged from a low of <0.6 µg/g in all samples from Agat Marina to a high of 324 µg/g in sediments from the inner Agana Boat Basin, adjacent to the refueling station (Denton *et al.* 1997). Levels exceeding 100 µg/g were also found at Apra Harbor adjacent to Hotel Wharf, the central portion of Commercial Port, and the northern end of Dry Dock Island. Biota were collected for analysis from within the vicinity of each of these sites during the current work.

1.8.1 Pb in Algae:

Algae have a high affinity for lead, and levels in excess of 100 µg/g have been reported in tropical species from relatively contaminated waters (Burdon-Jones *et al.* 1975, Agadi *et al.* 1978). The highest level reported to date is 1200 µg/g in the green alga, *Enteromorpha* sp., from a severely polluted fjord on the West Coast of Norway (Stenner and Nickless 1974).

Lead concentrations determined in *Padina* sp. from Guam harbors, during the current work, ranged from <0.25-8.07 µg/g and are relatively low by world standards (Table 5). The lowest levels were found at Agat Marina and the outer Agana Boat Basin area. The highest levels occurred in samples removed from the submerged concrete foundations of the refueling station at the Cocos Island ferry terminal, at Merizo.

1.8.2 Pb in Sponges:

Virtually nothing is known about the elemental composition of sponges despite the group's widespread geographic distribution. At the time of writing, we were unable to locate a single reference that dealt with lead in any of the 4000+ marine species described to date. Reviewing the data presented here for Guam sponges, it is clear that several species demonstrate relatively high concentration factors for this element (Table 9). Moreover there is some consistency in the data, highlighting Apra Harbor as a lightly lead-enriched area.

1.8.3 Pb in Corals:

The coelenterates, in contrast to the porifera, have received some attention from environmental chemists interested in their mineral content, and lead levels ranging from <2.0-42 µg/g appear in the literature for corals from the Caribbean (Livingston and Thompson 1971). A temperate species of soft coral, *Alcyonium digitatum*, taken from the Irish Sea reportedly contained 24.0 µg/g lead (Riley and Segar 1970). These data contrast markedly with the findings of Denton and Burdon-Jones (1986b), who were unable to detect lead in 10 soft coral and 3 hard coral species from the Australian Great Barrier Reef. In the present study, we were also unable to determine detectable quantities of lead in any of the hard and soft corals analyzed (Table 10).

1.8.4 Pb in Sea Cucumbers:

From the literature, it would seem that echinoderms are unable to regulate lead levels in their tissues and therefore may serve as potentially useful indicators of environmental contamination by this metal. Stenner and Nickless (1974) reported lead levels of up to 460 µg/g in various echinoderms from the West Coast of Norway. Matsumoto (1964) gave values of up to 14.4 µg/g wet weight in *Holothuria* sp. from lead-contaminated coastal waters of Japan, while Denton (unpublished data) found 3.8 µg/g in the same genera from a residential beach in Townsville, Australia. In contrast, *Stichopus variagatus*, from pristine waters of the Great Barrier Reef, contained <1.0 µg/g of lead in their body wall muscle (Burdon-Jones and Denton 1984a). Similarly low concentrations were found in both species of sea cucumber taken from Guam harbors during the present study (Table 11).

1.8.5 Pb in Mollusks:

Bivalves derive their metal loads primarily via the ingestion of food and suspended particulates, and are generally considered to be excellent indicators of heavy metal pollution (Phillips 1980). However, the utility of oysters as indicators of lead pollution is still a matter of some debate. The published data for lead in oyster tissues currently ranges from <0.1-84 µg/g, with the great majority of figures being less than 10 µg/g (Eisler 1981) in keeping with the results presented here (Table 12). It certainly seems like oysters have bioindicator potential for lead, although the work of Denton and Burdon-Jones (1981) suggests otherwise. These researchers examined the uptake and depuration kinetics of lead in the black-lip oyster, *Saccostrea echinata*. They found this bivalve's affinity for lead to be much lower than that shown for cadmium and mercury. Moreover, the biological half-life of lead in this species was relatively short, in the order of 30 days. It was concluded, therefore, that *S. echinata* was not a particularly sensitive indicator of lead. Moreover, its usefulness as a long-term integrator of this element was questionable in areas where ambient levels fluctuated widely. This latter failing could certainly account for the high variability noted in specimens collected from Agana Boat Basin during the current study.

The utility of the chamids as indicators of lead pollution is also suspect, based largely on their poor sensitivity and lack of response in areas of known lead-enrichment (Burdon-Jones and Klumpp 1979). Spondylids, on the other hand, are excellent candidates and readily respond to changes in ambient lead availability. They also have a high affinity for lead, concentrating it almost exclusively in the enlarged kidney in much the same way as tridacnid clams (see Denton and Heitz 1992, 1993). Previous studies with *Spondylis ducalis* from Australian waters have clearly shown that lead concentrations in the kidney of this species are highly correlated with distance from the coast. Specimens collected from patch reef areas 3, 24 and 42 km offshore, for example, contained mean renal lead levels of 40.3, 18.8 and 15.8 µg/g respectively (Burdon-Jones and Klumpp 1979).

Mean lead levels in whole soft tissue homogenates of *S. ducalis* from remote locations of the Great Barrier Reef were understandably lower and ranged from 1.63-5.50 µg/g (Burdon-Jones and Denton 1984a). In the present study, lead levels in whole soft tissues of *S. multimaricatus* from Agat Marina were of a similar order and ranged from 1.8-6.3 µg/g (Table 13). Predictably, levels were considerably higher in specimens from the inner portion of Agana Boat Basin and clearly identify this area as a zone of lead-enrichment.

Whether hepatic lead levels in octopus are reflective of this element's ambient availability is uncertain at this stage. Certainly the high value of 24.6 µg/g determined in the Apra Harbor specimen during the current work is appreciably higher than those recorded in the same tissue of cuttlefish and squid from Townsville coastal waters (Table 5). Fortunately, lead does not appear to accumulate in the edible portions of cephalopod mollusks.

1.8.6 Pb in Crustaceans:

Crustaceans tend to accumulate lead in their exoskeleton more so than their soft parts. As a consequence, levels found in edible tissues are typically low (Fowler 1977). In his review of the available data, Eisler (1981) cites values in crustacean muscle tissue ranging from <0.5-3.4 µg/g wet weight with the vast majority falling between 0.5-1.0 µg/g wet weight (Hall *et al.* 1978). This translates to an approximate range of 2.5-5.0 µg/g on a dry weight basis and is considerably higher than found in mantis shrimp during the present study (Table 14)

1.8.7 Pb in Ascidians:

The elemental composition of tunicates remains a little worked area despite the discovery of high vanadium concentrations in ascidians at the turn of the century. Papadopoulou and Kanias (1977) attempted to revive interest in the group, from a monitoring perspective, with their work on *Ciona intestinalis* and *Microcosmus sulcatus*, two ascidians that are reasonably well represented in temperate and Mediterranean waters. Their work highlighted relatively high affinities for certain elements in both species, although the indicator ability of each remains to be established. Lead levels reported by these authors ranged from 0.52-1.9 µg/g. Interestingly, an almost identical range was determined here in local ascidians from Apra Harbor (Table 14).

1.8.8 Pb in Fish:

Lead was not detected in the axial muscle of any fish analyzed during the present investigation and was only rarely seen in the livers. The limit of analytical detection was 0.5 µg/g or better in the great majority of samples analyzed. Concentrations at or close to this detection limit have been reported for tropical species from other areas of the world (Babji *et al.* 1979, Powell *et al.* 1981, Phillips *et al.* 1982, Denton and Burdon-Jones 1986c).

It is generally acknowledged that human activities influence the lead content of marine teleosts. Thus, Halcrow *et al.* (1973) found levels of 5.8-15.0 µg/g in the muscle tissue of eight demersal fish species from the polluted waters of the Firth of Clyde, in Scotland (UK). Reported levels for lead in the axial muscle of fish from less contaminated areas are, however, generally lower. For example, Portmann (1972) published mean lead levels ranging from <0.5-0.99 µg/g wet weight in various commercial fish species from UK coastal waters. Likewise, Eisler (1981) concluded that lead levels in the majority of fish analyzed from U.S. coastal waters were 0.3-0.7 µg/g wet weight.

In Australia, a similar range of means (0.4-0.71 µg/g wet weight) was given for 9 commercial fish species from New South Wales (Bebbington *et al.* 1977). Somewhat higher mean values of 1.55-2.24 µg/g wet weight were found in 12 species of fish from Cockburn Sound (Plaskett and Potter 1979).

1.8.9 Concluding Remarks:

In light of discussions presented above, it is clear that some mild lead-enrichment has occurred in the sediments and certain biota of Agana Boat Basin and Apra Harbor. However, the data indicate that such enrichment is generally localized and has not significantly impacted upon the quality of edible resources inhabiting these waters.

1.9 Tin (Sn):

Naturally occurring inorganic tin is relatively harmless to aquatic organisms. In contrast, organotin compounds like tributyl tin (TBT), a modern-day biocide in antifouling paints, are extremely toxic (UNEP 1985, Bryan and Langston 1992). All forms of tin are relatively insoluble in seawater. Inorganic tin concentrations in uncontaminated waters are commonly around 0.01 µg/l (Förstner and Wittman 1979). TBT is usually of the same order but may exceed 0.6 µg/l in harbors and marinas (Langston *et al.* 1987, Waldock *et al.* 1987). In extreme cases identified in England and Denmark, concentrations of up to 3 µg/l have been detected (Muller *et al.* 1989).

Natural tin concentrations in uncontaminated, non-mineralized sediments usually lie between 0.1-1.0 µg/g, and in geologically enriched areas may exceed 1000 µg/g (Bryan *et al.* 1985, Bryan and Langston 1992). Typical surface sediment values for TBT range from 0.005-0.05 µg/g and usually account for less than 5% of the total tin present (Brian and Langston 1992). An all time high of 38 µg/g TBT was found in sediments from Suva Harbor, Fiji (Stewart and de Mora 1992).

Baseline levels of tin in marine carbonate sediments from Guam were estimated to be less than 0.1 µg/g. Total tin levels in local harbor sediments mostly ranged between 1-3 µg/g although levels between 10 and 45 µg/g were occasionally observed (Denton *et al.* 1997). Levels of TBT and other organotin compound in local harbor sediments, although currently unknown, are assumed to be extremely high in places. For example, an earlier investigation revealed total tin concentrations of 148-1055 µg/g in sediments adjacent to a US naval ship repair and maintenance facility, in the inner Apra Harbor area (Belt Collins, Hawaii 1993). Undoubtedly, these high values are related to the sandblasting and repainting of naval docks and vessels with organotin-based anti-fouling paints.

Total tin levels found in biota from Guam harbors during the current work are discussed below. The fact, that little to no comparative information exists for several groups examined, highlights the need for reliable baseline data for this element in tropical marine ecosystems.

1.9.1 Sn in Algae:

Freshwater macrophytes biomagnify tin over aqueous levels achieving experimental concentration factors in the order of 90,000 for inorganic tin (Wong *et al.* 1984) and 30,000 for TBT (Maguire *et al.*1984). In contrast, concentration factor estimates for marine algae, from field data, are about an order of magnitude lower (Smith and Burton 1972, Bryan and Gibbs 1991).

Black and Mitchell (1952) measured total tin concentrations of 0.10-2.2 µg/g in 5 species of seaweed from Argyllshire (UK). Smith and Burton (1972) gave a slightly narrower range of 0.1-0.65 µg/g for brown algae from Southampton waters (UK). More recently, Langston *et al.* (1987) analyzed benthic organisms from the contaminated waters of Poole Harbor in Dorset, England, and reported total tin levels of 0.11-1.7 µg/g in the brown alga, *Fucus vesiculosus*. These authors also measured TBT and found that it accounted for around 40% of the total tin present.

In the current study, concentrations of total tin in *Padina* sp. from all harbor sites were consistently below a detection limit of 0.01 µg/g suggesting relatively low levels of dissolved tin in these areas (Table 8). However, marine algae are generally considered to be poor accumulators (and indicators) of tin, probably because of their ability to metabolize organic and inorganic forms of this element (Bryan and Langston 1992).

1.9.2 Sn in Sponges:

Surprisingly high total tin levels were found in a number of sponges analyzed during the present investigation, especially those taken from Agana Boat Basin, Agat Marina and the Merizo Pier area (Table 9). However, in the absence of any comparative date for sponges from elsewhere in the world, it is difficult to draw any satisfactory conclusions from these observations. Nevertheless, some degree of tin-enrichment is indicated in all three areas relative to the Apra Harbor sites. It should be noted here that organotin compounds are highly lipid soluble and sponges are relatively rich in lipids.

1.9.3 Sn in Corals:

Tin concentrations measured in soft and hard corals during the present study reinforce the harbor differences noted above with sponges (Table 10). The data also clearly show that soft corals have a greater affinity for this element than do their reef-building relatives. Comparative data for this group is limited to the work of Livingston and Thompson (1971) who failed to find tin in 34 species of hard coral from the Caribbean. However, their work was compromised by a relatively high detection limit of 5 µg/g.

1.9.4 Sn in Sea Cucumbers:

Data for both species of sea cucumber, examined here, clearly indicate tin-enrichment at all sites other than those in Apra Harbor (Table 11). Both muscle and hemal system portrayed similar distribution patterns for this element, although concentrations were generally much higher in the latter tissue. An exhaustive literature search failed to find any reference to tin in sea cucumbers from other areas of the world. Nonetheless, levels encountered here are among the highest ever reported for invertebrates in general (Bryan 1976, Eisler 1981). We strongly suspect that they reflect organotin uptake from the ingestion of contaminated sediments. Natural tin is strongly sorbed to aquatic sediments and as such is relatively unavailable to the benthos. Even when sedimentary concentration exceed 1000 µg/g, levels in the biota rarely pass 10 µg/g (Bryan and Langston 192). In contrast, organotin compounds are lipid soluble and are readily transferred across biological membranes.

1.9.5 Sn in Mollusks:

Certain bivalves have a high affinity for tin, reflecting their inability to metabolize both inorganic and organic forms of this element. For example, specimens of the long-neck clam, *Mya arenaria*, from Poole Harbor were found to contain total tin concentrations of 7.62-21.4 µg/g. Apparently, organotin compounds (TBT and DBT) accounted for around 95% of total residues (Langstone *et al.* 1987). Even higher TBT levels, 36.8 µg/g were found in this species from the Itchen Estuary, in the south of England (Bryan and Gibbs (1991).

Oysters have a somewhat lower affinity for tin than *M. arenaria*. For example, maximum total tin and TBT levels in *Crassostrea gigas* from the heavily contaminated waters of Arcachon Bay, on the French coast, ranged from 0.7-7.0 µg/g and 0.4-1.6 µg/g respectively (Alzeui *et al.* 1986). A higher TBT range of 0.27-0.33 µg/g wet weight (~1.4-1.7 µg/g on a dry weight basis) was reported by Thain and Waldock (1986) for *Ostrea edulis* from the polluted Crouch estuary, in eastern England. Control oysters from uncontaminated sites contained 0.1 µg/g wet weight (~0.4 µg/g dry weight). In the current study, total tin levels in oysters from Guam harbors ranged from <0.1-0.57 µg/g (Table12) and are, therefore, among the lowest reported in the literature for this group. Interestingly, the highest levels encountered throughout the study were in specimens collected from Apra Harbor in direct contrast to that observed with the invertebrate groups discussed above.

No baseline data exists for tin in chamid and spondylid bivalve mollusks. Levels encountered in both groups during the current work were similar to those in oysters (Table 13). They also compare reasonably well with levels found in other bivalves (0.23-0.67 µg/g) analyzed by Smith in the early seventies (Smith and Burton 1972). These particular specimens were taken from Southampton waters (UK) at about the time that organotin compounds were gaining popularity, as an alternative to copper and other heavy metals, in anti-fouling paints. It seems unlikely, therefore, that they would have been severely contaminated with TBT.

1.9.6 Sn in Crustaceans:

Crustaceans possess the necessary enzymes to break down organotin compounds fairly rapidly and, therefore, would not be expected to accumulate high concentrations of this element under typical harbor conditions. Levels found in mantis shrimp from Apra Harbor during the present study tend to confirm this (Table 14). However, relatively high total tin levels of 0.6-2.0 µg/g wet weight (~3.0-10 µg/g dry weight) were found in the edible tissues of several crustacean species analyzed by Hall *et al.* (1978).

1.9.7 Sn in Ascidians:

Total tin levels in the majority of ascidians analyzed during the current work were below an analytical detection limit of 0.01 µg/g. Detectable concentrations ranged from 0.01-0.13 µg/g (Table 14). Comparable tin data for this group is restricted to one publication by Smith (1970) who reported a total tin concentration of 15 µg/g in the internal organs of the ascidian, *Ascidia mentula*, from Southampton waters.

1.9.8 Sn in Fish:

Organotin compounds are rapidly metabolized and excreted in fish. Consequently, they generally do not accumulate in the tissues of this group (Bryan and Langston 1992). According to Eisler (1981), mean total tin levels for most U.S. coastal water fish species are between 0.4-0.8 µg/g wet weight in muscle and 0.3-0.7 µg/g wet weight for liver. On a dry weight basis, this translates to ~1.6-3.2 µg/g for muscle and ~1.2-2.8 µg/g for liver, assuming a water content of 75% in both tissue. All total tin concentrations in the axial muscle of fish from Guam harbors fell below the national range, as did the great majority of liver values (Table 15).

1.9.9 Concluding Remarks:

From the data, it is clear that sponges, soft corals, and sea cucumbers can accumulate significant quantities of tin. Levels encountered in these organisms, equal or exceed those reported for other invertebrate groups from the TBT-enriched waters of Poole Harbor in the south of England. This strongly suggests that there is significant TBT contamination in the small boat harbors of Guam. Bryan and Langston (1992) point out that small boat harbors and marinas are generally more prone to TBT problems than larger ports and harbors because of the higher density of boating traffic and permanently moored water-craft.

1.10 Zinc (Zn):

Although zinc is not appreciably toxic, it is a ubiquitous contaminant and is sometimes released into the marine environment in substantial quantities (Bryan and Langston 1992). Its omnipresence makes it notoriously difficult to determine in seawater, defying even the most rigorous precautions against contamination. Inter-laboratory calibration exercises undertaken by IOC/UNESCO in the late 1970's and early 80's identified only a handful of laboratories throughout the world that were capable of undertaking such a task. Since then, the pioneering work of Bruland and co-worker, using ultra clean techniques, have reshaped ideas on realistic levels of this element in uncontaminated seawater.

Surface water concentrations of dissolved zinc in the open ocean are now known to be around 0.01 µg/l, several orders of magnitude lower than previously thought (Bruland *et al.* 1978, Bruland 1980). In nearshore waters they are generally higher and show greater variability. A mean value of 0.161 µg/l was reported by Bruland and Frank (1981) for uncontaminated waters of the NW Atlantic, while Denton and Burdon-Jones (1986d) recorded mean levels of 0.06-0.44 µg/l in Australian waters from the Great Barrier Reef. In harbor environments and polluted estuaries, levels are considerably higher, and typically range from 10-50 µg/l (Preston *et al.* 1972, Abdullah and Royle 1974, Zinde *et al.* 1976, Burdon-Jones *et al.* 1982, Scoullos and Dassenakis 1983). One of the highest soluble zinc levels recorded is 305 µg/l from Restronguet Creek, a tidal arm of a large Cornish estuary in the UK that drains an area of heavily mineralized Devonian rocks and ancient mine workings (Klumpp and Peterson 1979).

Sediments from uncontaminated waters typically contain zinc levels of 5-50 µg/g depending upon local geology (Moore 1991). Residues in excess of 3000 µg/g are frequently found in the vicinity of mines and smelters (Bryan *et al.* 1985) and in contaminated harbor environments (Poulton 1987, Logorburu and Canton 1991).

Zinc levels in Guam harbor sediments were shown to span two orders of magnitude, ranging from baseline levels of 1-5 µg/g at uncontaminated sites, to 552 µg/g at Hotel Wharf in Apra Harbor. Levels in excess of 100 µg/g were also found in the inner Agana Boat Basin, at Shell Fox-1 Fuel Pier, Commercial Port, and Dry Dock Island in Apra Harbor, and at the refueling station at the Cocos Island ferry terminal, in Merizo (Denton *et al.* 1997). Biota samples were collected in the vicinity of each of these sites. The data obtained are discussed below.

1.10.1 Zn in Algae:

Marine algae readily concentrate zinc. Among the brown algae, which are most commonly used as indicators of heavy metal pollution, levels ranging from several hundred to several thousand part per million (µg/g) have been recorded in species from severely polluted environments (Bryan and Hummerstone 1973a, Fuge and James 1973, Haug *et al.* 1974, Stenner and Nickless 1974, Melhuus *et al.* 1978). In clean environments, zinc levels are usually less than 10 µg/g. For example, mean levels of zinc in 48 species of algae from the Australian Great Barrier Reef were 2.0, 2.7, and 2.2 µg/g in brown, red, and green representatives respectively (Denton and Burdon-Jones 1986a).

Zinc levels previously reported for *Padina* sp. range from 3.98-9.5µg/g in *P. australasis* from the Australian Great Barrier Reef, to 440 µg/g in *P. tetrostromatica* from the relatively polluted upper reaches of Townsville Harbor (Table 5). In the current study, we found a relatively low mean zinc concentration of 11.0 µg/g in *Padina* sp. from the outer region of Agana Boat Basin (algae were absent from the relatively turbid waters of the inner harbor area). A marginally higher mean level of 18.7 µg/g was encountered in *Padina* sp. from Agat Marina. Clear evidence of zinc-enrichment was found in algae from Apra Harbor and at Merizo Pier, in the vicinity of the Cocos Island ferry terminal (Table 8).

Within Apra Harbor, mean levels of zinc in *Padina* sp. ranged from 45.8-182 µg/g, peaking at Commercial Port (site d). These values are very close to the range of means reported by Burdon-Jones *et al.* (1982) for *P. tetrstromatica* from the lower reaches of Townsville Harbor (Table 5). These authors sampled monthly over one year to establish seasonal variability and showed that zinc fluctuations in the algae (67.2-166 µg/g) mirrored those generally occurring in the surrounding water (0.8-15.0 µg/l). It may be inferred from these data that dissolved levels of zinc in the waters of Apra Harbor are of the same order.

1.10.2 Zn in Sponges:

Very few papers have focused on the elemental composition of sponges and fewer again have looked at zinc. Two reports were uncovered during the course of this work and are briefly reviewed here. The first report by Lowman *et al.* (1966) looks at metal levels in a number of organisms from Puerto Rico coastal waters. The sponges analyzed during the investigation, though not identified, yielded zinc concentrations of 63-180 µg/g. The second study was conducted by Ireland (1973) who conducted a heavy metal survey in a range of organisms from the polluted waters of Cardigan Bay, in Wales (UK). In the latter investigation, only one species of sponge, *Halichondria panicea,* was analyzed for zinc and levels reported ranged from 89-152 µg/g. It is difficult to draw conclusions from these limited data, although the similarity between the two data sets implies that zinc concentrations remain fairly constant in all species of sponge regardless of background levels in the surrounding water. The data

obtained during the current study tends to support this hypothesis despite the greater concentration range encountered (Table 9).

1.10.3 Zn in Corals:

Coelenterates have received slightly more attention than porifera from scientists interested in determining their elemental composition. Published values for zinc in this group range widely, extending from below analytical detection limits to levels greater than 100 µg/g (Eisler 1981). The highest reported zinc value to date is 603 µg/g for the sea anemone, *Actinia equina*, from the polluted waters of Cardigan Bay in Wales, UK (Ireland 1973).

Fewer studies have examined the trace metal accumulating capacity of soft corals. Riley and Segar (1970) reported zinc levels of 46 µg/g in *Alcyonium digitatum* from the Irish Sea. Burdon-Jones and Klumpp (1979) found lower values ranging from 0.4-19.3 µg/g in two species of soft coral from coastal waters near Townsville, Australia. These figures are similar to the range of 1.5-29.0 µg/g found in soft corals from the Great Barrier Reef (Denton and Burdon Jones 1986b).

In the present study, we observed zinc levels of 38.9-143 µg/g in *Sinularia* sp. from Guam harbors (Table 10). Mean levels reported earlier by Denton and Burdon-Jones (1984b) for this genus from the Great Barrier Reef ranged from 1.5-5.7 µg/g. Based on known inter-site difference in zinc availability, these authors concluded that soft corals show bioindicator potential for zinc. The data presented here strongly supports this assumption.

Hard corals also seem to have some bioindicator potential for zinc as evidenced by the work of Livingston and Thompson (1971) and Denton and Burdon-Jones (1986b). The former research team analyzed several species of hard coral from a geologically enriched area of the Caribbean and reported zinc levels ranging from <2.0-70.0 µg/g. In contrast, the latter group examined zinc concentrations in corals from minerally impoverished areas of the Great Barrier Reef and found 0.57-1.3 µg/g. In the present study, we found levels of zinc that ranged from a low of 1.29 µg/g in corals from the outer area of Agana Boat Basin to a high of 7.66 µg/g in specimens from Commercial Port in Apra Harbor (Table 10). These data suggest that hard corals have little to no control over zinc levels in their skeletal and soft tissues in contrast to earlier claims to the contrary by Brown and Holley (1982).

A comparison, of the metal sequestering capability of each group, clearly indicates that soft corals have a greater affinity for zinc (copper and cadmium) than hard corals. The tendency for trace metals in corals to be associated with organic molecules and the higher organic content of soft corals (40-50%) compared with hard corals (0.1-6.0%), is considered to be largely responsible for this (Tapiolas 1980, Brown and Holley 1982 Howard and Brown 1984). However, quantitative differences in the organic matrix, in addition to variations in food, feeding characteristics and colonial growth form, may also be important in determining differences between, as well as within, each group (Harriss and Almy 1964, Howard and Brown 1984, Lasker 1981, Schlichter 1982, St. John 1974).

1.10.4 Zn in Sea Cucumbers:

In echinoderms, zinc concentrations in excess of 100 µg/g are not unusual. For example, Leatherland and Burton (1974) reported levels of 220 µg/g in the starfish, *Asterias rubens*, and Thompson and Paton (1979) found 171 µg/g in the muscle tissue of sea cucumber, *Molpadia intermedia*. Eisler (1981) suggests that the high zinc concentrations among echinoderms reflect their inability to regulate tissue levels of this metal. Thus, they could well prove to be useful indicators of zinc contaminated waters.

Burdon-Jones and Denton (1984a) looked at zinc in the body wall of the sea cucumber, *Stichopus variegatus*, from Lizard Island, Orpheus Island and Heron Island on the Great Barrier Reef, and reported mean levels 7.4, 9.0 and 6.7 µg/g respectively. Zinc levels in sediments at Orpheus Island were ~16 µg/g compared with ~0.5 µg/g at the other two collection sites. As sea cucumbers derive their metal load predominantly from ingested sediments, it was reasoned that specimens from Orpheus Island would contain the highest tissue concentrations of zinc assuming they lacked any regulatory capacity for this element. However, the fact that there was no significant difference between data sets suggested otherwise.

In the current work, we noticed very little inter-site difference in the body wall zinc concentrations of both sea cucumber species analyzed (Table 11). This finding supports the argument for metabolic regulation for zinc, at least in this tissue. Levels showed little variability and ranged from 8.33-18.0 µg/g in *Bohadschia argus*, and 12.6-21.2 µg/g in *Holothuria atra*. Concentrations in the hemal system were appreciably higher, particularly in specimens from the Hotel Wharf and Commercial Port area of Apra Harbor, where sedimentary zinc levels are known to be relatively high. This implies that the hemal system would be a better candidate tissue for determining zinc abundance in the marine environment.

1.10.5 Zn in Mollusks:

It is evident from the literature that trace metal levels in bivalves are subject to considerable inter-specific variation and, in this regard, zinc is probably affected most. Oysters rank among the greatest accumulators of zinc and levels reported in the literature range from less than 100 µg/g in clean waters to 100,000 µg/g in areas impacted by metal mining, smelting, or refining activities (Eisler 1981).

Levels in oysters from harbor locations typically range between 1,000-10,000 µg/g (Table 5). Hence, the high levels of zinc found in oysters during the present study are to be expected given the nature of the environment from which they were collected.

The utility of oysters as biomonitors of zinc and copper abundance in marine and estuarine environments is unequivocally established (Phillips 1980). For this reason, they rank among the most popular choice of sentinel species for pollution monitoring programs. Burdon-Jones *et al.* (1977) examined zinc levels in *Saccostrea amasa* from Townsville Harbor and reported mean monthly levels of 1,916-9,073 µg/g. The same species from an offshore location on the Great Barrier Reef contained much lower levels of 54.4-130 µg/g (Burdon Jones and Denton 1984a). In both cases, tissue concentrations of zinc were between 10^5 and 10^6 times higher

than ambient seawater levels. As a monitoring tool, then, oysters readily provide a first order approximation of zinc availability in their aqueous environment.

Although chamids accumulate respectable levels of zinc in their tissues, there is, as yet, no evidence to indicate they have bioindicator potential for this element. On the contrary, Burdon-Jones and Klumpp (1979) failed to establish any connection between zinc levels in *Chama iostoma* and proximity to land-based pollution sources of this element. Data from the current study is also non-supportive of their utility as sentinel species.

The spondylids, unlike the chamids, are reasonably sensitive indicators of zinc abundance although work still needs to be done to determine their uptake and depuration kinetics for this and other metals of environmental importance. The large spondylid kidney is the primary site of zinc accumulation, just as in tridacnid clams. Renal zinc levels in both groups are generally correlated with available levels in the surrounding water column (Burdon-Jones and Klumpp 1979, Denton and Heitz 1991), and turnover times are relatively long, in the order of 6 months. The sensitivity of this organism as an indicator of zinc is clearly demonstrated by comparing levels found in *S. multimuricatus* during the current study, with related species taken from the Australian Great Barrier Reef (Table 5).

Zinc levels in cephalopod mollusks also tend to be reasonably high, particularly in liver tissue. However, a comparative analysis of published data for cephalopods from other parts of the world suggests that zinc is regulated by these organisms. For example, the hepatic zinc concentration of 573 µg/g in octopus from Apra Harbor, is just outside the upper range of values (247-449 µg/g) found in the livers of squid from California coastal waters (Martin and Flegal 1975). Likewise, the concentration determined in the tentacles of our octopus (69.5 µg/g) was very close to that found in the edible tissue of squid taken from outer continental shelf waters off the southern Texas coast (Horowitz and Presley 1977). Additional examples are presented in Table 5.

1.10.6 Zn in Crustaceans:

Zinc concentrations in decapod crustaceans are generally high, although variations within and between species, as well as between tissues, are often considerable. The hepatopancreas and ovary typically contain the highest zinc levels within individuals, with reported levels ranging from 24-169 µg/g wet weight (~100-700 µg/g on a dry weight basis). Levels within edible body and tail muscle are lower and less variable, and usually lie between 20-100 µg/g (Eisler 1981). Zinc concentrations determined in the tissues of mantis shrimp during the current work fit reasonably well with these data ranges (Table 14).

The work of Pequegnat (1969) suggested that zinc was unregulated by crustaceans, and was accumulated in excess of each organism's immediate needs at rates determined by its ambient availability. However, in an important series of papers, Bryan has amply demonstrated otherwise (Bryan 1964, 1966, 1967, 1968, 1971, 1976). It is now well known that, essential elements, like copper, manganese and zinc are regulated to some extent by crabs, lobster and crayfish as well as amphipods and other crustacean species (Phillips 1980). Thus, crustaceans are not suitable as biological indicators for these metals.

1.10.7 Zn in Ascidians:

Zinc concentrations in ascidians are of the same order as those found in many other soft-bodied invertebrate groups. Levels reported by Papadopoulu and Kanias (1977) for two species of ascidians from the Mediterranean ranged from 100-180 µg/g. Levels recorded here for Apra Harbor specimens were somewhat lower, extending from 15.2-95.8 µg/g. No obvious parallels were apparent with zinc levels in sediments.

1.10.8 Zn in Fish:

Zinc levels in teleosts are generally lower than in most invertebrate groups and probably reflect their ability to regulate tissue levels of this metal within certain limits (Phillips 1980). It is, therefore, not surprising that during the present investigation there was no consistent evidence to suggest zinc levels varied between trophic levels, or between harbor sites. However, the data did show that inter-specific variations of zinc in liver tissue frequently span an order of magnitude or more. It was also evident that hepatic zinc concentrations generally bore no relationship to levels present in muscle tissue.

Zinc concentrations in axial muscle showed relatively little inter- or intra-specific variation and ranged from 8.4-48.9 µg/g for all samples. However, out of the 74 specimens analyzed, only 15% had concentrations above 20 µg/g (mostly from Apra Harbor). The great majority of samples yielded values between 10 and 20 µg/g. Denton and Burdon-Jones (1986c) noted similar findings with fish from the Great Barrier Reef. In their study, axial muscle concentrations of zinc ranged from 4.3-41.8 µg/g in 190 individuals, representing 50 different species. However, zinc concentrations exceeded 20 µg/g in only 8 % of samples analyzed while 16% gave values of less than 10 µg/g.

On a fresh weight basis, the results of the current study also compare favorably with those reported by Powell *et al.* (1981) for 8 tropical marine species from Bougainville Island, Papua New Guinea.

As mentioned above, it is now generally believed that fish actively regulate zinc concentrations in their muscle tissue (Cross *et al.* 1973, Bryan 1976) and, as a result, do not reflect changes in ambient available changes of this element in their environment (Phillips 1980). Therefore, it is noteworthy that generally higher zinc concentration ranges to those presented here have been reported in species from relatively polluted areas of the world (Halcrow *et al.* Eustace 1974, Sims and Presley 1976, Plaskett and Potter 1979) which infers that regulation of this element may not be complete.

1.10.9 Concluding Remarks:

Clear indications of mild to moderate zinc-enrichment of the biota are evident at all four harbor locations. Although contamination by this metal is widespread within Apra Harbor, it is predominantly confined to the inner section of Agana Boat Basin, the refueling station at Agat Marina, and adjacent to the Cocos Island ferry terminal at Merizo Pier.

Table 8

Heavy Metals in Seaweed From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Statistic	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_20
Padina sp.	Agana Boat Basin	18-Dec. '98	mean	0.89	32.2	0.26	0.68	1.53	< 0.002	1.18	0.46	<0.01	11.0	86
			range	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
			n	1	1	1	1	1	1	1	1	1	1	
Padina sp.	Apra Harbor (a)	5-June '98	mean	nc	6.38	0.17	0.62	2.66	0.007	2.55	1.84	nc	45.8	83
			range	<0.11-<0.11	5.79-7.09	0.15-0.18	0.57-0.67	2.59-2.71	nd	2.45-2.74	1.76-1.93	all <0.01	45.1-46.8	
			n	3	3	3	3	3	1	3	3	3	3	
Padina sp.	Apra Harbor (c)	3-June '98	mean	nc	7.34	0.17	1.36	5.42	0.009	1.09	6.04	nc	66.9	85
			range	<0.11-<0.11	7.06-7.58	0.15-0.19	1.31-1.39	5.35-5.46	nd	1.01-1.16	5.59-6.48	all <0.01	65.0-68.4	
			n	3	3	3	3	3	1	3	3	3	3	
Padina sp.	Apra Harbor (d)	9-June '98	mean	nc	33.2	0.18	2.05	33.9	0.026	1.63	4.96	nc	182	81
			range	<0.11-<0.12	30.0-38.1	0.15-0.21	1.97-2.10	29.8-36.6	nd	1.46-1.76	4.67-5.34	all <0.01	176-192	
			n	3	3	3	3	3	1	3	3	3	3	
Padina sp.	Apra Harbor (e)	9-June '98	mean	nc	27.5	0.5	2.9	14.5	0.014	3.0	5.1	nc	119	86
			range	<0.11-<0.12	24.0-35.9	0.49-0.49	2.84-2.98	13.9-15.1	nd	2.89-3.17	4.03-5.82	all <0.01	114-122	
			n	3	3	3	3	3	1	3	3	3	3	
Padina sp.	Apra Harbor (f)	12-June '98	mean	nc	18.3	0.2	2.8	6.3	0.007	1.8	2.6	nc	73.6	90
			range	<0.10-<0.10	17.7-18.8	0.20-0.22	2.80-2.86	6.21-6.36	nd	1.68-1.86	2.58-2.66	all <0.01	72.3-74.9	
			n	2	2	2	2	2	1	2	2	2	2	
Padina sp.	Agat Marina	21-Dec. '98	mean	< 0.08	20.5	0.09	2.67	4.07	< 0.002	2.85	< 0.25	<0.01	18.7	81
			range	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
			n	1	1	1	1	1	1	1	1	1	1	
Padina sp.	Merizo Pier	22-Dec. '98	mean	< 0.08	17.4	0.07	14.1	27.7	0.003	2.28	8.07	<0.01	78.3	83
			range	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
			n	1	1	1	1	1	1	1	1	1	1	

* Hg concentrations as µg/g wet weight; mean = geometric mean; n = number of replicates analyzed; nc = not calculable; nd = no data

Table 9

Heavy Metals in Sponges From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_20
SPONGES													
Callyspongia diffusa	Agat Marina	21-Dec. '98	< 0.11	< 0.01	0.86	9.72	40.4	0.014	6.04	0.45	23.6	62.5	86
Cinachyra sp.	Agana Boat Basin	18-Dec. '98	0.39	< 0.01	0.26	0.98	46.2	0.027	0.40	1.46	5.73	26.4	64
Cinachyra sp.	Merizo Pier	22-Dec. '98	0.11	0.01	0.20	1.11	15.0	0.023	0.87	< 0.72	10.9	22.2	71
Clathria vulpina ?	Agat Marina	21-Dec. '98	< 0.08	< 0.01	0.46	2.02	30.3	0.005	5.37	< 0.25	13.5	200	85
Clathria vulpina ?	Merizo Pier	22-Dec. '98	< 0.11	< 0.01	0.33	0.45	15.6	0.007	0.70	< 0.34	17.0	178	86
Dysidea sp.	Apra Harbor (c)	3-June '98	0.47	6.39	0.23	2.20	72.9	0.015	1.65	2.58	0.03	62.7	71
Dysidea sp.	Apra Harbor (d)	9-June '98	0.33	10.5	0.28	2.24	73.1	0.059	0.62	4.37	0.04	75.6	75
Dysidea sp.	Apra Harbor (f)	12-June '98	< 0.11	6.90	0.15	1.70	21.3	0.010	1.61	2.50	< 0.01	25.8	65
Dysidea sp.	Agat Marina	21-Dec. '98	< 0.10	< 0.01	0.20	4.29	20.2	0.007	3.81	< 0.30	17.7	47.5	84
Liosina cf. granularis	Apra Harbor (b)	5-June '98	0.15	39.7	0.50	24.9	72.4	0.008	9.04	68.3	< 0.01	275	76
Liosina cf. granularis	Apra Harbor (e)	9-June '98	< 0.10	47.7	0.18	15.1	40.3	0.051	8.93	52.0	< 0.01	232	80
Stylotella aurantium	Apra Harbor (b)	5-June '98	< 0.09	6.25	0.33	1.90	23.5	0.021	0.79	2.70	< 0.01	61.2	83
Stylotella aurantium	Apra Harbor (e)	9-June '98	< 0.12	5.96	0.22	2.60	21.0	0.043	1.71	3.02	< 0.01	53.3	84
Stylotella aurantium	Apra Harbor (e)	9-June '98	< 0.10	6.42	0.11	2.43	17.7	0.053	1.15	2.92	< 0.01	70.8	84
Stylotella aurantium	Merizo Pier	22-Dec. '98	< 0.11	< 0.01	0.20	1.33	19.4	0.027	2.01	< 0.33	16.1	83.5	82
UNIDENTIFIED SPONGES													
Brown Wart Sponge	Apra Harbor (e)	9-June '98	0.14	19.8	0.23	17.3	34.9	0.012	10.6	20.3	< 0.01	131	78
Brown Wart Sponge	Apra Harbor (f)	12-June '98	0.24	5.91	0.21	13.5	31.5	0.005	7.04	23.7	< 0.01	144	73
Orange Wart Sponge	Apra Harbor (e)	9-June '98	< 0.10	37.9	0.24	2.27	7.86	0.031	12.61	7.24	< 0.01	34.5	83
Yellow Bread Sponge	Agat Marina	21-Dec. '98	< 0.08	< 0.01	0.14	1.10	6.2	0.004	0.66	< 0.26	6.45	102	86
Yellow Bread Sponge (red outside)	Apra Harbor (c)	3-June '98	< 0.10	43.1	0.14	0.45	17.0	0.087	35.0	1.20	0.01	47.4	84

* = Hg concentrations as µg/g wet weight

Table 10

Heavy Metals in Corals From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_20
SOFT CORALS													
Simularia sp.	Apra Harbor (c)	3-June '98	< 0.11	2.33	0.13	0.31	0.89	0.007	0.53	< 0.34	0.13	143	72
Simularia sp.	Apra Harbor (e)	9-June '98	< 0.12	1.60	0.16	0.27	0.44	0.013	0.70	< 0.37	0.24	76.3	60
Simularia sp.	Agana Boat Basin	18-Dec. '98	2.69	0.01	0.10	< 0.15	0.98	0.004	0.80	< 0.28	10.5	74.5	84
Simularia sp.	Merizo Pier	22-Dec. '98	< 0.10	< 0.01	< 0.05	< 0.16	0.60	0.022	0.24	< 0.30	7.12	38.9	65
HARD CORALS													
Acropora formosa	Apra Harbor (e)	6-June '98	< 0.11	0.14	0.09	0.27	< 0.10	0.017	2.12	< 0.32	< 0.01	1.69	16
Fungia concinna	Apra Harbor (c)	3-June '98	0.24	0.25	0.08	0.34	1.06	< 0.011	< 0.17	< 0.34	0.06	3.14	16
Fungia echidata	Apra Harbor (e)	6-June '98	0.14	0.19	0.10	0.24	0.49	0.007	0.27	< 0.31	< 0.01	1.76	18
Herpolitha limax	Apra Harbor (c)	3-June '98	< 0.12	0.17	0.09	0.29	0.85	< 0.005	< 0.18	< 0.36	< 0.01	2.21	14
Herpolitha limax	Apra Harbor (e)	6-June '98	1.17	0.20	0.08	0.25	1.52	0.015	< 0.15	< 0.30	< 0.01	4.14	16
Pocilopora damicornis	Agana Boat Basin	18-Dec. '98	< 0.10	< 0.01	< 0.06	< 0.17	< 0.11	0.006	< 0.16	< 0.32	0.16	1.29	10
Pocilopora damicornis	Apra Harbor (c)	3-June '98	0.18	67.1	0.07	< 0.12	0.11	< 0.006	0.29	< 0.33	< 0.01	7.16	21
Pocilopora damicornis	Apra Harbor (d)	6-June '98	0.26	0.84	0.24	0.33	< 0.10	< 0.007	0.24	< 0.31	< 0.01	7.66	29
Pocilopora damicornis	Apra Harbor (f)	12-June '98	< 0.11	0.41	0.09	0.14	0.15	< 0.005	0.21	< 0.34	< 0.01	6.97	17
Pocilopora damicornis	Agat Marina	21-Dec. '98	< 0.07	< 0.01	< 0.04	< 0.12	0.24	0.005	< 0.11	< 0.23	0.63	3.26	12
Pocilopora damicornis	Merizo Pier	22-Dec. '98	< 0.12	< 0.01	< 0.06	< 0.19	< 0.13	0.004	< 0.18	< 0.36	0.37	3.81	14

* = Hg concentrations as µg/g wet weight

Table 11

Heavy Metals in Sea Cucumbers From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Tissue	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_20
Bohadschia argus	Agana Boat Basin	18-Dec. '98	M	< 0.10	< 0.01	0.08	< 0.13	0.89	0.007	0.28	< 0.37	14.5	12.5	86
			H	< 0.12	< 0.01	0.18	7.55	2.25	0.096	0.39	< 0.33	40.3	58.3	83
Bohadschia argus	Apra Harbor (b)	5-June '98	M	< 0.13	14.7	0.12	< 0.17	0.63	0.005	1.38	< 0.33	0.26	13.8	87
			H	< 0.13	32.6	0.33	7.28	2.84	0.221	0.43	0.58	3.27	374	78
Bohadschia argus	Apra Harbor (c)	12-June '98	M	< 0.12	17.7	0.11	0.43	0.63	0.005	1.04	< 0.31	0.11	18.0	87
			H	< 0.14	42.8	0.39	31.88	4.15	0.459	1.21	< 0.38	5.25	206	87
Bohadschia argus	Apra Harbor (e)	9-June '98	M	< 0.09	7.81	0.11	0.23	2.26	0.005	1.07	0.56	0.12	13.8	87
			H	< 0.11	16.6	0.32	8.28	39.1	0.301	0.48	0.88	1.72	41.4	80
Bohadschia argus	Agat Marina	21-Dec. '98	M	< 0.09	< 0.01	0.08	< 0.13	0.66	0.001	1.01	< 0.36	7.25	8.33	86
			H	< 0.14	0.15	0.28	12.58	3.15	0.006	0.90	< 0.37	45.9	76.3	85
Bohadschia argus	Agat Marina	21-Dec. '98	M	< 0.09	< 0.01	0.06	< 0.12	0.69	0.003	0.70	< 0.35	19.3	16.6	87
			H	< 0.12	0.20	0.24	6.27	3.45	0.070	0.50	< 0.32	51.9	96.8	84
Bohadschia argus	Merizo Pier	22-Dec. '98	M	< 0.10	< 0.01	0.09	< 0.14	0.59	0.003	1.12	< 0.39	14.8	11.0	88
			H	< 0.09	< 0.01	0.20	10.11	3.47	0.058	0.62	< 0.26	38.5	40.6	84
Holothuria atra	Agana Boat Basin	18-Dec. '98	M	0.24	< 0.01	0.06	< 0.13	1.40	0.008	< 0.19	< 0.36	10.6	12.6	87
			H	0.72	< 0.01	0.12	3.14	6.37	0.091	< 0.43	< 0.72	18.3	117	88
Holothuria atra	Apra Harbor (e)	9-June '98	M	< 0.12	13.6	0.07	0.25	0.71	0.008	< 0.19	< 0.32	0.11	15.5	89
			H	< 0.35	7.24	0.25	2.21	4.70	0.049	< 0.54	< 0.92	1.63	120	91
Holothuria atra	Apra Harbor (g)	12-June '98	M	< 0.10	23.2	0.04	< 0.13	1.18	0.007	< 0.15	< 0.26	0.16	17.9	89
			H	4.90	28.3	0.26	8.58	5.19	0.088	< 0.49	< 0.84	6.54	180	85
Holothuria atra	Agat Marina	21-Dec. '98	M	< 0.10	< 0.01	0.07	< 0.14	1.71	0.014	< 0.22	< 0.40	21.5	17.0	90
			H	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Holothuria atra	Agat Marina	21-Dec. '98	M	< 0.16	< 0.01	< 0.07	< 0.23	1.27	0.022	< 0.34	< 0.63	9.76	15.4	90
			H	< 0.17	0.18	0.09	0.88	3.69	0.072	< 0.28	< 0.47	11.9	141	93
Holothuria atra	Merizo Pier	22-Dec. '98	M	< 0.11	< 0.01	0.07	< 0.16	2.51	0.008	< 0.23	< 0.43	10.7	21.2	86
			H	< 0.11	0.03	0.10	2.85	3.81	0.016	< 0.18	< 0.30	17.8	253	85

M = body wall muscle tissue; H = hemal system; * = Hg concentrations as µg/g wet weight; nd = no data

Table 12

Heavy Metals in Bivalve Mollusks From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site) Date	Statistic	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_20
OYSTERS													
Saccostrea cucullata	Apra Harbor (c)	mean	nc	12.4	0.56	nc	916	0.056	0.65	0.60	0.43	2933	86
	5-June '98	range	<0.14-0.61	8.33-21.8	0.51-0.69	<0.28-<0.49	661-1911	0.043-0.078	<0.36-1.21	<0.33-1.14	0.24-0.70	2262-4722	78-90
		n*	5	5	5	5	5	4	5	5	5	5	5
Saccostrea cucullata	Merizo Pier	mean	4.48	26.5	0.69	1.12	654	0.02	1.37	nc	nc	1153	86
(Juveniles)	22-Dec '98	range	4.09-4.91	21.3-32.9	0.61-0.77	1.03-1.21	598-715	nd	1.25-1.50	<0.31-0.38	<0.01-<0.01	1086-1225	85-87
		n*	2	2	2	2	2	1	2	2	2	2	2
Striostrea cf. mytiloides	Agana Boat Basin	mean	0.56	21.1	0.58	1.81	1968	0.092	1.28	2.79	nc	5130	82
	30-Jan '99	range	0.13-2.96	16.5-35.5	0.36-0.78	0.84-9.04	500-3047	0.080-0.149	0.37-3.60	0.72-12.2	<0.01-0.09	2002-8375	79-86
		n	13	13	13	13	13	4	13	13	13	13	13
Striostrea cf. mytiloides	Apra Harbor (a)	mean	0.14	19.3	0.73	nc	1381	0.039	0.65	0.57	0.34	6367	81
	5-June '98	range	<0.09- 0.47	13.6-25.1	0.51-0.99	<0.19-0.71	878-2076	0.031-0.053	0.45-0.91	<0.27-0.93	0.23-0.57	4014-9789	71-83
		n	8	8	8	8	8	6	8	8	8	8	8
Striostrea cf. mytiloides	Apra Harbor (e)	mean	0.17	12.2	0.31	nc	777	0.033	0.73	nc	0.04	3931	84
	9-June '98	range	<0.08-0.30	9.48-15.0	0.23-0.37	<0.15-0.49	496-1483	0.022-0.043	0.43-2.56	<0.17-<0.24	<0.01-0.08	2148-5643	78-83
		n	10	10	10	10	10	9	10	10	10	10	10
Striostrea cf. mytiloides	Apra Harbor (f)	mean	0.37	14.1	0.43	0.24	1071	0.037	1.03	nc	0.18	4225	84
		range	<0.11-1.34	12.2-18.9	0.39-0.60	<0.18-0.89	629-2971	0.031-0.048	0.68-1.43	<0.21-0.62	0.11-0.27	2800-6280	81-88
		n	10	10	10	10	10	9	10	10	10	10	10
Striostrea cf. mytiloides	Agat Marina	mean	0.13	33.2	0.70	1.74	795	0.017	2.01	nc	0.02	3944	81
	21-Dec '98	range	<0.10-0.20	28.7-38.4	0.56-1.04	1.54-2.01	689-962	0.016-0.022	1.64-2.67	<0.30-<0.70	0.01-0.05	2492-5393	79-84
		n	4	4	4	4	4	3	4	4	4	4	4
Striostrea cf. mytiloides	Merizo Pier	mean	< 0.09	27.7	0.60	2.17	815	nd	2.73	6.48	< 0.02	3571	84
	22-Dec '98	range	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
		n	1	1	1	1	1	nd	1	1	1	1	1

* Hg concentrations as µg/g wet weight; mean = geometric mean; n = number of individuals analyzed; n* = number of pooled samples analyzed (5 oysters per pool); nc = not calculable; nd = no data;

Table 13

Heavy Metals in Bivalve Mollusks From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site) Date	Statistic	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_20
CHAMIDS													
Chama brassica	Apra Harbor (d)	mean	0.25	35.3	0.41	5.09	8.76	0.10	18.9	0.71	0.09	141	86
	9-June '98	range	<0.12-0.58	23.6-51.6	0.23-0.68	3.97-6.22	6.84-11.2	0.033-0.312	14.9-25.1	<0.30-2.03	0.03-0.23	79.4-387	85-87
		n	3	3	3	3	3	2	3	3	3	3	3
Chama lazarus	Apra Harbor (b)	mean	nc	54.1	0.13	1.04	5.95	0.054	2.79	nc	0.03	97	84
	5-June '98	range	<0.10-<0.12	43.0-61.9	0.09-0.16	0.55-2.51	4.42-6.94	0.053-0.055	2.44-3.58	<0.28-<0.35	0.01-0.05	62.7-202	83-86
		n	3	3	3	3	3	2	3	3	3	3	3
Chama lazarus	Apra Harbor (c)	mean	nc	29.2	0.21	1.3	7.3	0.3	2.2	nc	0.010	78.3	80-85
	3-June '98	range	<0.10-<0.10	28.4-30.0	0.21-0.21	1.13-1.42	6.99-7.57	0.064-1.041	1.98-2.53	<0.29-<0.30	<0.01-0.03	50.1-122	83
		n	2	2	2	2	2	2	2	2	2	2	2
Chama lazarus	Apra Harbor (d)	mean	nc	131	0.30	2.77	13.4	0.076	2.52	0.64	0.05	103	86
	9-June '98	range	<0.10-0.23	73.6-331	0.18-0.75	1.94-2.90	8.55-129	0.036-0.193	1.49-7.81	<0.31-0.94	<0.01-0.37	70.1-161	84-87
		n	5	5	5	5	5	4	5	5	5	5	5
Chama lazarus	Apra Harbor (e)	mean	nc	31.9	0.11	1.04	6.57	0.037	1.67	nc	nc	82.2	83
	9-June '98	range	<0.11-<0.11	21.6-66.8	0.09-0.15	0.60-1.36	5.35-8.14	0.020-0.229	1.30-3.19	<0.31-<0.31	<0.01-<0.01	46.2-137	82-84
		n	4	4	4	4	4	4	4	4	4	4	4
Chama lazarus	Apra Harbor (f)	mean	nc	70	0.19	1.91	5.83	0.058	2.48	nc	0.01	102	84
	12-June '98	range	<0.10-<0.12	67.5-104	0.11-0.35	1.38-2.78	5.17-6.52	0.030-0.150	1.78-3.85	<0.30-<0.34	<0.01-0.03	61.8-197	82-86
		n	5	5	5	5	5	4	5	5	5	5	5
Chama lazarus	Merizo Pier	mean	0.11	152	0.18	0.57	7.19	0.018	2.59	nc	0.02	170	81
	22-Dec. '98	range	<0.11-0.22	103-225	0.18-0.19	0.48-0.67	5.35-9.67	nd	1.90-3.53	<0.35-<0.67	<0.02-0.05	127-227	77-84
		n	2	2	2	2	2	1	2	2	2	2	2
SPONDYLIDS													
Spondylus ? multimuricatus	Agana Boat Basin	mean	1.01	44.4	5.95	6.34	331	0.001	15.1	79.5	0.31	492	82.3
	18-Dec. '98	range	0.41-1.73	33.0-52.3	5.30-6.89	2.93-9.55	271-432	0.001-0.001	13.7-18.0	72.8-88.6	0.28-0.33	404-730	79-85
		n	3	3	3	3	3	2	3	3	3	3	3
Spondylus ? multimuricatus	Agat Marina	mean	nc	88.0	5.64	3.27	153	0.003	33.8	2.88	0.11	448	86
	21-Dec. '98	range	<0.10-0.26	46.7-195	3.92-6.76	0.56-6.07	52.5-328	0.002-0.004	23.0-65.2	1.76-6.32	0.07-0.19	213-858	83-88
		n	10	10	10	10	10	5	10	10	10	10	10

* Hg concentrations as µg/g wet weight; mean = geometric mean; n = number of individuals analyzed; nc = not calculable; nd = no data;

Table 14

Heavy Metals in Octopus, Mantis Shrimp and Ascidians From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Tissue	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_20
OCTOPUS														
Octopus cyanea	Apra Harbor (c)	6-June '98	T	< 0.12	96.4	0.06	< 0.16	12.1	0.047	< 0.18	< 0.31	0.17	69.5	80
			HS	4.40	44.3	7.82	1.87	5680	0.242	4.70	24.79	0.77	573	68
MANTIS SHRIMP														
Gonodactylus sp.	Apra Harbor (e)	9-June '98	M	0.27	5.06	0.36	0.57	11.0	0.075	< 0.23	< 0.39	0.09	125	81
			G	1.43	4.58	9.11	0.91	3195	0.085	< 0.81	< 1.38	0.25	148	75
ASCIDIANS														
Ascidia sp.	Apra Harbor (b)	5-June '98	W	0.33	3.92	0.23	1.03	5.58	0.013	0.60	0.54	0.01	22.8	95
Ascidia sp.	Apra Harbor (e)	9-June '98	W	< 0.49	3.05	0.36	5.08	3.48	0.038	< 0.71	< 1.47	0.13	95.8	93
Ascidia sp.	Apra Harbor (e)	9-June '98	W	< 0.13	2.74	0.08	1.41	3.10	0.011	0.84	0.64	< 0.01	15.2	95
Rhopalaea sp.	Apra Harbor (b)	9-June '98	W	< 0.81	3.59	0.44	9.65	9.87	0.011	2.95	2.21	< 0.01	34.1	95
Rhopalaea sp.	Apra Harbor (c)	3-June '98	W	< 0.27	2.31	0.20	3.08	8.57	0.009	0.89	2.91	< 0.01	21.6	95
Rhopalaea sp.	Apra Harbor (d)	9-June '98	W	0.27	2.85	0.13	1.82	6.66	0.007	1.64	1.94	< 0.01	27.6	95
Rhopalaea sp.	Apra Harbor (e)	9-June '98	W	< 0.28	2.84	0.28	3.35	6.46	0.017	1.52	1.06	0.01	20.7	95

* = Hg concentrations as µg/g wet weight, T = tentacle; L = liver; M = tail muscle; G = gonad; W = whole

Table 15

Heavy Metals in Tissues of Fish From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_20
Acanthurus xanthopterus	Agana Boat Basin	18-Dec '98	36	M	< 0.10	8.17	< 0.04	< 0.13	0.30	0.165	< 0.20	< 0.37	< 0.01	8.41	69
				L	< 0.10	12.1	0.72	< 0.15	20.4	1.028	< 0.16	0.50	0.13	426	56
Acanthurus xanthopterus	Agana Boat Basin	31-Dec '98	22	M	< 0.09	9.08	< 0.04	< 0.13	0.39	0.024	< 0.20	< 0.36	< 0.01	12.1	77
				L	< 0.20	2.29	1.44	< 0.31	17.4	0.180	< 0.33	10.8	0.14	485	73
Acanthurus xanthopterus	Agana Boat Basin	30-Dec '98	18	M	< 0.08	7.61	< 0.03	< 0.11	0.42	0.017	< 0.17	< 0.32	0.02	8.76	78
				L	< 0.30	1.49	0.21	< 0.46	17.2	0.169	< 0.48	1.32	0.07	290	74
Acanthurus xanthopterus	Agana Boat Basin	31-Dec '98	14.5	M	< 0.13	10.1	0.06	0.32	0.40	0.065	< 0.27	< 0.50	< 0.01	10.9	78
				L	< 0.90	0.54	< 0.46	< 1.39	10.4	0.333	< 1.39	< 2.70	nd	49.3	wet
Acanthurus xanthopterus	Apra Harbor (c)	3-Jun '98	38.0	M	< 0.07	2.24	< 0.04	< 0.17	0.62	0.265	< 0.16	< 0.35	0.06	8.31	71
				L	< 0.10	2.77	0.18	< 0.14	5.33	1.060	< 0.16	< 0.27	0.28	394	50
Acanthurus xanthopterus	Apra Harbor (c)	3-Jun '98	30.5	M	< 0.06	3.78	< 0.03	< 0.15	3.28	0.067	< 0.15	< 0.32	0.11	12.7	76
				L	0.45	2.37	0.32	< 0.19	97.2	0.356	< 0.22	< 0.38	0.63	435	63
Acanthurus xanthopterus	Apra Harbor (c)	3-Jun '98	29.0	M	< 0.08	6.38	< 0.04	< 0.18	0.51	0.060	< 0.17	< 0.37	0.13	12.4	76
				L	< 0.09	1.25	0.16	< 0.13	7.01	0.123	< 0.15	0.32	0.21	277	58
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun '98	16.5	M	< 0.09	9.00	< 0.05	< 0.22	1.72	0.018	< 0.21	< 0.45	0.20	13.5	81
				L	< 0.53	3.38	0.48	< 0.73	319	0.111	< 0.82	< 1.40	1.91	407	80
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun '98	15.5	M	< 0.09	3.42	< 0.05	< 0.22	2.86	0.014	< 0.21	< 0.45	0.09	11.5	80
				L	< 1.74	0.31	< 0.69	< 2.43	42.9	0.092	< 3.65	< 6.78	nd	47.9	wet
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun '98	12.8	M	< 0.12	2.56	< 0.06	< 0.30	4.00	0.037	< 0.28	< 0.61	0.22	17.7	81
				L	< 0.87	4.31	0.71	< 1.19	9.90	0.053	< 1.35	< 2.30	1.84	214	83
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun '98	11.0	M	<0.17	6.30	< 0.09	< 0.41	5.03	0.035	< 0.39	< 0.84	0.40	14.5	80
				L	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Balistoides viridescens	Merizo Pier	22-Dec '98	18.5	M	0.281	52.4	0.07	< 0.17	0.79	0.048	< 0.26	< 0.48	< 0.01	24.3	82
				L	< 0.18	8.88	0.71	< 0.27	3.43	0.053	< 0.29	< 0.48	< 0.01	392	53
Bolbometopon muricatum	Apra Harbor (c)	3-Jun '98	52.0	M	< 0.08	4.81	< 0.04	< 0.18	2.08	0.022	< 0.17	< 0.37	0.05	20.6	77
				L	< 0.12	5.12	0.06	< 0.16	5.39	0.020	< 0.18	< 0.31	0.18	28.9	30

M = muscle tissue; L = liver tissue; * = Hg concentrations as µg/g wet weight; nd = no data; wet = analysis performed on wet tissue

Table 15 (cont.)

Heavy Metals in Tissues of Fish From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_2O
Caranx ignobilis	Agana Boat Basin	30-Dec '98	26.5	M	<0.09	1.60	<0.04	<0.13	0.6	0.068	<0.19	<0.36	< 0.01	13	78
				L	< 0.57	3.04	0.31	< 0.87	12.2	0.112	< 0.92	< 1.54	0.07	89.9	67
Caranx melampygus	Apra Harbor (b)	5-Jun '98	26.5	M	< 0.07	0.90	< 0.03	< 0.16	1.42	0.660	< 0.15	< 0.33	0.10	14.0	74
				L	< 0.25	2.35	0.54	< 0.34	13.6	0.553	< 0.38	< 0.66	1.18	102	63
Caranx melampygus	Apra Harbor (e)	9-Jun '98	33.0	M	< 0.07	0.95	< 0.03	< 0.15	1.22	0.385	< 0.15	< 0.32	0.13	17.6	76
				L	< 0.31	3.21	0.53	< 0.42	25.2	0.557	< 0.48	< 0.81	0.43	154	71
Caranx sexfasciatus	Agana Boat Basin	30-Dec '98	25	M	< 0.09	3.02	< 0.04	< 0.12	0.64	0.062	< 0.19	< 0.35	< 0.01	13.5	77
				L	< 0.13	7.15	0.48	< 0.20	11.6	0.158	< 0.21	< 0.36	0.01	92.7	68
Caranx sexfasciatus	Agana Boat Basin	30-Dec '98	23	M	< 0.09	1.58	< 0.04	< 0.13	0.67	0.151	< 0.19	< 0.36	< 0.01	11.7	78
				L	< 0.29	2.81	1.80	< 0.45	10.6	0.227	< 0.48	< 0.79	0.29	112	77
Caranx sexfasciatus	Apra Harbor (c)	3-Jun '98	22.0	M	< 0.07	4.93	< 0.03	< 0.17	3.24	0.069	< 0.16	< 0.34	0.13	10.8	76
				L	< 0.71	1.78	< 0.29	< 1.00	3.42	0.069	< 1.50	< 2.78	nd	25.4	wet
Caranx sexfasciatus	Apra Harbor (d)	9-Jun '98	17.0	M	< 0.09	24.2	< 0.05	< 0.22	3.42	0.137	< 0.21	< 0.45	0.09	13.6	77
				L	< 0.47	2.69	4.76	< 0.64	16.1	0.089	< 0.72	< 1.23	9.67	136	66
Cephalopholis sonnerati	Merizo Pier	22-Jan '99	16.5	M	<0.15	2.98	<0.06	<0.20	0.45	0.026	<0.31	<0.57	< 0.01	12.4	74
				L	< 1.56	0.46	< 0.65	< 1.95	3.32	0.010	< 1.95	< 3.78	nd	23.7	wet
Cheilinus chlorourus	Agat Marina	22-Jan '99	22.5	M	< 0.09	2.48	< 0.05	< 0.14	0.51	0.033	< 0.13	< 0.26	0.01	12.0	77
				L	< 2.80	0.79	< 1.44	< 4.33	8.66	0.182	< 4.33	< 8.41	nd	27.8	wet
Cheilinus fasciatus	Apra Harbor (c)	3-Jun '98	24.5	M	< 0.08	4.92	< 0.04	< 0.20	1.85	0.140	< 0.19	< 0.41	0.01	13.4	80
				L	< 0.16	5.41	0.31	< 0.22	5.40	2.197	< 0.25	< 0.42	0.38	83.4	59
Cheilinus fasciatus	Apra Harbor (c)	3-Jun '98	24.5	M	< 0.10	6.52	< 0.05	< 0.23	0.64	0.244	< 0.22	< 0.47	0.04	12.5	80
				L	0.31	4.06	0.35	< 0.35	35.9	1.405	< 0.40	< 0.68	0.69	202	54
Cheilinus fasciatus	Apra Harbor (c)	3-Jun '98	19.0	M	< 0.08	18.7	< 0.04	< 0.20	0.62	0.152	< 0.19	< 0.40	0.01	10.1	77
				L	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Cheilinus trilobatus	Merizo Pier	22-Dec '98	19.5	M	< 0.08	1.65	< 0.03	< 0.11	0.32	0.021	< 0.17	< 0.32	< 0.01	11.0	76
				L	< 0.29	3.81	1.51	< 0.45	9.86	0.060	< 0.48	< 0.80	< 0.01	76.4	51
Cheilinus trilobatus	Merizo Pier	22-Dec '98	19	M	0.10	2.48	< 0.03	< 0.12	0.31	0.023	< 0.18	< 0.33	< 0.01	11.9	76
				L	< 0.18	1.87	0.83	< 0.28	3.78	0.051	< 0.30	< 0.50	< 0.01	31.8	41

M = muscle tissue; L = liver tissue; * = Hg concentrations as µg/g wet weight; nd = no data; wet = analysis performed on wet tissue

Table 15 (cont.)

Heavy Metals in Tissues of Fish From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_2O
Ctenochaetus binotatus	Apra Harbor (d)	9-Jun '98	21.0	M	< 0.07	24.1	< 0.03	< 0.16	0.72	0.101	< 0.15	< 0.32	0.10	9.21	76
				L	< 0.23	13.0	0.35	< 0.31	61.8	0.672	< 0.35	1.66	1.70	466	71
Ctenochaetus striatus	Apra Harbor (e)	9-Jun '98	12.5	M	< 0.16	0.63	< 0.08	< 0.37	1.71	0.013	< 0.35	< 0.76	0.16	10.0	80
				L	< 0.49	1.42	0.66	< 0.66	30.3	0.050	< 0.75	2.08	0.74	540	77
Ctenochaetus striatus	Apra Harbor (f)	12-Jun '98	13.0	M	< 0.12	1.62	< 0.06	< 0.28	2.40	0.018	< 0.26	< 0.57	0.19	11.3	75
				L	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Ctenochaetus striatus	Agat Marina	22-Jan '99	12.5	M	< 0.21	5.17	< 0.11	< 0.34	0.51	0.003	< 0.32	< 0.65	0.07	11.8	75
				L	< 1.41	0.15	1.67	< 2.19	7.07	0.478	< 2.19	< 4.24	nd	192	wet
Epibulus insidiator	Apra Harbor (c)	3-Jun '98	24.5	M	<0.07	5.13	<0.04	<0.17	2.97	0.361	<0.16	<0.36	0.10	14.2	77
				L	<0.28	3.10	0.20	<0.38	7.97	0.758	<0.43	<0.73	1.39	42.9	52
Epibulus insidiator	Apra Harbor (e)	12-Jun '98	16.0	M	< 0.08	5.38	< 0.04	< 0.20	2.67	0.177	< 0.19	< 0.41	0.06	11.2	78
				L	< 0.38	1.66	0.21	< 0.52	11.6	0.308	< 0.59	< 1.01	0.83	73.3	57
Epinephelus merra	Merizo Pier	22-Dec '98	24	M	< 0.09	4.03	< 0.04	< 0.13	0.37	0.116	< 0.19	< 0.35	< 0.01	13.8	76
				L	< 1.04	0.93	2.74	< 1.61	5.96	0.761	< 1.61	< 3.12	nd	53.3	wet
Gerres argyreus	Agana Boat Basin	30-Dec '98	24	M	< 0.11	7.30	< 0.04	0.58	0.33	0.116	< 0.22	< 0.41	< 0.01	34.9	74
				L	< 0.40	2.03	0.66	< 0.62	5.42	0.110	< 0.66	< 1.10	0.21	52.8	75
Gerres argyreus	Agana Boat Basin	30-Dec '98	15.5	M	< 0.18	5.68	< 0.07	< 0.25	0.52	0.082	< 0.38	< 0.70	< 0.01	48.9	79
				L	< 3.67	2.74	< 1.88	< 5.66	3.00	0.119	< 5.66	< 11.0	nd	73.0	wet
Gerres argyreus	Apra Harbor (d)	9-Jun '98	16.5	M	< 0.11	15.9	< 0.06	< 0.26	1.48	0.154	< 0.25	< 0.54	0.17	34.2	77
				L	4.09	3.35	1.00	< 1.36	8.27	0.105	< 1.54	< 2.63	2.20	127	57
Gerres argyreus	Apra Harbor (d)	9-Jun '98	15.0	M	< 0.15	8.00	< 0.07	< 0.35	1.74	0.056	< 0.33	< 0.71	0.11	31.8	80
				L	< 3.26	0.99	< 1.30	< 4.56	< 3.59	0.101	< 6.84	< 12.7	nd	52.5	wet
Gerres argyreus	Apra Harbor (d)	9-Jun '98	14.5	M	< 0.14	4.17	< 0.07	< 0.32	0.93	0.104	< 0.30	< 0.66	0.11	25.1	80
				L	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Gymnothorax javanicus	Apra Harbor (b)	5-Jun '98	60.0	M	< 0.08	4.25	< 0.04	< 0.19	0.70	0.580	< 0.18	< 0.39	0.12	31.7	79
				L	< 0.15	4.38	0.17	< 0.21	16.9	0.426	< 0.24	< 0.41	0.71	88.7	74

M = muscle tissue; L = liver tissue; * = Hg concentrations as µg/g wet weight; nd = no data; wet = analysis performed on wet tissue

Table 15 (cont.)

Heavy Metals in Tissues of Fish From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_20
Letognathus equulus	Agat Marina	22-Jan '99	14	M	< 0.18	1.39	< 0.10	< 0.29	0.91	0.029	< 0.27	< 0.55	0.04	24.3	78
				L	< 2.26	0.55	< 1.16	< 3.49	2.46	0.055	< 3.49	< 6.77	nd	30 0	wet
Lethrinus rubrioperculatus	Agat Marina	21-Dec '98	24.5	M	< 0.12	1.25	< 0.05	0.225	0.42	0.214	< 0.25	< 0.46	< 0.01	11.7	74
				L	< 0.28	1.41	1.90	< 0.44	9.13	0.190	< 0.46	< 0.77	0.02	52.6	51
Lethrinus rubrioperculatus	Merizo Pier	22-Dec '98	20.5	M	< 0.10	16.8	< 0.04	< 0.15	0.50	0.042	< 0.22	< 0.41	< 0.01	13.1	75
				L	< 0.16	7.04	0.87	< 0.25	71.7	0.086	< 0.27	< 0.45	< 0.01	375	61
Lutjanus kasmira	Merizo Pier	22-Dec '98	13.5	M	< 0 08	6.98	< 0.13	< 0.47	0.77	0.025	< 0.70	< 1.30	< 0 01	14.7	81
				L	2.32	18.2	0.83	< 1.58	6.86	0.122	< 1.67	< 2.78	0.11	61.2	45
Monodactylus argenteus	Agana Boat Basin	18-Dec '98	14.5	M	< 0.13	5.83	< 0.05	< 0.18	0.80	0.042	< 0.26	< 0.49	< 0.01	22.2	75
				L	< 0.45	2.20	0.53	< 0.69	8.29	0.196	< 0.73	< 1.22	0.09	69.8	32
Monodactylus argenteus	Apra Harbor (d)	9-Jun '98	17.8	M	< 0.09	7.21	< 0.04	< 0.21	1.55	0.253	< 0.19	< 0.42	0.16	18.9	76
				L	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Monodactylus argenteus	Apra Harbor (d)	9-Jun '98	17.0	M	< 0.07	17.7	< 0 03	< 0.16	1.77	0.195	< 0.15	< 0.32	0.07	24.7	77
				L	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Monodactylus argenteus	Apra Harbor (d)	9-Jun '98	17.0	M	< 0 09	21.3	< 0 04	< 0.21	1.09	0.284	< 0.20	< 0.42	0.03	24.8	70
				L	5.14	9.45	0.67	< 0.15	2.57	0.123	< 0.17	< 0.29	0.60	28.2	41
Monodactylus argenteus	Apra Harbor (d)	9-Jun '98	17.0	M	< 0.07	5.10	< 0.04	< 0.17	1.63	0.265	< 0.16	< 0.35	0.20	16.4	74
				L	0.85	2.52	4.15	< 0.75	6.05	0.084	< 0.85	< 1.45	2.37	75.0	49
Monodactylus argenteus	Apra Harbor (d)	9-Jun '98	16.8	M	< 0.11	13.7	< 0.05	< 0.25	2.72	0.180	< 0.24	< 0.52	0.11	25 0	74
				L	1.31	12.4	1.38	< 0.28	3.28	0.097	< 0.32	< 0.55	0.38	39.8	53
Monodactylus argenteus	Apra Harbor (d)	9-Jun '98	16.5	M	< 0.07	10.9	< 0.03	< 0.17	0.93	0.135	< 0.16	< 0.34	0.13	16.1	74
				L	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Naso annulatus	Apra Harbor (e)	12-Jun '98	13.5	M	< 0.22	1.64	< 0.11	< 0.52	7.76	0.018	< 0.49	< 1.07	0.28	26.1	81
				L	< 1.75	0.36	< 0.70	< 2.45	9.44	0.084	< 3.67	< 6.28	nd	35.8	wet
Naso unicornis	Apra Harbor (a)	5-Jun '98	18.5	M	< 0.07	0.87	< 0.04	< 0.17	2.11	0.015	< 0.16	< 0.35	0.13	13.3	81
				L	2.16	3.89	0.89	< 0.48	337	0.071	< 0.54	< 0.92	1.27	12.4	79
Naso unicornis	Apra Harbor (b)	5-Jun '98	25.0	M	< 0.07	2.50	< 0.04	< 0.17	1.33	0 012	< 0.16	< 0.34	0.26	20.6	79
				L	2.43	5.89	1.97	< 0.26	1920	0.085	< 0.29	< 0.50	1.40	219	75

M = muscle tissue; L = liver tissue; * = Hg concentrations as µg/g wet weight; nd = no data; wet = analysis performed on wet tissue

Table 15 (cont.)

Heavy Metals in Tissues of Fish From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_20
Odemus niger	Agat Marina	22-Jan '99	17	M	< 0.12	47.3	< 0.07	< 0.20	0.75	0.027	< 0.19	< 0.38	< 0.01	15.7	78
				L	< 0.23	26.4	60.4	< 0.38	40.3	0.197	< 0.36	< 0.72	0.06	438	58
Parupeneus barberinus	Merizo Pier	22-Dec '98	26	M	0.20	15.5	< 0 04	< 0.15	0.47	0.066	< 0.22	< 0.42	< 0.01	10.1	76
				L	< 0.23	18.4	2.87	< 0.35	33.8	0.042	< 0.37	3.85	< 0.01	108	74
Parupeneus barberinus	Merizo Pier	22-Dec '98	16	M	< 0.16	33.9	< 0.07	< 0.23	0.41	0.062	< 0.35	< 0.64	< 0.01	11.1	76
				L	< 2.63	9.78	< 1.35	< 4.07	4.55	0.057	< 4.07	< 7.90	nd	25.2	wet
Parupeneus cyclostomus	Merizo Pier	22-Dec '98	25	M	< 0.11	15.7	< 0.04	< 0.16	0.58	0.063	< 0.23	< 0.43	< 0.01	9.55	77
				L	1.28	4.92	0.94	< 0.42	17.4	0.068	< 0.44	< 0.74	< 0.01	65.6	70
Parupeneus multifasciatus	Merizo Pier	22-Dec '98	17.5	M	< 0.11	77.6	< 0.04	< 0.15	0.65	0.061	< 0.22	< 0.42	< 0.01	12.7	88
				L	< 1.39	13.04	< 0.71	< 2.15	3.80	0.109	< 2.15	< 4.18	nd	24.8	wet
Saurida gracilis	Agana Boat Basin	30-Dec '98	23	M	< 0.10	2.80	< 0.04	0.239	0.43	0.099	< 0.21	< 0.39	< 0.01	16.9	74
				L	1.00	0.69	0.22	< 0.46	33.4	0.143	< 0.49	< 0.81	0.19	133	53
Saurida gracilis	Agana Boat Basin	30-Dec '98	19.5	M	< 0.14	10.8	< 0.06	< 0.19	0.47	0.025	< 0.29	< 0.54	< 0.01	12.6	74
				L	< 1.39	9.52	< 0.71	< 2.15	65.1	0.053	< 2.15	< 4.18	nd	116	wet
Saurida gracilis	Agana Boat Basin	30-Dec '98	16.5	M	<0.23	9.44	<0 09	<0.33	0.4	0.024	<0.49	<0.92	< 0.01	13	79
				L	< 2 21	9.14	< 1.13	< 3.41	41.7	0.048	< 3.41	< 6.62	nd	38.3	wet
Saurida gracilis	Agana Boat Basin	30-Dec '98	15.5	M	< 0.21	8.23	< 0.08	< 0.29	0.33	0.034	< 0.44	< 0 81	< 0.01	11.4	73
				L	< 3.00	9.25	< 1.54	< 4.64	64.4	0.041	< 4.64	< 9.00	nd	57.5	wet
Saurida gracilis	Agat Marina	31-Dec '98	20	M	< 0.11	10.3	< 0.04	< 0.15	0.40	0.027	< 0.23	< 0.42	< 0.01	13.1	76
				L	< 0.40	7.47	0.29	< 0.61	89.8	0.637	< 0.65	< 1.08	0.55	212	50
Saurida gracilis	Agat Marina	31-Dec '98	19	M	< 0 11	14.2	0 05	< 0.15	0.29	0.027	< 0.23	< 0.42	< 0.01	11.5	76
				L	< 1 48	11.7	0.94	< 2.29	30 0	0.052	< 2.29	< 4.44	nd	43 0	wet
Saurida gracilis	Agat Marina	31-Dec '98	17.5	M	< 0.19	12.0	< 0.08	< 0.27	0.47	0.017	< 0.41	< 0.76	< 0.01	12.1	73
				L	< 1.34	7.96	< 0.69	< 2.08	39.9	0.018	< 2.08	< 4.03	nd	39.2	wet
Saurida nebulosa	Apra Harbor (b)	5-Jun '98	21.5	M	< 0 07	1.20	< 0 04	< 0.18	0.79	1.157	< 0.17	< 0.36	0.18	11.3	79
				L	< 1.09	0.14	< 0.43	2.50	67.5	0.556	< 2.28	< 4.23	nd	54.1	wet
Saurida nebulosa	Merizo Pier	22-Dec '98	16.5	M	< 0.20	7.12	< 0.08	0.480	0.54	0.011	< 0.42	< 0.78	< 0.01	12.5	68
				L	< 2.30	1.78	< 1.18	< 3.56	51.7	0.012	< 3.56	< 6.91	nd	43.9	wet

M = muscle tissue; L = liver tissue; * = Hg concentrations as µg/g wet weight; nd = no data; wet = analysis performed on wet tissue

Table 15 (cont.)

Heavy Metals in Tissues of Fish From Guam Harbor Waters (data as µg/g dry wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	Ag	As	Cd	Cr	Cu	Hg*	Ni	Pb	Sn	Zn	% H_2O
Scarus sordidus	Apra Harbor (e)	12-Jun '98	16.0	M	< 0.11	0.80	< 0.06	< 0.27	4.89	0.021	< 0.25	< 0.55	0.14	10.4	78
				L	< 0.38	1.34	0.21	< 0.52	3.13	0.072046	< 0.58	< 1.00	0.31	22.0	38
Scarus sordidus	Apra Harbor (e)	9-Jun '98	15.0	M	< 0.10	0.88	< 0.05	< 0.23	2.37	0.019	< 0.22	< 0.48	0.12	10.9	78
				L	0.51	1.53	0.13	< 0.38	5.01	0.024	< 0.43	< 0.74	0.28	30.7	37
Scarus sordidus	Apra Harbor (e)	12-Jun '98	14.0	M	< 0.13	0.92	< 0.07	< 0.31	1.80	0.024	< 0.29	< 0.64	0.11	10.7	78
				L	< 0.23	1.75	0.13	< 0.31	3.56	0.036	< 0.36	< 0.61	0.18	29.3	38
Siganus spinus	Agana Boat Basin	18-Dec '98	15	M	< 0.14	1.37	< 0.06	< 0.20	0.32	0.009	< 0.29	< 0.55	< 0.01	10.3	77
				L	1.70	0.41	0.28	< 0.96	188	0.010	< 1.02	< 1.70	0.08	167	21
Sufflamen chrysoptera	Apra Harbor (e)	12-Jun '98	17.0	M	< 0.14	18.4	< 0.07	< 0.33	1.65	0.226	< 0.31	< 0.67	0.25	27.5	80
				L	< 0.18	13.5	0.33	< 0.24	1.61	0.227	< 0.28	< 0.47	0.23	78.2	37
Valamugil engeli	Apra Harbor (c)	3-Jun '98	37.5	M	< 0.08	1.45	< 0.04	< 0.18	1.35	0.027	< 0.17	< 0.37	0.15	10.5	77
				L	< 0.20	5.97	0.34	< 0.27	73.1	0.064	< 0.30	< 0.52	0.92	187	74

M = muscle tissue; L = liver tissue; * = Hg concentrations as µg/g wet weight; nd = no data; wet = analysis performed on wet tissue

2. PCBs in Harbor Biota

PCBs consist of 209 theoretically possible congeners having different toxic and biologic responses. Approximately half this number accounts for almost all of the environmental contamination attributable to PCBs. Based on potential toxicity, environmental prevalence and abundance in animal tissues, the number of environmentally threatening PCBs reduces to about 36 (McFarland and Clarke 1989).

The aqueous solubilities of individual PCBs range from 1-5 mg/l for monochlorobiphenyls to low-µg/l, or less, for the more highly chlorinated congeners (Opperhuizen *et al.* 1988, Patil 1991). However, it is most unlikely that these solubility limits would ever be approached in natural waters, even in highly contaminated environments, because of the hydrophobic nature of PCBs coupled with their high affinity for suspended particulates, sediments, and biota.

PCBs are ubiquitous contaminants and occur in all environmental compartments. Levels in open ocean waters are highly variable with reported levels ranging from <2-6 pg/l in the Arctic Ocean (Hargrave *et al.* 1992), up to 590 pg/l in the northwestern Pacific Ocean (Tanabe *et al.* 1984). PCB concentrations in marine coastal waters that are distanced from potential sources of local contamination are normally in the low ng/l range (Niimi 1996). The highest waterborne concentrations of PCB occur near point-source discharges, with concentrations in the range of 50-500 ng/l (Tanabe *et al.* 1989, El-Gendy *et al.* 1991).

World baseline levels for PCBs in clean coastal sediments are <1 ng/g whereas, in heavily contaminated environments, levels as high as 61,000 ng/g have been reported (Nisbet 1976). PCB concentrations (based on a 20-congener calibration standard) in Guam harbor sediments were previously found to range from <1 ng/g at Agat Marina, up to 549 ng/g at the western end of Commercial Port, in Apra Harbor. Localized pockets of PCB contamination were also encountered here, in sediments from Hotel Wharf (162 ng/g) and Dry Dock Island (153 ng/g). Long *et al.* (1995) estimated that adverse biological effects frequently occur in biota exposed to sedimentary PCB levels exceeding 180 ng/g. Thus, there are discrete areas of PCB contamination in Apra Harbor sediments that are of environmental concern.

Outside the Apra Harbor area, the highest PCB concentration was found in sediments from the inner Agana Boat Basin area (64 ng/g). Elsewhere, levels encountered were mostly below 10 ng/g (Denton *et al.* 1997).

Tables 16-22 summarizes the PCB data found in biota during the present study. Each table presents concentrations found at 9 levels of chlorination (PCB homologues Cl_2-Cl_{10}) within each group of organisms. These values were derived using the 20-congener standard mix described earlier, and were summed to provide total congener estimates (Σ_{20}PCB). If no congeners were detected then all estimates were set to zero.

Where possible, the data are discussed below with reference to PCB levels found in the same or related species from elsewhere in the world. It is noteworthy that a large proportion of the published information centers on edible species of mollusk, crustaceans, and fish. Very little information of this nature exists for the other invertebrate groups considered here. As a

general rule of thumb, however, PCB concentrations in marine organisms from relatively uncontaminated regions are in the low ng/g range

All referenced data included in the following discussions are expressed on a wet weight basis unless indicated otherwise.

2.1 PCBs in Algae:

Σ_{20}PCB concentrations determined in *Padina* sp. during the current work ranged from non-detectable to 1.85 ng/g (Table 16). In all cases, the lower chlorinated homologues (Cl_2-Cl_4) predominated. Amico *et al.* (1982) noted similar findings with macrophytes from Sicilian waters. They suggested that the inability of algae to metabolize the lower chlorinated PCB congeners was the primary reason for this. There are, of course, other equally plausible explanations. For example, since algae derive PCBs from the water column by direct partitioning, it seems reasonable to assume that the lower chlorinated PCBs would be preferentially accumulated over their higher chlorinated counterparts by virtue of their higher water solubilities and, hence, greater abundance in the hydrosphere.

Macroalgae have been used very little as bioindicators of PCBs, compared with their frequent use for studies of trace metals (Phillips 1986a). The reasons for this are not entirely clear because the group, as a whole, demonstrates a marked bioaccumulation capacity for PCBs and possess no apparent regulatory mechanisms for these compounds. One of the best known studies supporting this group's bioindicator potential is that of Amico and co-workers cited above. In this study, PCBs were measured in a variety of seaweeds from the east coast of Sicily. Concentrations ranged from 37-591 ng/g dry weight (~4-60 ng/g on a wet weight basis) and there were no major differences between the taxonomic groups studied. The highest concentrations were found in algae from an area near Syracuse that was allegedly polluted by nearby industrial activity (Amico *et al.* 1982). Pavoni *et al.* (1990) conducted a similar study on seaweeds in the Lagoon of Venice, in the Adriatic Sea, and reported PCB levels ranging from 13-120 µg/g dry weight. Levels encountered in both of these studies are appreciably higher than we found here in *Padina* sp.

More recently, Hope *et al.* (in press) monitored the same 20 congeners as we did in a range of biota from Midway Atoll, a national wildlife refuge, in the north Pacific. An overall average Σ_{20}PCB concentration of 44.6 ng/g dry weight was given for the brown alga, *Dictyota acutiloba.* This translates to ~4.5 ng/g wet weight and is a little over twice the highest Σ_{20}PCB concentration given here for *Padina* sp. In the same paper, Hope and colleagues reported Σ_{20}PCB levels in sediments of 1-2 ng/g, indicative of a relatively clean environment.

2.2 PCBs in Sponges:

Remarkably high Σ_{20}PCB concentrations of 712-9,740 ng/g were found in the sponge *Dysidea* sp. from Apra Harbor (sites c, d and f). This particular sponge has a lipid content of around 20-30%, which is at least an order of magnitude higher than most other invertebrate species. Thus, a high bioaccumulation capacity for PCBs and other lipophilic substances is not altogether unexpected. Nevertheless, it would be interesting to expand the database for *Dysidea* sp. and include representatives from more remote locations.

Residue profiles for *Dysidea* are shown in Fig. 6 and are dominated by Cl_4-Cl_7 homologues. This isomeric group is found in high proportions in the commercial PCB mixture Aroclor 1254 (Hutzinger *et al.* 1974, Brownawell and Farrington 1986). The data therefore implies the existence of one or more point sources of PCB in waters bounded by the Shell Fox-1 Fuel Pier (site c), Commercial Port (side d), and Echo Wharf (site f). The data obtained earlier with sediments, certainly support this conclusion (Denton *et al.* 1997).

Σ_{20}PCB concentrations in all other species of sponge examined, although generally high, were less than 100 ng/g (Table 17). No comparable data for sponges were found in the literature at the time of writing this report. Clearly, sponges are very responsive to ambient changes in PCB concentrations and further work should be directed towards their use as bioindicators of these compounds.

2.3 PCBs in Soft Corals:

Soft corals, like sponges, are rich in triglycerides and also demonstrate a high accumulation capacity for PCBs. Σ_{20}PCB concentrations in *Simularia* sp. ranged from a low of 3.72 ng/g, at Agat Marina, to a high of 4,103 ng/g at site c, in Apra Harbor. The latter value confirms the occurrence of elevated PCB concentrations in the vicinity of the Shell Fox-1 Fuel Pier. Residues in *Simularia* sp. from this site were dominated by the mid-range homologues common to Aroclor 1254 (Fig. 7). No comparable data for soft corals were found in the literature at the time of writing this report.

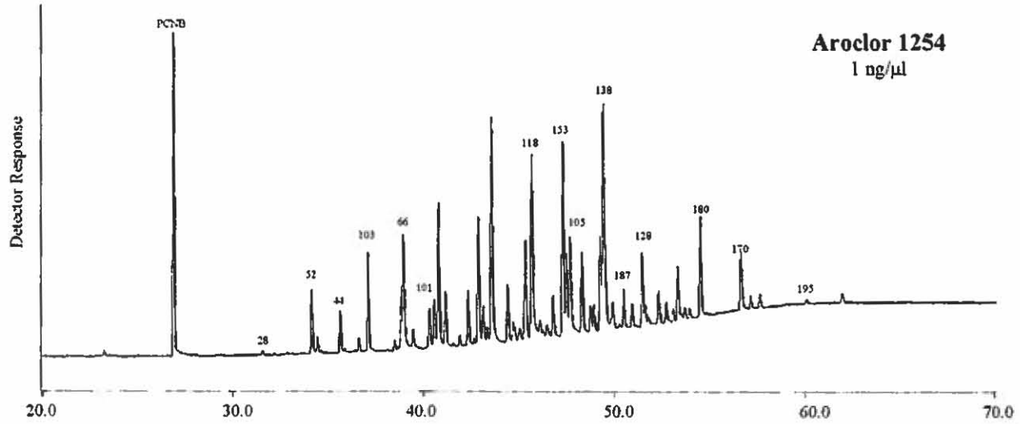
2.4 PCBs in Sea Cucumbers:

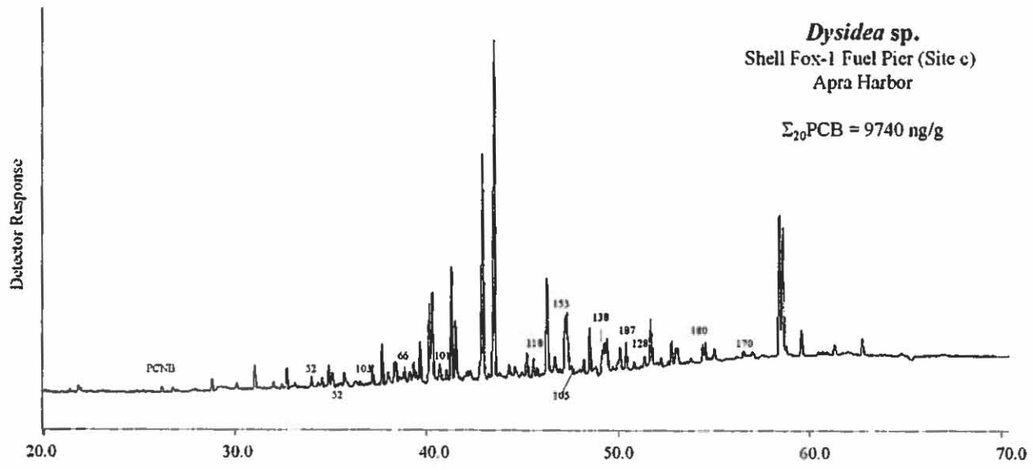
The current work revealed that PCBs in sea cucumbers are tissue dependent and appreciably more concentrated in the hemal system than the body wall muscle (Table 18). In *Bohadschia argus,* for example, Σ_{20}PCB concentrations ranged from 0.03-12.8 ng/g in muscle, compared with 0.28-66.5 ng/g in the hemal system. Overall, levels in both tissues were highest in the Apra Harbor specimens and were dominated by Cl_4-Cl_7 homologues (Fig. 8). Comparable ranges were determined in *Holothuria atra,* apart from a very high value of 1279 ng/g in the hemal system of one specimen from Merizo Pier.

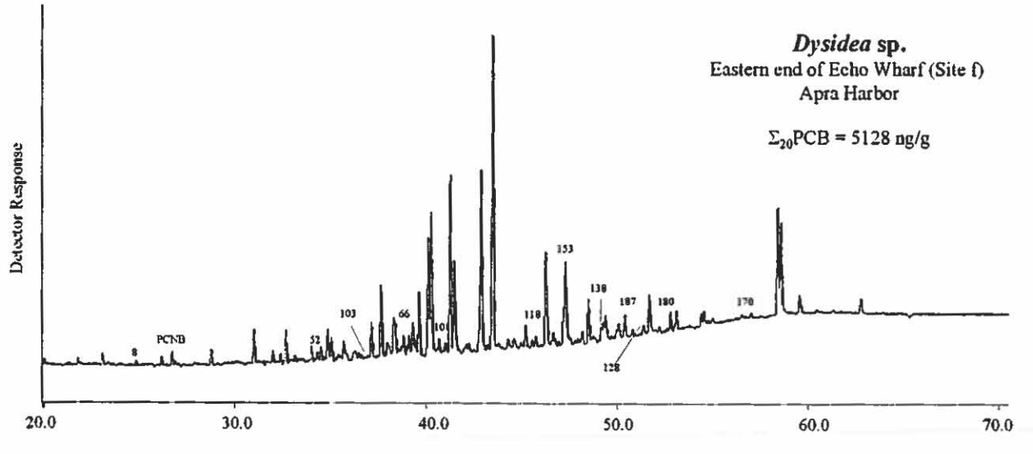
Very little attention has been focused on echinoderms as indicators of PCBs. Everaarts *et al.* (1998) measured levels of 7 chlorobiphenyls in an unnamed brittle star, from the east coast of Africa, and reported Σ_7PCB concentrations of 0.07-0.15 ng/g. Bright *et al.* (1995) considered several Arctic invertebrates to monitor 47 PCB congeners in biota from Cambridge Bay, NWT. Apparently the bay received local sources of PCBs in runoff from contaminated terrestrial sites. Σ_{47}PCB concentrations measured in sea urchins by these authors ranged from <1.0-210 ng/g.

Hope and co-workers looked at PCBs in *Bohadschia obesus* and *Holothuria atra* from Midway Atoll and are the only other investigators known to have examined PCBs in sea cucumbers from the Pacific. Σ_{20}PCB estimates derived from their data were 183 and 9.36 ng/g dry weight (~37 and 2 ng/g on a wet weight basis) for each species respectively (Hope *et al.* in press). Allowing for the fact that analysis was conducted on whole specimens, these values compare reasonably well with those determined by us during the current study.

Figure 6. Polychlorinated Biphenyls in Sponges from Apra Harbor

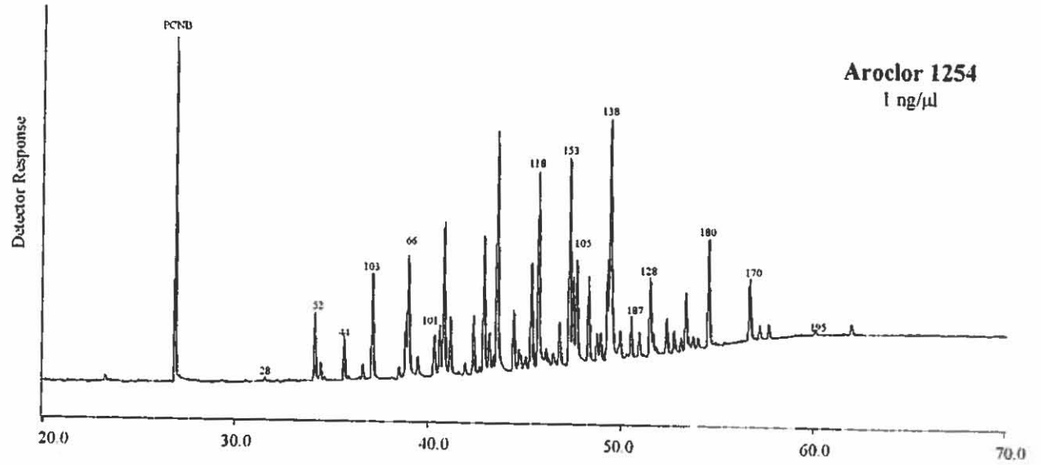


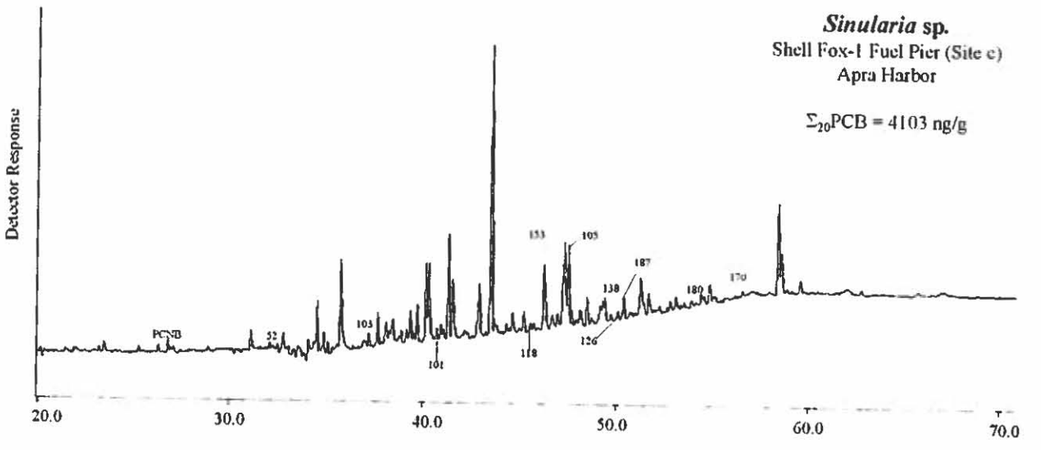


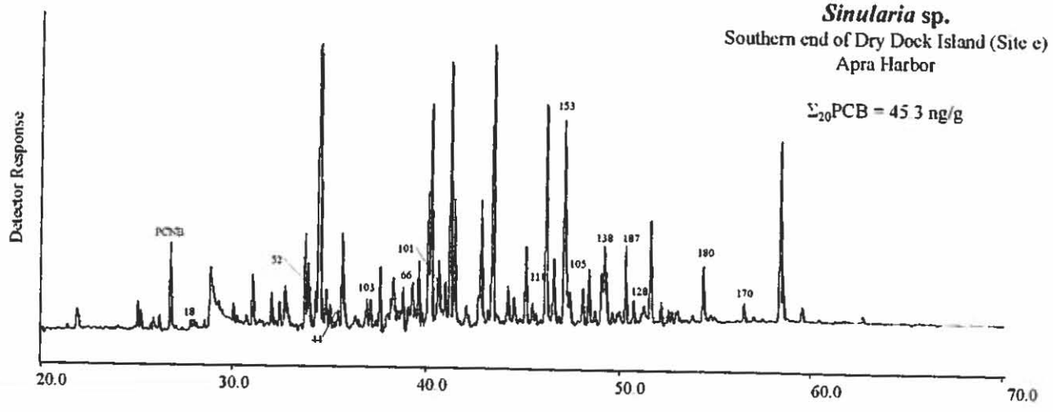


Time (min)

Figure 7. Polychlorinated Biphenyls in Soft Corals from Apra Harbor

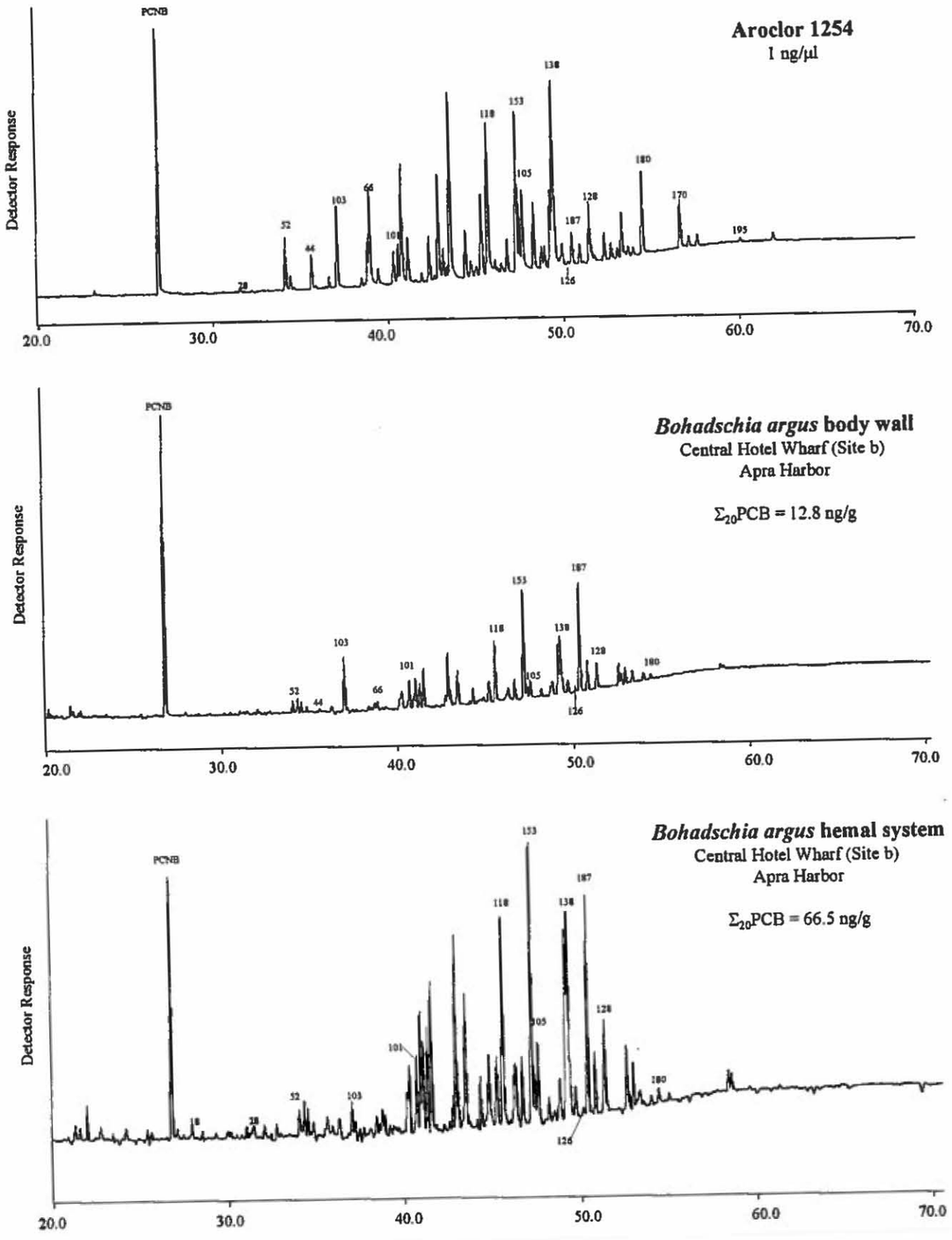






Time (min)

Figure 8. Polychlorinated Biphenyls in Sea Cucumber from Apra Harbor



2.5 PCBs in Mollusk:

Next to fish, bivalve mollusks are the most commonly used indicators of PCBs in aquatic environments (Phillips 1980). Both the U.S. National Status and Trends (NS&T) program and the International 'Mussel Watch' (IMW) program center on the use of mussels and oysters for monitoring PCBs and other contaminants in aquatic environments.

The NS&T program collects bivalves annually from numerous sites on the Atlantic, Pacific and Gulf coasts of the U.S., including Alaska, the Hawaiian Islands, and Puerto Rico. According to a recent report by Sericano *et al.* (1995), PCBs have been detected in all oyster samples since the program began in 1986. Average concentrations up to 1993, ranged from 100-630 ng/g dry weight at 15 sites, and from 10-100 ng/g dry weight at all the rest. Total PCB levels exceeding 1,000 ng/g dry weight have been reported in oysters from two IMW sampling locations in South America (Sericano *et al.* 1995). It should be mentioned here, that the NS&T criteria for estimating 'total' PCB is twice the sum of all detectable chlorobiphenyls of an 18-congener calibration standard (O'Connor 1998).

In the present study, Σ_{20}PCB concentrations in oysters ranged from a low 1-2 ng/g at Agat Marina and Merizo Pier, to a high of 47 ng/g in one specimen from Dry Dock Island (site e) in Apra Harbor (Table 19). Σ_{20}PCB levels of 10-15 ng/g were present in pooled oyster composites from beneath the Shell Fox-1 Fuel Pier (site c) as well as from Agana Boat Basin. Concentration differences between oyster composites revealed within-site variability factors of 3.2, 1.4 and 6.5 at Apra Harbor sites a, e, and f. Geometric mean Σ_{20}PCB concentrations in oysters at these sites were calculated at 4.6, 39.8, and 7.42 ng/g respectively. The relatively high levels determined in oysters from Dry Dock Island support our earlier findings of PCB enrichment in the sediments from around this area (Denton *et al.* 1997).

No comparative data were found for PCBs in chamids or spondylids outside of this study. From the limited data presented here, it appears that chamids have a lower affinity for PCBs than oysters. In contrast, spondylids and oysters seem to demonstrate similar accumulation capacities for these compounds and both highlight PCB-enrichment in the Dry Dock Island area (Table 20).

Limited data exists for PCBs in cephalopods. Kawano *et al.* (1986) determined up to 17 ng/g (as Aroclor 1254) in whole squid from the Pacific Ocean and Bering Sea, while Everaarts *et al.* (1998) reported a mean Σ_7PCB concentration of 3.0 ng/g for cuttlefish (*Sepia* sp.) from east African waters. In an earlier study, Monod *et al.* (1995) examined 6 chlorobiphenyls in octopus from Saint Paul and Amsterdam Islands, in the central southern Indian Ocean, and reported low Σ_6PCB concentrations of 8.1-19.2 ng/g dry weight. This is about 2-4 ng/g wet weight, assuming octopus is 80% water. The Σ_{20}PCB concentration determined in tentacles of octopus from Apra Harbor during the current study was 8.78 ng/g (Table 21). Interestingly, the 6 congeners that Monod and co-workers focused on accounted for almost 70% of total residues quantified.

The very high Σ_{20}PCB levels in the liver of the Apra Harbor octopus (1271 ng/g) no doubt reflects the high fat content of this tissue and, hence, its ability to store relatively high concentrations of lipophilic xenobiotics like PCBs.

2.6 PCBs in Crustaceans:

Crustaceans are a comparatively well worked group in terms of their PCB content and are frequently incorporated into marine pollution monitoring programs. While some notable PCB levels have been documented in representatives of this group, metabolic transformations of some of the lower chlorinated congeners has been demonstrated in certain members, and this could account for some of the large residue differences often observed between species (Porte and Albaigés 1993). For example, shrimp (*Parapenaeus longirostris*) sampled throughout the Mediterranean contained PCBs in muscle tissue that rarely exceeded concentrations of 30 ng/g. In contrast, mean levels reported for crabs (*Carcinus mediterraneus*) from the same sites were as high as 1,448 ng/g (Fowler 1987). As a general rule, however, PCB levels in shrimp, crabs and lobsters, from relatively uncontaminated waters, usually fall well under 10 ng/g (Monod *et al.* 1995, Everaarts, 1998). Baseline data for PCBs in stomatopod crustaceans from similar environments are currently unavailable, but, in all probability, levels are lower than the value of 38.2 ng/g determined in the tail muscle of mantis shrimp during the current investigation (Table 21).

2.7 PCBs in Ascidians:

Σ_{20}PCB concentrations determined in ascidians from Apra Harbor during the present study were low and ranged from 0.10-3.0 ng/g. Comparable data for ascidians from elsewhere, were not forthcoming at the time of writing this report. However, a total PCB concentration of 49 ng/g dry weight was reported by Contardi *et al.* (1979) for the salp, *Pyrosoma atlanticum*, from the Ligurian Sea. This translates to ~2.5 ng/g on a wet weight basis, assuming 95% water content, and is within the range of values reported here (Table 21).

2.8 PCBs in Fish:

Marine fish are a valuable source of high quality protein to people all over the world. Their importance in this regard has been a primary driving force behind the extensive monitoring of edible species for PCB residues over the last 20 years. In more recent times, the popularity of fish as sentinel organisms for PCBs, has added greatly to the volume of published information that currently exists for this group.

A compilation of the reported data for PCBs in fish muscle is given in Table 6. From these data, it is apparent that the flesh of marine fish from relatively uncontaminated waters usually contains PCBs in the low ng/g range. On the other hand, fish from PCB contaminated environments may contain levels two to three orders of magnitude higher.

PCBs found in fish during the present study are summarized in Table 22. A total of 75 specimens were analyzed of which 40 were from Apra Harbor, 15 from Agana Boat Basin, 8 from Agat Marina, and 12 from Merizo Pier. Σ_{20}PCB concentrations in axial muscle ranged from 0.09-85 ng/g overall. Thirteen fish from Apra Harbor contained levels greater than 20 ng/g. A further 13 fish contained levels between 10 and 20 ng/g and were predominantly from Apra Harbor and Agana Boat Basin. A similar number contained between 5 and 10 ng/g while levels ranging from 1-5 ng/g occurred in 23 fish, with representatives from all four harbors. All the rest had levels of less than 1 ng/g and were exclusively from Agat Marina and Merizo Pier.

Several workers have explored the potential of fish liver as an indicator tissue for PCBs (Marthinsen *et al.* 1991, Pereira *et al.* 1994, and Brown *et al.* 1998). For this reason, the livers of 20 fish were analyzed during the present investigation. In all cases, Σ_{20}PCB concentrations greatly exceeded those found in axial muscle (Table 22). Such differences between the two tissues simply reflect the liver's higher lipid content (>50% in some cases) which greatly enhances its capacity to act as a reservoir for refractory, lipophilic compounds like PCBs.

During the course of the current work, hepatic Σ_{20}PCB concentrations exceeding 10,000 ng/g were found in two fish from Apra Harbor. The first fish, *Caranx melampygus*, a relatively large carnivorous species from Dry Dock Island (site e), contained 17,009 ng/g in its liver. A slightly lower value of 11,346 ng/g was measured in *Monodactylus argentius*, a small omnivorous species captured at the western end of Commercial Port (site d). Chromatograms from both fish were not too far removed from the commercial PCB mixture, Aroclor 1260, as shown in Figs. 9-10. It is noteworthy that PCB profiles resembling this Aroclor were previously identified in sediments from the Dry Dock Island area (Denton *et al.* 1997).

In sharp contrast to the two fish described above, *C. melampygus* taken from the Hotel Wharf area contained PCB residues in its axial muscle that were proportionately similar to Aroclor 1254. Once again, attention is drawn to the fact that we previously observed a PCB signature similar to that of Aroclor 1254 in sediments from around this area. The axial muscle chromatograms of *C. melampygus* from both sites are presented together in Fig. 11 for comparative purposes.

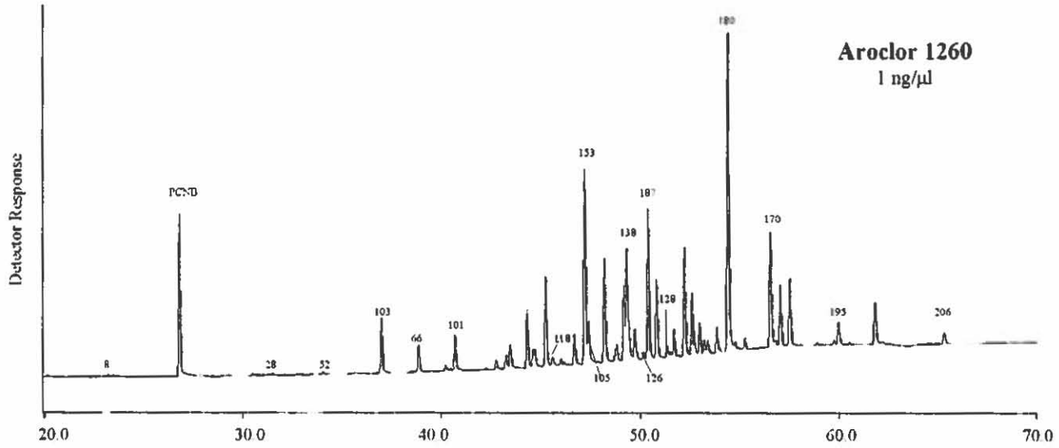
Comparably high hepatic PCB concentrations have been reported by others and, in all instances, were related to elevated environmental levels of these compounds. For example Marthinsen *et al.* (1991) found 6-8,320 ng/g in two fish species from the mouth of the Glomma, the largest river in Norway. Similarly, levels exceeding 10,000 ng/g dry weight were reported by Brown *et al.* (1998) for livers of three species of fish from various locations along the U.S. Pacific coast.

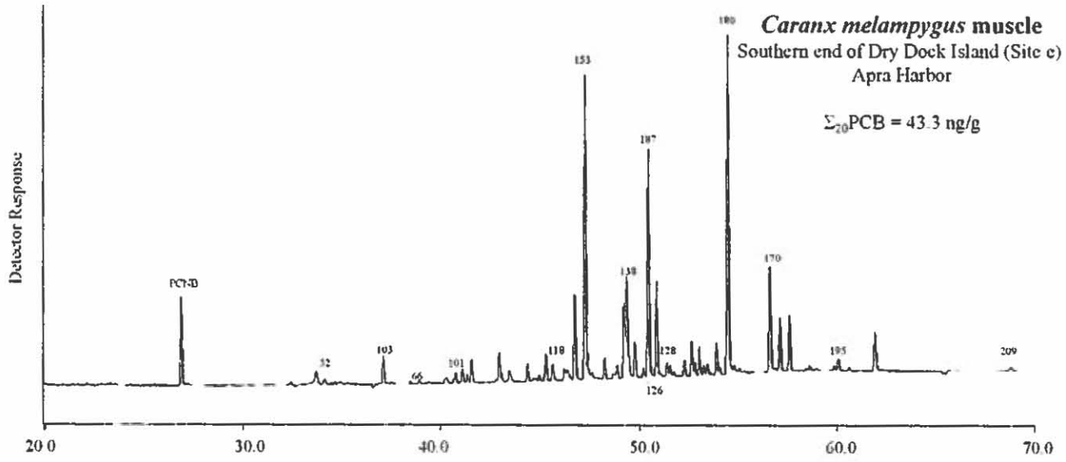
2.9 Concluding Remarks:

From the preceding discussions, it is evident that the PCB-enrichment noted earlier in sediments from certain locations in Apra Harbor is also reflected in the biota. However, a comparative analysis of the data with levels found in similar and related species elsewhere, generally indicates only mild enrichment extending to moderate, in certain species at localized sites in and around the Commercial Port and Dry Dock Island areas.

It is clear, from the literature and from the current work, that PCB concentrations in aquatic organisms can vary by up to a factor of 10^5 depending upon the species, the location and the tissue examined. The wide range of values reported here, especially for organisms from the same site, largely reflects inter-specific differences in lipid content. Species with the highest lipid content can be expected to accumulate the largest amounts of PCBs. Thus, species differences in bioaccumulation capacities appear considerable, when PCB concentrations are determined on wet weight basis; however when based on lipid weights they are far less variable (Phillips 1986a). Future monitoring programs are, therefore, recommended to express the data on both a fresh weight and lipid weight basis.

Figure 9. Polychlorinated Biphenyls in Fish from Apra Harbor





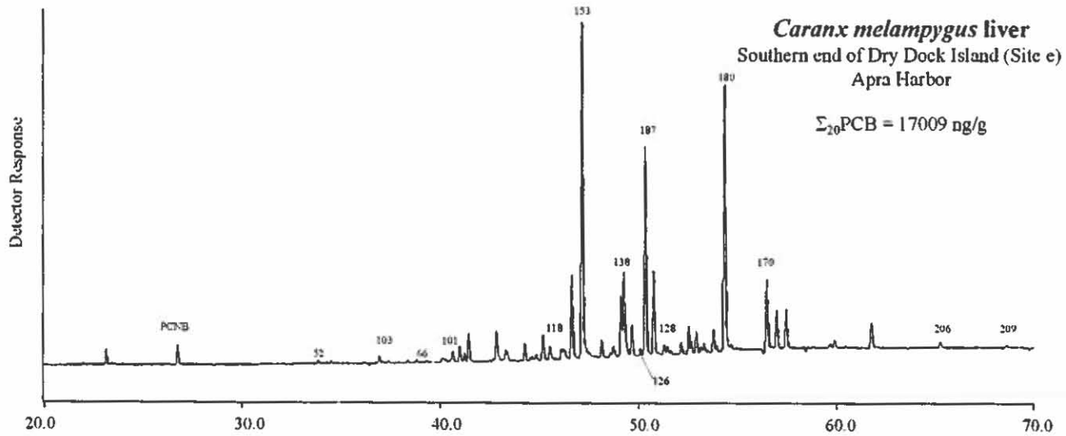
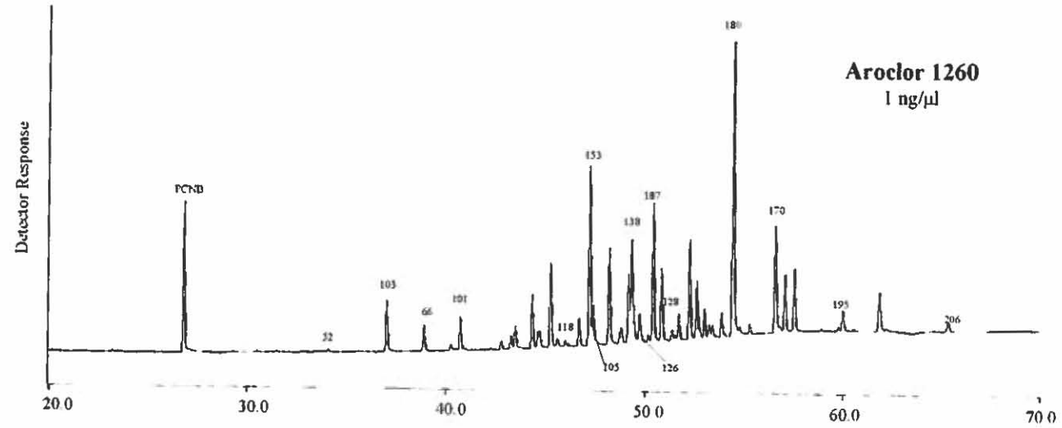
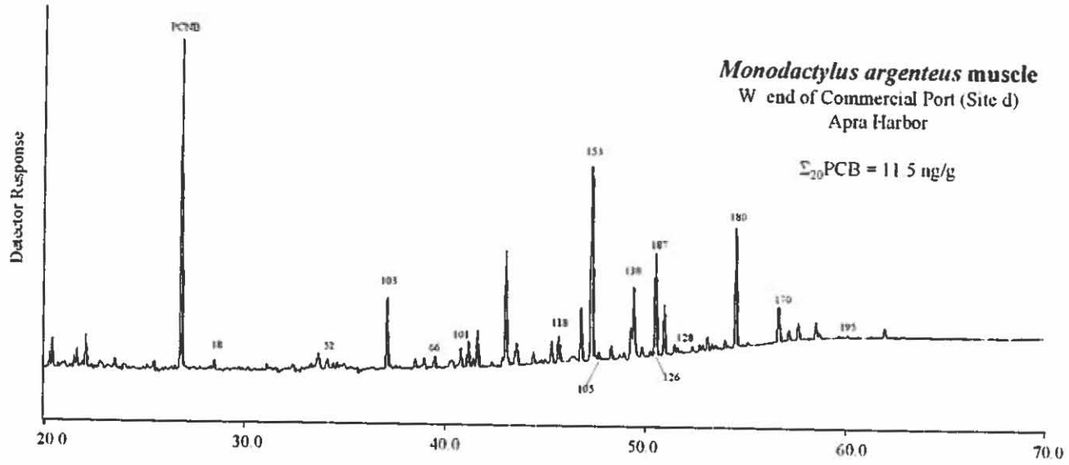


Figure 10. Polychlorinated Biphenyls in Fish from Apra Harbor





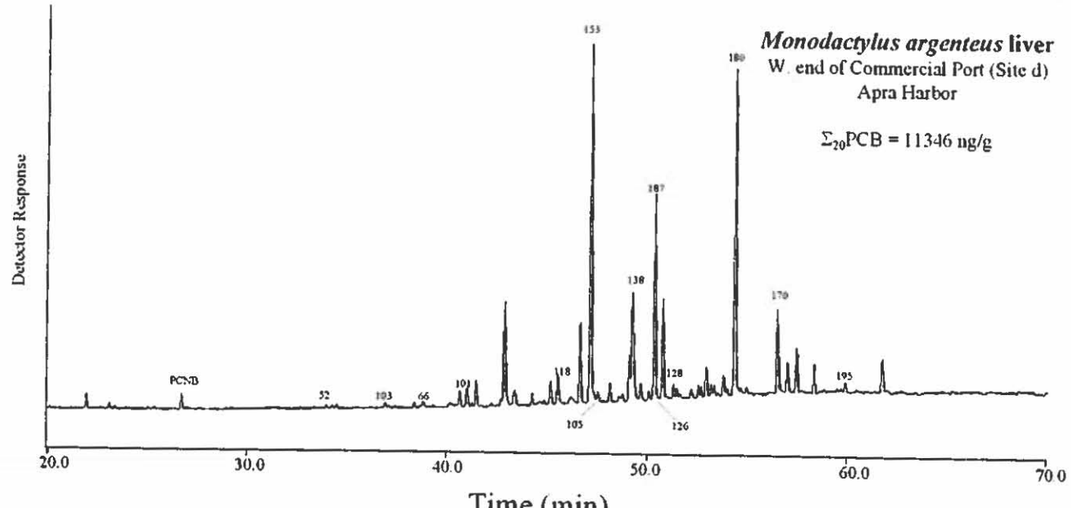


Figure 11. Polychlorinated Biphenyls in Fish from Apra Harbor

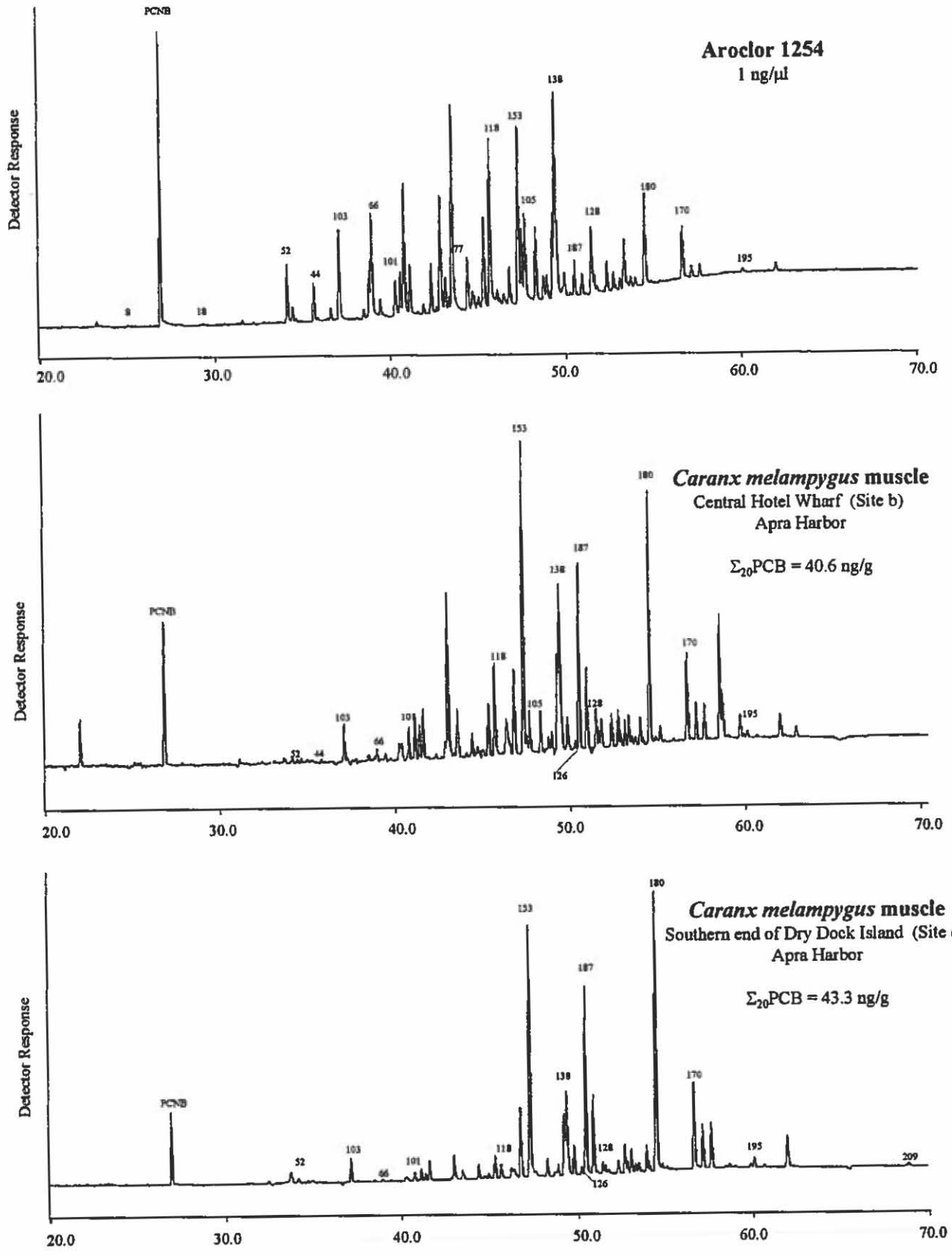


Table 16

PCB Homologues in Seaweed From Guam Harbor Waters (data as ng/g wet wt.)

Species	Location (site)	Date	Cl_2B	Cl_3B	Cl_4B	Cl_5B	Cl_6B	Cl_7B	Cl_8B	Cl_9B	$Cl_{10}B$	$\Sigma_{20}PCB$
Padina sp.	Agana Boat Basin	18-Dec-98	0.69	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.00	0.74
Padina sp.	Apra Harbor (a)	5-Jun-98	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0
Padina sp.	Apra Harbor (c)	3-Jun-98	0.00	0.00	0.00	0.05	0.23	0.16	0.00	0.00	0.00	0.44
Padina sp.	Apra Harbor (d)	9-Jun-98	0.00	0.00	0.00	0.53	0.46	0.30	0.00	0.00	0.00	1.85
Padina sp.	Apra Harbor (e)	9-Jun-98	0.00	0.54	0.56	0.17	0.53	0.57	0.00	0.00	0.00	1.81
Padina sp.	Apra Harbor (f)	12-Jun-98	0.47	0.00	0.00	0.00	0.16	0.13	0.00	0.00	0.00	0.76
Padina sp.	Agat Marina	21-Dec-98	0.39	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.39
Padina sp.	Merizo Pier	22-Dec-98	0.59	0.34	0.00	0.10	0.16	0.07	0.00	0.00	0.00	1.26

Table 17

PCB Homologues in Sponges and Soft Corals From Guam Harbor Waters (data as ng/g wet wt.)

Species	Location (site)	Date	Cl_2B	Cl_3B	Cl_4B	Cl_5B	Cl_6B	Cl_7B	Cl_8B	Cl_9B	$Cl_{10}B$	$\Sigma_{20}PCB$
SPONGES												
Callyspongia diffusa	Agat Marina	21-Dec-98	0.00	0.00	0.53	0.68	0.42	0.41	0.00	0.00	0.00	2.04
Clathria vulpina ?	Agat Marina	21-Dec-98	0.71	0.00	0.00	0.41	0.19	6.67	0.00	0.00	0.00	7.98
Clathria vulpina ?	Merizo Pier	22-Dec-98	0.00	0.00	0.00	0.11	0.22	0.27	0.04	0.00	0.00	0.65
Dysidea sp.	Apra Harbor (c)	3-Jun-98	367	0.00	2096	3126	2497	1632	22.00	0.00	0.00	9740
Dysidea sp.	Apra Harbor (d)	9-Jun-98	7.92	0.00	197	308	146	52.7	0.00	0.00	0.00	712
Dysidea sp.	Apra Harbor (f)	12-Jun-98	610	0.00	1937	1285	605	692	0.00	0.00	0.00	5128
Liosina cf. granularis	Apra Harbor (b)	5-June '98	0.11	0.00	0.91	1.35	3.15	3.53	0.09	0.00	0.17	9.32
Liosina cf. granularis	Apra Harbor (e)	9-Jun-98	0.00	0.12	0.58	2.65	14.9	18.7	0.53	0.10	0.43	38.0
Stylotella aurantium	Apra Harbor (b)	5-Jun-98	0.00	0.00	0.00	2.10	7.78	7.52	0.25	0.00	0.24	17.9
Stylotella aurantium	Apra Harbor (e)	9-Jun-98	0.00	0.00	0.00	3.81	24.7	27.1	0.86	0.00	0.61	57.0
Stylotella aurantium	Merizo Pier	22-Dec-98	1.00	1.10	1.44	0.13	1.12	9.01	0.39	0.00	0.48	14.7
UNIDENTIFIED SPONGES												
Brown Wart Sponge	Apra Harbor (e)	9-Jun-98	0.17	0.16	1.24	1.44	8.05	9.18	0.45	0.00	0.14	20.8
Brown Wart Sponge	Apra Harbor (f)	12-Jun-98	0.13	0.08	1.13	1.38	7.28	7.14	0.13	0.00	0.20	17.5
Orange Wart Sponge	Apra Harbor (e)	9-Jun-98	0.00	0.00	0.00	7.49	42.1	46.6	2.02	0.00	0.62	98.9
Yellow Bread Sponge	Agat Marina	21-Dec-98	0.00	0.00	0.92	1.28	0.53	0.29	0.00	0.00	0.00	3.02
Yellow Sponge (red outside)	Apra Harbor (c)	3-Jun-98	0.24	3.55	5.91	18.1	36.8	22.2	0.26	0.00	0.00	87.1
SOFT CORALS												
Sinularia sp.	Apra Harbor (c)	3-Jun-98	0.00	154	1881	1057	292	720	0.00	0.00	0.00	4103
Sinularia sp.	Apra Harbor (e)	9-Jun-98	0.48	1.73	7.64	10.4	18.1	6.96	0.04	0.00	0.00	45.3
Sinularia sp.	Agana Boat Basin	18-Dec-98	0.25	0.09	0.10	0.12	0.09	3.07	0.00	0.00	0.00	3.72
Sinularia sp.	Merizo Pier	22-Dec-98	2.93	3.39	2.05	0.27	2.88	60.4	0.00	0.77	0.00	72.7

Table 18

PCB Homologues in Sea Cucumbers From Guam Harbor Waters (data as ng/g wet wt.)

Species	Location (site)	Date	Tissue	Cl_2B	Cl_3B	Cl_4B	Cl_5B	Cl_6B	Cl_7B	Cl_8B	Cl_9B	$Cl_{10}B$	$\Sigma_{20}PCB$
Bohadschia argus	Agana Boat Basin	18-Dec-98	M	0.00	0.00	0.00	0.05	0.09	0.12	0.00	0.00	0.00	0.26
			H	0.00	0.00	0.41	2.61	4.66	2.67	0.00	0.00	0.00	10.4
Bohadschia argus	Apra Harbor (b)	5-Jun-98	M	0.00	0.13	1.13	4.67	4.91	2.00	0.00	0.00	0.00	12.8
			H	0.42	0.85	3.78	28.0	25.5	7.92	0.00	0.00	0.00	66.5
Bohadschia argus	Apra Harbor (c)	12-Jun-98	M	0.00	0.00	0.00	0.47	0.99	0.69	0.00	0.00	0.00	2.15
Bohadschia argus	Apra Harbor (e)	9-Jun-98	M	0.00	0.00	0.00	0.43	1.52	1.10	0.00	0.00	0.00	3.05
			H	0.16	0.14	1.24	7.86	32.1	21.5	0.00	0.00	0.00	63.0
Bohadschia argus	Agat Marina	21-Dec-98	M	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.03
			H	0.00	0.00	0.00	0.00	0.15	0.12	0.00	0.00	0.00	0.28
Bohadschia argus	Merizo Pier	22-Dec-98	M	0.00	0.00	0.00	0.26	0.40	0.20	0.00	0.00	0.00	0.86
			H	0.10	0.00	0.32	4.13	3.31	0.86	0.00	0.00	0.00	8.71
Holothuria atra	Agana Boat Basin	18-Dec-98	M	0.00	0.00	0.00	0.07	0.32	0.54	0.00	0.00	0.00	0.94
			H	1.04	0.00	0.20	2.14	6.13	11.9	0.00	0.00	0.00	21.4
Holothuria atra	Apra Harbor (g)	12-Jun-98	M	0.00	0.00	0.00	0.43	1.55	0.79	0.00	0.00	0.00	2.77
			H	0.00	0.00	0.00	1.40	7.16	2.50	0.00	0.00	0.00	11.1
Holothuria atra	Merizo Pier	22-Dec-98	M	0.00	0.00	0.45	4.17	4.90	0.95	0.00	0.00	0.00	10.5
			H	2.60	4.33	25.2	646	597	4.05	0.00	0.00	0.00	1279
Holothuria atra	Apra Harbor (e)	9-Jun-98	M	0.00	0.00	0.39	2.11	9.59	5.49	0.00	0.00	0.00	17.6
			H	0.00	0.00	2.03	2.93	6.02	1.78	0.00	0.00	0.00	12.8
Holothuria atra	Agat Marina	21-Dec-98	M	0.00	0.00	0.00	0.00	0.09	0.17	0.00	0.00	0.00	0.27
Holothuria atra	Agat Marina	21-Dec-98	M	0.00	0.00	0.00	0.00	0.05	0.09	0.00	0.00	0.00	0.14
			H	0.00	0.00	0.00	0.06	0.18	0.22	0.00	0.00	0.00	0.46

M = body wall muscle tissue; H = hemal system

Table 19

PCB Homologues in Bivalve Mollusks From Guam Harbor Waters (data as ng/g wet wt.)

Species	Location (site)	Date	Pool	Cl_2B	Cl_3B	Cl_4B	Cl_5B	Cl_6B	Cl_7B	Cl_8B	Cl_9B	$Cl_{10}B$	$\Sigma_{20}PCB$
OYSTERS													
Saccostrea cucculata	Apra Harbor (c)	5-Jun-98	10	0.00	0.00	0.00	2.69	8.96	2.56	0.00	0.00	0.00	14.2
*Saccostrea cucculata**	Merizo Pier	22-Dec-98	7	0.00	0.00	0.82	0.22	0.12	0.14	0.00	0.00	0.00	1.30
Striostrea mytiloides	Agana Boat Basin	18-Dec-98	4	0.48	0.00	0.95	5.19	6.36	1.75	0.00	0.00	0.00	14.7
Striostrea mytiloides	Apra Harbor (a)	5-Jun-98	2	0.00	0.00	0.00	1.37	5.92	1.24	0.00	0.00	0.00	8.54
Striostrea mytiloides	Apra Harbor (a)	5-Jun-98	2	0.00	0.00	0.00	0.91	3.14	0.73	0.00	0.00	0.00	4.79
Striostrea mytiloides	Apra Harbor (a)	5-Jun-98	1	0.00	0.00	0.00	1.27	3.93	0.77	0.00	0.00	0.00	5.97
Striostrea mytiloides	Apra Harbor (a)	5-Jun-98	2	0.00	0.00	0.00	0.44	1.70	0.41	0.05	0.00	0.00	2.60
Striostrea mytiloides	Apra Harbor (a)	5-Jun-98	5	0.00	0.00	0.00	0.60	2.13	0.49	0.00	0.00	0.00	3.22
Striostrea mytiloides	Apra Harbor (c)	5-Jun-98	1	0.00	0.00	0.00	3.31	5.86	1.11	0.00	0.00	0.00	10.3
Striostrea mytiloides	Apra Harbor (e)	9-Jun-98	4	0.00	0.00	0.00	6.09	20.14	8.29	0.00	0.00	0.00	34.5
Striostrea mytiloides	Apra Harbor (e)	9-Jun-98	6	0.00	0.00	0.00	5.88	26.96	6.00	0.00	0.00	0.00	38.8
Striostrea mytiloides	Apra Harbor (e)	9-Jun-98	1	0.00	0.00	0.00	9.31	30.10	7.52	0.04	0.00	0.03	47.0
Striostrea mytiloides	Apra Harbor (f)	12-Jun-98	3	0.00	0.00	0.00	0.81	6.19	1.15	0.00	0.00	0.00	8.15
Striostrea mytiloides	Apra Harbor (f)	12-Jun-98	1	0.00	0.00	0.00	2.12	10.18	2.19	0.03	0.00	0.00	14.5
Striostrea mytiloides	Apra Harbor (f)	12-Jun-98	1	0.00	0.00	0.00	0.16	1.46	0.53	0.04	0.00	0.04	2.23
Striostrea mytiloides	Apra Harbor (f)	12-Jun-98	1	0.00	0.00	0.00	1.97	7.85	1.71	0.00	0.00	0.00	11.5
Striostrea mytiloides	Agat Marina	21-Dec-98	2	0.00	0.00	0.00	0.08	0.32	0.80	0.00	0.00	0.00	1.20

* juveniles

Table 20

PCB Homologues in Bivalve Mollusks From Guam Harbor Waters (data as ng/g wet wt.)

Species	Location (site)	Date	Pool	Cl_2B	Cl_3B	Cl_4B	Cl_5B	Cl_6B	Cl_7B	Cl_8B	Cl_9B	$Cl_{10}B$	$\Sigma_{20}PCB$
CHAMIDS													
Chama brassica	Apra Harbor (d)	9-Jun-98	2	0.00	0.00	1.04	0.81	1.05	0.46	0.00	0.00	0.00	3.36
Chama brassica	Apra Harbor (f)	12-Jun-98	2	0.00	0.00	0.00	0.07	0.67	0.47	0.00	0.00	0.00	1.21
Chama lazarus	Apra Harbor (b)	5-Jun-98	3	0.00	0.00	0.00	0.07	0.31	0.28	0.00	0.00	0.08	0.66
Chama lazarus	Apra Harbor (b)	5-Jun-98	3	0.00	0.11	0.17	0.46	0.56	0.42	0.08	0.00	0.00	1.87
Chama lazarus	Apra Harbor (c)	5-Jun-98	1	0.00	0.11	0.09	0.19	0.34	0.23	0.00	0.00	0.00	0.95
Chama lazarus	Apra Harbor (c)	5-Jun-98	1	0.18	0.11	0.00	0.15	0.23	0.15	0.00	0.00	0.00	0.82
Chama lazarus	Apra Harbor (d)	9-Jun-98	2	0.15	0.00	0.00	0.24	0.32	0.16	0.00	0.00	0.00	0.88
Chama lazarus	Apra Harbor (d)	9-Jun-98	2	0.00	0.00	0.60	0.40	0.47	0.28	0.05	0.00	0.00	1.78
Chama lazarus	Apra Harbor (d)	9-Jun-98	2	0.00	0.00	0.00	1.72	0.00	0.00	0.00	0.00	0.00	1.72
Chama lazarus	Apra Harbor (e)	9-Jun-98	2	0.00	0.00	0.17	0.15	1.05	0.96	0.03	0.00	0.00	2.36
Chama lazarus	Apra Harbor (e)	9-Jun-98	2	0.00	0.00	0.56	0.82	3.55	2.89	0.10	0.00	0.00	8.0
Chama lazarus	Apra Harbor (f)	12-Jun-98	2	0.11	0.12	0.10	0.08	0.52	0.39	0.00	0.00	0.00	1.32
Chama lazarus	Apra Harbor (f)	12-Jun-98	2	0.10	0.00	0.16	0.16	0.68	0.40	0.00	0.00	0.00	1.50
Chama lazarus	Merizo Pier	22-Dec-98	1	0.00	0.00	0.00	0.00	0.00	0.29	0.00	0.00	0.00	0.29
SPONDYLIDS													
Spondylus ? multimuricatus	Agana Boat Basin	18-Dec-98	2	0.15	1.84	1.14	2.29	4.22	1.64	0.02	0.00	0.00	11.3
Spondylus ? multimuricatus	Apra Harbor (e)	9-Jun-98	1	1.92	2.25	2.05	2.81	12.61	8.73	0.08	0.00	0.08	30.5
Spondylus ? multimuricatus	Apra Harbor (e)	9-Jun-98	1	0.97	0.00	3.16	4.81	22.65	12.49	0.11	0.00	0.00	44.2
Spondylus ? multimuricatus	Agat Marina	21-Dec-98	4	0.36	0.00	2.52	0.19	0.50	0.63	0.00	0.00	0.00	4.19

Table 21

PCB Homologues in Octopus, Mantis Shrimp and Ascidians From Guam Harbor Waters (data as ng/g wet wt.)

Species	Location (site)	Date	Tissue	Cl_2B	Cl_3B	Cl_4B	Cl_5B	Cl_6B	Cl_7B	Cl_8B	Cl_9B	$Cl_{10}B$	$\Sigma_{20}PCB$
OCTOPUS													
Octopus cyanea	Apra Harbor (c)	6-Jun-98	T	0.19	0.19	0.00	1.70	3.75	2.87	0.07	0.00	0.00	8.78
			L	0.30	6.59	15.6	42.9	770	436	0.00	0.00	0.00	1271
MANTIS SHRIMP													
Gonodactylus sp.	Apra Harbor (e)	9-Jun-98	M	0.00	0.29	0.20	1.84	21.41	14.51	0.00	0.00	0.00	38.2
ASCIDIANS													
Ascidia sp.	Apra Harbor (e)	9-Jun-98	W	0.00	0.00	0.00	0.41	0.75	1.17	0.04	0.00	0.00	2.38
Rhopalaea	Apra Harbor (b)	5-Jun-98	W	0.00	0.00	0.00	0.00	0.07	0.03	0.00	0.00	0.00	0.10
Rhopalaea	Apra Harbor (c)	3-Jun-98	W	0.41	0.00	0.00	0.41	0.84	0.67	0.00	0.00	0.06	2.39
Rhopalaea	Apra Harbor (d)	9-Jun-98	W	0.00	0.00	0.00	0.99	1.12	0.89	0.00	0.00	0.00	3.00

T = tentacle; L = liver; M = tail muscle; W = whole

Table 22

PCB Homologues in Tissues of Fish From Guam Harbor Waters (data as ng/g wet wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	Cl_2B	Cl_3B	Cl_4B	Cl_5B	Cl_6B	Cl_7B	Cl_8B	Cl_9B	$Cl_{10}B$	$\Sigma_{20}PCB$
Acanthurus xanthopterus	Agana Boat Basin	18-Dec-98	36.0	M	0.98	2.78	2.33	5.11	4.97	4.87	0.13	0.00	0.00	21.16
				L	1.85	25.8	20.4	32.8	24.8	12.4	0.51	0.27	0.00	119
Acanthurus xanthopterus	Agana Boat Basin	30-Dec-98	22.0	M	0.00	0.20	0.37	0.09	0.41	0.46	0.00	0.00	0.00	1.53
Acanthurus xanthopterus	Agana Boat Basin	30-Dec-98	18.0	M	0.57	0.75	1.54	1.91	0.22	1.13	0.00	0.00	0.00	6.11
Acanthurus xanthopterus	Agana Boat Basin	30-Dec-98	14.5	M	0.00	0.28	0.27	0.13	0.25	0.27	0.00	0.00	0.00	1.22
Acanthurus xanthopterus	Apra Harbor (c)	3-Jun-98	38.0	M	0.00	1.91	18.8	38.7	19.2	6.08	0.19	0.00	0.00	85.0
				L	0.00	0.00	35.9	95.8	43.4	8.68	0.58	0.00	0.00	184
Acanthurus xanthopterus	Apra Harbor (c)	3-Jun-98	30.5	M	0.00	0.14	0.97	3.13	5.48	4.20	0.10	0.00	0.06	14.1
				L	0.00	1.46	7.04	26.8	43.0	23.7	0.60	0.54	0.39	103
Acanthurus xanthopterus	Apra Harbor (c)	3-Jun-98	29.0	M	0.38	0.00	0.66	3.02	10.4	7.64	0.13	0.00	0.00	22.2
				L	0.00	2.94	25.8	93.5	288	201	3.97	0.00	0.00	615
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun-98	16.5	M	0.36	0.00	0.43	0.25	0.95	0.88	0.00	0.00	0.00	2.86
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun-98	15.5	M	0.44	0.28	1.24	0.94	1.06	0.91	0.15	0.00	0.14	5.15
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun-98	12.8	M	0.19	0.00	0.58	0.11	0.61	0.84	0.04	0.00	0.09	2.45
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun-98	11.0	M	0.21	0.00	0.53	0.72	2.60	2.81	0.06	0.00	0.20	7.13
Balistoides viridescens	Merizo Pier	22-Dec-98	18.5	M	0.00	0.00	0.00	0.26	0.67	0.22	0.00	0.00	0.00	1.15
				L	1.09	4.62	12.1	75.0	128	31.1	1.52	0.77	0.00	255
Bolbometopon muricatum	Apra Harbor (c)	3-Jun-98	52.0	M	0.27	0.00	0.00	0.47	1.13	0.62	0.00	0.00	0.00	2.50
				L	0.00	3.82	21.6	29.8	272	296	0.00	0.00	0.00	623
Caranx ignobilis	Agana Boat Basin	18-Dec-98	26.5	M	0.62	0.24	1.01	3.84	6.03	3.93	0.14	0.00	0.00	15.8
				L	0.70	2.24	5.78	32.7	42.5	27.5	0.94	1.42	0.64	114
Caranx melampygus	Apra Harbor (b)	5-Jun-98	26.5	M	0.00	0.00	1.02	9.64	18.02	11.72	0.15	0.00	0.06	40.6
Caranx melampygus	Apra Harbor (e)	9-Jun-98	33.0	M	0.00	0.00	0.00	2.42	20.44	19.87	0.33	0.00	0.22	43.3
				L	0.00	0.20	3.22	47.5	10996	5955	3.16	2.23	1.53	17009
Caranx sexfasciatus	Agana Boat Basin	30-Dec-98	25.0	M	0.39	0.32	0.36	2.19	4.86	3.02	0.08	0.00	0.00	11.23
Caranx sexfasciatus	Agana Boat Basin	30-Dec-98	23.0	M	0.40	0.38	0.39	0.99	1.60	1.32	0.00	0.00	0.00	5.08
Caranx sexfasciatus	Apra Harbor (c)	3-Jun-98	22.0	M	0.16	0.05	0.24	2.08	8.52	5.76	0.07	0.00	0.00	16.9
Caranx sexfasciatus	Apra Harbor (d)	9-Jun-98	17.0	M	0.00	0.00	0.93	8.27	8.95	3.12	0.05	0.00	0.00	21.3

M = muscle tissue; L = liver tissue;

Table 22 (cont.)

PCB Homologues in Tissues of Fish From Guam Harbor Waters (data as ng/g wet wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	Cl_2B	Cl_3B	Cl_4B	Cl_5B	Cl_6B	Cl_7B	Cl_8B	Cl_9B	$Cl_{10}B$	$\Sigma_{20}PCB$
Cephalopholis sonnerati	Merizo Pier	22-Dec-98	16.5	M	0.38	0.00	0.00	0.00	0.09	0.10	0.00	0.00	0.00	0.57
Cheilinus chlorourus	Agat Marina	22-Jan-98	22.5	M	0.15	0.00	0.26	0.00	0.21	0.22	0.00	0.00	0.00	0.84
Cheilinus fasciatus	Apra Harbor (c)	3-Jun-98	24.5	M	0.00	0.07	0.10	0.38	0.68	2.64	0.04	0.00	0.15	4.07
Cheilinus fasciatus	Apra Harbor (c)	3-Jun-98	24.5	M	0.23	0.40	0.20	1.16	1.76	3.23	0.05	0.00	0.09	7.12
Cheilinus fasciatus	Apra Harbor (c)	3-Jun-98	19.0	M	0.00	0.08	0.00	0.28	0.54	1.45	0.02	0.00	0.06	2.43
				L	0.00	2.69	5.56	21.2	38.0	85.9	0.00	0.00	0.00	153
Cheilinus trilobatus	Merizo Pier	22-Dec-98	19.5	M	0.00	0.00	0.00	0.08	0.09	0.09	0.00	0.00	0.00	0.26
Cheilinus trilobatus	Merizo Pier	22-Dec-98	19.0	M	0.22	0.00	0.21	0.40	0.52	0.18	0.00	0.00	0.00	1.53
Ctenochaetus binotatus	Apra Harbor (d)	9-Jun-98	21.0	M	0.13	0.12	1.99	4.86	5.69	3.52	0.07	0.00	0.04	16.4
				L	0.36	0.00	1.47	5.37	5.84	2.65	0.08	0.00	0.00	15.8
Ctenochaetus striatus	Apra Harbor (e)	9-Jun-98	12.5	M	0.00	0.00	0.00	0.95	10.6	11.3	0.49	0.18	0.60	24.2
Ctenochaetus striatus	Apra Harbor (f)	12-Jun-98	13.0	M	0.38	0.09	0.37	0.62	3.13	2.52	0.06	0.00	0.00	7.17
Ctenochaetus striatus	Agat Marina	22-Jan-98	12.5	M	0.47	0.00	0.00	0.00	0.15	0.22	0.00	0.00	0.00	0.83
Epibulus insidiator	Apra Harbor (c)	3-Jun-98	24.5	M	0.29	0.07	0.59	8.68	12.8	4.61	0.09	0.00	0.00	27.1
Epibulus insidiator	Apra Harbor (e)	12-Jun-98	16.0	M	0.33	0.19	0.17	1.89	18.6	15.0	0.39	0.00	0.22	36.8
Epinephelus merra	Merizo Pier	22-Dec-98	24.0	M	0.00	0.00	0.00	0.10	0.25	0.25	0.00	0.00	0.00	0.59
Gerres argyreus	Agana Boat Basin	30-Dec-98	24.0	M	0.41	1.43	1.43	1.56	2.74	1.57	0.00	0.00	0.00	9.13
				L	2.19	78.8	58.0	84.0	369	38.8	0.56	0.53	0.00	632
Gerres argyreus	Agana Boat Basin	30-Dec-98	15.5	M	0.00	0.00	0.00	0.25	0.83	0.60	0.00	0.00	0.00	1.67
Gerres argyreus	Apra Harbor (d)	9-Jun-98	16.5	M	0.37	0.00	0.78	5.92	5.79	1.84	0.00	0.00	0.00	14.7
Gerres argyreus	Apra Harbor (d)	9-Jun-98	15.0	M	0.00	0.00	0.00	1.98	3.98	2.72	0.00	0.00	0.00	8.67
Gerres argyreus	Apra Harbor (d)	9-Jun-98	14.5	M	0.00	0.00	0.00	0.71	1.67	1.12	0.00	0.00	0.00	3.49
Gymnothorax javanicus	Apra Harbor (a)	5-Jun-98	60.0	M	0.19	0.00	0.60	1.09	2.80	1.63	0.02	0.00	0.00	6.33
				L	0.00	0.00	1.18	5.01	12.8	7.75	0.00	0.00	0.18	27.0
Leiognathus equulus	Agat Marina	22-Jan-98	14.0	M	0.00	0.00	0.00	0.84	8.99	8.13	0.20	0.00	0.00	18.2
Lethrinus rubrioperculatus	Agat Marina	21-Dec-98	24.5	M	0.00	0.14	0.00	0.78	2.14	2.14	0.07	0.00	0.00	5.27
				L	0.00	5.89	0.00	24.2	71.0	46.2	1.55	0.00	0.00	149
Lethrinus rubrioperculatus	Merizo Pier	22-Dec-98	20.5	M	0.40	0.00	0.00	0.60	0.59	0.15	0.00	0.00	0.00	1.74

M = muscle tissue; L = liver tissue;

Table 22 (cont.)

PCB Homologues in Tissues of Fish From Guam Harbor Waters (data as ng/g wet wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	Cl_2B	Cl_3B	Cl_4B	Cl_5B	Cl_6B	Cl_7B	Cl_8B	Cl_9B	$Cl_{10}B$	$\Sigma_{20}PCB$
Lutjanus kasmira	Merizo Pier	22-Dec-98	13.5	M	0.68	0.11	0.20	0.00	0.43	0.39	0.00	0.00	0.00	1.81
Monodactylus argenteus	Agana Boat Basin	18-Dec-98	14.5	M	0.41	1.02	2.54	2.64	2.09	1.54	0.03	0.00	0.00	10.3
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	17.8	M	0.60	0.08	0.80	1.89	5.21	2.84	0.03	0.00	0.00	11.5
				L	0.26	1.20	10.9	61.3	6394	4875	1.70	0.98	0.68	11346
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	17.0	M	0.00	0.19	2.22	5.87	8.97	3.76	0.04	0.00	0.00	21.0
				L	0.36	0.86	7.38	56.7	2722	1038	1.03	0.71	0.51	3827
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	17.0	M	0.21	0.19	2.06	5.18	18.5	9.82	0.13	0.00	0.00	36.1
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	17.0	M	0.74	0.13	1.40	3.47	6.93	2.91	0.00	0.00	0.00	15.6
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	17.0	M	0.74	0.13	1.40	3.47	6.93	2.91	0.00	0.00	0.00	15.6
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	16.8	M	0.00	0.55	3.29	5.20	6.21	2.50	0.04	0.00	0.00	17.8
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	16.5	M	0.31	0.61	2.34	5.55	9.99	5.33	0.07	0.00	0.00	24.2
				L	1.10	3.13	19.5	78.4	171	112	1.50	1.67	1.34	390
Naso annulatus	Apra Harbor (e)	12-Jun-98	13.5	M	0.00	0.00	1.03	0.22	2.25	2.43	0.07	0.00	0.07	6.08
Naso unicornis	Apra Harbor (a)	5-Jun-98	18.5	M	1.57	0.06	0.25	0.46	1.81	1.33	0.02	0.00	0.00	5.51
Naso unicornis	Apra Harbor (a)	5-Jun-98	25.0	M	0.19	0.00	0.38	0.36	1.30	0.82	0.00	0.00	0.00	3.06
Odenus niger	Agat Marina	22-Jan-98	17.0	M	0.00	0.00	0.00	0.00	0.33	0.27	0.00	0.00	0.00	0.61
Parupeneus barberinus	Merizo Pier	22-Dec-98	26.0	M	0.30	0.12	0.29	0.47	0.44	0.67	0.00	0.00	0.00	2.30
Parupeneus barberinus	Merizo Pier	22-Dec-98	16.0	M	0.21	0.00	0.23	0.14	0.28	0.10	0.00	0.00	0.00	0.96
Parupeneus cyclostomus	Merizo Pier	22-Dec-98	25.0	M	0.33	0.09	0.00	0.60	1.34	0.54	0.00	0.00	0.00	2.90
Parupeneus multifasciatus	Merizo Pier	22-Dec-98	17.5	M	0.00	0.00	0.00	0.66	1.63	0.74	0.00	0.00	0.00	3.02
Saurida gracilis	Agana Boat Basin	30-Dec-98	23.0	M	0.00	0.00	0.00	0.20	0.75	0.69	0.00	0.00	0.00	1.64
				L	0.00	12.1	21.9	77.0	203	105	2.98	0.00	0.45	423
Saurida gracilis	Agana Boat Basin	30-Dec-98	19.5	M	0.00	0.00	0.00	0.22	0.83	0.55	0.00	0.00	0.00	1.60
Saurida gracilis	Agana Boat Basin	30-Dec-98	16.5	M	0.00	0.00	0.00	0.56	2.13	1.39	0.03	0.00	0.00	4.11
Saurida gracilis	Agana Boat Basin	30-Dec-98	15.5	M	0.00	0.00	0.00	0.87	2.41	1.43	0.03	0.00	0.00	4.75
Saurida gracilis	Agat Marina	31-Dec-98	20.0	M	0.00	0.00	0.00	0.00	0.32	0.46	0.00	0.00	0.00	0.78
Saurida gracilis	Agat Marina	31-Dec-98	19.0	M	0.26	0.00	0.26	0.11	0.42	0.59	0.00	0.00	0.00	1.63
Saurida gracilis	Agat Marina	31-Dec-98	17.5	M	0.00	0.00	0.00	0.00	0.09	0.12	0.00	0.00	0.00	0.21
Saurida nebulosa	Apra Harbor (a)	5-Jun-98	21.5	M	0.00	0.00	0.28	1.56	7.68	7.62	0.16	0.00	0.14	17.4
Saurida nebulosa	Merizo Pier	22-Dec-98	16.5	M	0.00	0.00	0.00	0.00	0.03	0.06	0.00	0.00	0.00	0.09

M = muscle tissue; L = liver tissue;

Table 22 (cont.)

PCB Homologues in Tissues of Fish From Guam Harbor Waters (data as ng/g wet wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	Cl_2B	Cl_3B	Cl_4B	Cl_5B	Cl_6B	Cl_7B	Cl_8B	Cl_9B	$Cl_{10}B$	$\Sigma_{20}PCB$
Scarus sordidus	Apra Harbor (e)	12-Jun-98	16.0	M	0.00	0.00	0.00	0.00	0.71	2.69	0.10	0.00	0.00	3.50
Scarus sordidus	Apra Harbor (e)	9-Jun-98	15.0	M	0.00	0.00	0.00	0.00	0.43	0.95	0.04	0.00	0.08	1.50
				L	3.25	0.00	10.7	5.45	28.0	153	8.12	14.6	10.8	234
Scarus sordidus	Apra Harbor (e)	12-Jun-98	14.0	M	0.20	0.21	0.36	0.71	0.61	1.42	0.17	0.00	0.19	3.86
Siganus spinus	Agana Boat Basin	18-Dec-98	15.0	M	0.34	0.58	1.00	1.16	1.62	0.83	0.00	0.00	0.00	5.54
Sufflamen chrysopiera	Apra Harbor (e)	12-Jun-98	17.0	M	0.00	0.00	0.00	1.54	16.5	19.1	0.06	0.00	0.38	37.6
				L	1.11	0.00	26.1	107	64.4	129	4.15	39.2	19.4	390
Valamugil engeli	Apra Harbor (b)	5-Jun-98	37.5	M	0.35	0.20	0.80	1.27	1.53	1.52	0.04	0.00	0.06	5.79
				L	0.84	4.32	9.32	21.0	25.9	16.6	0.34	0.38	0.36	79.1

M = muscle tissue; L = liver tissue;

3. Polycyclic Aromatic Hydrocarbons (PAHs) in Harbor Biota

PAHs are a group of aromatic hydrocarbons made up of two or more fused benzene rings. They are released into the environment from both natural and anthropogenic sources, although the latter are far more important in terms of global contributions to the environment. True PAHs contain only hydrogen and carbon atoms and are differentiated here from polycyclic aromatic compounds that contain other atoms such as nitrogen, oxygen or sulfur (McElroy *et al.* 1989).

Primary anthropogenic sources of PAHs include the burning of fossil fuels (pyrogenic PAHs) and accidental petroleum discharges (petrogenic PAHs). The widespread occurrence of PAHs in the environment is largely a result of the former source, i.e., the incomplete combustion of coal, oil, petroleum and wood (Jacobs 1995). Pyrogenic PAHs are predominantly unsubstituted and often referred to as 'pure' or 'parent' compounds. They consist largely of the higher molecular weight, 4-6 ring compounds. In contrast, petrogenic PAHs are predominantly low molecular weight congeners and are commonly characterized by the presence of alkylated derivatives of parent compounds with 2-4 aromatic rings (Law and Biscaya 1994).

Ecotoxicological interest in PAHs has grown in recent years, particularly in light of fairly strong evidence linking them with liver neoplasms and other abnormalities in demersal fish species (Malins *et al.* 1984, 1988). Several of the higher molecular weight compounds are metabolically transformed in many organisms, into potent carcinogens, teratogens and/or genotoxic metabolites (Cerniglia and Heitkamp 1989).

PAHs are relatively insoluble in seawater and rapidly become associated with suspended sediments upon entry into the marine environment. Consequently, in nearshore waters most PAHs are deposited in bottom sediments fairly close to their point of entry (Phillips *et al.* 1992). Aqueous solubilities generally decrease with increased molecular weight and range from around 30 mg/l for naphthalene to about 0.3 µg/l for benzo(g,h,i)perylene at 25°C (Readman *et al.* 1982, Eisler 1987). PAHs with more than seven aromatic rings are virtually insoluble, have extremely limited biological availability and, consequently, are of limited environmental significance (Neff 1979).

Concentrations of individual PAHs in the open ocean are usually in the sub-nanogram per liter range. Law *et al.* (1997) measured 15 unsubstituted PAHs in seawater from around England and reported total quantifiable concentrations of <1-15 ng/l in offshore samples. In coastal and estuarine waters, levels were between 2-3 orders of magnitude higher again. Dissolved PAH fractions were generally dominated by the more soluble, low molecular weight congeners, while the heavier compounds tended to predominate in the particulate fraction.

Total PAH levels in uncontaminated sediments are generally less than 5 ng/g (Pierce *et al.* 1986, Van Fleet *et al.* 1986) although background levels of 10-15 ng/g have been reported for some unimpacted, deep-sea sediments (Hites *et al.* 1980). PAH concentrations in sediments from the Great Barrier Reef, Australia, were always <0.8 ng/g, except in small areas close to sites frequently visited by powerboats; in those instances, total PAH levels exceeded 13.4 µg/g (Smith *et al.* 1985).

In highly contaminated waters, notably estuaries, ports and harbors, sedimentary PAHs may exceed concentrations of 1000 µg/g. Sediments collected near a coking facility in Nova Scotia in 1980, for example, contained total PAH levels of up to 2,830 µg/g (Eisler 1987). An all time high of 6,000 µg/g was reported for sediments from the creosote-contaminated waters of Eagle Harbor in Puget Sound (Swartz *et al.* 1989).

We previously measured 16 individual PAHs in Guam harbor sediments and found total quantifiable levels ranging from non-detectable to 10.7 µg/g. According to the United Nations Environment Program (UNEP 1994), total PAH levels of ~0.5 µg/g constitute a moderate degree of contamination whereas levels exceeding 10 µg/g are classified as highly contaminated. In our study, only samples from Hotel Wharf and the Shell Fox-1 Fuel Pier in Apra Harbor fell into the latter category. Moderate contamination was encountered around the Commercial Port and Dry Dock Island areas. All other Apra Harbor sites were classified as either lightly contaminated or clean (Denton *et al.* 1997).

According to Long *et al.* (1995), sediments with total PAH concentrations of 4 µg/g, or less, pose minimal risk of adverse biological effects to resident biota. From this it would appear that levels encountered in and around Hotel Wharf and the Shell Fox-1 Fuel Pier in Apra Harbor are also significant from an environmental toxicity standpoint.

In the present study, we determined the same 16 PAHs in biotic representatives from several sites, including those mentioned above. The findings of the study are summarized in Tables 23-29, together with the sum totals for all detectable residues (Σ_{16}PAH) for each organism or tissue analyzed. Non-detectable residues were set to zero during the summing process.

The data are briefly reviewed in the context of previously published information from elsewhere. Unfortunately, little or no comparative data exists for several of the invertebrate groups considered here. Nevertheless, an overall review of the literature indicates that total PAH concentration in excess of 100 µg/g dry weight are not unusual in aquatic organisms living close to point sources of PAH, such as petroleum drilling activities, oil spills or chronic fuel leakages. In contrast, organisms from remote or relatively unpolluted areas generally contain levels in the low ng/g range (Onuska 1989). Reported values for individual PAHs range from ~0.01-5,000 ng/g dry weight (McElroy *et al.* 1989). In general, the highest tissue concentrations are displayed by organisms with high lipid content, poor PAH metabolizing capabilities, and distribution patterns coincident with the location of PAH sources (Kennish 1998).

All referenced data included in the following discussions are expressed on a wet weight basis unless indicated otherwise.

3.1 PAHs in Algae:

Algae rapidly accumulate dissolved PAHs from the water column, attaining steady state concentrations usually within 24 h (Neff 1979). Bioconcentration factors of 10^3, or more, are not uncommon and reflect this group's inability to effectively metabolize PAHs (Eisler 1987). Experimental evidence suggests that uptake is related more to adsorption rather than absorption processes (Leversee *et al.* 1981). As a result, depuration is primarily the result of

slow partitioning from surface adsorption sites back into the water column once ambient PAH levels subside (Kauss *et al.* 1973, Soto *et al.* 1975).

Algae are particularly useful indicators of petroleum spillages. Such events are typically characterized by an abundance of the more water soluble, low molecular weight PAHs in the water column. These are highly available to algae and tend to dominate tissue profiles for some time after the spill has passed (Farrington *et al.* 1983, Jones *et al.* 1986, and Murray *et al.* 1991). In contrast, the more hydrophobic, high molecular weight members are rapidly scavenged from solution by suspended particles and their biological availability is considerably reduced (Readman *et al.* 1984).

In the current study, only very low levels of some of the higher molecular weight PAHs were detected in *Padina* sp. from Commercial Port (site d), Dry Dock Island (site e), and Echo Wharf (site f). Σ_{16}PAH concentrations ranged from 30-41 ng/g and are presumably a reflection of pyrogenic PAH contributions from the engine exhaust streams of watercraft in the area. The absence of detectable 2- and 3-ring PAHs indicated that significant fuel spills had not occurred at these sites in the recent past. At all other sites, levels of all PAHs examined were below the limits of analytical detection (Table 23).

Few studies have focused on the PAH content of algae. Harrison *et al.* (1975) published a maximum value of 60 ng/g for total PAHs in marine algae from Greenland. This value is not too far removed from the maximum Σ_{16}PAH concentration reported here for *Padina* sp. In an earlier series of studies, Mallet and coworkers looked at benzo(*a*)pyrene levels in marine algae from Greenland and French Mediterranean coastal waters and found levels ranging from undetectable to 60 ng/g dry weight (Mallet 1961, Mallet *et al.* 1963, Perdriau 1964). The highest value reported by these researchers translates to ~15 ng/g on a wet weight basis and is approximately half the maximum benzo(*a*)pyrene concentration determined in *Padina* sp. during the present study.

Levels of this particular PAH are usually no more than 1 or 2 ng/g in marine organisms from remote locations. In large harbors and marinas, they are typically higher and are frequently associated with creosoted wharf pilings, domestic and industrial sewage discharges, shipping wastes, crude oil and refined petroleum spills, engine exhausts, and stormwater runoff from sealed roads and other bituminous surfaces (Neff 1979).

3.2 PAHs in Sponges:

Σ_{16}PAH concentrations in the sponges analyzed were at least an order of magnitude higher than in *Padina* sp. Presumably, this reflects the relatively high lipid content of the various representatives looked at within this group. The fact that sponges have very limited PAH metabolizing capabilities may also be a contributing factor here (Kurelec *et al.* 1985).

PAH profiles were largely dominated by 4-6 ring compounds of pyrogenic origin (Table 24). Low levels of the 3-ringed PAH, anthracene, were detected in several species of sponge from Apra Harbor. However, this low molecular weight congener is a product of combustion and is not present in petroleum (Hellou 1996). The dominance of pyrogenic PAHs in the area is, therefore, confirmed and further supported by the absence of other low molecular weight

congeners, apart from phenanthrene, a 3-ringed compound and common component of both petrogenic and pyrogenic PAHs (Hellou 1996).

We were unable to locate any comparative data for sponges from elsewhere at the time of compiling this report.

3.3 PAHs in Corals:

Σ_{16}PAH concentrations in the soft coral *Simularia* sp. were of the same order as determined in *Padina* sp., apart from one sample taken from underneath the Shell Fox-1 Fuel Pier, in Apra Harbor (site c). This particular specimen had a total quantifiable PAH concentration of 117 ng/g. Its PAH profiles were dominated by anthracene, fluorene and chrysene, three common constituents of fossil fuel combustion (Table 24).

No comparative PAH data was found for soft corals from other parts of the world.

3.4 PAHs in Sea Cucumbers:

A limited number of PAHs were detected in sea cucumbers from Apra Harbor and the Merizo Pier area, although there was no consistency in residue patterns between sites. Total quantifiable concentrations were relatively low and ranged from 26-83 ng/g (Table 25).

Aquatic organisms can acquire PAHs from water, food and sediments. Direct uptake from water is generally considered to be more efficient than from food or sediment. In fact, sediment bound PAHs have only limited biological availability. Consequently, benthic organisms, like sea cucumbers, rarely contain higher levels of PAHs than the sediment in their immediate surroundings, even in highly polluted waters (Neff 1979). Moreover, there is now evidence to suggest that higher invertebrates like echinoderms, arthropods and annelids, can metabolize PAHs, whereas lower invertebrates like coelenterates and sponges generally cannot (James 1989). The fact that we were unable to detect any PAHs in the majority of sea cucumbers analyzed is, therefore, not surprising.

Remarkably little attention has been directed towards the PAH assimilating capacity of echinoderms considering the intimate contact these organisms have with marine sediments. Mallet *et al.* (1963) was unable to detect benzo(*a*)pyrene in an unidentified sea cucumber from the west coast of Greenland. However, they reported a maximum value of 126 ng/g dry weight for this PAH in an unidentified starfish from the North Sea coast of France. In the present study detectable levels of benzo(*a*)pyrene were only found in the hemal system of *Holothuria atra* from the Port Authority Beach area. In this particular instance a value of 58 ng/g was recorded. This equates to ~387 ng/g when recalculated on a dry weight basis and is relatively high for an aquatic organism.

3.5 PAHs in Mollusks:

From a PAH monitoring standpoint, bivalve mollusks have received far more attention than any other invertebrate group. Their popularity stems from the fact that they can rapidly accumulate PAHs and have little capacity for PAH metabolism (McElroy *et al.* 1989, Hellou 1996). Moreover, they have the advantage of being sessile and attached; hence tissue concentrations are a reflection of levels in their immediate surroundings. Mussels and oysters

are the most commonly used indicator species in PAH surveillance studies and recent data from the NS&T and IMW 'Mussel Watch' programs indicates that Σ_{18}PAH (and Σ_{20}PCB) levels in both bivalves from the same sites agree within a factor of two (O'Connor 1992).

O'Connor (1998) recently summarized the NS&T 1988-96 'Mussel Watch' data for 18-24 PAH congeners in oysters and mussels from 287 U.S. coastal sites. Annual median total PAH concentrations ranged from 62-503 ng/g dry weight over the nine-year period.

Earlier, Sericiano *et al.* (1995) produced a more comprehensive breakdown of the NS&T and IMW data for bivalves from the North, Central and South American coasts, between 1986-1993. It transpired that samples from five out of 51 NS&T sites from the Gulf of Mexico contained Σ_{18}PAH concentrations between 1,100 and 3,700 ng/g dry weight. A further 18 sites yielded samples with levels ranging between 100-1,000 ng/g dry weight. Bivalves from all other sites in this region contained total PAH levels of <100 ng/g dry weight. PAH levels in the bivalves from 71 out of 76 IMW sites in Central and South America also fell within the latter range. The highest value of 1600 ng/g dry weight was measured in samples collected near a local port in Punta Arenas, Chile.

In the present investigation, PAHs were detected in 53% of oyster samples analyzed (Table 26). Total quantifiable levels ranged from 15-78 ng/g and were highest in samples collected from underneath the Shell Fox-1 Fuel Pier (site c). Phenanthrene and fluoranthene were the most commonly detected congeners. Benzo(*a*)pyrene was identified only once, in oysters from Agana Boat Basin, and at a relatively low concentration of 10 ng/g.

To permit comparisons with the NS&T and IMW data, the current findings were recalculated on a dry weight basis and ranged from ~100-520 ng/g. These values are very close to the annual median ranges for U.S. coastal waters cited above and are well within the range of values determined by both programs.

Total PAH levels in oysters from clean environments are usually less than 10 ng/g on a fresh weight basis. This is inferred from the work of Pendoley (1992) who examined 16 parent PAHs and 8 alkalyted derivatives of napthalene and phenanthrene in oysters from a remote offshore location in Western Australia. Total quantifiable levels of pure and alkylated PAHs were 4.6 and 135 ng/g respectively and were classed as being representative of an unpolluted environment.

In a more recent investigation, Michel and Zengel (1998) measured 14 pure and 20 alkylated PAHs in the oysters from Acajutla, El Salvador, following two oil spill incidences. They reported total PAH concentrations ranging from a low of 37 ng/g dry weight (~6 ng/g wet weight) in specimens from clean areas, up to 18,000 ng/g dry weight at the most heavily impacted sites. Residues were primarily of petrogenic origin in all instances.

Clearly then, while PAH levels in oysters from Guam harbors are not exactly representative of pristine conditions, they fall a long way short of those encountered in bivalves from heavily polluted waters (see Table 7).

No comparative data exists to evaluate the PAH levels found in chamids and spondylids during the present investigation (Table 27). The limited data we have suggests that their affinities for PAHs compare reasonably well with those of oysters. However, it is well known that different species of mollusks can take up different types and levels of PAHs from their environment (Boehm *et al.* 1982).

The highest PAH levels recorded here for chamids were in specimens from the western end of Commercial Port (site d). At this site, total quantifiable levels ranged from 63-783 ng/g with an overall geometric mean value of 235 ng/g. Such high sample variability may reflect individual differences in size and/or physiological condition related to gonad development and spawning. These variables were not accounted for during this preliminary study.

Tissue PAH profiles in chamids from site d were dominated by phenanthrene, anthracene, fluoranthene, chrysene, benzo(*k*)fluoranthene and benzo(*a*)pyrene. The absence of the low molecular weight homologues, in addition to the fact that phenanthrene/anthracene ratios were less than 10, indicates that residues were primarily of pyrolytic origin (Benlahcen *et al.* 1997).

Although numerous studies have focused on PAH levels in bivalves, we were unable to locate any that dealt specifically with cephalopods. Suffice to say, the single octopus taken from Apra Harbor during the present study contained no quantifiable levels of PAHs in either tissue analyzed (Table 28.). We therefore suspect that the appropriate metabolic processes are sufficiently well developed in this organism to maintain PAHs at very low levels. The squid, *Ilex illecebrosus*, is certainly able to rapidly transform PAHs into polar metabolites (Payne 1976), but whether all cephalopod mollusks can do the same remains to be established.

3.6 PAHs in Crustaceans:

Crustaceans generally show better PAH metabolizing capabilities than mollusks and other lower invertebrates (James 1989, Kennish 1998). However, excretion is relatively slow and so tissue residues tend to build up when ambient concentrations are elevated. The work of Sirota *et al.* (1983) admirably demonstrates this. These researchers measured total PAHs in the American lobster, *Homarus americanus*, living in the vicinity of the Nova Scotia coking facility mentioned earlier. It will be recalled that sedimentary PAH levels peaked at 2,830 µg/g. Lobsters exposed to such unusually high concentrations accumulated levels ranging from 1.91-2.67 µg/g and 57.3-88.1 µg/g in their tail muscle and hepatopancreas respectively. Levels in control specimens taken some distance from the facility were 1-2 orders of magnitude lower.

Total PAH levels reported for crustaceans from other areas are highly variable and range from <100->6,000 ng/g dry weight in whole specimens (see Table 7). Among the highest levels encountered in edible tissue was a value of 1600 ng/g for the rock crab, *Cancer irroratus* from the New York Bight area (Humason and Gadbois 1982).

In view of the above, it is significant to note that we were unable to detect any PAH residues in the tail muscle of the stomatopod, *Gonodactylus* sp. from Apra Harbor (Table 28). This burrowing predatory species might be expected to reflect the PAH loading of the bottom sediments in which it lives, although sediment-sorbed PAHs have limited bioavailability as

mentioned earlier. Nevertheless, the stark absence of PAHs in the tail muscle of this specimen deserves further investigation to determine possible links between habitat and/or effective PAH metabolism.

3.7 PAHs in Ascidians:

Almost nothing is known about the PAH accumulation characteristics of tunicates. What limited data there is suggests that certain species can metabolize these compounds while others clearly cannot (Kurelec *et al* 1977). In the present study, we were unable to detect any PAH residues in the ascidian analyzed, apart from very low levels of anthracene (3 ng/g) and benzo(*k*)fluoranthene (9 ng/g) in *Rhopalaea* sp. from site d. The fact that ascidians are approximately 95% water could possibly account for their apparent lack of sensitivity to environmental PAHs although metabolic process cannot be overruled.

3.8 PAHs in Fish:

Fish have a well-developed enzyme system that rapidly transforms PAHs into water-soluble metabolites. Consequently, they accumulate these contaminants only when exposed to heavily contaminated environments or chronic leakages (see Table 7). Even then, they are able to depurate 99% of all accumulated PAHs within 24 h of uptake, once returned to clean water (Varanasi *et al.* 1989). For these reasons, PAH levels in fish axial muscle are commonly close to or below the limits of analytical detection, even in moderately polluted waters.

The results of the present survey are, therefore, encouraging. Out of 75 fish analyzed, quantifiable levels of PAHs were detected in the axial muscle of only 10 specimens. Levels ranged from 4-64 ng/g with a median value of 20 ng/g. Tissue PAH profiles varied between species but, in general, were dominated by phenanthrene, followed in decreasing frequency of detection by: benzo(*g,h,i*)perylene > dibenz(*a,h*)anthracene > anthracene > acenaphthene and fluorene (Table 29). This ranking suggests exposure to PAHs of predominantly pyrogenic origin, with minor contribution from petrogenic sources. PAHs were not detected in any of the fish livers examined.

3.9 Concluding Remarks:

This preliminary survey generally indicates low level movement of PAHs into the biota of each harbor studied. The biota from Apra Harbor are particularly clean when compared with levels found in related species from similar sized ports elsewhere in the world. This is somewhat surprising considering the intensity of military and commercial shipping activities that go on here on a day-to-day basis. No doubt, current harbor policies aimed at preventing petroleum spillage and oil/water discharges from boats and ships in the area have much to do with this. Also, PAH degradation and volatilization rates are higher here compared with cooler regions, and, in all probability, is paralleled by higher PAH turnover rates in the local biota. Thus, the impact of a small spill on tissue PAH residues will very likely be short-lived, as will the telltale PAH signatures in the bioindicators of choice, once conditions return to normal.

Table 23

PAHs in Seaweed From Guam Harbor Waters (data as µg/g wet wt.)

Species	Location (site)	Date	NAP	ACY	ACE	FLR	PHE	ANT	FLU	PYR	BAA	CHR	BBF	BKF	BAP	DBA	BPE	INP	Σ_{16}PAH
Padina sp.	Agana Boat Basin	18-Dec-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Padina sp.	Apra Harbor (a)	5-Jun-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Padina sp.	Apra Harbor (c)	3-Jun-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Padina sp.	Apra Harbor (d)	9-Jun-98	BDL	BDL	BDL	BDL	BDL	BDL	0.016	BDL	BDL	BDL	BDL	0.010	0.012	BDL	BDL	BDL	0.037
Padina sp.	Apra Harbor (e)	9-Jun-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.005	BDL	BDL	BDL	0.036	BDL	BDL	0.041
Padina sp.	Apra Harbor (f)	12-Jun-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.030	BDL	BDL	BDL	0.030
Padina sp.	Agat Marina	21-Dec-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Padina sp.	Merizo Pier	22-Dec-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC

BDL = below detection limits; NC = not calculable

PAH Abbreviations (in order of molecular weight):

NAP	Naphthalene	BAA	Benz(a)anthracene
ACY	Acenaphthylene	CHR	Chrysene
ACE	Acenaphthene	BBF	Benzo(b)fluoranthene
FLR	Fluorene	BKF	Benzo(k)fluoranthene
PHE	Phenanthrene	BAP	Benzo(a)pyrene
ANT	Anthracene	BPE	Benzo(g,h,i)perylene
FLU	Fluoranthene	INP	Indeno(1,2,3-cd)pyrene
PYR	Pyrene	DBA	Dibenz(a,h)anthracene

Table 24

PAHs in Sponges and Soft Corals From Guam Harbor Waters (data as µg/g wet wt.)

Species	Location (site)	Date	NAP	ACY	ACE	FLR	PHE	ANT	FLU	PYR	BAA	CHR	BBF	BKF	BAP	DBA	BPE	INP	Σ_{16}PAH
SPONGES																			
Callyspongia diffusa	Agat Marina	21-Dec-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.002	BDL	BDL	BDL	BDL	BDL	BDL	0.074	0.075
Clathria vulpina ?	Agat Marina	21-Dec-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Clathria vulpina ?	Merizo Pier	22-Dec-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Dysidea sp.	Apra Harbor (c)	3-Jun-98	BDL	BDL	BDL	BDL	0.005	BDL	0.042	BDL	BDL	0.083	BDL	BDL	0.035	0.449	0.084	0.024	0.722
Dysidea sp.	Apra Harbor (d)	9-Jun-98	BDL	BDL	BDL	BDL	0.022	0.015	0.026	BDL	BDL	0.228	BDL	BDL	BDL	BDL	BDL	BDL	0.291
Dysidea sp.	Apra Harbor (f)	12-Jun-98	BDL	BDL	BDL	BDL	BDL	0.006	BDL	BDL	BDL	BDL	0.103	BDL	BDL	0.201	0.025	0.008	0.343
Liosina cf. granularis	Apra Harbor (b)	5-June '98	BDL	BDL	BDL	BDL	BDL	0.007	BDL	0.006	0.006	0.028	0.041	0.011	BDL	0.330	0.165	BDL	0.595
Liosina cf. granularis	Apra Harbor (e)	9-Jun-98	BDL	BDL	BDL	BDL	BDL	0.004	0.008	0.012	0.008	0.014	0.016	0.007	BDL	0.274	0.046	BDL	0.387
Stylotella aurantium	Apra Harbor (b)	5-Jun-98	BDL	BDL	BDL	BDL	BDL	BDL	0.006	0.003	BDL	0.008	0.007	BDL	BDL	0.180	BDL	BDL	0.204
Stylotella aurantium	Apra Harbor (e)	9-Jun-98	BDL	BDL	BDL	BDL	BDL	0.016	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.151	0.044	BDL	0.211
Stylotella aurantium	Merizo Pier	22-Dec-98	BDL	BDL	BDL	BDL	BDL	BDL	0.006	BDL	0.030	0.546	BDL	BDL	BDL	BDL	BDL	BDL	0.582
UNIDENTIFIED SPONGES																			
Brown Wart Sponge	Apra Harbor (e)	9-Jun-98	BDL	BDL	BDL	BDL	BDL	0.013	BDL	BDL	0.001	0.015	BDL	BDL	BDL	0.061	BDL	BDL	0.091
Brown Wart Sponge	Apra Harbor (f)	12-Jun-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.003	BDL	BDL	BDL	0.141	BDL	BDL	0.143
Orange Wart Sponge	Apra Harbor (e)	9-Jun-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.009	BDL	BDL	0.037	BDL	BDL	0.046
Yellow Bread Sponge	Agat Marina	21-Dec-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Yellow Sponge (red outside)	Apra Harbor (c)	3-Jun-98	BDL	BDL	BDL	BDL	BDL	BDL	0.017	0.010	0.001	BDL	0.047	0.023	0.030	0.144	0.020	0.020	0.312
SOFT CORALS																			
Sinularia sp.	Apra Harbor (c)	3-Jun-98	BDL	BDL	BDL	BDL	BDL	0.003	0.014	BDL	BDL	0.101	BDL	BDL	BDL	BDL	BDL	BDL	0.117
Sinularia sp.	Apra Harbor (e)	9-Jun-98	BDL	BDL	BDL	BDL	BDL	BDL	0.007	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.007
Sinularia sp.	Agana Boat Basin	18-Dec-98	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.024	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.024
Sinularia sp.	Merizo Pier	22-Dec-98	BDL	BDL	BDL	BDL	BDL	BDL	0.041	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.041

BDL = below detection limits; NC = not calculable

PAH Abbreviations (in order of molecular weight):

NAP	Naphthalene	BAA	Benz(a)anthracene
ACY	Acenaphthylene	CHR	Chrysene
ACE	Acenaphthene	BBF	Benzo(b)fluoranthene
FLR	Fluorene	BKF	Benzo(k)fluoranthene
PHE	Phenanthrene	BAP	Benzo(a)pyrene
ANT	Anthracene	BPE	Benzo(g,h,i)perylene
FLU	Fluoranthene	INP	Indeno(1,2,3-cd)pyrene
PYR	Pyrene	DBA	Dibenz(a,h)anthracene

Table 25

PAHs in Sea Cucumbers From Guam Harbor Waters (data as µg/g wet wt.)

Species	Location (site)	Date	Tissue	NAP	ACY	ACE	FLR	PHE	ANT	FLU	PYR	BAA	CHR	BBF	BKF	BAP	DBA	BPE	INP	Σ_{16}PAH
Bohadschia argus	Agana Boat Basin	18-Dec-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
			H	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Bohadschia argus	Apra Harbor (b)	5-Jun-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
			H	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Bohadschia argus	Apra Harbor (c)	12-Jun-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.006	0.002	BDL	BDL	BDL	BDL	0.059	BDL	BDL	0.067
Bohadschia argus	Apra Harbor (e)	9-Jun-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
			H	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Bohadschia argus	Agat Marina	21-Dec-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
			H	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Bohadschia argus	Merizo Pier	22-Dec-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
			H	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Holothuria atra	Agana Boat Basin	18-Dec-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
			H	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Holothuria atra	Apra Harbor (g)	12-Jun-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
			H	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.016	BDL	0.008	BDL	BDL	0.058	BDL	BDL	BDL	0.083
Holothuria atra	Merizo Pier	22-Dec-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
			H	BDL	BDL	BDL	BDL	0.015	BDL	0.011	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.026
Holothuria atra	Apra Harbor (e)	9-Jun-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.035	BDL	BDL	0.035
			H	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Holothuria atra	Agat Marina	21-Dec-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Holothuria atra	Agat Marina	21-Dec-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
			H	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC

M = body wall muscle tissue; H = hemal system; BDL = below detection limits; NC = not calculable

PAH Abbreviations (in order of molecular weight):

NAP	Naphthalene	BAA	Benz(a)anthracene
ACY	Acenaphthylene	CHR	Chrysene
ACE	Acenaphthene	BBF	Benzo(b)fluoranthene
FLR	Fluorene	BKF	Benzo(k)fluoranthene
PHE	Phenanthrene	BAP	Benzo(a)pyrene
ANT	Anthracene	BPE	Benzo(g,h,i)perylene
FLU	Fluoranthene	INP	Indeno(1,2,3-cd)pyrene
PYR	Pyrene	DBA	Dibenz(a,h)anthracene

Table 26

PAHs in Bivalve Mollusks From Guam Harbor Waters (data as µg/g wet wt.)

Species	Location (site)	Date	Pool	NAP	ACY	ACE	FLR	PHE	ANT	FLU	PYR	BAA	CHR	BBF	BKF	BAP	DBA	BPE	INP	Σ_{16}PAH
OYSTERS																				
Saccostrea cucculata	Apra Harbor (c)	5-Jun-98	10	BDL	BDL	BDL	BDL	0.022	0.003	0.036	BDL	BDL	BDL	BDL	0.011	BDL	BDL	BDL	BDL	0.073
*Saccostrea cucculata**	Merizo Pier	22-Dec-98	7	BDL	BDL	BDL	BDL	0.013	BDL	0.021	BDL	BDL	0.007	BDL	BDL	BDL	BDL	BDL	BDL	0.041
Striostrea mytiloides	Agana Boat Basin	18-Dec-98	4	BDL	BDL	BDL	BDL	0.004	BDL	0.021	BDL	BDL	BDL	BDL	0.012	0.010	BDL	BDL	BDL	0.048
Striostrea mytiloides	Apra Harbor (a)	5-Jun-98	2	BDL	BDL	BDL	0.022	0.014	BDL	0.013	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.049
Striostrea mytiloides	Apra Harbor (a)	5-Jun-98	2	BDL	BDL	BDL	BDL	BDL	BDL	0.007	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.017
Striostrea mytiloides	Apra Harbor (a)	5-Jun-98	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Striostrea mytiloides	Apra Harbor (a)	5-Jun-98	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.015	BDL	BDL	BDL	BDL	BDL	0.015
Striostrea mytiloides	Apra Harbor (a)	5-Jun-98	5	BDL	BDL	BDL	BDL	BDL	BDL	0.009	BDL	BDL	BDL	0.008	BDL	BDL	BDL	BDL	BDL	0.017
Striostrea mytiloides	Apra Harbor (e)	5-Jun-98	1	BDL	BDL	BDL	BDL	0.037	BDL	0.041	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.078
Striostrea mytiloides	Apra Harbor (e)	9-Jun-98	4	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Striostrea mytiloides	Apra Harbor (e)	9-Jun-98	6	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Striostrea mytiloides	Apra Harbor (e)	9-Jun-98	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Striostrea mytiloides	Apra Harbor (f)	12-Jun-98	3	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Striostrea mytiloides	Apra Harbor (f)	12-Jun-98	1	BDL	BDL	BDL	BDL	BDL	BDL	0.005	0.005	BDL	BDL	0.009	BDL	BDL	BDL	BDL	BDL	0.019
Striostrea mytiloides	Apra Harbor (f)	12-Jun-98	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Striostrea mytiloides	Apra Harbor (f)	12-Jun-98	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Striostrea mytiloides	Agat Marina	21-Dec-98	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC

* juveniles

M = muscle tissue; L = liver tissue; BDL = below detection limits; NC = not calculable

PAH Abbreviations (in order of molecular weight):

NAP	Naphthalene	BAA	Benz(a)anthracene
ACY	Acenaphthylene	CHR	Chrysene
ACE	Acenaphthene	BBF	Benzo(b)fluoranthene
FLR	Fluorene	BKF	Benzo(k)fluoranthene
PHE	Phenanthrene	BAP	Benzo(a)pyrene
ANT	Anthracene	BPE	Benzo(g,h,i)perylene
FLU	Fluoranthene	INP	Indeno(1,2,3-cd)pyrene
PYR	Pyrene	DBA	Dibenz(a,h)anthracene

Table 27

PAHs in Bivalve Mollusks From Guam Harbor Waters (data as μg/g wet wt.)

Species	Location (site)	Date	Pool	NAP	ACY	ACE	FLR	PHE	ANT	FLU	PYR	BAA	CHR	BBF	BKF	BAP	DBA	BPE	INP	Σ_{16}PAH
CHAMIDS																				
Chama brassica	Apra Harbor (d)	9-Jun-98	2	BDL	BDL	BDL	BDL	0.049	0.035	0.043	BDL	BDL	0.030	BDL	0.052	0.050	BDL	BDL	BDL	0.259
Chama brassica	Apra Harbor (f)	12-Jun-98	2	BDL	BDL	BDL	BDL	BDL	0.020	BDL	0.005	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.024
Chama lazarus	Apra Harbor (b)	5-Jun-98	3	BDL	BDL	BDL	BDL	BDL	BDL	0.014	BDL	BDL	BDL	0.009	BDL	BDL	0.026	BDL	BDL	0.048
Chama lazarus	Apra Harbor (b)	5-Jun-98	3	BDL	BDL	BDL	BDL	0.005	BDL	0.025	BDL	0.004	BDL	0.008	0.007	BDL	0.073	BDL	BDL	0.122
Chama lazarus	Apra Harbor (c)	5-Jun-98	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.003	BDL	BDL	0.007	BDL	BDL	BDL	BDL	BDL	0.010
Chama lazarus	Apra Harbor (c)	5-Jun-98	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Chama lazarus	Apra Harbor (d)	9-Jun-98	2	BDL	BDL	BDL	BDL	0.027	0.004	BDL	BDL	BDL	BDL	BDL	0.019	0.013	BDL	BDL	BDL	0.063
Chama lazarus	Apra Harbor (d)	9-Jun-98	2	BDL	BDL	BDL	BDL	0.071	0.013	0.025	BDL	BDL	0.021	BDL	0.030	0.028	0.021	0.028	BDL	0.238
Chama lazarus	Apra Harbor (d)	9-Jun-98	2	BDL	BDL	BDL	BDL	0.259	BDL	0.315	BDL	BDL	0.044	BDL	0.071	0.047	0.000	0.047	BDL	0.783
Chama lazarus	Apra Harbor (e)	9-Jun-98	2	BDL	BDL	BDL	BDL	BDL	BDL	0.007	0.005	0.001	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.012
Chama lazarus	Apra Harbor (e)	9-Jun-98	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Chama lazarus	Apra Harbor (f)	12-Jun-98	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.009	BDL	BDL	BDL	BDL	BDL	0.009
Chama lazarus	Apra Harbor (f)	12-Jun-98	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.153	BDL	BDL	BDL	0.153
Chama lazarus	Merizo Pier	22-Dec-98	1	BDL	BDL	BDL	BDL	0.004	BDL	0.012	0.009	BDL	0.003	BDL	BDL	BDL	BDL	BDL	BDL	0.028
SPONDYLIDS																				
Spondylus ? multimuricatus	Agana Boat Basin	18-Dec-98	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Spondylus ? multimuricatus	Apra Harbor (e)	9-Jun-98	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.011	BDL	BDL	BDL	0.011
Spondylus ? multimuricatus	Apra Harbor (e)	9-Jun-98	1	BDL	BDL	BDL	BDL	BDL	BDL	0.008	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.008
Spondylus ? multimuricatus	Agat Marina	21-Dec-98	4	BDL	BDL	BDL	BDL	0.014	0.003	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.016

BDL = below detection limits; NC = not calculable

PAH Abbreviations (in order of molecular weight):

NAP	Naphthalene	BAA	Benz(a)anthracene
ACY	Acenaphthylene	CHR	Chrysene
ACE	Acenaphthene	BBF	Benzo(b)fluoranthene
FLR	Fluorene	BKF	Benzo(k)fluoranthene
PHE	Phenanthrene	BAP	Benzo(a)pyrene
ANT	Anthracene	BPE	Benzo(g,h,i)perylene
FLU	Fluoranthene	INP	Indeno(1,2,3-cd)pyrene
PYR	Pyrene	DBA	Dibenz(a,h)anthracene

Table 28

PAHs in Octopus, Mantis Shrimp and Ascidians From Guam Harbor Waters (data as μg/g wet wt.)

Species	Location (site)	Date	Tissue	NAP	ACY	ACE	FLR	PHE	ANT	FLU	PYR	BAA	CHR	BBF	BKF	BAP	DBA	BPE	INP	Σ_{16}PAH
OCTOPUS																				
Octopus cyanea	Apra Harbor (c)	6-Jun-98	T	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
			L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
MANTIS SHRIMP																				
Gonodactylus sp.	Apra Harbor (e)	9-Jun-98	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
ASCIDIANS																				
Ascidia sp.	Apra Harbor (e)	9-Jun-98	W	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Rhopalaea	Apra Harbor (b)	5-Jun-98	W	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Rhopalaea	Apra Harbor (c)	3-Jun-98	W	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Rhopalaea	Apra Harbor (d)	9-Jun-98	W	BDL	BDL	BDL	BDL	BDL	0.003	BDL	BDL	BDL	BDL	BDL	0.009	BDL	BDL	BDL	BDL	0.012

T = tentacle; L = liver; M = tail muscle; W = whole; BDL = below detection limits; NC = not calculable

PAH Abbreviations (in order of molecular weight):

NAP	Naphthalene	BAA	Benz(a)anthracene
ACY	Acenaphthylene	CHR	Chrysene
ACE	Acenaphthene	BBF	Benzo(b)fluoranthene
FLR	Fluorene	BKF	Benzo(k)fluoranthene
PHE	Phenanthrene	BAP	Benzo(a)pyrene
ANT	Anthracene	BPE	Benzo(g,h,i)perylene
FLU	Fluoranthene	INP	Indeno(1,2,3-cd)pyrene
PYR	Pyrene	DBA	Dibenz(a,h)anthracene

Table 29

PAHs in Tissues of Fish From Guam Harbor Waters (data as µg/g wet wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	NAP	ACY	ACE	FLR	PHE	ANT	FLU	PYR	BAA	CHR	BBF	BKF	BAP	DBA	BPE	INP	Σ_{16}PAH
Acanthurus xanthopterus	Agana Boat Basin	18-Dec-98	36.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Acanthurus xanthopterus	Agana Boat Basin	30-Dec-98	22.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Acanthurus xanthopterus	Agana Boat Basin	30-Dec-98	18.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Acanthurus xanthopterus	Agana Boat Basin	30-Dec-98	14.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Acanthurus xanthopterus	Apra Harbor (c)	3-Jun-98	38.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Acanthurus xanthopterus	Apra Harbor (c)	3-Jun-98	30.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Acanthurus xanthopterus	Apra Harbor (c)	3-Jun-98	29.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun-98	16.5	M	BDL	BDL	0.008	0.009	0.044	0.003	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.064
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun-98	15.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun-98	12.8	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Acanthurus xanthopterus	Apra Harbor (f)	12-Jun-98	11.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Balistoides viridescens	Merizo Pier	22-Dec-98	18.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Bolbometopon muricatum	Apra Harbor (c)	3-Jun-98	52.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Caranx ignobilis	Agana Boat Basin	18-Dec-98	26.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Caranx melampygus	Apra Harbor (b)	5-Jun-98	26.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Caranx melampygus	Apra Harbor (e)	9-Jun-98	33.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Caranx sexfasciatus	Agana Boat Basin	30-Dec-98	25.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Caranx sexfasciatus	Agana Boat Basin	30-Dec-98	23.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Caranx sexfasciatus	Apra Harbor (c)	3-Jun-98	22.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Caranx sexfasciatus	Apra Harbor (d)	9-Jun-98	17.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC

M = muscle tissue; L = liver tissue; BDL = below detection limits; NC = not calculable

Table 29 (cont.)

PAHs in Tissues of Fish From Guam Harbor Waters (data as µg/g wet wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	NAP	ACY	ACE	FLR	PHE	ANT	FLU	PYR	BAA	CHR	BBF	BKF	BAP	DBA	BPE	INP	Σ_{16}PAH
Cephalopholis sonnerati	Merizo Pier	22-Dec-98	16.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Cheilinus chlorourus	Agat Marina	22-Jan-98	22.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Cheilinus fasciatus	Apra Harbor (c)	3-Jun-98	24.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Cheilinus fasciatus	Apra Harbor (c)	3-Jun-98	24.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Cheilinus fasciatus	Apra Harbor (c)	3-Jun-98	19.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Cheilinus trilobatus	Merizo Pier	22-Dec-98	19.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Cheilinus trilobatus	Merizo Pier	22-Dec-98	19.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Ctenochaetus binotatus	Apra Harbor (d)	9-Jun-98	21.0	M	BDL	BDL	BDL	BDL	0.005	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.005
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Ctenochaetus striatus	Apra Harbor (e)	9-Jun-98	12.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Ctenochaetus striatus	Apra Harbor (f)	12-Jun-98	13.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Ctenochaetus striatus	Agat Marina	22-Jan-98	12.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Epibulus insidiator	Apra Harbor (c)	3-Jun-98	24.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Epibulus insidiator	Apra Harbor (e)	12-Jun-98	16.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Epinephelus merra	Merizo Pier	22-Dec-98	24.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Gerres argyreus	Agana Boat Basin	30-Dec-98	24.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Gerres argyreus	Agana Boat Basin	30-Dec-98	15.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Gerres argyreus	Apra Harbor (d)	9-Jun-98	16.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Gerres argyreus	Apra Harbor (d)	9-Jun-98	15.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Gerres argyreus	Apra Harbor (d)	9-Jun-98	14.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Gymnothorax javanicus	Apra Harbor (a)	5-Jun-98	60.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Leiognathus equulus	Agat Marina	22-Jan-98	14.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Lethrinus rubrioperculatus	Agat Marina	21-Dec-98	24.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Lethrinus rubrioperculatus	Merizo Pier	22-Dec-98	20.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC

M = muscle tissue; L = liver tissue; BDL = below detection limits; NC = not calculable

Table 29 (cont.)

PAHs in Tissues of Fish From Guam Harbor Waters (data as μg/g wet wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	NAP	ACY	ACE	FLR	PHE	ANT	FLU	PYR	BAA	CHR	BBF	BKF	BAP	DBA	BPE	INP	Σ_{16}PAH
Lutjanus kasmira	Merizo Pier	22-Dec-98	13.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Monodactylus argenteus	Agana Boat Basin	18-Dec-98	14.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	17.8	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	17.0	M	BDL	BDL	BDL	BDL	0.007	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.007
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	17.0	M	BDL	BDL	BDL	BDL	0.004	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.004
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	17.0	M	BDL	BDL	BDL	BDL	0.009	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.010	0.019	BDL	0.037
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	17.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	16.8	M	BDL	BDL	BDL	BDL	0.006	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.006
Monodactylus argenteus	Apra Harbor (d)	9-Jun-98	16.5	M	BDL	BDL	BDL	BDL	0.009	0.003	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.011
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Naso annulatus	Apra Harbor (e)	12-Jun-98	13.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Naso unicornis	Apra Harbor (a)	5-Jun-98	18.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.030	BDL	0.030
Naso unicornis	Apra Harbor (a)	5-Jun-98	25.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Odenus niger	Agat Marina	22-Jan-98	17.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Parupeneus barberinus	Merizo Pier	22-Dec-98	26.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Parupeneus barberinus	Merizo Pier	22-Dec-98	16.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Parupeneus cyclostomus	Merizo Pier	22-Dec-98	25.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.012	0.024	BDL	0.036
Parupeneus multifasciatus	Merizo Pier	22-Dec-98	17.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.015	0.046	BDL	0.061
Saurida gracilis	Agana Boat Basin	30-Dec-98	23.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Saurida gracilis	Agana Boat Basin	30-Dec-98	19.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Saurida gracilis	Agana Boat Basin	30-Dec-98	16.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Saurida gracilis	Agana Boat Basin	30-Dec-98	15.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Saurida gracilis	Agat Marina	31-Dec-98	20.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Saurida gracilis	Agat Marina	31-Dec-98	19.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Saurida gracilis	Agat Marina	31-Dec-98	17.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Saurida nebulosa	Apra Harbor (a)	5-Jun-98	21.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Saurida nebulosa	Merizo Pier	22-Dec-98	16.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC

M = muscle tissue; L = liver tissue; BDL = below detection limits; NC = not calculable

Table 29 (cont.)

PAHs in Tissues of Fish From Guam Harbor Waters (data as μg/g wet wt.)

Species	Location (site)	Date	Fork Length (cm)	Tissue	NAP	ACY	ACE	FLR	PHE	ANT	FLU	PYR	BAA	CHR	BBF	BKF	BAP	DBA	BPE	INP	Σ_{16}PAH
Scarus sordidus	Apra Harbor (e)	12-Jun-98	16.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Scarus sordidus	Apra Harbor (e)	9-Jun-98	15.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Scarus sordidus	Apra Harbor (e)	12-Jun-98	14.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Siganus spinus	Agana Boat Basin	18-Dec-98	15.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Sufflamen chrysoptera	Apra Harbor (e)	12-Jun-98	17.0	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
Valamugil engeli	Apra Harbor (b)	5-Jun-98	37.5	M	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC
				L	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	NC

M = muscle tissue; L = liver tissue; BDL = below detection limits; NC = not calculable

PAH Abbreviations (in order of molecular weight):

NAP	Naphthalene	BAA	Benz(a)anthracene
ACY	Acenaphthylene	CHR	Chrysene
ACE	Acenaphthene	BBF	Benzo(b)fluoranthene
FLR	Fluorene	BKF	Benzo(k)fluoranthene
PHE	Phenanthrene	BAP	Benzo(a)pyrene
ANT	Anthracene	BPE	Benzo(g,h,i)perylene
FLU	Fluoranthene	INP	Indeno(1,2,3-cd)pyrene
PYR	Pyrene	DBA	Dibenz(a,h)anthracene

GENERAL CONCLUSIONS

This study though preliminary in nature, has produced a considerable bank of data upon which planners, regulators, water quality managers, and researchers can draw upon when dealing with related environmental problems. It clearly identifies areas of contaminant enrichment within biotic components of Guam's harbor environments, and provides a useful database with which future levels can be compared and evaluated. In addition, the study has identified a number of potentially useful bioindicator organisms for future monitoring purposes, and has assessed their current contamination status by reference to levels found in similar and related species from other parts of the world. It is hoped that the study will serve as a catalyst for more detailed investigations of spatial and temporal trends in contaminant levels for all of Guam's nearshore waters, and in representatives of the biotic resources that inhabit them. Such data is imperative if we are to achieve sustainability of our fragile coastal ecosystems and preserve the integrity of species frequently harvested for human consumption. To this end, some final comments are directed towards bioindicator use and the implementation of a suitable monitoring program for our coastal waters. The public health considerations relating to levels of certain contaminants determined in edible species during the course of this investigation are also briefly addressed together with recommendations for future work.

1. The Implementation of a Marine Monitoring Program Using Biological Indicators: Some Preliminary Considerations

The use of aquatic biota to monitor pollutant levels in aquatic environments started about 40 years ago with investigations into the abundance of radionuclides in the environment (e.g., Seymour 1966). Over the last two decades, the technique has been adapted to the study of stable heavy metals, persistent organochlorines like DDT and PCBs, and more recently, hydrocarbons. It is during this latter period that we have largely come to grips with many of the problems that rendered much of the earlier work invalid. Problems related to the use of inappropriate organisms, the timing and frequency of sampling events, and undue attention to biological variable such as growth and reproductive status, have all taken their toll on the usefulness of data produced by the early pioneers in this field. There are now a number of treatises available that deal with essential design imperatives for aquatic monitoring programs and we aim only to summarize the major points here. For further information the reader is referred to the excellent reviews of Phillips (1977, 1978, 1980, 1986a) and Phillips and Segar (1986).

1.1 Species Selection:

The basic premise underlying the bioindicator concept is that contaminants accumulate in the tissues of the bioindicator organism at rates that are proportional to concentrations in the surrounding water. Tissue residue levels are, therefore, a time-averaged indication of each contaminant's biological availability at that particular location and point in time.

According to Butler *et al.* (1971), Haug *et al* (1974), and Phillips (1977), an ideal indicator has the following attributes:

- It should accumulate the pollutant without being killed by the levels encountered in the environment

- It should be sedentary in order to be representative of the area in which it is collected
- It should be abundant throughout the study area, easily recognized, and readily sampled
- It should be of sufficient size to provide adequate tissue for analysis
- It should be relatively long-lived to permit sampling over several months or years
- It should be amenable to translocation
- It should demonstrate a simple correlation between pollutant levels accumulated in its tissues and the average pollutant concentration in the surrounding water.

The latter prerequisite is of overriding importance here because it requires that the bioindicator of choice possesses little or no ability to metabolically regulate pollutant levels in its tissues. Another highly desirable characteristic is that the bioindicator should exhibit a high concentration capacity for the contaminant in question. Some of the early studies with heavy metals were compromised by insufficient attention to metabolic control and the flawed assumption that high tissue concentrations of a particular element were a sign of bioindicator potential. Crustaceans for example are naturally high in copper and zinc and regulate tissue levels of both metals within relatively narrow limits (Bryan 1964). Hence, they are of no practical use as indicators for these elements. Zinc regulation has also been observed in a number of other invertebrate groups that accumulate this metal to relatively high levels (Bryan and Hummerstone 1973b, Phillips and Yim 1981, Klumpp and Burdon-Jones, 1982).

During the present study, we have also seen that fish and various invertebrate species have the capacity to rapidly metabolize and excrete PAHs from their tissues. Thus, they lack the sensitivity required to identify low-level environmental enrichment by these contaminants. Even with highly recalcitrant compounds like PCBs, certain bivalves show a preferential accumulation of the lower chlorinated congeners, while others rapidly eliminate them from their tissues (Denton 1974, Courtney and Denton 1974, Langston 1978a and b). Thus, it is important to tailor the choice of organism to the precise requirements of the monitoring program for PCBs, if the lower chlorinated congeners are of specific interest.

Clearly then, a number of considerations present themselves when selecting a suitable bioindicator. Some of these considerations are common to all contaminant groups examined here while others are more specific. For example, heavy metals are naturally occurring, and different species have evolved widely differing capacities to accumulate them. Even closely related species sometimes have metal profiles that are very different from one another. Some metals are biologically essential and are regulated in certain species but not in others. Again such differences can occur within, as well as, between biotic groups. The simple fact of the matter is that no single organism will satisfy the monitoring needs for all heavy metals of environmental interest. Moreover, comparing metal concentrations between closely related species can, at best, only provide an approximation of actual differences in elemental abundance between locations.

For persistent organochlorine compounds like PCBs, the situation is somewhat different. These are not naturally occurring and are certainly not biologically essential. Consequently

their uptake is purely a passive process and amounts found in the biota are largely a function of an organism's lipid content and composition. Crucial factors that affect PCB levels within and between species are largely those that influence cyclical events of lipid deposition and metabolism, and are primarily related to the interactive effects of season and sexual development. Needless to say, these variables are equally important from a heavy metal and PAH monitoring perspective. Choosing the correct bioindicator organism or suit of organisms, and refining sampling parameters and protocols is, therefore, of paramount importance, if spatial and temporal differences in pollution abundance are to be accurately assessed.

In temperate regions, a considerable amount of research has focused on the bioindicator ability of a select group of organisms (mostly brown algae, bivalve mollusk especially mussels and oysters, and various fish). In contrast, relatively little attention has been directed towards the utility of tropical species for monitoring purposes. As a consequence, preliminary monitoring programs, like the one undertaken here, may be forced to include hitherto 'untested' species that are only distantly related to well-established monitoring organisms from other regions of the world. This particular problem is compounded by the fact that, while species diversity is characteristically high in the tropical waters, the abundance of any one species is often not very great.

This was certainly evident during the present investigation. The oysters, for example, were not found in abundance outside of Apra Harbor. This was indeed unfortunate because these bivalves are excellent bioindicators of all three contaminant groups. Likewise the distribution of chamids and spondylids was found to be patchy, and available numbers were clearly insufficient to support the requirements of a long-term monitoring program in each of the harbors studied.

Locally, there are a number of other bivalves that could be considered for monitoring purposes, although they too are either absent or in low abundance in Guam harbors. One such example is the mussel, *Modiolus auriculatus*. This particular species occurs intertidally and on reef flats all around the island and is particularly abundant in Tumon Bay, Tanguisson Beach and Cocos Island lagoon. The cockle, *Gafrarium tumidum*, is another example and is relatively abundant in the mangroves of Sasa Bay. Its close relative, *Gafrarium pectinatum*, is widely distributed in sandy deposits of back-reef areas, and the wedge-clam, *Tellina palatum*, commonly occurs in sea-grass meadows. The availability of each of these species would certainly support a transplant-monitoring program providing of course that their bioindicator potential had been firmly established beforehand.

The tridacnid clams are another group that merit special mention here. These organisms are common inhabitants of coral reefs throughout the Indo-Pacific and are particularly sensitive bioindicators of heavy metal pollution (Kristoforova *et al.* 1979, Denton and Heitz 1991, 1993, Dight and Gladstone 1994). They have also been used as indictors of PCBs and PAHs in Australian waters (Olafson 1978, Smith *et al.* 1984, Smillie and Waid 1985).

T. maxima is commonly found on reefs around Guam, although not in the numbers that would support a regular monitoring program. However, culturing techniques are well established for

this group and large numbers are being raised in hatcheries throughout the Pacific for commercial purposes, as well as for restocking depleted reefs. Hatchery stocks are very amenable to transplantation and certain members have been shown to tolerate harbor conditions, seemingly without any adverse effect (Denton and Heitz 1991, 1993). Given the close proximity of Guam to Japan and the Asia market, a tridacnid clam hatchery on Guam, is a very attractive possibility both from a commercial and an environmental monitoring standpoint.

Other potentially useful candidate species for pollution monitoring purposes on Guam include the brown alga *Padina*. This particular genus is relatively widespread in local waters and its indicator capacity, at least for heavy metals, has been firmly established (Burdon-Jones *et al.* 1982, Denton and Burdon-Jones 1986). Moreover, there do not appear to be major inter-specific differences in metal uptake for this genus and so identification to species in the field is not critical.

Algae are an important component of any pollution-monitoring program because they reflect the availability of the soluble contaminant fraction and do not respond to fractions associated with sediments or suspended particulates. Together with bivalves, they can, therefore, provide the investigator with a greater understanding of contaminant movement and partitioning within aquatic ecosystems.

The soft corals have received some attention as bioindicators of certain heavy metals although evidence attesting to their reliability in this regard remains inconclusive (Denton and Burdon-Jones 1986). Nevertheless, they are a very common component of local reefs, and certain genera like *Sarcophyton* and *Sinularia* are readily identifiable. The current work identified *Sinularia* as a promising indicator for tin, zinc, PCBs and PAHs. We also consider this genus to be a probable indicator of arsenic, and a possible indicator of cadmium and chromium (see Table 30).

The chief disadvantage of using soft corals as an indicator organism appears to be one of species identification. The systematics of the group as a whole is not particularly well documented. Identification to genera can be accomplished relatively easily in the field, as mentioned above, but species determination, if at all possible, requires verification by spicule examination. The failure to distinguish between different species of the same genus could, therefore, compromise inter-site comparisons in contaminant abundance. However, the monitoring of within-site temporal trends is still possible, if tissue samples are repeatedly taken from the same colony over an extended period of time.

Of the less well known bioindicators examined here, the sponge, *Dysidea* sp shows promise for monitoring arsenic, copper, tin, and zinc. Their high fat content renders them excellent accumulators of lipophilic contaminants like PCBs and PAHs (Table 30). However, species identification in the field remains a problem.

The sea cucumbers are an obvious choice for future monitoring purposes, although their bioindicator potential for all three contaminant groups has yet to be unequivocally established. This notwithstanding, they appear to show excellent promise for the monitoring of arsenic,

Table 30

Evalution of the Bioindicator Potential of the Various Organisms Analyzed During this Study

Biotic Group	Ag	As	Cd	Cr	Cu	Hg	Ni	Pb	Sn	Zn	PCBs	PAHs
Brown Algae	5	3	5	5	5	5	5	5	3	5	5	5
Sponge	2	4	0	0	4	2	0	3	4	4	4	4
Soft Corals	1	3	2	2	0	0	0	0	4	4	4	4
Hard Corals	1	2	2	3	3	0	2	3	1	3	3	0
Sea Cucumbers (muscle)	0	4	2	3	0	1	0	2	4	1	3	3
Sea Cucumbers (hemal system)	0	4	2	0	0	3	0	2	4	3	4	3
Bivalve 1: Oysters	5	0	5	0	5	5	5	1	1	5	5	5
Bivalve 2: Chamids	0	0	0	0	1	4	0	1	1	1	4	4
Bivalve 3: Spondylids	3	0	0	0	4	0	0	4	1	4	4	4
Octopus (muscle)	1	0	0	0	1	4	0	1	1	1	3	1
Octopus (liver)	0	0	0	0	2	4	0	2	1	1	4	2[a]
Stomatopod (tail muscle)	0	0	3	0	0	4	0	0	1	1	3	1
Ascidians	0	0	0	3	0	0	0	0	0	2	3	0
Fish (muscle)	1	3	3	1	1	5	1	1	1	1	5	1
Fish (liver)	0	0	0	1	1	1	1	1	1	1	5	5[a]

The following numerical ranking was formulated, based on current findings and supportive evidence from the literature for similar or related species:
5 = bioindicator potential unequivocally established; 4 = promising bioindicator potential demonstrated; 3 = probable bioindicator potential demonstrated
2 = possible bioindicator potential demonstrated; 1 = limited bioindicator potential (e.g., due to excessive variability, or low a accumulation capacity
associated with restricted uptake and/or rapid turnover rate); 0 = insufficient data available to evaluate bioindicator potential; a = determined as PAH metabolites in bile

tin, and PCB, and very likely have good bioindicator potential for chromium, mercury, zinc, and PAHs. *Holothuria atra*, is particularly abundant around much of Guam. The feeding sorties of this species are restricted to within relatively small areas and so tissue contaminant levels should be reasonably representative of the collection sites. The tagging and transplanting of these organisms also offers an attractive means of monitoring the biological availability of sediment-bound pollutants in areas where they are not common.

The utility of fish as bioindicators of mercury and PCBs is now well established and further supported by the data presented during the present work. In selecting any particular species of fish for monitoring purposes, it is important that its migratory habits are known. It cannot be assumed that contaminant levels in a fish are representative of their capture site, particularly if it is a migratory species. Usual candidates are demersal species or territorial species with restricted ranges. One such candidate identified during the present survey was the lizard-fish, *Saurida gracilis*. This pisciverous species is extremely common and easily captured by hook and line. Moreover, it has a relatively large liver that adequately supports the tissue requirements for analysis.

1.2 Sample Variability:

How well a bioindicator reflects changes in the ambient availability of a contaminant is determined largely by the degree of variability encountered in the population sampled. The more variable the tissue levels, the less reliable the organism becomes, and the greater the number of individuals required to detect a given level of change. Such variability can essentially be divided into two broad categories, namely that which can be reduced or eliminated by the investigator, as opposed to that which cannot. Controllable variations include parameters such as the age/size, growth, fitness, sex and reproductive condition of the individuals sampled, in addition to differences related to their position on the shore and/or in the water column. Uncontrollable variations may be ascribed to regional and seasonal differences in temperature and salinity, and includes the inherent, natural variability normally encountered between individuals of the same species as a result of subtle variations in genetic make-up, metabolic efficiencies, health and well-being. Failure to address these variables during the initial design phase of a monitoring program can produce data that are extremely noisy and often highly misleading.

1.3 Program Design:

Pollution monitoring programs involving the use of bioindicators generally have one or both of the following objectives:

- To identify spatial difference in contaminant abundance within an area or region, including the delineation of 'hot-spots'
- To evaluate short- and long-term temporal changes in contaminant abundance within any particular site or area

Both objectives are separate from one another and have specific requirements (Phillips and Segar 1986). For example, if the primary goal is to delineate spatial difference in contaminant bioavailability, it is important to adopt a synchronous sampling regime to ensure that temporal fluctuations in pollutant availability at each of the sites studied do not interfere with the data.

On the other hand, monitoring temporal trends in pollutant abundance within any particular site requires a sampling frequency that is determined by the biological half-life of the contaminant of interest if an uninterrupted record of its biological availability is to be obtained. In addition, the influence of seasonal changes in temperature, salinity and reproductive status on pollutant levels within the bioindicator needs to be addressed in order to identify 'real' changes in a contaminant's availability.

Both objectives also have a number of common requirements that must be met in order to optimize the survey design. For example, it is customary to standardize on a specific size or size range of individuals in order to eliminate any possible age-dependant variability in contaminant levels (e.g., mercury in fish). This can be done in one of two ways, either by selecting a specific size range, or by taking what is available and normalizing the data to a specific size by regression techniques. Another requirement common to both monitoring objectives calls for the standardization of collection sites on the shore or in the water column, and this is particularly important in areas receiving freshwater inflow or in waters that are highly stratified. Finally, it is necessary to identify the bioindicator's inherent variability in tissue pollutant levels in order to optimize sample size for the desired resolution.

1.4 Site Selection:

For monitoring the spatial and temporal variability in pollutant abundance in Guam's nearshore waters, a number of sites ranging from 'suspected as contaminated' to 'control' or 'background' should be chosen. The selection of potential study sites can be based on a number of criteria, including the following:

- Existence of previous data
- Proximity to important fisheries and other edible marine resources
- Proximity to potential sources of contamination (marinas, harbor activities, discharges from stormwater outlets, sewage treatment plants etc)
- Proximity to population centers
- Proximity to popular tourist and recreational fishing areas
- Proximity to major river mouths

The control site should be located offshore (e.g., Double Reef) away from the influence of short-term fluctuations attributable to coastal activities. The distance between sites will vary according to monitoring needs. However, sites are normally much closer together for hot-spot delineation than they are for monitoring trends at more remote locations.

2. Evaluation of Data in Relation to Current Food Standards

Some brief comments are appropriate here regarding contaminant levels measured in edible fish and shellfish during the present study, in relation to national and international food standards. All standards included in the following discussion are given on a wet weight basis.

Food standards in the U.S. are under the jurisdiction of the U.S. Food and Drug Administration (FDA) with non-regulatory technical guidance provided by the U.S. EPA. Current standards for metals and PCBs are listed in Table 31 along with those from various other countries. There are no national or international food standards for PAHs at this time (Law *et al.* 1997).

Table 31

Compilation of Legal Limits for Hazardous Metals and PCBs in Fish and Fishery Products[a] (all values as µg/g wet weight)

Country	As	Cd	Cr	Cu	Hg	Ni	Pb	Sn	Zn	PCBs
Australia	1	2	-	10 (70)	0.5[b]	-	0.5	150	150 (1000)	0.5
Brazil	-	-	-	-	0.5	-	-	-	-	-
Canada	3.5	-	-	-	0.5	-	0.5	-	-	-
Chile	1	0.5	-	10	-	-	2.0	-	100	-
Denmark	-	-	-	-	0.5	-	-	-	-	-
Ecuador	1	-	-	10	1.0	-	5.0	-	-	-
Finland	5	-	-	-	1.0	-	2.0	-	-	-
France	-	-	-	-	0.5, 0.7	-	-	-	-	-
Germany	-	0.5	-	-	1.0	-	0.5	-	-	-
Greece	-	-	-	-	0.7	-	-	-	-	-
Hong Kong	1.4	2	1	-	0.5	-	6.0	-	-	-
India	1	-	-	10	0.5[b]	-	5.0	-	50	-
Israel	-	-	-	-	0.5	-	-	-	-	-
Italy	-	-	-	-	0.7[b]	-	2.0	-	-	-
Japan	-	-	-	-	0.3-0.4	-	-	-	-	-
Korea	-	-	-	-	0.5	-	-	-	-	-
Netherlands	-	0.5-1.0	-	-	1.0[b]	-	0.5, 2.0	-	40	-
New Zealand	1	1	-	30	0.5[b]	-	2.0	-	-	-
Philippines	30	-	-	-	0.5	-	0.5	-	-	-
Poland	4	-	-	10-30	-	-	1.0-2.0	-	30-50	-
Spain	-	-	-	-	0.5	-	-	-	-	-
Sweden	-	-	-	-	1.0[b]	-	1.0-2.0	-	-	-
Switzerland	-	0.1	-	-	0.5	-	1.0	-	-	-
Thailand	2	-	-	20	0.5	-	1.0	-	-	-
United Kingdom	1	-	-	20	0.5	-	2.0-10	-	50	-
United States	76, 86[d]	3, 4[d]	12, 13[d]	-	1.0[c]	70-80[d]	1.5-1.7	-	-	2.0
U.S.S.R	-	-	-	-	-	-	-	-	-	-
Venezuela	0.1	0.1	-	10	0.1-0.5	-	2.0	-	-	-
Zambia	3.5-5.0	-	-	100	0.2-0.3	-	0.5-10	-	100	-

a = modified after Nauen 1983 (unmodified table cited in USEPA 1989). Note: Food standards are continually being updated and those listed above may not be current for countries other than the United States and Australia
b = as total mercury; c = as organic mercury; d = non-enforceable U.S. FDA guidance levels for crustacans (lower value) and mollusks (higher value) (U.S. FDA 1998); Australian values in parenthesis are for oysters; dashes indicate no data

It can be seen from Table 3, that the only enforceable heavy metal standard for seafood in the U.S. is that for mercury. An 'action level' is currently set at 1.0 µg/g and is for organic (methyl) mercury rather than total mercury. There is some controversy over this limit, with U.S. EPA maintaining that it should be 3-5 times lower to adequately protect consumers. As a consequence the standard is currently being re-evaluated (USFDA 1998).

A number of other countries have set lower limits for mercury. Japan for example, exercises a 0.3 µg/g standard for total mercury while the maximum permissible level in Australia and Canada is 0.5 µg/g. In our study, only four out of 75 fish analyzed exceeded 0.3 µg/g. Of these, three were above 0.5 µg/g and only one was higher than 1.0 µg/g. Interestingly, all four fish were captured in Apra Harbor.

The only other enforceable FDA food standard that is applicable to this study is the 2.0 µg/g tolerance level established for total PCBs. This standard is approximately one order of magnitude higher than the highest value determined during the present study, assuming that total PCBs are roughly equivalent to twice the sum of all detectable congeners (Σ_{20}PCB). Germany and Sweden have set identical limits to the U.S standard. However, the recently introduced Australian standard for PCBs in fish is significantly lower and stands at 0.5 µg/g (NFA 1992, cited in Roach and Runcie 1998).

The U.S. FDA has recently prepared a series of non-enforceable guidelines for arsenic (total), cadmium, chromium, lead and nickel in shellfish (crustaceans and mollusks). Proposed 'levels of concern' are listed in Table 31 and assume a shellfish consumption of 15 g/person/day. One has to wonder at the adequacy of these standards for populations that rely heavily on the sea for their primary source of protein. Fortunately, levels of all five elements determined in edible species from Guam were well below the FDA proposed limits, with the possible exception of arsenic in octopus – a popular food on Guam. This single specimen from Apra Harbor contained 19.3 µgAs/g wet weight in its tentacles. Persons consuming in excess of 60 g of octopus on a daily basis could, therefore, be at risk of deleterious health effects.

Oysters are another group of mollusks that are commonly consumed locally. Indeed, they are a favored dish in many parts of the world, including the U.S. The absence of an FDA food standards for copper and zinc is, therefore, surprising in light of these organisms' exceptional ability to accumulate both elements. Oysters from Agana Boat Basin and Apra Harbor were heavily contaminated with copper and zinc and frequently contained levels of both elements in excess of the appropriate current Australian food standards (see Table 31).

3. Recommendations for Future Work

This preliminary investigation generally suggests that Guam's harbor environments are relatively clean by world standards. However, there is evidence of small localized hot-spots for several metals and PCBs in Agana Boat Basin and Apra Harbor. We strongly suspect there are others, particularly in the inner Apra Harbor area where high levels of several heavy metals, including tin, are known to exist. Other areas of suspected enrichment include the anchorage and mooring facilities abutting the Piti Channel, and in Sasa Bay. The mangroves in Sasa Bay were total destroyed by an oil spill a number of years ago and, despite intensive

cleanup and replanting activities in this area, the underlying sediments remain heavily contaminated. The extent of the PAH contamination here is unknown but is likely to be considerable. We also know very little about contaminant levels residing in sediments and biota outside of the harbor environments. In all probability they are low, although certain areas close to river mouths may be considerably enriched. The Pago River mouth is an obvious focal point for future monitoring studies in view of the drainage waters it receives from the Ordot landfill. Likewise, for coastal areas close to sewer outlets and wastewater discharges in Agat, Merizo, Yona, Tamuning, and Agana. We also need to establish baseline contaminant levels for our cleaner, relatively unimpacted stretches of coastline. Without such vital information, the effects of future developments in these areas will be difficult to assess.

BIBLIOGRAPHY

Abdullah, M.I. and L.G. Royle (1974). A Study of the Dissolved and Particulate Trace Erlements in the Bristol Channel. *Journal of the Marine Biological Association of the UK*, 54: 581-597.

Agadi, V.V., N.B. Bhosle and A.G. Untawale (1978). Metal Concentration in Some Seaweeds of Goa (India). *Botanica Marina*, XXI: 247-250.

Ahsanullah, M. (1976). Acute Toxicity of Cadmium and Zinc to Seven Invertebrate Species from Western Port, Victoria. *Australian Journal of Marine and Freshwater Research*, 27:187-196.

Albaiges, J., J. Algaba, P. Arambarri, F. Cabrera, G. Baluja, L. M. Hernandez and J. Castro-viejo (1987a). Budget of Organic and Inorganic Pollutants in the Donano Natural Park (Spain). *Science of the Total Environment*, 63: 13-28.

Albaiges, J., A. Farran, M. Soler and A. Gallifa (1987b). Accumulation and Distribution of Biogenic and Pollutant Hydrocarbons, PCBs, and DDT in Tissues of Western Mediterranean Fishes. *Marine Environmental Research*, 22: 1-18.

Alexander, G.V. and D.R. Young (1976). Trace Metals in Southern California Mussels. *Marine Pollution Bulletin*, 7: 7-9.

Alzieu, C., J. Sanjuan, J. P. Deltreil, and M. Borel (1986). Tin Contamination in Arachon Bay: Effects on Oyster Shell Anomalies. *Marine Pollution Bulletin*, 17: 494-498.

Amesbury, S.S., F.A. Cushing, R.K. Sakamoto (1986). Guide to the Coastal Resources of Guam: Vol. 3. Fishing on Guam. *University of Guam Marine Laboratory Contribution Number 225*. University of Guam Press. 109 pp.

Amico, V., G. Orient, M. Piattelli and C. Tringali (1979). Concentrations of PCB, BHCs and DDTs Residues in Seaweeds of the East Coast of Sicily. *Marine Pollution Bulletin*, 10: 177.

Babji, A.S., M.S. Embong and W.W. Woon (1979). Heavy Metal Contents in Coastal Water Fishes of West Malaysia. *Bulletin of Environmental Contamination and Toxicology*, 23: 830-836.

Baker, C.W. (1977). Mercury in Surface Waters of Seas Around the United Kingdom. *Nature*, 270: 230-232.

Ballschmiter, K. and M. Zell (1980). Analysis of Polychlorinated Biphenyls (PCB) by Glass Capillary Gas Chromatography. *Fresenius Z. Analytical Chemistry*, 302: 20-31.

Ballschmiter, K., W. Schäfer and H. Buchert (1987). Isomer-Specific Identification of PCB Congeners in Technical Mixtures and Environmental Samples by HRGC-ECD and HRGC-MSD. *Fresenius Z. Analytical Chemistry*, 326: 253-257.

Barnard, L.A., I.G. Macintyre and J.W. Pierce (1974). Possible Environmetal Index in Tropical Reef Corals. *Nature*, 252: 219-220.

Baumard, P., H. Budzinski, P. Garrigues, J. C. Sorbe, T. Burgeot and J. Bellocq (1998). Concentrations of PAHs (Polycyclic Aromatic Hydrocarbons) in Various Marine Organisms in Relation to Those in Sediments and to Trophic Level. *Marine Pollution Bulletin*, 36: 951-960.

Bebbington, G.N., N.J. Mackay, R. Chvojka, R.J. Williams, A. Dunn and E.H. Auty (1977). Heavy Metals, Selenium and Arsenic in Nine Species of Australian Commercial Fish. *Australian Journal of Marine and Freshwater Research*, 28: 277-286.

Beckett, J.S. amd H.C. Freeman (1974). Mercury in Swordfish and Other Pelagic Species from the Western Atlantic Ocean. *Proceedings of the International Billfish Symposium, Pt. 2. U.S. Department of Commerce, NOAA Technical Report NMFS SSRF*, 154-159.

Belt Collins Hawaii (1993). Final Environmental Impact Statement of Proposed Facilities Development and Relocation of Navy Activities to the Territory of Guam from th Republic of the Philippines. *U.S. Navy Pacific Division Naval Facilities Engineering Command.* Prepared by the U.S. Navy in Cooperation with the U.S. Air Force and the U.S. Army Corps of Engineers, July 1993.

Benlahcen, K.T., A. Chaoui, H. Budzinski, J. Bellocq and P. Garrigues (1997). Distribution and Sources of Polycyclic Aromatic Hydrocarbons in Some Mediterranea Coastal Sediment. *Marine Pollution Bulletin*, 34: 298-305.

Benoit, G., J.M. Schwantes, G.S. Jacinto and M.R. Goud-Collins (1994). Preliminary Study of the Redistribution and Transformation of HgS from Cinnabar Mine TailingsDeposited in Honda Bay, Palawan, Philippines. *Marine Pollution Bulletin*, 28: 754-759.

Benson, A.A. (1983). Arsenic Metabolism in *Tridacna. XV Pacific Science Congress*, New Zealand, February 1983 (Abstract only).

Bernhard, M. and A. Zattera (1975). Major Pollutants in the Marine Environment. In: *Marine Pollution and Marine Waste Disposal*. (Pearson and Frangipane (eds.)). Pergamon Press New York. Pp. 195-300.

Bertine , K.K. and E.D. Goldberg (1972). Trace Elements in Clams, Mussels, and Shrimp. *Limnology and Oceanography*, 17: 877-884.

Beukema, A.A., G.P. Hekstra and C. Venema (1986). The Netherlands Environmental Policy for the North Sea and Wadden Sea. *Environmental Monitoring and Assessment*, 7: 117-155.

Black, W.A.P. and R.L. Mitchell (1952). Trace Elements in the Common Brown Algae and in Seawater. *Journal of the Marine Biological Association of the UK*, 30: 575-583.

Bligh, E.G. and F.A.G. Armstrong (1971). Marine Mercury pollution in Canada. A Preliminary Report. *International Council for the Exploration of the Sea,*, 27 September-October 6, Paper C.M. 1971/E:34.

Boehm, P. D. and P. Hirtzer (1982). Gulf and Atlantic Survey for Selected Organic Pollutants in Fish. NOAA Technical Report NMFS (National Marine Fisheries Service) Circular. NEC-13, 101pp.

Boehm, P.D., D.L. Fiest, P. Hirzer, L. Scott, R. Norstrom and R. Engelhardt (1982). A Biochemical Assessment of the BIOS Experimental Spills: Transport, Pathways and Fate of Petroleum in Benthic Animals. In: *Proceedings of the Arctic Marine Oil Spill Program Technical Seminar*, Edmonton, Alberta, Canada, June 15-17 1982. Pp.581-618.

Bohn, A, and B.W. Fallis (1978). Metal Concentrations (As, Cd, Cu, Fe, and Zn) in Shorthorn Sculpins, *Myoxocephalus scorpius* (Linnaeus), and Arctic Char, *Salvelinus alpinus* (Linnaeus) from the Vicinity of Strathcona Sound, Northwest Territories. *Water Research*, 12:659-663.

Bok, C.S. and W.M. Keong (1976). Heavy Metals in Marine Biota from Coastal Waters around Singapore. *Journal of the Singapore National Academy of Science.* 5: 47-53.

Bowen, H.J.M. (1979). *Environmental Chemistry of the Elements*. Academic Press, New York and London

Boyden, C. (1975). Distribution of Some Trace Metals in Poole Harbour, Dorset. *Marine Pollution Bulletin*, 6: 180-187.

Boyle, E.A., S.S. Huested and S.P. Jones (1981). On the Distribution of Copper, Nickel, and Cadmium in the Surface Waters of the North Atlantic and the North Pacific Ocean. *Journal of Geophysical Research*, 86: 8048-8066.

Bright, D.A., S.L. Grundy and K.J Reimer (1995). Differential Bioaccumulation of Non-*Ortho*-Substituted and Other PCB Congeners in Coastal Arctic Invertebrates and Fish. *Environmental Science and Technology*, 29: 2504-2512.

Brooks, R.R. and M.G. Rumsby (1965). The Biogeochemistry of Trace Element Uptake by Some New Zealand Bivalves. *Limnology and Oceanography*, 10: 521-527.

Brown, L.S. and M.C. Holley (1982). Metal Levels Associated with Tin Dredging and Smelting and their Effect Upon Intertidal Reef Flats at Ko Phuket, Thailand. *Coral Reefs*, 1: 131-137.

Brown, D.W., B.B McCain, B.H. Horness, C.A. Sloan, K.L Sloan, K.T. Tilbury, S.M., S.M. Pierce, D.G. Burrows, S-L. Chan, J.T. Landahl and M. Krahn (1998). Status, Correlations and Temporal Trends of Chemical Contaminants in Fish and Sediment from Selected Sites on the Pacific Coast of the USA. *Marine Pollution Bulletin*, 37: 67-85.

Brownawell, B.J. and Farrington (1986). Biogeochemistry of PCBs in Interstitial Waters of a Coastal Marine Sediment. *Geochimica et Cosmochimica Acta*, 50: 157-169.

Bruland, K.W. (1980). Oceanographic Distribution of Cadmium, Zinc, Nickel, and Copper in the North Pacific. *Earth Planet Scientific Letters*, 4: 176-198.

Bruland, K.W. and R.P. Franks (1981). Mn, Ni, Cu, Zn, and Cd in the Western North Atlantic. In: *Trace Metals in Seawater*, (C.W. Wong, E. Boyle, K.W. Bruland, J.D. Burton and E.D. Goldberg (eds.)). Plenum Press New York, 1983. Pp. 395-414.

Bruland, K.W., G.A. Knauer and J.H. Martin (1978). Zinc in Northeast Pacific Waters. *Nature*, 271: 741-743.

Bryan, G. (1964). Zinc Regulation in the Lobster, *Homerus vulgaris*, I. Tissue Zinc and Copper Concentrations, *Journal of the Marine Biological Association of the UK*, 44: 549-563

Bryan, G. (1966). The Metabolism of Zinc and Zn^{65} in Crabs, Lobsters, and Freshwater Crayfish. In: *Radioecological Concentration Processes* (B Aberg and F.P. Hungate (eds.)). Pergamon Press, Oxford. Pp. 1005-1016.

Bryan, G. (1967). Zinc Regulation in the Freshwater Crayfish (Including Some Comparative Copper Analysis). *Journal of Experimental Biology*, 46: 281-296.

Bryan, G. (1968). Concentrations of Zinc and Copper in the Tissues of Decapod Crustaceans, *Journal of the Marine Biological Association of the UK*, 48: 303-321.

Bryan, G. (1971). The Effects of Heavy Metals (Other than Mercury) on Marine and Estuarine Organisms. *Proceedings of the Royal Society of London, Series B.* 177: 389-410.

Bryan, G. (1976). Heavy Metal Contamination in the Sea. In: *Marine Pollution* (R. Johnson (ed.)). Academic Press, London • New York • San Francisco 185-302.

Bryan, G. and P.E. Gibbs (1991). Impact of Low Concentrations of Tributyltin on Marine Organisms: A Review. In: *Ecotoxicology of Metals: Current Conceptsand Applications* (M.C. Newman and A.W. McIntosh (eds.)). Lewis Publishers, Ann Arbor, Boca Raton, Boston. Pp323-361.

Bryan, G. and L.G. Hummerstone (1973a). Brown Seaweed as an Indicator of Heavy Metals in Estuaries in South-West England, *Journal of the Marine Biological Association of the U.K.*, 53: 705-720.

Bryan, G. and L.G. Hummerstone (1973b). Adaptation of the Polychaete *Neries diversicolor* to Estuarine Sediments Containing High Concentrations of Zinc and Cadmium. *Journal of the Marine Biological Association of the U.K.*, 53: 839-857.

Bryan, G. and L.G. Hummerstone (1977). Indicators of Heavy Metal Contamination in the Looe Estuary (Cornwall) with Particular Regard to Silver and Lead. *Journal of the Marine Biological Association of the United Kingdom*, 57: 75-92.

Bryan, G.W. and W.J. Langston (1992). Bioavailability, Accumulation and Effects of Heavy Metals in Sediments with Special reference to United Kingdom Estuaries: A Review. *Environmental Pollution*, 76: 89-131.

Bryan G.W., W.G. Langston, L.G. Hummerstone and G.R. Burt (1985). A Guide to the Assessment of Heavy-Metal Contamination in Estuaries Using Biological Indicators. *Marine Biological Association of the United Kingdom, Occasional Publication*, Number 4, 92 pp.

Bryan, G. and H. Uysal (1978). Heavy Metals in the Burrowing Bivalve *Scrobicularia plana* from the Tamar Estuary in Relation to Environmental Levels. *Journal of the Marine Biological Association of the United Kingdom*, 58: 89-108.

Buddemeier, R.W. (1978). Schlerochronology: A Data Source for Reef Systems Models. *Atolls Research Bulletin*, 220: 25-33.

Buggiani, S.S. and C. Vannuchi (1980). Mercury and Lead Concentrations in Some Species of Fish from the Tuscan Coast (Italy). *Bulletin of Environmental Contamination and Toxicology*, 25: 90-92.

Burdon-Jones and Denton (1984a). Metals in Marine Organisms from the Great Barrier Reef Province. Part 1, Baseline Survey. *Final Report to the Australian Marine Science Technologies Committee, Canberra, Australia.* 155 pp.

Burdon-Jones and Denton (1984b). Metals in Marine Organisms from the Great Barrier Reef Province. Part 2, Regional and Seasonal Variations. *Final Report to the Australian Marine Science Technologies Committee, Canberra, Australia.* 161 pp.

Burdon-Jones, C., G.R.W. Denton, G.B. Jones and K.A. McPhie (1975). Long-Term Sub-Lethal Effects of Metals on Marine Organism. Part I Baseline Survey. *Final Report to the Water Quality Council of Queensland, Australia.* 105 pp.

Burdon-Jones, C., G.R.W. Denton, G.B. Jones and K.A. McPhie (1977). Long-Term Sub-Lethal Effects of Metals on Marine Organism. Part 2 Regional and Seasonal Variations. *Final Report to the Water Quality Council of Queensland, Australia*. 45 pp.

Burdon-Jones, C., G.R.W. Denton, G.B. Jones and K.A. McPhie (1982). Regional and Seasonal Variations of Trace Metals in Tropical Phaeophyceae from North Queensland. *Marine Environmental Research*, 7: 13-30.

Burdon-Jones, C. and D. Klumpp (1979). Identification of Sentinel Organisms for Monitoring Metals in Tropical Waters. *Final Report to the Department of Science and the Environment, Australia*. 51 pp.

Burnett, M., A. Settle, D. Ng and C.C. Patterson (1977). Impact of Man on Coastal Marine Ecosystems. In: *Lead in the Environment*, (M. Branica and Z. Konrad (eds.)), Pergamon Press, Oxford, New York, Toronto, Sydney, Paris, Frankfurt, 1980. Pp. 7-13.

Butler, P.A., L. Andrén, G.J. Bond, A. Jernlöv and D.J. Reish (1971). Monitoring Organisms. In: *FAO Technical Conference on Marine Pollution and its Effect on Living Resources and Fishing*, Rome 1970. *Supplement 1: Report of the Seminar on Methods of Detection, Measurement and Monitoring of Pollutants in the Marine Environment.* FAO Fisheries Report No. 99, Supplement 1: 101-112.

Butterworth, J. P. Lester, and G, Nickless (1972). Distribution of Heavy Metals in the Severn Estuary. *Marine Pollution Bulletin*, 3: 72-74.

Cerniglia, C.E. and M.A. Heitkamp (1989). Microbial Degradation of Polycyclic Aromatic Hydrocarbons (PAH) in the Aquatic Environment. . In: *Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment* (U. Varanasi (ed.)), Press Inc. Pp. 41-68.

Cheevaparnapivat, V. and P. Menasveta (1979). Total and Inorganic Mercury in Marine Fish of the Upper Gulf of Thailand. *Bulletin of Environmental Contamination and Toxicology*, 23: 291-299.

Cocchieri, R. A., A. Arnese and A. Minicucci (1990). Polycyclic Aromatic Hydrocarbons in Marine Organisms from Italian Central Mediterranean Coasts. *Marine Pollution Bulletin*, 21: 15-18.

Contardi, V., R. Capelli, T. Pellacani and G. Zanicchi (1979). PCBs and Chlorinated Pesticides in Organisms from the Ligurian Sea. *Marine Pollution Bulletin*, 10: 307.

Courtney W.A.M. and G.R.W. Denton (1976). Persistence of Polychlorinated Biphenyls in the Hard-Clam (*Mercenaria mercenaria*) and the Effect upon the Distribution of these Pollutants in the Estuarine Environment. *Environmental Pollution*, 10: 55-64.

Culkin F. and J.P. Riley (1958). The Occurrence of Galium in Marine Organisms. *Journal of the Marine Biological Association of the UK*, 37: 607-615.

Cutshall, N. and R. Holton (1972). Metal Analysisin I.D.O.E. Baseline Studies. In: *Proceedings of the I.D.O.E. Workshop on Baseline Studies*, Brookhaven National Laboratory 24-26 May, 1972.

Denton, G.R.W. (1974). *The Uptake and Elimination of Polychlorinated Biphenyls (PCB) in the American Hard-Shell Clam, Mercenaria mercenaria.* Ph.D. Thesis London 1974. 194 pp.

Denton, G.R.W. and W.G. Breck (1981). Mercury in Tropical Marine Organisms from North Queensland. *Marine Pollution Bulletin*, 12: 116-121.

Denton, G.R.W. and C. Burdon-Jones (1982). Influence of Temperature and Salinity on the Uptake, Distribution and Depuration of Mercury Cadmium and Lead by the Black-Lip Oyster, *Saccostrea echinata*. *Marine Biology*, 64: 317-326.

Denton, G.R.W. and C. Burdon-Jones (1982). The Influence of Temperature and Salinity Upon the Acute Toxicity of Heavy metals to the Banana Prawn (*Panaeus murguiensis* de Man). *Chemistry in Ecology*, 1: 131-143.

Denton, G.R.W. and C. Burdon-Jones (1986a). Trace Metals in Algae from the Great Barrier Reef. *Marine Pollution Bulletin*, 17: 98-107.

Denton, G.R.W. and C. Burdon-Jones (1986b). Trace Metals in Corals from the Great Barrier Reef. *Marine Pollution Bulletin*, 17: 209-213.

Denton, G.R.W. and C. Burdon-Jones (1986c). Trace Metals in Fish from the Great Barrier Reef. *Marine Pollution Bulletin*, 17: 201-209.

Denton, G.R.W. and C. Burdon-Jones (1986d). Environmental Effects on Toxicity of Heavy Metals to Two Species of Tropical Marine Fish from Northern Australia. *Chemistry in Ecology*, 2: 233-249.

Denton, G.R.W. and C. Burdon-Jones (1986e). Trace Metals in Seawater from the Great Barrier Reef. *Marine Pollution Bulletin*, 17: 96-98.

Denton, G.R.W. and Heitz, L.F. 1991. *Tridacna:* Sentinels of Heavy Metal Pollution in Torres Strait Waters - a Critical Evaluation. In: *Sustainable Development for Traditional Inhabitants of the Torres Strait Region.* (D. Lawrence and T. Cansfield-Smith (eds.)) Great Barrier Reef Marine Park Authority Workshop Series No. 47. Australian Government Publishing Service, Canberra, Australia. Pp. 311-331

Denton, G.R.W. and Heitz, L.F. 1993. Heavy Metal Uptake and Loss in the Burrowing Clam *Tridacna crocea:* Implications from a Public Health and Mariculture Viewpoint. In: *Biology and Mariculture of Giant Clams.* (W.K. Fitt (ed.)). ACIAR Monograph Series No. 47. Pirie Printers, Canberra, Australia. Pp. 119-132

Denton G.R.W., H.R. Wood, L. P. Concepcion, H.G. Siegrist, V.S. Eflin, D.K. Narcis and G.T Pangelinan (1997). Analysis of In-Place Contaminants in Marine Sediments from Four Harbor Locations on Guam. A Pilot Study. *WERI Technical Report No. 81,* 120 pp.

De Voogt, P., D.E. Wells, L. Reutergårdh and U.A Th. Brinkman (1990). Biological Activity, Determination and Occurrence of Planar, Mono- and Di-Ortho PCBs. *International Journal of Environmental Analytical Chemistry,* 40: 1-46.

Dight, I. and W. Gladstone (1994). The Torres Strait Baseline Study. *Final Report of the Pilot Study.* Great Barrier Reef Marine Park Authority, Townsville, Australia

Dujmov, J. and P. Sucevic (1989). Contents of Polycyclic Aromatic Hydrocarbons in the Adriatic Sea Determined by UV-Fluorescence Spectroscopy. *Marine Pollution Bulletin,* 20: 405-409.

Einga, H. (1977). Studies on the Accumulation of Heavy Metals in Fish from Japan Seas. *Journal of the Mara Medical Association,* 28: 362-368.

Eisler, R. (1981). *Trace Metal Concentrations in Marine Organisms.* Pergamon Press, New York • Oxford • Toronto • Sydney • Paris • Frankfurt. 685 pp.

Eisler, R. (1987). Polycyclic Aromatic Hydrocarbon Hazards to Fish, Wildlife and Invertebrates: A Synoptic Review. *U.S. Fish and Wildlife Service Biological Report 85(1.11).* 81 pp.

El Deeb, K. Z. and E. H. El Ebiary (1988). Total Aromatic Hydrocarbon Content in the Muscle and Liver Lipid Extracts of Two Seabream Fishes from the Arabian Gulf. *Arabian Gulf Journal of Scientific Research and Agricultural. Biological Science,* 86: 139-151.

El-Gendy, K.S., A.A. Abdalla, H.A. Aly, G. Tantawy and A.H El-Sebae (1991). Residue Levels of Chlorinated Hydrocarbon Compounds in Water and Sediments from Nile Branches in the Delta, Egypt. *Journal of Environmental Science and Health Part B, Pesticides, Food Contamination and Agricultural Wastes.* 26: 15-36.

El Nabawi, A., B. Heinzow and H. Kruse (1987). Residue Levels of Organochlorine Chemicals and Polychlorinated Biphenyls in Fish from the Alexandria Region, Egypt. *Archives of Environmental Contamination and Toxicology,* 16: 689-696.

Eustace, I.J (1974). Zinc, Cadmium, Copper and Manganese in Species of Finfish and Shellfish Caught in the Derwent Estuary. *Australian Journal of Marine and Freshwater Research,* 25: 209-220.

Everaarts, J.M. E.M. Van Weerlee, C.V. Fischerm and Th. J. Hillebrand (1998). Polychlorinated Biphenyls and Cyclic Pesticides in Sediments and Macro-Invertebrates from the Coastal Zone and Continental Slope of Kenya. *Marine Pollution Bulletin,* 36: 492-500.

Farrington, J.W., E.D. Goldberg, R.W. Riseborough, W.R. Martin and V.T. Bowen (1983). U.S. "Mussel Watch" 1976-1978: An Overview of the Trace Metal, DDE, PCB Hydrocarbon and Artificial Radionuclide Data. *Environmental Science and Technology,* 17: 490-496.

Feldman, C. (1974). Preservation of Dilute Mercury Solutions. *Analytical Chemistry,* 46: 99-102.

Forster, W.O., E.D. Wood and F. Padovani (1972). A Study program to Identify Problems Related to Oceanic Environmental Quality in the Caribbean. In: *Proceedings of the I.D.O.E. Workshop on Baseline Studies,* Brookhaven National Laboratory, 24-26 May 1972.

Förstner, U (1980). In: *Chemistry and Biogeochemistry of Estuaries,* (E. Olausson and I. Cato (eds.)). John Wiley & Sons, Chichester, U.K. p. 37.

Förstner, U. and G.T.W. Wittman (1979). *Metal Pollution in Aquatic Environments.* Springer, New York. 486 pp.

Fowler, S. (1977). Trace Elements in Zooplankton Particulate Products. *Nature,* 269: 51-53.

Fowler, S. (1987). PBBs and the Environment: The Mediterranean Marine Ecosystem. In: *PCBs and the Environment Volume III* (J.S. Waid (ed.)). CRC Press Inc. Boca Raton, Florida. Pp. 209-239.

Fowler, S.W., J.W. Readman, B. Oregioni, J.P. Villeneuve and K. McKay (1993). Petroleum Hydrocarbons and Trace Metals in Nearshore Gulf Sediments and Biota Before and after the 1991 War: An Assessment of Temporal and Spatial Trends. *Marine Pollution Bulletin* 27: 171-182.

Fowler, S.W. and B. Oregioni (1976). Trace Metals in Mussels from the N.W. Mediterranean. *Marine Pollution Bulletin,* 7: 26-29.

Fowler, S.W. and M.Y. Unlu (1978). Factors Affecting Bioaccumulation and Elimination of Arsenic in the Shrimp, *Lysmata seticaudata. Chemosphere,* 9: 711-720.

Frankin, A. (1987). The Concentration of Metals, Organochlorine Pesticide and PCB Residues in Marine Fish and Shellfish: Results from MAFF Fish and Shellfish Monitoring Programmes. 1977-1984. *Aquatic Environmental Monitoring Report. MAFF Directorate of Fisheries Research* 16.

Fuge, R. and K.H. James (1973). Trace Metal Concentrations in Brown Seaweeds, Cardigan Bay, Wales. *Marine Chemistry,* 1: 281-293.

Fukai, R. (1965). Analysis of Trace Amounts of of Chromium in Marine Organisms by the Isotope Dilution of Cr-51. In: *Radiochemical Methods of Analysis*, International Atomic Energy Agency of Vienna. Pp. 335-351.

Giouranovits-Psyllidou, R., E. Georgakopoulos-Gregoriades and V. Vassilopoulou (1994). Monitoring of Organochlorine Residues in Red Mullet (*Mullus barbatus*) from Greek waters. *Marine Pollution Bulletin*, 28: 121-123.

Goldberg, E.D. V.T. Bowen, J.W. Farrington, G.Harvey, J.H. Martin, P.L. Parker, R.W. Risebrough, W. Robertson, E. Schneider and E. Gamble (1978). The Mussel Watch. *Environmental Conservation*, 5: 101-125.

GREG (1993) NOAA's National Status and Trends, Mussel Watch Project, Analytical Data Year VII, Geochemical and Environmental Research Group, Texas A&M University, College Station, Texas, USA.

GREG (1994) NOAA's National Status and Trends, Mussel Watch Project, Analytical Data Year VIII, Geochemical and Environmental Research Group, Texas A&M University, College Station, Texas, USA.

GREG (1995) NOAA's National Status and Trends, Mussel Watch Project, Analytical Data Year IX, Geochemical and Environmental Research Group, Texas A&M University, College Station, Texas, USA.

Greig, R.A. (1979). Trace Metal Uptake in Three Species of Mollusks. *Bulletin of Environmental Contamination and Toxicology*, 22: 643-647. *Archives of Environmental Contamination and Toxicology*, 6: 395-409.

Greig, R.A. and D. Wenzloff (1978). Metal Accumulation and Depuration in the American Oyster, *Crassostrea virginica*. *Bulletin of Environmental Contamination and Toxicology*, 20: 499-504.

Greig, R.A., D. Wenzloff, A. Adams, B. Nelso, C. Shelpuk (1977). Trace Metals in Organisms from Ocean Disposal Sites of the Middle Eastern United States.

Grimanis, A.P., D. Zafiropoulos and M. Vassilaki-Grimani (1978). Trace Elements in the Flesh and Liver of Two Fish Species from Polluted and Unpolluted Areas of the Aegean Sea. *Environmental Science and Technology*, 12: 723-726.

Gryzhanková, L.N., G.N. Sayenko, A.V Karyakin and N.V. Laktionova (1973). Concentrationsof some Metals in the Algae of the Sea of Japan. *Oceanology*, 13: 206-210.

Hall, R.A., E.G. Zook, and G.M. Meaburn (1978). National Marine Fisheries Service Survey of Trace Elements in the Fisheries Resources. *U.S. Department of Commerce NOAA Technical Report NMFS SSRF-721*. 313 pp.

Halcrow, W., D.W. Mackay and I. Thornton (1973). The Distribution of Trace Metals in Fauna in the Firth of Clyde in Relation to the Disposal of Sewage Sludge. *Journal of the Marine Biological Association of the U.K.*, 53: 721-739.

Hargrave, B.T., G.C. Harding, W.P. Vass, P.E. Erickson, B.R. Fowler and V. Scott (1992). Organochlorine Pesticides and Polychlorinated Biphenyls in the Arctic Ocean Food Web. *Archives of Environmental Contamination and Toxicology*, 22: 41-54.

Harris, R.C. and C.C. Almy, Jr. (1964). A Preliminary Investigation into the Incorporation and Distribution of Minor Elements in the Skeletal Material of Scleractinian Corals. *Bulletin of Marine Science of the Gulf and Caribbean*, 14: 418-423.

Haug, A., S. Melsom, and S. Omang (1974). Estimation of Heavy Metal Pollution in Two Norwegian Fjord Area by Analysis of the Brown Alga, *Ascophyllum nodosum*. *Environmental Pollution*, 7: 179-192.

Hellou, J. (1996). Polycyclic Aromatic Hydrocarbons in Marine Mammals, Finfish, and Mollusca. In: *Environmental Contaminants in Wildlife. Interpreting Tissue Concentrations* (W. N. Beyer, G.H. Heinz and A.W. Redmon-Norwood). SETAC Special Publication Series, CRC, Lewis Publishers, Boca Raton, New York, London, Tokyo. Pp. 229-250.

Hellou, J., J. F. Payne and C. Hamilton (1993). GC-MS Analysis of Polycyclic Aromatic Compounds in Cod (*Gadus morhua*) from the Northwest Atlantic. *Environmental Pollution*, 85: 197-202.

Hellou, J., J.F. Payne, C. Upshall, L.L. Fancey and C. Hamilton (1994). Bioaccumulation of Aromatic Hydrocarbons from Sediments: A Dose Response Study with Flounder (*Pseudopleuronectes americanus*). *Archives of Environmental Contamination and Toxicology*, 27: 477-485.

Hites, R.A., R.E. Laflamme and J.G. Windsor, Jr. (1980). Polycyclic Aromatic Hydrocarbons in Marine/Aquatic Sediments, Their Ubiquity. In: *Petroleum in the Marine Environment*. American Chemical Society, Washington, DC.

Holden A. (1973). Mercury in Fish and Shellfish, A Review. *Journal of Food Technology*, 8: 1-25.

Holden, A. (1986). The Reliability of PCB Analysis. In: *PCBs in the Environment* Vol. 1, (J.S. Waid (ed.)). CRC Press, Boca Raton • Ann Arbor • Boston. Pp. 65-78.

Hope, B., S. Scatolini, E. Titus and J. Cotter (1999). Distribution Patterns of Polychlorinated Biphenyl Congeners in Water, Sediment, and Biota from Midway Atoll (North Pacific Ocean). *Marine Pollution Bulletin* (in press).

Hornung, H., D. Ravi and B.S Trugalz (1981). The Ocurrence of Mercury in Marine Algae and Some Gastropod Molluscs of the Mediterranean Shoreline of Israel. *Marine Pollution Bulletin*, 12: 387-389.

Horowitz A. and B.J. Presley (1977). Trace Metal Concentrations and Partitioning in Zooplankton, Neuston, and Benthos from the South Texas Outer Continental Shelf. *Archives of Environmental Contamination and Toxicology*, 5: 241-255.

Howard, L.S. and B.E. Brown (1984). Heavy Metals in Reef Corals. *Oceanography and Marine Biology Annual Reviews*, 22: 195-210.

Humason, A.W. and D.F. Gadbois (1982). Determination of Polynuclear Aromatic Hydrocarbons in the New York Bight Area. *Mimeo MSG-506. National Marine Fisheries Service*, Gloucester, Massachusetts.

Hutzinger, O., S. Safe and V. Zitko (1974). *The Chemistry of PCBs.* CRC Press: Cleveland Ohio.

Ireland, M.P. (1973). Result of Fluvial ZincPollution on the Zinc Content of Littoral and Sublittoral Organisms in Cardigan Bay, Wales. *Environmental Pollution*, 4: 27-35.

Irukayama, K. (1967). The Pollution of Minimata bay and Minimata Disease. *Proceedings of the 3rd International Conference on Water Pollution Research*, Munich 1966, 3: 153-180.

Irukayama, K., T. Kondo, F. Kai and M. Fujiki (1961). Studies on the Origin of the Causative Agent of Minimata Disease. I. Organic Mercury Compounds in the Fish and Shellfish from Minimata Bay. *Kumamoto Medical Journal*, 14: 158-169.

Jacob, J. (1995). Method development for the Determination of Polycyclic Aromatic Hydrocarbons (PAHs) in Environmental Matrices. In: *Quality Assurance for Environmental Analysis-Method Evaluation within the Measurements and Testing Program (BCR).* (P. Quevauviller, E. Maier and B. Griepink, (eds.)). Elsevier, Amsterdam.

James, M.O. (1989). Biotransformation and Disposition of PAH in Aquatic Invertebrates. In: *Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment* (U. Varanasi (ed.)), CRC Press Inc. Pp. 69-91.

Jones, K.C. (1986). The Distribution and Partitioning of Silver and Other Heavy Metals in Sediments Associated with an Acid Mine Drainage System. *Environmental Pollution*, 12: 249-263.

Jones, D.M., S.J. Rowland and A.G. Douglas (1986). An Examination of the Fate of Nigerian Crude Oil in the Surface Sediments of the Humber Estuary by Gas Chromatography and Gas Chromatography-Mass Spectroscopy. *International Journal of Environmental Analytical Chemistry*, 24: 222-247.

Jones, A.M., Y. Jones and W.D.P. Stewart (1972). Mercury in Marine Organisms from the Tay Region. *Nature*, 238: 164-165.

Kawano, M., S. Matsushita, T. Inoue, H. Tanaka and R. Tatsukawa (1986). Biological Accumulation of Chlordane Compounds in Marine Organisms from the Northern North Pacific and Bering Sea. *Marine Pollution Bulletin*, 17: 512-516.

Kennish, M.J. (1998). *Pollution Impacts on Marine Biotic Communities.* CRC Press, Boca Raton, New York. 310 pp.

Khristoforova, N.K., N.N. Bogdanova and A.I. Obukhov (1979). The Content of Certain Metals in Soft Tissues of the Bivalve Mollusc *Tridacna squamosa* from Islands of the Tropical Zone of the Pacific Ocean in Connection with Environmental Conditions. (Russian, English Summary). *Biologiya morya (Marine Biology Vladivostok)*, 3: 67-73.

Kim, C.Y. (1972). Studies on the Contents of Mercury, Cadmium, Lead and Copper in Edible Seaweeds in Korea. *Bulletin of the Korean Fisheries Society*, 5: 8-96.

Klumpp, D. and C. Burdon-Jones (1982). Investigation of the Potential of Bivalve Molluscs as Indicators of Heavy Metal Levels in Tropical marine Waters. *Australian Journal of Marine and Freshwater Research*, 33: 285-300.

Klumpp, D. and P.J. Peterson (1979). Arsenic and Other Trace Elements in the Waters and Organisms of an Estuary in SW England. *Environmental Pollution*, 19: 11-20.

Knauer, G.A. (1976). Immediate Industrial Effects on Sediment Mercury Concentrations in a Clean Coastal Environment. *Marine Pollution Bulletin*, 7: 112-115.

Knauer, G.A. (1977). Immediate Industrial Effects on Sediment Metals in a Clean Coastal Environment. *Marine Pollution Bulletin*, 8: 249-254.

Kurelec, B., S. Britvic, M. Rijavec, W.E.G. Muller and R.K. Zahn (1977). Benzo(*a*)pyrene Monooxygenase Induction in Marine Fish – Molecular Response to Oil Pollution. *Marine Biology*, 44: 211-216. Kurelec, B., S. Britvic and R.K. Zahn (1985).

Kurelec, B., S. Britvic and R.K. Zahn (1985). The Activation of Aromatic Amines in Some Marine Invertebrates. *Marine Environmental Research*, 17: 141

Langston, W.J. (1984). Availability of Arsenic to Estuarine and Marine Organisms: A Field and Laboratory Evaluation. *Marine Biology*, 80: 143-154.

Langston, W.J. (1985). Assessment of the Distribution and Availability of Arsenic and Mercury in Estuaries. In: *Estuarine Management and Quality Assessment* (J.G. Wilson and W. Halcrow (eds.)). Plenum Press, New York. Pp. 131-146.

Langston, W.J., G.R. Burt and M. Zhou (1987). Tin and Organotin in Water, Sediments and Benthic Organisms of Poole Harbour. *Marine Pollution Bulletin*, 18: 634-639.

Lasker, H.R. (1981). A Comparison of the Particulate Feeding Abilities of Three Species of Gorgonian Soft Coral. *Marine Ecology Progress Series*, 5: 61-67.

Law, R.J., V.J. Dawes, R.J. Woodhead and P. Matthiessen (1997). Polycyclic Aromatic Hydrocarbons (PAH) in Seawater around England and Wales. *Marine pollution Bulletin*, 34: 306-322.

Law, R.J. and J.L Biscaya (1994). Polycyclic Aromatic Hydrocarbons (PAH) – Problems and Progress in Sampling, Analysis and Interpretation. *Marine Pollution Bulletin*, 29: 235-241.

Leatherland, T.M. and J.D. Burton (1974). The Occurrence of Some Trace Metals in Coastal Organisms with Particular Reference to the Solent Region. *Journal of the Marine Biological Association of the U.K.*, 54: 457-468.

Leatherland, T.M. J.D. Burton, F. Culkin, M.J. McCartney and R.J. Morris (1973). Concentrations of Some Trace Metals in Coastal Organisms and of Mercury in Northeast Atlantic Ocean Water. *Deep Sea Research*, 20: 679-685.

Legoburu I. and L. Canton (1991). Heavy Metal Concentrations in Sedimentsfrom Pasajes Harbour, Spain. *Marine Pollution Bulletin*, 22: 207-209.

Leversee, G.J., J.P. Geisy, P.F. Landrum, S. Bartell, S. Gerould, M. Bruno, A. Spacie, J. Bowling, J. Haddock, and T. Fannin (1981). Disposition of Benzo(a)pyrene in Aquatic Systems Components: Periphyton, Chironomids, Daphnia, Fish. In: *Chemical Analysis and Biological Fate: Polynuclear Aromatic Hydrocarbons* (M. Cooke and A.J. Dennis (eds.)). Fifth International Symposium. Battelle Press, Columbus, Ohio. Pp. 357-366.

Levine E.P. (1961). *Science, N.Y.* 133: 1352.

Livingston, H.D. and Thompson, G. (1971). Trace Elements in Some Modern Corals. *Limnology and Oceanography*, 16: 786-795.

Long, E.R., D.D. McDonald, S.L. Smith and F.D. Calder (1995). Incidence of Adverse Biological Effects within Ranges of Chemical Concentrations in Marine and Estuarine Sediments. *Environmental Management*, 19: 81-97.

Louma, S. and D.J.H. Phillips (1988). Distribution, Variation, and Impact of Trace Elementsin San Francisco Bay. *Marine Pollution Bulletin*, 19: 413-425.

Lunde, G. (1977) Occurrence and Transformation of Arsenic in the Marine Environment. *Environmental Health Perspectives*, 19: 47-52.

Mackay, D., W. Shiu and K.C. Ma (1992). *Illustrated Handbook of Phisico-Chemical Properties and Environmental Fate for Organic Chemicals, Volume II Polynuclear Aromatic Hydrocarbons, Polychlorinated Dioxins, and Dibenzofurans.* Lewis Publishers, Boca Raton. 597 pp.

Mackay, N.J., R.J. Williams, J.L Kacprzac, M.N Kazacos, J. Collins and E.H. Auty (1975a). Heavy Metals in Cultivated Oysters (*Crassostrea commercialis = Saccostrea cucullata*) from the Estuaries of New South Wales. *Australian Journal of Marine and Freshwater Research*, 26: 31-46.

Mackay, N.J., M.N. Kazacos, R.J. Williams and M.I. Leedow (1975b). Selenium and Heavy Metals in Black marlin. *Marine Pollution Bulletin*, 6: 57-60

Maguire, R.J., P.T.S. Wong and J.S. Rhamey (1984). Accumulation and Metabolism of Tri-n-Butyl Cation by a Green Alga, *Ankistrodesmus falcatus*. *Canadian Journal of Fisheries and Aquatic Sciences*. 41: 537-540.

Malins, D. C., M. M. Krahn, M. S. Myers, L. D. Rhodes, D. W. Brown, C. A. Krone, B. B. McCain and S. -L. Chan (1985). Toxic Chemicals in Sediment and Biota from a Creosote-polluted Harbor: Relationships with Hepatic Neoplasms and Other Hepatic Lesions in English sole (*Parophys vetulus*). *Carcinogenesis* (*London.*), 6: 1463-1469.

Malins, D.C., B.B. McCain, D.W. Brown, S-L. Chan, M.S. Myers, J.T. Landahl, P.G. Prohaska, A.J. Friedman, L.D. Rhodes, D.G. Burrows, W.D. Gronlund and H.O. Hodgins (1984). Chemical Pollutants in Sediments and Diseases of Bottom-Dwelling Fish in Puget Sound, Washington. *Environmental Science and Technology*, 18: 705-713.

Malins, D.C., B.B. McCain, J.T. Landahl, M.S. Myers, M.M Krahn, D.W. Brown, S-L. Chan and W.T. Roubal (1988). Neoplastic and Other Diseases in Fish in Relation to Toxic Chemicals: An Overview. *Aquatic Toxicology*, 11: 43-67.

Mallet, L. (1961). Investigation of Benzo-3,4-Pyrene-Type Polycyclic Aromatic Hydrocarbons in the Fauna of Marine Environments (the Channel, Atlantic and Mediterranean). *C.R. Academy of Science (Paris) Series D*, 253: 168-170 (French).

Mallet, L., V. Perdriau and S. Perdriau (1963). Extent of Pollution by Polycyclic Aromatic Hydrocarbons of the Benzo-3,4-Pyrene-Type in the North Sea and the Glacial Arctic Ocean. *Bulletin of the Academy of Natural Medicine (Paris) Series D.* 264: 969-971 (French).

Marthinsen, I., G. Staveland, J.U. Skaare, K.I. Ugland and A. Haugland (1991). Levels of Environmental Pollutants in Male and Female Flounder (*Platichthys flesus* L.) and Cod (*Gadus morhua*) Caught During the Year 1988 Near or in the Waterway of Glomma, the Largest River in Norway. I. Polychlorinated Biphenyls. *Archives of Environmental Contamination and Toxicology*, 20: 353-360.

Marcus, J. M. and T. P. Stokes (1985). Polynuclear Aromatic Hydrocarbons in Oyster Tissue Around Three Coastal Marinas. *Bulletin of Environmental Contamination and Toxicology*,. 35: 835-844.

Martin, J.H. and Flegal (1975). High Copper Concentrations in Squid Livers in Association with Elevated Levels of Silver, Cadmium and Zinc. *Marine Biology*, 30: 51-55.

Matida, Y. and H. Kumada (1969). Distribution of Mercury in Water, Bottom Mud, and Aquatic organisms of Minimata Bay, the River Agano, and Other Waste Bodies in Japan. *Bulletin of the freshwater Fisheries Laboratory of Tokyo*, 19: 73-93.

Matsumoto, T., M. Satake, Y. Yamamoto and S. Haruna (1964). On the Microconstituent Elements in Marine Invertebrates. *Journal of the Oceanography Society of Japan*, 20: 15-19.

McElroy, A.E., J.W. Farrington and J.M. Teal (1989). Bioavailability of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment. In: *Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment* (U. Varanasi (ed.)), CRC Press Inc. Pp. 1-39.

McFarland, V.A. and J.U. Clarke (1989). Environmental Occurrence, Abundance, and Potential Toxicity of Polychlorinated Biphenyl Congeners: Considerations for a Conger Specific Analysis. *Environmental Health Perspectives*, 81: 225-239.

Melhuus, A., K.L. Seip, H.M. Seip and S. Myklestad S. (1978). A preliminary Study of the Use of Benthic Algae as Biological Indicators of Heavy Metal Pollution in Sorfjorden, Norway. *Environmental Pollution*, 15: 103-122.

Michel, J. and S. Zengel (1998). Monitoring of Oysters and Sediments in Acajutla, El Salvador. *Marine Pollution Bulletin*, 36: 256-266.

Miyake, Y. and Y. Suzuki (1983). The Concentrations and Chemical Forms of Mercury in Waters of the Western North Pacific. *Deeo Sea Research*, 30: 615-627.

Mix, M. C. and R. L. Schaffer (1983). Concentrations of Unsubstituted Polynuclear Aromatic Hydrocarbons of Bay Mussels (*Mytilus edulis*) from Oregon, USA. *Marine Environmental Research*, 9: 193-209.

Monod, J-L., P.M. Arnaud and A. Arnoux (1995). PCB Congeners in the Marine Biot of Saint Paul and Amsterdam Islands, Southern Indian Ocean. *Marine Pollution Bulletin*, 30: 272-274.

Moore, J.W. (1991). *Inorganic Contaminants of Surface Waters. Research and Monitoring Priorities*. Springer-Verlag: New York • Berlin • Heidelberg • London • Paris • Tokyo • Hong Kong • Barcelona. 334 pp.

Muller M.D., L. Renberg and G. Rippen (1989). Tributyl Tin in the Environment – Sources, Fate, and Determination. An Assessment of Present Status and Research Needs. *Chemosphere*, 18: 2015-2042.

Murray, A. J. and J. E. Portmann (1984). Metals an Organochlorine Pesticide and PCB Residues in Fish and Shellfish in England and Wales in 1976 and Trends Since 1970. *Aquatic Environmental Monitoring Report. MAFF Directorate of Fisheries Research* 10.

Murray, A.P., B.J. Richardson and C.F. Gibbs (1991). Bioconcentration Factors for Petroleum Hydrocarbons, PAHs, LABs and Biogenic Hydrocarbons in the Blue Mussel. *Marine Pollution Bulletin*, 12: 595-603.

Myklestad, S., I. Eidie and S. Melsom (1978). Exchange of Heavy Metalsin *Ascophyllum nodosum* (L) Le Jol in situ by Means of Transplanting Experiments. *Environmental Pollution*, 16: 277-284.

Naidu, S. and R.J. Morrison (1994). Contamination of Suva Harbor. *Marine Pollution Bulletin*, 29: 126-30.

Nakayama, E., H. Tokoro, T. Kuwamoto and T. Fujinaga (1981). Dissolved State of Chromium in Seawater *Nature*, 290: 768-770.

Neff, J.M. (1979). *Polycyclic Aromatic Hydrocarbons in the Aquatic Environment – Sources Fates and Biological Effects.* Applied Science Publishers, London.

NFA (1992). *Australian Food Standards Code*. National Food Authority, Australian Government Publishing Service, Canberra.

Nicholson, G. J., T. Theodoropoulos and G. J. Fabris (1994). Hydrocarbons, Pesticides, PCB and PAH in Port Phillip Bay (Victoria) Sand Flathead. *Marine Pollution Bulletin*, 28: 115-120.

Niimi, A. (1996). PCBs in Aquatic Organisms. In: *Environmental Contaminants in Wildlife. Interpreting Tissue Concentrations.* (W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood (eds.)). SETAC Special Publication Series, CRC, Lewis Publishers, Boca Raton, New York, London, Tokyo. Pp. 117-152.

Nisbet I.C.T. (1976). Criteria Document for PCBs. *Report No. 440/9-76-021.* U.S. Environmental Protection Agency, Washington, D.C.

Nishigaki, S., Y. Tamura. T. Maki, H. Yamada, K. Toba, Y. Shimamura and Y. Kimura (1973). Investigations of Mercury Levels in Tuna, Marlin and Marine Products. *A Reort of the Tokyo MetrapolitanResearch laboratory of Public Health,* 24: 239-248.

NOAA (1993a). National Status and Trends Program for Marine Environmental Quality. Sampling and Analytical Methods of the National Status and Trends Program, National Benthic Surveillance and Mussel Watch Projects 1984-1992. Volume I Overview and Summary of Methods. *National Oceanographic and Atmospheric Administration Technical Memorandum NOS ORCA 71.* July 1993. 117 pp.

NOAA (1993b). National Status and Trends Program for Marine Environmental Quality. Sampling and Analytical Methods of the National Status and Trends Program, National Benthic Surveillance and Mussel Watch Projects 1984-1992. Volume II. Comprehensive Descriptions of Complementary Measurements. *National Oceanographic and Atmospheric Administration Technical Memorandum NOS ORCA 71.* July 1993. 101 pp.

NOAA (1993c). National Status and Trends Program for Marine Environmental Quality. Sampling and Analytical Methods of the National Status and Trends Program, National Benthic Surveillance and Mussel Watch Projects 1984-1992. Volume III. Comprehensive Descriptions of Elemental Analytical Methods. *National Oceanographic and Atmospheric Administration Technical Memorandum NOS ORCA 71.* July 1993. 219 pp.

NOAA (1993d). National Status and Trends Program for Marine Environmental Quality. Sampling and Analytical Methods of the National Status and Trends Program, National Benthic Surveillance and Mussel Watch Projects 1984-1992. Volume IV. Comprehensive Descriptions of Trace Organic Analytical Methods. *National Oceanographic and Atmospheric Administration Technical Memorandum NOS ORCA 71.* July 1993. 181 pp.

Noddack, I. and W. Noddack (1939). Die Haufigkeiten der Schwermetalle in Meerestieren. *Ark. Zool.*, 32A: 1-35.

Nauen, C.E. (1983). A Compilation of Legal Limits for Hazardous Substances in Fish and Fisheries Products. *FAO Fisheries Circular No. 764.* Food and Agriculture Organization (FAO) of the United Nations, Rome, Italy. 102 pp.

O'Connor, T.P. (1992). *Mussel Watch: Recent Trends in Coastal Environmental Quality.* NOAA, Rockvill, MD.

O'Connor, T.P. (1998). Mussel Watch Results from 1986 to 1996). *Marine Pollution Bulletin,* 37: 14-19.

Olaffson, R.W. (1978). Effect of Agricultural Activity on Organochlorine Pesticides in Hard Corals, Fish and Molluscs from the Great Barrier Reef. *Marine Environmental Research,* 1: 87-95.

Onuska, F.I. (1989). Analysis of Polycyclic Aromatic Hydrocarbons in Environmental Samples. In: *Analysis of Trace Organics in the Aquatic Environment.* CRC Press Inc. Boca Raton, Florida. Pp. 205-241.

Opperhuizen, A., A.P.C. Gobas and J.M. D. Vander Steen (1988). Aqueous Solubility of Polychlorinated Biphenyls Related to Molecular Structure. *Environmental Science and Technology,* 22: 638-646.

Pantil, G.S. (1991). Correlation of Aqueous Solubility and Octonol-Water Partition Coefficient Based on Molecular Structure. *Chemosphere,* 22: 723-738.

Papadopoulu, C. (1973). The Elementary Composition of Marine Invertebrates as a Contribution to the Sea Pollution Investigation. *Proceedings of the MAMBO Meeting,* Castellabate, Italy, June 18-22, 1973. 18 pp.

Papadopoulu, C., A.P. Grimanis and I. Hadzistelios (1973). Mercury and Arsenic in a Fish Collected in Polluted and Non-Polluted Sea Waters. *Thalassia Jugoslavica,* 9: 211-218.

Papadopoulu, C., I. Hadzistelios and A.P. Grimanis (1972). Schedule of Elements Distribution in the Main Organs of Fish *Pagellus erythrinus. Hellenic Oceanography and Limnology,* XI: 9 ff.

Papadopoulu, C. and G.D. Kanias (1977). Tunicate Species as Marine Pollution Indicators. *Marine Pollution Bulletin,* 8: 229-231.

Papadopoulu, C., G.D. Kanias and E.M. Kassimati (1976). Stable Elements of Radioecological Importance in Certain Echinoderm Species. *Marine Pollution Bulletin,* 7: 143-144.

Parvaneh, V (1979). An Investigation into the Mercury Contamination of Persian Gulf Fish, *Bulletin of Environmental Contamination and Toxicology,* 23: 357-359.

Pastor, D., J. Boix, V. Fernandez, and J. Albaiges (1996). Bioaccumulation of Organochlorinated Contaminants in Three Estuarine Fish Species (*Mullus barbatus, Mugil cephalus and Dicentrarcus labrax*). *Marine Pollution Bulletin,* 32: 257-262.

Pavoni, B., C. Calvo, A. Sfriso and A.A Orio (1990). Time Trend of PCB Concentrations in Surface Sediments from a Hypertrophic, Macroalgae Populated Area of the Lagoon of Venice. *Science of the Total Environment,* 91: 13-21.

Payne, J.F. (1976). Field Evaluations of Benzopyrene Hydroxylase Induction as a Monitor for Marine Petroleum Pollution. *Science,* 191: 945-946.

Pendoley, K. (1992). Hydrocarbons in Rowley Shelf (Western Australia) Oysters and Sediments. *Marine Pollution Bulletin*, 24: 210-215.

Pequegnat, J.E., S.W. Fowler and L.F. Small (1969). Estimates of the Zinc Requirements of Marine Organisms. *Journal of the Fisheries Research Board of Canada*, 26: 145-150.

Perdriau, J. (1964). Marine Pollution by Carcinogenic Benzo-3,4-Pyrene-Type Hydrocarbons – Biological Incidences. Part II. *Cah. Oceanography*, 16: 205-229 (French).

Pereira, W.E., F.D. Hostettler, J.R. Cashman and R.S. Nishioka (1994). Occurrence and Distribution of Organochlorine Compounds in Sediment and Livers of Striped Bass (*Morone saxatilis*) from the San Francisco Bay-Delta Estuary. *Marine Pollution Bulletin*, 28: 434-441.

Phillips, D.J.H. (1977). Use of Biological Indicator Organisms to Quantitate Organochlorine Pollutants in Aquatic Environments – A Review. *Environmental Pollution*, 16: 167-229.

Phillips, D.J.H. (1978). The Use of Biological Indicator Organisms to Monitor Trace Metal Pollution in Marine and Estuarine Environments – A Review. *Environmental Pollution*, 13: 281-317.

Phillips, D.J.H. (1979). The Rock Oyster *Saccostre glomerata* as an Indicator of Trace Metaals in Hong Kong. *Marine Biology*, 53: 353-360.

Phillips, D.J.H. (1980). *Quantitative Aquatic Biological Indicators.* Pollution Monitoring Series (Professor Kenneth Mellanby: advisory editor). Applied Science Publishers Ltd., London. 488 pp.

Phillips, D. J. H. (1985). Organochlorines and Trace Metals in Green-Lipped Mussels *Perna viridis* from Hong Kong Waters: A Test of Indicator Ability. *Marine Ecology Progress Series*, 21: 251-258.

Phillips, D.J.H. (1986a). Use of Organisms to Quantify PCBs in Marine and Estuarine Environments. In: *PCBs and the Environment Volume II* (J.S. Waid (ed.)). CRC Press Inc. Boca Raton, Florida. Pp.127-181.

Phillips, D. J. H. (1986b). Organochlorines in Green-Lipped Mussels (*Perna viridis*) from Hong Kong Waters. In: *Proceedings, Second International Workshop on the Marine Flora & Fauna of Hong Kong & Southern China*, Hong Kong 1986.

Phillips, D.J.H., G.B. Thompson, K.M. Gabuji and C.T. Ho (1982). Trace Metals of Toxicological Significance to Man in Hong Kong Seafood. *Environmental Pollution (Series B)*, 3:27-45.

Phillips, D.J.H., B. J. Richardson A.P. Murray and J.G Fabris (1992). Trace Metals, Organochlorines and Hydrocarbons in Port Phillip Bay, Victoria: A Historical Review. *Marine Pollution Bulletin*, 25: 200-217.

Phillips, D.J.H. and D.A. Segar (1986). Use of Bio-Indicators in Monitoring Conservative Contaminants. *Marine Pollution Bulletin*, 17: 10-17.

Phillips, D.J.H. and W.W.S. Yim (1981). A Comparative Evaluation of Oysters, Mussels and Sediments as Indicators of Trace Metals In Hong Kong Waters. *Marine Ecology Progress Series*, 6: 285-293.

Pierce, R.H., R.C. Brown, E.S. Van Fleet and R.M. Joyce (1986). Hydrocarbon Contamination and Coastal Development. In: *Organic Marine Geochemistry* (M.L. Sohn (ed.)). ACM Symposium Series No. 305. American Chemical Society, NY. Pp. 229-246.

Plaskett, D. and I.C. Potter (1979). Heavy Metal Concentrations in the Muscle Tissue of 12 Species of Teleost from Cockburn Sound, Western Australia. *Australian Journal of Marine and Freshwater Research*, 30: 607-616.

Porte, C. and J. Albaigés (1993). Bioaccumulation Patterns of Hydrocarbons and Polychlorinated Biphenyls in Bivalves, Crustaceans, and Fishes. *Archives of Environmental Contamination and Toxicology*, 26: 273-281.

Portmann, J.E. (1972). The Levels of Certain Metals in Fish from Coastal Water Around England and Wales. *Aquiculture*, 1: 91-96.

Poulton, D.J. (1987). Trace Contaminant Status of Hamilton Harbor. *Journal of Great Lakes Research*, 13: 193-201.

Powell, J.H., R.E. Powell and D.R. Fielder (1981). Trace Element Concentrations in Tropical Marine at Bougainville Island, Papua New Guinea. *Water, Air, Soil Pollution*, 16: 143-158.

Preston, A, D.F. Jeffries, J.W.R. Dutton, B.R. Harvey and A.K Steele (1972) British Isles Coastal Waters: The Concentrations of Selected Heavy Metals in Seawater, Suspended Matter and Biological Indicators – A Pilot Survey. *Environmental Pollution*, 3: 69-82.

Rainbow, P.S., and S.L. White (1989). Comparative Strategies of Heavy Metal Accumulation by Crustaceans; Zinc, Copper and Cadmium in a Decapod, an Amphipod and Barnacle. *Hydrobiologia*, 174: 245-?

Rainio, K., R. R. Linko and L. Ruotsila (1986). Polycyclic Aromatic Hydrocarbons in Mussel and Fish from the Finnish Archipelago Sea. *Buletin of Environmental Contamination and Toxicology*, 37: 337-343.

Ratkowsky, D.A., S.J. Thrower, I.J. Eustace and J. Olley (1974). A Numerical Study of the Concentration of Some Heavy Metals in Tasmanian Oysters. *Journal of the Fisheries Research Board of Canada*, 31: 1165-1171.

Readman, J.W., R.F.C. Matoura, M.M. Rhead and L. Brown (1982). Aquatic Distribution and Heterotrophic Degradation of Polycyclic Aromatic Hydrocarbons (PAH) in the Tamar Estuary. *Estuarine, Coastal and Shelf Science*, 14: 369-389.

Readman, J.W., R.F.C. Matoura and M.M. Rhead (1984). The Physico-Chemical Speciation of Polycyclic Aromatic Hydrocarbons (PAHs) in Aquatic Systems. *Fesenius Z. Analytical Chemistry*, 319: 126-131.

Rebbert, R.E., S.N. Chesler, F.R Guenther, B.J. Koster, R.M. Parris, M.M. Shantz and S.A. Wise (1992). Preparation and Analysis of River Sediment Standard Reference Material for the Determination of Trace Organic Constituents. *Fresenius Z. Analytical Chemistry*, 342: 30-38.

Reid, R.G.B., P.V. Fankboner and D.G. Brand (1984). Studies of the Physiology of the Giant Clam *Tridacna gigas* Linné – II. Kidney Function. *Comparative Biochemistry and Physiology*, 78A: 103-108.

Renzoni, A., E. Bacci and L. Falciai (1973). Mercury Concentrations in the Water, Sediments and Fauna of an Area of the Tyrrhenian Coast. In: *6th International Symposium on Medical Oceanography*, Portoroz, Yugoslavia, September 26-30, 1973. Pp 17-45.

Riley, J.P. and R. Chester (1971). *Introduction to Marine Chemistry*. Academic Press, New York and London.

Riley, J.P. and D.A. Segar (1970). The Distribution of the Major Ions and Some Minor Elements in Marine Animals, I. Echinoderms and Coelenterates. *Journal of the Marine Biological Association of the United Kingdom*, 50: 721-730.

Rivers, J.B., J.E. Pearson and C.D Schultz (1972). Total and Organic Mercury in Marine Fish. *Bulletin of Environmental Contamination and Toxicology*, 8: 257-266.

Roach, A.C. and J. Runcie (1998). Levels of Selected Chlorinated Hydrocarbons in Edible Fish Tissues from Polluted Areas in the Georges/Cook Rivers and Sydney Harbour, New South Wales, Australia. *Marine Pollution Bulletin*, 36: 323-344.

Robertson, D.E., L.G. Rancitelli, J.C. Langford and R.W. Perkins (1972). Battelle-Northwest Contribution to the I.D.O.E. Workshop Baseline Study. In: *Proceedings of the I.D.O.E. Workshop on Baseline Studies*, Brookhaven National Laboratory 24-26 May, 1972.

Roth, I. and Hornung, H. (1977). Heavy metal Concentrations in Water, Sediments and Fish from Mediterranean Coastal Area, Israel. *Environmental Science and Technology*, 11: 265-269.

Salihoglu, I., C. Saydam and A. Yilmaz (1987). Long Term Impact of Dissolved Dispersed Petroleum Hydrocarbons (DDPH) in Gulf of Iskenderun. *Chemosphere* 16: 381-394.

Sbriz, L., M. R. Aquino, N. M. Alberto de Rodriguez, S. W. Fowler and J. L. Sericano (1998). Levels of Chlorinated Hydrocarbons and Trace Metals in Bivalves and Nearshore Sediments from the Dominican Republic. *Marine Pollution Bulletin*, 36: 971-979.

Schafer, H.A. and W. Bascom (1976). Sludge in Santa Monica Bay. In: *South CarolinaCoastal Water Research Project, El Segundo, Annual Report*. Pp. 77-82.

Scnantz, M.M., R. Parris, J. Kurz, K. Ballschmiter and S.A Wise (1993). Comparison of Methods for the Gas-Chromatographic Determination of PCB Congeners and Chlorinated Pesticides in Marine Reference Materials. *Fresenius Z. Analytical Chemistry*, 346: 766-778.

Schlichter, D. (1982) Epidermal Nutrition of the Alcyonarian, *Heteroxenia fuscescens* (Ehrb.): Adsorption of Dissolved Organic Material and Lost Endogenous Photosynthates. *Oecologia*, 53: 40-49.

Schultz, C. and D. Crear (1976). The Distribution of Total and Organic Mercury in Seven Tissues of the Pacific Blue Marlin, *Makaira nigricans*. *Pacific Science*, 30: 101-107.

Scoullos, M. and Dassenakis (1983). Trace Metals in a Tidal Mediterranean Embayment. *Marine Pollution Bulletin*, 14: 24-29.

Segar, D.A., J.D. Collins and J.P. Riley (1971). The Distribution of the Major Ions and Some Minor Elements in Marine Animals. Part II. Molluscs. *Journal of the Marine Biological Association of the United Kingdom*, 51: 131-136.

Sericano, J.L., T.L. Wade, T.J. Jackson, J.M. Brooks, B.W. Tripp, J.W. Farrington, L.D. Mee, J.W. Readmann, J-P Villeneuve and E.D. Goldberg (1995). Trace Organic Contamination in the Americas: An Overview of the US National Status and Trends and the International 'Mussel Watch' Programmes. *Marine Pollution Bulletin*, 3: 214-225.

Seymour, A.H. (1966). Accumulation and Loss of Zinc-65 by Oyster in a Natural Environment. In: *Disposal of Radioactive Wastes into Seas, Oceans and Surface Waters*. Vienna Atomic Energy Commission. Pp. 605-619.

Shafer, M.M. (1995). Sampling and Analytical Techniques for Silver in Natural Waters. *Proceedings, 3rd International Conference, Transport, Fate and Effects of Silver in the Environment*. Washington, DC, USA, August 6-9, pp. 99-108.

Shaw, G. R. and D. W. Connell (1982). Factors Influencing Concentrations of Polychlorinated Biphenyls in Organisms from an Estuarine Ecosystem. *Australian Journal of Marine and Freshwater Research*, 33: 1057-1070.

Scribner, E. A., S. Fredrickson, A. Kastle, K. W. McDougall, E. G. Moodie and R. J. Williams (1987). Organochlorine Pesticide and Polychlorinated Biphenyl (PCB) Residues in Fish and Other Aquatic Organisms in New South Wales. Part II. Marine and Estuarine Waters. *Miscellaneous Bulletin 5*. Department of Agriculture New South Wales.

Sims, R.R. Jr. and B.J. Presley (1976). Heavy Metal Concentrations in Organisms from an Actively Dredged Texas Bay. *Bulletin of Environmental Contamination and Toxicology*, 16: 520-527.

Sirota, G.R., J.F. Uthe, A. Sreedharan, R. Matheson, C.J. Musial and K. Hamilton (1983). Polynuclear Aromatic Hydrocarbons in Lobster (*Homarus americanus*) and Sediments in the Vicinity of a Coking Facility. In: *Polynuclear Aromatic Hydrocarbons: Formation, Metabolism and Measurement* (M. Cooke and N.J. Dennis (eds.)). Battelle Press, Columbus, Ohio. Pp. 1123-1136.

Sivalingam, P.M. (1978). Biodeposited Trace Metals and Mineral Content Studies of Some Tropical Marine Algae. *Botanica Marina*, XXI: 327-330.

Sivalingam, P.M. (1980). Mercury Contamination in Tropical Algal Species of the Island of Penang, Malaysia. *Marine Pollution Bulletin*, 11: 106-107.

Skei, J.M., M. Suanders and N.B. Pierce (1976). Mercury in Plankton from a Polluted Norwegian Fjord. *Marine Pollution Bulletin*, 7: 34-36.

Smiley, R. and J.S. Waid (1985). Polychlorinated Biphenyls and Organochlorine Compounds in Great Barrier Reef Biota. In: *Workshop on Contaminants in Waters of the Great Barrier Reef* (I.M. Dutton (ed.)). Workshop Series No. 5. Great Barrier Reef Marine Park Authority, Townsville, Australia.

Smith, J.D. (1970). *Nature*, 225: 103-104.

Smith, J.D., J. Bagg, and B.M. Bycroft (1984). Polycyclic Aromatic Hydrocarbons in the Clam *Tridacna maxima* from the Great Barrier Reef, Australia. *Environmental Science and Technology*, 18: 353-358.

Smith, J.D. and J.D. Burton (1972). The Occurrence and Distribution of Tin with Particular Reference to Marine Environments. *Geochimica et Cosmochimica Acta*, 36: 621-633.

Smith, J.D., J.Y. Hauser, and J. Bagg (1985). Polycyclic Aromatic Hydrocarbons in Sediments of the Great Barrier Reef Region, Australia. *Marine Pollution Bulletin*, 16: 110-114.

Somayajula, B.L. and K. Rama (1972). Mercury in Sea-Food from the Coast of Bombay. *Current Science* (India), 41: 207-208.

Sorentino, C. (1979). Mercury in Marine and Freshwater Fish of Papua New Guinea. *Australin Journal of Marine and Freshwater Research*, 30: 617-623.

Stewart, C. and S.J. de Mora (1992). Elevated tri(n-butyl)tin Concentrations in Shellfish and Sediments from Suva Harbor, *Environmental Technology*, 11: 565-570.

St. John, B.E. (1973). Trace Elements in Corals of the Coral Sea: Their Relationship to Oceanographic Factors. In: *Proceedings of the International Symposium on Oceanography of the South Pacific, Wellington.* (R. Fraser (ed.)). UNESCO. Pp. 149-158.

Stainton, M.P. (1971). Syringe Procedure for the Transfer of Nanogram Quantities of Mercury Vapor for Flameless Atomic Absorption Spectrophotometry. *Analytical Chemistry*, 43: 625-627.

Steimle, F. W., V. S. Zdauowicz and D. F. Gadbois (1990). Metals and Organic Contaminants in Northwest Atlantic Deep-Sea Tilefish Tissues. *Marine Pollution Bulletin*, 21: 530-535.

Stenner, R.D. and Nickless, G. (1974). Distributions of Some Heavy Metals in Organisms of Hardangerfjord and Skjerstadfjord, Norway. *Water, Air, and Soil Pollution*, 3: 279-291.

Stenner, R.D. and Nickless, G. (1975). Heavy Metals in Organisms of the Atlantic Coast of S.W. Spain and Portugal. *Marine Pollution Bulletin*, 6: 89-92.

Stevenson, R.A. and Ufret, S.L. (1966). Iron, Manganese and Nickel in Skeletons and Food of the Sea Urchins *Tripneustesesculentus* and *Echinometra lucunter. Limnology and Oceanography*, 11: 11-17.

Swartz,R.C., P.F. Kemp, D.W. Schultz, G.R. Ditsworth and R.J. Ozretich (1989). Acute Toxicity of Sediment from Eagle Harbor, Washington, to the Infaunal Amphipod, *Rhepoxynnius abronius. Environmental Toxicology and Chemistry*, 8:215-222.

Talbot, V.W., R.G. Magee and M. Hussain (1976). Cadmium in Port Phillip Bay Mussels. *Marine Pollution Bulletin*, 7: 84-86.

Tanabe, S., N. Kannan, N. Fukushima, T. Okamoto, T. Wakimoto and R. Tatsukawa (1989). Persistent Organochlorines in Japanese Coastal Waters: An Introspective Summary from a Far East Developed Nation. *Marine Pollution Bulletin*, 20: 344-352.

Tanabe, S., H. Tanaka and R. Tatsukawa (1984). Polychlorinated Biphenyls, ΣDDT, and Hexachlorocyclohexane Isomers in the Western North Pacific Ecosystem. *Archives of Environmental Contaminstion and Toxicology*, 13: 731-738.

Tapiolas, D.M. (1980). *Natural Products Derived from Soft Corals.* Unpublished B.Sc. (Hon.) Thesis. James Cook University of North Queensland. 115 pp.

Thain, J. E. and M. J. Waldock (1986). The Impact of Tributyl Tin (TBT) Antifouling Paints on Molluscan Fisheries. *Water Science Technology*, 18: 193-202.

Thompson, J.A.J. and D.W. Paton (1978). Heavy Metals in Benthic organisms from Point Grey Dumpsite – Vancouver, B.C. *A Preliminary Report, Institute of Oceanographic Sciences, Patricia Bay, Sidney, B.C., Canada*, PMCR 78-11: 18 pp.

Thrower, S.J. and I.J. Eustace (1973) Heavy Metal Accumulation in Oysters Grown in Tasmanian Waters. *Food Technology of Australia*, 25: 546-553.

Thurberg, F.P., A. Calabrese and M.A. Dawson (1974). Effects of Silver on Oxygen Consumption of Bivalves at Various Salinities. In: *Pollution and Physiology of Marine Organisms* (F.J. Vernberg and W.B. Vernberg (eds.)). Academic Press, New York. Pp. 67-68.

Tokuomi, H. (1969). Medical Aspects of Minimata Disease. *Revues in International Oceanographic Medicine*, 13: 5-35.

Turekian, K.K. and K.H. Wedepohl (1961). Distribution of the Elements in Some Major Units of the Earth's Crust. *Bulletin of the Geological Society of America*, 72: 175-192.

UNEP (1985). GESAMP: Cadmium, Lead and Tin in the Marine Environment. *United Nations Environment Program: Regional Seas Reports and Studies*, No. 92. 172pp.

UNEP (1994). Monitoring Program for the Eastern Adriatic Coastal Area. *MAP Technical Report Series No. 86.* United Nations Environment Program, Geneva.

USFDA (1998). Appendix 5: FDA and EPA Guidance Levels. In: *Fish and Fisheries Products Hazards and Controls Guide, Chapter 9: Environmental Chemical Contaminants and Pesticides (A Chemical Hazard).* U.S. Food & Drug Administration, Center for Food Safety and Applied Nutrition

USEPA (1986). Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish. A Guidance Manual. *U.S. Environmental Protection Agency, Offices of Marine and Estuarine Protection, and Water Regulations and Standards.* Document No. EPA-503/8-89-002, 132 pp.

USEPA (1995). SW-846 Test Methods for Evaluating Solid Waste Physical/Chemical Methods. Proposed Update III (January 1995). *Produced by the US Environmental Protection Agency, Office of Solid Waste.*

Van Fleet, E.S., R.M. Joyce and M.R. Sherwin (1986). Comparison of Anthropogenic Hydrocarbon Inputs to Two Subtropical Marine Estuaries. *The Science of the Total Environment*, 56: 221-230.

Varanasi, U., J.E. Stein and M. Nishimoto (1989). Biotransformation and Disposition of Polycyclic Aromatic Hydrocarbons (PAH) in Fish. In: *Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment* (U. Varanasi (ed.)), CRC Press Inc. Pp. 93-149.

Veek, H.H. and Turekian (1968). Cobalt, Silver and Uranium Concentrations in Reef-Building Corals in the Pacific Ocean. *Limnology and Oceanography*, 13: 304-308.

Waldock, M.J., J.E. Thain and M.E. Waite (1987). The Distribution and Potential Toxic Effects of TBT in UK Estuaries During 1986. *Applied Organometallic Chemistry*, 1: 287-301.

Watling, H.R. and R.J. Watling (1976a). Trace Metals in Oysters from Knysna Estuary. *Marine Pollution Bulletin*, 7: 45-48.

Watling, H.R. and R.J. Watling (1976b). Trace Metals in *Chromomytilus meridionalis*. *Marine Pollution Bulletin*, 7: 91-94.

White, S.L. and P.S. Rainbow (1982). Regulation and Accumulation of Copper, Zinc and Cadmium by the Shrimp, *Palaemon elegans*. *Marine Ecology Progress Series*, 8:95-?

Williams, R. J. and M. Krogh (1993). Coastal Survey of Fish for Organochlorine Compounds. *Phase 1 Situation Report*, Fisheries Research Institute, NSW Fisheries.

Windom, H.L. (1972). Arsenic, Cadmium, Copper, Lead, Mercury and Zinc in Marine Biota – North Atlantic Ocean. In: *Proceedings of the I.D.O.E. Workshop on Baseline Studies*, Brookhaven National Laboratory 24-26 May, 1972.

Windom, H.L. and R.G. Smith (1972). Distribution of Iron, Magnesium, Copper, Zinc, and Silver in Oyster along the Georgia Coast. *Journal of the Fisheries Research Board of Canada*, 29: 450-452.

Windom, H.L., R.Stickney, R. Smith, D. White and F. Taylor (1973). Arsenic Cadmium, Copper, Lead, Mercury, and Zinc in Some Species of North Atlantic Finfish. *Journal of the Fisheries Research Board of Canada*, 30: 275-279.

Wise, S.A., M.M. Shantz, B.J. Koster, R. Demiralp, E.A Mackey, R. Greenberg, M. Burow, P. Ostapczuk and T.I Lillistolen (1993). Development of Frozen Whale Blubber and Liver Reference Materials for the Measurement of Organic and Inorganic Contaminants. *Fresenius Z. Analytical Chemistry*, 345: 270-277.

Wong, P.T.S., R.J. Maguire, Y.K. Chau and O. Kramer (1984). Uptake and Accumulation of Inorganic Tin by a Freshwater Alga, *Ankistrodesmus falcatus*. *Canadian Journal of Fisheries and Aquatic Sciences*, 41: 1570-1574.

Yeates, P.A. and J.M. Bewers (1987). Evidence for Anthropogenic Modifications of Global Transport of Cadmium. In: *Cadmium in the Aquatic Environment.* (J.O. Nriagu and J.B. Sprague (eds.)). Wiley, New York. Pp. 19-34.

Yanni, S. and K. Sachs (1978). Mercury Compounds in Some Eastern Mediterranean Fishes, Invertebrates, and their Habitats. *Environmental Research,* 16: 408-418.

Yilmaz, K., A. Yilmaz, S. Yemenicioglu, M. Sur, I. Salihoglu, Z. Karabulut, F. Telli Karakoc, E. Hatipoglu, A. F. Gaines, D. Phillips and A. Hewer (1998). Polynuclear Aromatic Hydrocarbons (PAHs) in the Eastern Mediterranean Sea. *Marine Pollution Bulletin,* 36: 922-925.

Yoshinari, T. and V. Subramanian 1976. Adsorption of Metals by Chitin. In: *Environmental Biogeochemistry. Vol. 2. Metals Transfer and Ecological Mass Balances.* Ann Arbor Science Publications, Ann Arbor, Michigan. Pp. 541-555.

Young, D. and J Means (1987). Progress Report on Preliminary Assessment of Findings of the Benthic Surveillance Project, 1984. In: *National Status and Trends Program for Marine Environmental Quality.* National Oceanic and Atmospheric Administration (NOAA), Rockville, MD; U.S. Geological Survey, National Water Summary.

Zingde, M.D., S.Y.S. Singbal, C.F. Moraes and C.F.G. Reddy (1976). Arsenic, Copper, Zinc, and Manganese in the Marine Flora and Fauna of Coastal and Estuarine Waters around Goa. *Indian Journal of Marine Science,* 5: 212-217.