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ECOLOGICAL RISK ASSESSMENT OF THE MARINE SEDIMENTS AT THE UNITED HECKATHORN SUPERFUND SITE

32865354

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EXECUTIVE SUMMARY

From approximately 1947 to 1966, United Heckathorn and other companies operated a pesticide formulating plant in Richmond, California, on the shoreline of San Francisco Bay. The operations resulted in the release of DDT and other pesticides to the shoreline and the Bay. Today, in the waters of Richmond Harbor near the former plant, high levels of DDT and its metabolites, the sum of which are referred to as ΣDDT, remain in marine sediments. DDT and dieldrin, another banned organochlorine pesticide, bioaccumulate in marine organisms to the highest levels found in the State of California. The Environmental Protection Agency listed the site on the National Priorities List on March 14, 1990. This document presents the results of studies performed by EPA under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, 1980) to assess the threats posed to the marine environment by the contaminants released from United Heckathorn and to determine potential sediment cleanup levels that will protect the beneficial uses of San Francisco Bay.

The waters of Richmond Harbor are part of San Francisco Bay, the West Coast's second largest estuary. The estuary sustains a complex ecosystem containing thousands of species of fish, invertebrates, birds, mammals, insects, amphibians, plants and other life, as well as nearly half the waterfowl and shorebirds migrating along the Pacific flyway. Fish-eating birds, including cormorants, grebes, loons, kingfishers, and California brown pelicans, an endangered species, have been observed feeding in the most contaminated channels at the site.

The initial components of the study included a review of previous studies of species and contamination in the area. Highlights of this review include the findings that sediment concentrations of ΣDDT are elevated to acutely toxic levels in the Lauritzen Channel and decline by over four orders-of-magnitude to near background levels in the vicinity of Point Potrero. ΣDDT and dieldrin concentrations are extremely elevated in transplanted mussels and resident invertebrates in the Lauritzen Channel, and decline by two orders-of-magnitude in the Inner Richmond Harbor Channel. Fish caught in the Lauritzen Channel in 1986 contained extremely high levels of ΣDDT, which were comparable to the levels measured in 1960. Finally, a study of migratory waterfowl in San Francisco Bay found that only those which wintered in Richmond Harbor significantly accumulated metabolites of DDT. Although other chemicals are present in Richmond Harbor, they have not been found at levels notably above background or above levels which are likely to cause toxicity, in marked contrast to the pesticides ΣDDT and dieldrin, which are several orders-of-magnitude above background and were selected as the contaminants of concern for the study.

The next preliminary phase of the study was a review of the available standards, criteria, and scientific literature regarding ecological impacts of the contaminants of concern to deter-

mine as far as possible the contaminant levels in various media which could adversely impact sensitive organisms. This review provided an indication of which ecological receptors were likely to be the most sensitive and helped guide the selection of field and laboratory studies. EPA's Ambient Water Quality Criteria, the State of California's Water Quality Objectives for DDT and dieldrin in Enclosed Bays and Estuaries and the Water Quality Control Plan for the San Francisco Bay Basin (derived from EPA's Ambient Water Quality Criteria) were identified as applicable or relevant and appropriate to the site. The marine chronic objective for DDT is based upon preventing bioaccumulation in fish to levels that may result in harm to sensitive fish-eating marine birds.

The major phase of this study involved field and laboratory measurements of contaminant concentrations in various media and the performance of standard benthic tests for determining existing impacts from contaminated sediments. The study focused on the two DDT compounds (4,4'-DDT and 2,4'-DDT), their metabolites (4,4'-DDD, 2,4'-DDD, 4,4'-DDE, and 2,4'-DDE), and dieldrin. The studies included bulk sediment toxicity testing, benthic community analyses, bioaccumulation testing, and chemical analyses in sediments, interstitial waters, surface waters, and tissues of infaunal organisms, mussels, and fish and shellfish collected in trawls. An additional goal of this study was the development of relationships between sediment contaminant concentrations and effects on key ecological receptors or pollutant concentrations in other media. These relationships were then used to predict the exposure and effects of ΣDDT and dieldrin to water-column and sediment-associated organisms and the sediment cleanup concentrations which would result in the attainment of water quality objectives, would protect contaminant levels in fish and shellfish tissues, and reduce toxic effects on benthic organisms.

The major results of the studies are:

- Mean ΣDDT concentrations in unfiltered surface water from the Lauritzen, Santa Fe and lower Richmond Inner Harbor Channels were 50, 9 and 1 ng/L (nanogram per liter, or part per trillion), respectively. The mean dieldrin concentrations in unfiltered water were 18, 2 ng/L and non-detectable, respectively. These results indicate that the EPA/State's Water Quality Criteria are violated in the Lauritzen and Santa Fe Channels but are achieved in the lower Inner Harbor Channel. Analysis of water-to-sediment ratios indicates that the Lauritzen is a source of contamination to the other channels.

- Sediment concentrations of ΣDDT declined from over 50,000 μg/kg (parts per billion, dry weight) in the Lauritzen Channel to 12 μg/kg near Point Potrero. Dieldrin concentrations declined from 570 μg/kg in the Lauritzen to non-detectable levels in the Inner Harbor Channel. These

results are consistent with those of previous researchers. The strong concentration gradient indicates that Lauritzen Channel is a source for ΣDDT and dieldrin to Santa Fe and Richmond Inner Harbor Channels and, hence, to San Francisco Bay.

- In 28-day bioaccumulation tests using Macoma nasuta, steady-state tissue levels of ΣDDT of over 70,000 μg/kg (dry) and about 1,500 μg/kg dieldrin were obtained with Lauritzen Channel sediments. Tissue levels declined to 80 μg/kg ΣDDT and undetectable levels of dieldrin in clams exposed to sediments near Point Potrero. These results are consistent with those of previous researchers. Further studies revealed that the tissue concentrations obtained at 28 days were approximately half those obtained after a 90-day exposure, which was sufficient to reach or closely approach steady-state.

- Tissue residues of ΣDDT and dieldrin measured in field-collected benthic infauna were as high as 46,000 and 2,500 μg/kg (dry), respectively, in the Lauritzen Channel. Residues were about two orders-of-magnitude lower in benthic organisms from the Inner Harbor Channel.

- Tissue residues of ΣDDT and dieldrin measured in mussels (Mytilus galloprovincialis) were 2,900 and 97 μg/kg (wet) in the Lauritzen Channel, and declined to 40 and 4 μg/kg in the Richmond Inner Harbor Channel. These results are consistent with those of the State Mussel Watch program.

- Tissue residues of ΣDDT in whole surfperch were approximately 7,500 μg/kg (wet) in the Lauritzen Channel, 920 μg/kg in the Santa Fe Channel, and 100 μg/kg in the Richmond Inner Harbor Channel. Dieldrin levels were roughly 390, 40, and 2 μg/kg in the respective channels. The contaminant concentrations in fish from the Lauritzen Channel are in the same range as those measured in the 1960s and exceed the levels which may cause adverse impacts to sensitive fish-eating birds by orders-of-magnitude. A sensitive bird which had no other source of DDT in its diet and which consumed as little as <1% of its diet from the Lauritzen Channel could be adversely affected. These concentrations may also cause direct chronic effects such as reduced fry survival in fish. The results for the Santa Fe Channel are an order-of-magnitude lower but still exceed levels which may cause adverse impacts to sensitive fish-eating birds. A sensitive bird which consumed more than about 15% of its diet from the Santa Fe Channel might be adversely affected.

- Sediment toxicity tests using the amphipod Eohaustorius estuarius indicated significant acute toxicity in sediments from the Lauritzen Channel. Toxicity at one of the Laurit-

zen stations ranked among the most toxic sediments ever tested at any site. Sediments from the Santa Fe Channel displayed lower but significant toxicity relative to the amphipod's native Yaquina Bay, Oregon, control sediment but were not significantly different from those found in the Inner Harbor Channel or other San Francisco Bay locations. ΣDDT was determined to be the primary cause of toxicity in the Lauritzen Channel.

- An analysis of the benthic community indicated that 10 of 16 measures of community structure were significantly related to ΣDDT concentrations. Based on the Infaunal Index, a composite measure of the abundance of pollutant-sensitive and pollutant-tolerant taxa, the Lauritzen Channel would have to be cleaned-up to remediate the localized sites with "changed" benthic communities.

- An analysis of benthic infauna indicated that amphipod abundance (with the exception of the pollutant-tolerant Grandidierella japonica) was inversely related to ΣDDT concentration. The minimum ecological effects concentration for amphipods was determined to be 100 μg ΣDDT/gOC (100 parts per million organic carbon).

- A hierarchial regression analysis indicated that reductions in the number of amphipods and the Infaunal Index were significantly related to ΣDDT sediment concentrations even after accounting for grain size and sediment organic content (TOC). Dieldrin did not significantly improve the regression.

Overall, the results indicate that the gross contaminant levels in the Lauritzen Channel threaten a variety of ecological receptors at various trophic levels, including benthic and water column organisms and fish-eating birds. Effects are likely to be much less severe in the Santa Fe Channel, although the contaminant levels in fish are still significantly higher than the levels which may threaten sensitive fish-eating birds. In the Richmond Inner Harbor Channel, the ΣDDT residues in whole fish (115 μg/kg wet) fall between the levels which are the basis of the EPA/state chronic marine water quality criteria intended to protect marine birds (150 ppb) and the National Academy of Sciences recommendation (50 ppb) for protecting marine birds.

It appears, then, that the most sensitive ecological receptors to sediment organochlorines in Richmond Harbor are fish-eating marine birds. The only contaminated medium for which regulatory criteria were identified is surface water. Non-regulatory or surrogate criteria were also identified for fish and shellfish tissues, benthic communities and sediments. It was found that surface water concentrations were consistent during different tidal cycles and seasons in each of the three channels sampled, allowing the prediction of the sediment concentration

required to achieve a protective water concentration. In addition, the concentrations measured in the water column and the concentrations measured in whole fish were found to agree remarkably well with the concentrations predicted by the EPA marine chronic Water Quality Criteria and State Water Quality Objectives. This demonstrates that ΣDDT present in surface waters is bioavailable and that it accumulates as predicted by the applicable criteria.

The analysis of surface water pesticide concentrations in the three channels indicates that the concentrations in the Santa Fe and Richmond Inner Harbor Channels are likely elevated by approximately an order-of-magnitude over the concentrations which would result from the respective local sediment concentrations, due to the flux of contaminated water from the Lauritzen Channel. This can confound the calculation of potential remediation goals in the Santa Fe and Richmond Inner Harbor Channels, making them overly conservative, but it also indicates that remediation of the Lauritzen would have beneficial effects throughout the Inner Harbor.

The final goal of the ecological assessment was to provide sufficient information to develop site remediation goals for contaminated sediments which would be protective of human health and the environment. As indicated above, it was determined that the minimum ecological effects concentration for benthic amphipods was 100 μg ΣDDT/g organic carbon, which is equivalent to 1900 μg/kg (dry wt.) at 1.9% organic carbon. Sediment concentrations exceeding this value might cause local chronic adverse impacts to benthic organisms. EPA has reviewed data for other DDT contaminated sites, and found a similar threshold for benthic amphipods. Cleaning-up sediment to this ΣDDT concentration would reduce dieldrin concentrations below its proposed sediment quality criterion of 17 μg/g organic carbon.

Analysis of the Lauritzen Channel data indicates that the marine chronic Water Quality Criteria and State Objective of 1 ng/L for ΣDDT is likely to be achieved if the average channel sediment concentration is below about 1,000 μg/kg ΣDDT (dry wt.) and the human health criteria and objective of 0.6 ng/L is likely to be achieved if the sediment concentration is below 590 μg/kg. The NAS action level for ΣDDT for fish-eating birds, which is not a regulatory criteria, is likely to be achieved if the average channel sediment concentration is below 420 μg/kg. In contrast to the minimum ecological effects concentration for amphipods, the potential sediment remediation concentrations to achieve surface water criteria are a result of the particular hydrologic conditions in the Richmond Harbor and are not likely to be applicable to other ecosystems.

1.0 INTRODUCTION

From approximately 1947 to 1966, United Heckathorn and other companies operated a pesticide formulating plant in Richmond, California, on the shoreline of San Francisco Bay. The operations resulted in the release of DDT and other pesticides to the shoreline and the Bay. In the late 1960's, DDT was implicated in the rapid decline and endangerment of several predatory birds, particularly bald eagles, peregrine falcons and brown pelicans. The unintended toxicity of DDT to birds was promoted by its persistence in the environment and its tendency to biomagnify in the food chain. The use of DDT was banned in the United States in 1973. Today, in the waters of Richmond Harbor near the former plant, high levels of DDT remain in marine sediments, and DDT and dieldrin, another banned organochlorine pesticide, bioaccumulate in marine organisms to the highest levels found in the State of California.

This document presents the results of studies performed by the Environmental Protection Agency under the Comprehensive Environmental Response Compensation and Liability Act (CERCLA, commonly referred to as Superfund) to assess the threats posed to the environment by the contaminants released from United Heckathorn, and to determine potential cleanup levels which will protect the beneficial uses of San Francisco Bay.

1.1. SCOPE OF INVESTIGATION

The primary goal of this ecological assessment is to determine whether contaminants from United Heckathorn threaten the marine environment, and if so, to determine potential cleanup levels for pesticides in marine sediments which will protect the organisms which reside in and feed off the sediments and waters of Richmond Harbor.

The approach taken to determining whether the contamination presents a threat to the environment focused on addressing the following basic concerns for major components of the harbor ecosystem:

1) Are marine sediments in Richmond Harbor contaminated at levels which may be toxic to benthic organisms or affect benthic community structure?

2) Does the water column contain contaminants at concentrations which may be toxic to sensitive marine organisms?

3) Have contaminants bioaccumulated in fish and shellfish to levels which may be harmful to the organisms themselves, or to predators such as birds, marine mammals and humans?

The initial components of the study included a review of previous studies in Richmond Harbor, and of the available stan-

dards, criteria, and scientific literature regarding ecological impacts of the contaminants of concern to determine as far as possible the contaminant levels in various media which could adversely impact sensitive organisms. This review provided a conceptual model of which ecological receptors were likely to be the most sensitive and helped guide the selection of field and laboratory studies.

The major phase of the study involved field and laboratory measurements of contaminant concentrations in various media and the performance of standard benthic tests for determining impacts from contaminated sediments. The studies included bulk sediment toxicity testing, benthic community analyses, laboratory bioaccumulation testing, and chemical analyses in sediments, surface waters, and tissues of field-collected benthos, fish, and shellfish. An additional goal was the determination of the relationship between sediment contaminant concentrations and the concentrations in other media so that, for example, a sediment cleanup concentration could be determined which would result in the attainment of San Francisco Bay Water Quality Objectives or a protective contaminant level in pelagic fish or mussel tissues.

Finally, the field and laboratory results were compared with the available criteria and toxicity information to determine which ecological receptors are likely to be threatened in various media and areas of the harbor, which is the most sensitive, and what sediment concentration must be achieved to protect the most sensitive receptor.

1.2. OVERVIEW OF REPORT

This ecological assessment is part of EPA's overall Remedial Investigation of the United Heckathorn site. Other parts of the Remedial Investigation include a human health risk assessment, an investigation to map the areal and vertical extent of sediment contamination in Richmond Harbor, and limited screening surveys in other media. The Remedial Investigation is being conducted under CERCLA and the National Contingency Plan (NCP).

EPA relied on a number of guidance documents in planning, conducting and reporting the results of this ecological assessment, particularly Risk Assessment Guidance for Superfund, Volume II, Environmental Evaluation Manual (U.S. EPA, 1989a). Additional guidance documents (U.S. EPA, 1989b, 1991c, 1992a, and 1992b) are listed in the Reference Section.

The sections of this report are organized as follows:

Chapter 2 - Site Description, provides a description of the site's regional setting and a discussion of the site history including probable major sources of releases to the environment, government involvement, major contamination investigations, and remedial activities to date. This is followed by brief descriptions of natural resources, and areas assessed in the study.

Chapter 3 - Nature and Extent of Contamination, contains a review of previous investigations of contamination of sediment and biota in Richmond Harbor. These studies were used to identify the contaminants of concern which are the focus of this study.

Chapter 4 - Ecological Receptors and Exposure Mechanisms, contains a description of the aquatic habitats at the site and species which are known or likely to be present. This is followed by a discussion of environmental fate and chemical transport and exposure pathways.

Chapter 5 - Criteria and Toxicity Assessment, discusses various standards, criteria, and other information regarding toxicity which are used to assess ecological threats and determine likely sensitive ecological endpoints for field measurement and preliminary remediation goals for various media.

Chapter 6 - General Study Methods, describes field sampling and laboratory procedures, and general analytical techniques.

Chapter 7 - Physical Characteristics of Study Sites, contains a description of the physical characteristics of the sediments and environment at the study sites.

Chapter 8 - Exposure Assessment, discusses the results of the analyses of sediment, interstitial waters, and the water column, and bioaccumulation in infauna, fishes, and megafauna.

Chapter 9 - Effects Assessment, discusses the results of the benthic survey and toxicity tests, and the potential for deleterious residue levels in the prey of fish eating birds.

Chapter 10 - Risk Characterization and Cleanup Goals, summarizes the observed or threatened adverse effects to ecological receptors in the Richmond Harbor channels, and the water and sediment concentrations required to achieve various ecological criteria.

2.0 SITE DESCRIPTION

This chapter provides a description of the site's regional setting and a discussion of the site history including probable major sources of releases to the environment, government involvement, major contamination investigations, and remedial activities to date. This is followed by a brief description of natural resources, and areas assessed in the study. More detailed discussions of the results of previous studies, aquatic habitats and species likely to be present are provided in subsequent chapters.

2.1. REGIONAL SETTING

Richmond Harbor is located on the eastern shore of Central San Francisco Bay, as shown in Figure 2-1. The approximate location of the former United Heckathorn facility on the eastern shoreline of the Lauritzen Channel in the Richmond Inner Harbor, is shown in Figure 2-2. The Lauritzen Channel is bounded to the south by the Santa Fe Channel, which links it to the Richmond Inner Harbor Channel and San Francisco Bay. The present harbor channels are the result of dredging and filling which began in the early 1930s. Although the channel widths are variable, the Lauritzen is roughly 1,500 feet (~457 meters) long, and 200 ft (~61 m) wide, the Santa Fe is roughly 4,500 ft (~1,370 m) long and 400 ft (~122 m) wide, and the Inner Harbor Channel is roughly 6,000 ft (~1,829 m) long and 1,000 ft (~305 m) wide. The total area of the Richmond Inner Harbor is roughly one quarter of a square mile (~0.65 square kilometer).

Richmond Harbor is highly industrialized and its shoreline is dominated by petroleum and shipping terminals. Running clockwise around the Inner Harbor from Point Potrero are the Port of Richmond's Marine Terminal for automobiles, Arco, Unocal, GATX, Gold Bond, Castrol, Texaco, the Levin Richmond Terminal, Time Oil, the Port of Richmond's Terminals 2 and 3, and an abandoned automobile manufacturing plant. Other facilities include a small yacht harbor, a number of boat yards, a lumber yard, a sand and gravel company, and the California Oils Corporation. The shoreline throughout the Inner Harbor is armored by concrete, rip rap, rubble, and vertical piers.

The harbor channel depths have been maintained by dredging at approximately 35 ft (~10.7 m, ~5.8 fathoms). The Lauritzen Channel varies in depth from approximately 10 ft (~3 m) at its northern end to 40 ft (~12.2 m) at its mouth. Portions of the channel have been periodically dredged. The most recent maintenance dredging occurred in December, 1984 and January 1985, and included a berth area about 120 ft (~36.6 m) in width on the eastern side of the channel, from the Santa Fe northward about 850 ft (~259 m). The berth was dredged to a depth of approximately 41 ft (~12.5 m).

At the mouth of the Inner Harbor Channel, roughly 1 1/2 miles (~2.4 km, ~1.3 nautical mile) from the location of the former United Heckathorn facility, lies Brooks Island. It is

accessible only by boat, and is part of the East Bay Regional Parks District. The island is vegetated and surrounded by shallow mudflats.

2.2. SITE HISTORY

United Heckathorn was a pesticide formulator operating from approximately 1947 to 1966. Heckathorn received technical grade pesticides from chemical manufacturers, ground them in air mills, mixed them with other ingredients such as clays, and packaged them for final use. DDT was the primary pesticide processed. In 1960, approximately 500 (short) tons (~455 metric tons) of DDT and 20 to 25 (short) tons (~18.2 to 23 metric tons) of other chemicals were processed at the site per month (Calif. Reg. Wat. Poll. Contr. Bd., 1960). Although many pesticides were handled by United Heckathorn, only DDT and several other persistent organochlorine pesticides, including dieldrin, remain in significant concentrations (Levine-Fricke, 1990).

United Heckathorn was reported to have routinely washed out equipment containing pesticide residues. The wash water was permitted to either run through drains into the adjacent Lauritzen Channel, or to seep into the ground. After approximately 1960, settling tanks were used to remove pesticides prior to discharge, however California Department of Fish and Game staff observed overflow and leakage from the tanks to the Lauritzen Channel (Calif. Dept. Fish Game, 1960). A 1965 inspection again found discharges from overflowing tanks used to settle pesticide mill scrubber water (Calif. Dept. Fish Game, 1965).

United Heckathorn ceased operations in approximately 1966. The Heckathorn buildings were demolished prior to 1970, and the site has been used since for bulk storage of shipped materials. In 1981, Levin Metals Corporation purchased the site from Parr-Richmond Terminal Company. The Levin Richmond Terminal Corporation currently operates the site as a shipping and storage terminal for bulk materials.

In 1982, the site was listed by the California Department of Health Services as a State Superfund Site. In 1983, workers at the site laying piles and clearing soils encountered pesticide residues containing up to 77% DDT (Calif. Dept. Health Services, 1983). In 1986, during the excavation for a train scale at the site, an oily residue was encountered which contained up to 44% DDT (Harding Lawson Assoc., 1986a). Approximately 60 cubic yards (~46 cubic m) of this material was disposed of off-site in a hazardous waste landfill.

Harding Lawson Associates began the first extensive investigation of the site, on behalf of Levin, in 1983. The investigation included monitoring of groundwater, soils, air, and sediments and organisms in the Lauritzen Channel. The findings of this investigation are presented in Harding Lawson Associates's "Revised Draft Site Characterization and Remedial Action Plan, Former United Heckathorn Site," dated November 6, 1986 (Harding Lawson Assoc., 1986b).

The 1984-1985 survey by the California State Mussel Watch

(SMW), published in 1986, included the first sampling of Richmond Harbor (Calif. State Water Res. Contr. Bd., 1986). The results for DDT in the Santa Fe Channel were described as "by far the highest ever measured in mussels by the SMW program." The dieldrin measurement was also the highest ever measured by the State Mussel Watch. The 1985-1986 Mussel Watch included a new station in the Lauritzen Channel adjacent to the former United Heckathorn plant site (Calif. State Water Res. Contr. Bd., 1987). The results for total DDT were higher than those reported the previous year for the Santa Fe Channel and again "the highest ever measured in mussels in the history of the SMW Program." The report concluded: "The results also suggest that the Lauritzen Channel in the inner portion of the Harbor is the major source of DDT to the Harbor."

Levine-Fricke performed additional site characterization for its 1990 Remedial Investigation and 1991 Feasibility Study. One of the findings of Levine-Fricke's 1990 study was the presence of a highly concentrated pesticide deposit in the intertidal zone on the shoreline of the Lauritzen Channel.

EPA listed the site on the National Priorities List on March 14, 1990. In September, 1990, EPA ordered the immediate removal of the pesticide deposit found by Levine-Fricke. In November, 1990, the Levin Richmond Terminal Corporation, the Montrose Chemical Corporation of California, and the Rhone-Poulenc Basic Chemicals Company, removed approximately 800 cubic yards (~608 cu. m) of soils and sediments, containing up to 100% DDT, from the Lauritzen Channel shoreline. This removal revealed a layer of pesticide residue approximately three feet (~1 m) thick beneath the foundation of the former United Heckathorn building. This deposit was removed in April, 1991. All remaining stockpiles of contaminated soils and known soil pesticide "hot-spots" were removed from the site in several actions which concluded in May, 1993.

The U.S. Army Corps of Engineers has performed several studies in Richmond Harbor aimed primarily at assessing the quality of sediments which would be removed in routine harbor maintenance and in a proposed future deepening of the federal navigation channel. A 1990 study of chemical contamination in the channel done for the Corps by the Battelle Marine Sciences Laboratory found that pesticide contamination in sediment core samples generally dropped to non-detectable levels in the vicinity of Point Potrero (Brown *et al.*, 1990). A follow-up study by Battelle in 1992 examined sediment toxicity and bioaccumulation in the navigation channel (Pinza, *et al.*, 1992). While the study found little toxicity, the bioaccumulation of DDT in clams exposed to sediments taken from the Richmond Inner Harbor Channel was up to 125 times the level in clams exposed to sediments at the Alcatraz environs.

2.3. NATURAL RESOURCES

The waters of Richmond Harbor are part of San Francisco Bay,

California's largest estuary. The estuary sustains a complex ecosystem containing thousands of species of fish, invertebrates, birds, mammals, insects, plants and other life. The estuary also plays an important part of the larger ecology of the Pacific coast. Numerous migratory fish and birds use the Bay as a nursery and for spawning and wintering. The majority of the State's salmon pass through the Bay and delta, as well as nearly half the waterfowl and shorebirds migrating along the Pacific flyway.

Although Richmond Harbor encompasses only a fraction of a percent of the area of San Francisco Bay and contains a less diverse habitat than the Bay as a whole, its open-water channels support a wide variety of resident and migratory species of fish, shellfish and wildlife. Common organisms include clams, mussels, crabs, shrimp, marine and estuarine fish, such as anchovies, smelt, perch, croaker and gobies, and fish-eating birds, such as cormorants and grebes. Harbor seals and the endangered California brown pelican are also seen in the harbor. Brooks Island, at the mouth of the harbor, provides habitat for shorebirds and supports nesting black-crowned night-herons, caspian terns, snowy egrets and gulls.

2.4. AREAS ASSESSED

Previous studies were reviewed in order to determine the likely extent of contamination from United Heckathorn. Based on Battelle's chemical characterization of sediment cores in the federal navigation channel, it appears that DDT concentrations drop to very low or non-detectable levels in the vicinity of Point Potrero. Based on this information, the areas chosen for field studies were the Lauritzen, Santa Fe, and Richmond Inner Harbor Channels.

FIGURE 2-1. Site Location Map for Richmond Harbor (adapted from White *et al.*, 1993).

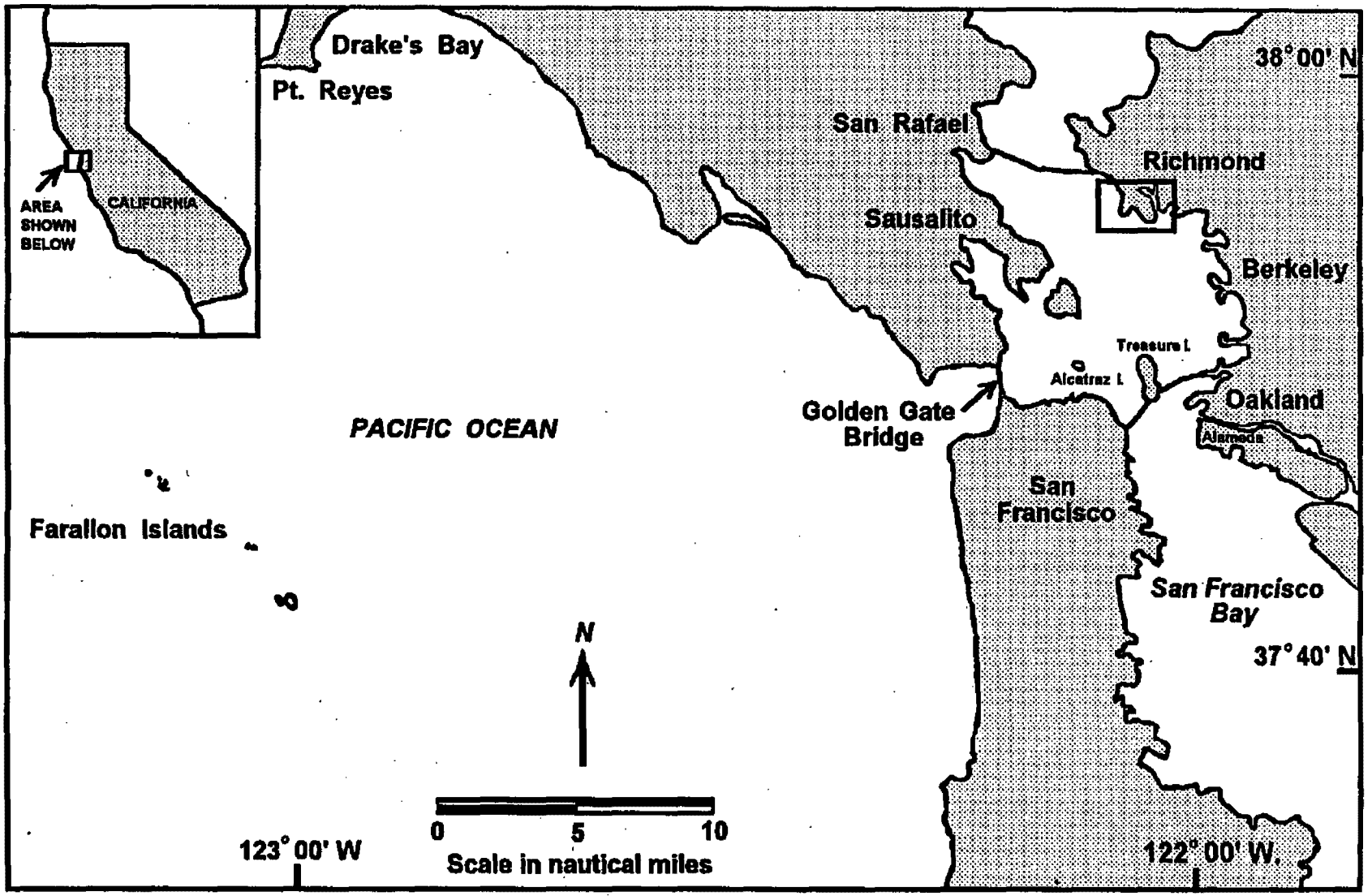

AREA SHOWN BELOW
CALIFORNIA
Drake's Bay
Pt. Reyes
San Rafael
Richmond
Sausalito
Berkeley
Treasure I.
Alcatraz I.
Oakland
Alameda
Golden Gate Bridge
PACIFIC OCEAN
San Francisco
Farallon Islands
San Francisco Bay
38° 00' N
37° 40' N
N
0
5
10
Scale in nautical miles
123° 00' W
122° 00' W.

FIGURE 2-2. Map of Richmond Harbor (adapted from White *et al.*, 1993).

TABLE 4-1. Birds known to nest in central or northern San Francisco Bay or likely to regularly feed in the immediate vicinity of Richmond Harbor.

Common Name Taxonomic Group	Nests in SF or San Pablo Bays	Feeds In/Around Richmond Harbor	Prey
Loons:			
Common Loon		Y	fish
Red-throated/Pacific Loon		Y	fish
Grebes:			
Pied-billed Grebe	Y	Y	fish
Eared Grebe		Y	fish
Horned Grebe		Y	fish
Western/Clark's Grebe			fish
Pelicans and Cormorants:			
White Pelican			fish
Brown Pelican (Endangered)		Y	fish
Double-crested Cormorant	Y	Y	fish
Brandt's Cormorant	Y		fish
Pelagic Cormorant	Y		fish
Herons, Egrets and Bitterns:			
*Great Blue Heron	Y	Y	mixed
Green-backed Heron	Y		mixed
*Black-crowned Night Heron	Y	Y	mixed
*Great Egret	Y	Y	mixed
*Snowy Egret	Y	Y	mixed
American Bittern	Y		mixed
Waterfowl:			
*Pintail	Y		vegy
*Northern Shoveler	Y		prim
*Mallard	Y		vegy
*Scoters (Surf and White-winged)		Y	invert
Ruddy Duck	Y		vegy
Scaups (Lesser and Greater)		Y	vegy
Green-winged Teal			vegy
Canvasback			invert
Redhead			invert
Mergansers (Common and Red-throated)			fish
Common Goldeneye		Y	invert
Bufflehead		Y	invert
American Widgeon			vegy
Canada Geese			vegy
Raptors and Owls:			
Osprey (special concern)			fish
Bald Eagle (Endangered)			omni
Peregrine Falcon (Endangered)	Y	Y	bird
Burrowing Owl (sp. concern)	Y		invert
Short-eared Owl (sp. concern)	Y		mamm
Rails and Coots:			
Black Rail	Y		invert
Clapper Rail (Endangered)	Y		invert
American Coot	Y	Y	vegy

(Cont'd.)

TABLE 4-1 (cont'd.).

Common Name Taxonomic Group	Nests in SF or San Pablo Bays	Feeds In/Around Richmond Harbor	Prey
Shorebirds:			
Snowy Plover (sp. concern)	Y		invert
*Semipalmated Plover	Y		invert
*Black-bellied Plover		Y	invert
*Killdeer	Y	Y	invert
*Western Sandpiper		Y	prim
*Least Sandpiper		Y	prim
*Dunlin		Y	prim
*Sanderling		Y	prim
*Dowitchers (Short- and Long-billed)		Y	prim
*Red Knot		Y	prim
Black Turnstone		Y	invert
Ruddy Turnstone		Y	invert
*Willet		Y	omni
*Marbled Godwit		Y	prim
Black-necked Stilt	Y		prim
*American Avocet	Y	Y	prim
*Greater Yellowlegs			invert
Lesser Yellowlegs			invert
Long-billed Curlew (special concern)			invert
Red-Necked Phalarope			prim
Water Pipit			invert
Black Oystercatcher	Y	Y	invert
Gulls, Terns, and other Seabirds:			
*Bonaparte's Gull		Y	omni
*California Gull	Y	Y	omni
Glaucous-winged Gull		Y	omni
Heermann's Gull		Y	omni
*Herring Gull	Y	Y	omni
*Mew Gull		Y	omni
*Ring-billed Gull		Y	omni
Thayer's Gull		Y	omni
*Western Gull	Y	Y	omni
*Caspian Tern	Y	Y	fish
*Elegant Tern		Y	fish
Least Tern (Endangered)	Y	Y	fish
*Forster's Tern	Y	Y	fish
Common Murre	Y		fish
Pigeon Guillemot	Y		fish

KEY:
fish = consumes primarily fish
mixed = consumes invertebrates and fish
vegy = consumes primarily algae and plant material
mamm = consumes primarily terrestrial mammals
bird = consumes primarily birds
omni = diet usually omnivorous/scavenger
invert = consumes primarily small to medium sized invertebrates
prim = consumes primarily very small invertebrates or plankton
* = species that are particularly abundant in the Richmond Harbor area relative to other members of their taxonomic group.
"Y" indicates species known to occur in the area, U.S. Fish & Wildlife Service.

TABLE 4-2. Physico-chemical properties and bioaccumulation potential of DDT, DDE, DDD, and dieldrin.

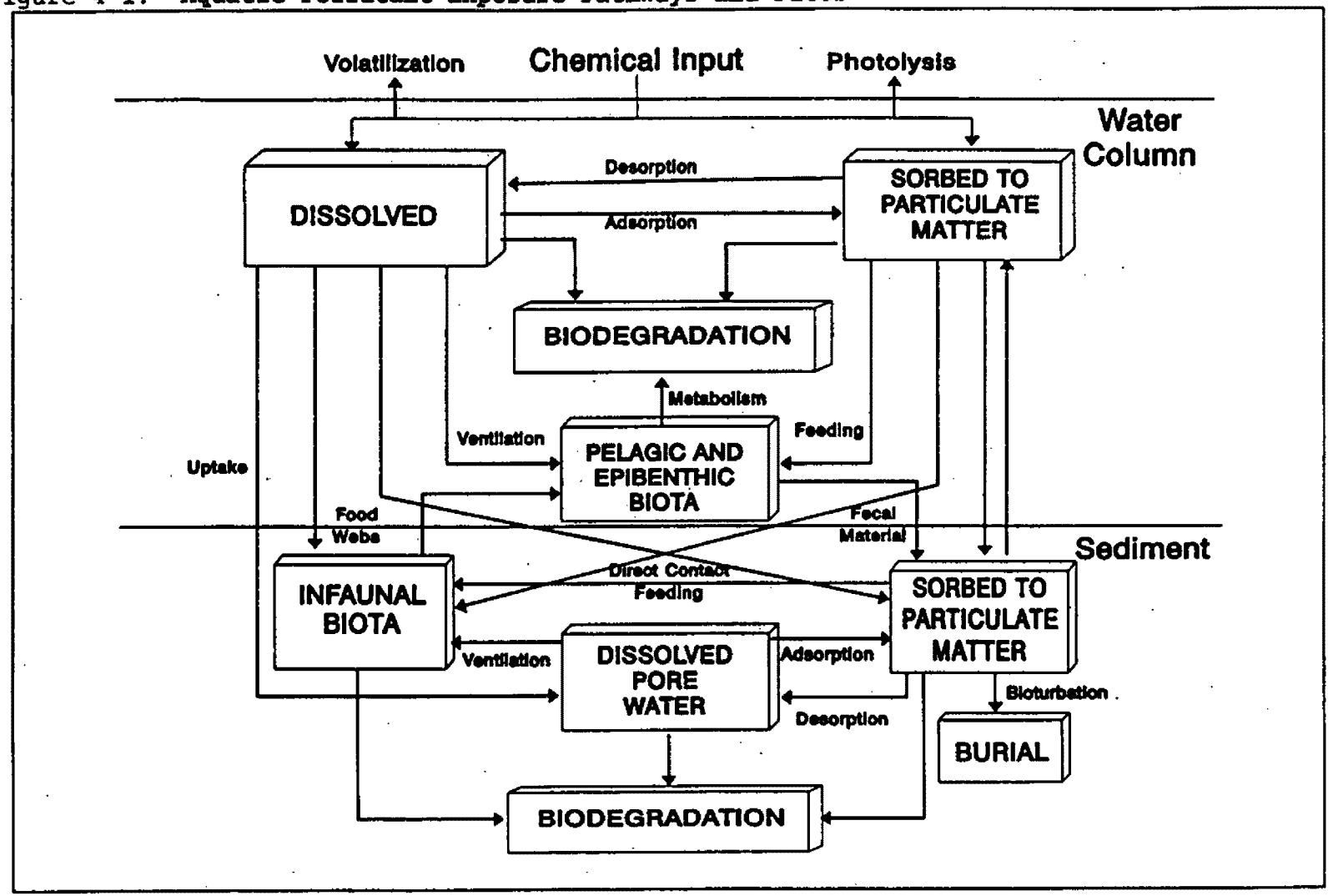
Compound	DDT	DDE	DDD	Dieldrin
Molecular Weight	354.5	318.0	320.1	380.9
Water Solubility[1] (ppb @ 25°C)	5.5 (4,4'-DDT) 26 (2,4'-DDT)	14 (4,4'-DDE)	20 (4,4'-DDD)	200
Vapor Pressure[2] (x 10^{-7} torr)	7.3 (4,4'-DDT) 55 (2,4'-DDT)	65 (4,4'-DDE) 62 (2,4'-DDE)	10 (4,4'-DDD) 19 (2,4'-DDD)	28
log K_{ow}[3]	6.91	6.96	6.22	5.40
Log BCF (Fish)	4.52[4] (Fathead minnow) 4.58[5] (trout)	4.71[4] (Fathead minnow) 4.58[5] (bluegill) 7.26[6] (trout)	5.24[5] (estimate)	4.1[7] (guppy)
Infaunal AFs[8]		1.2 (bivalves)	1.5 (bivalves) 4.5 (worms)	

1 = Weil *et al.*, 1974; 2 = Spencer, 1975; 3 = DeBruijn *et al.*, 1989; 4 = Veith *et al.*, 1979; 5 = BEIA, 1989; 6 = Oliver and Niimi, 1985; 7 = Reinert, 1972; 8 = Boese and Lee, 1992.

K_{ow} = octanol-water partitioning coefficient (see Section 8.1.2.)
BCF = tissue residue/water concentration (see Section 8.4.)
AF = (tissue residue/lipid conc.)/(sediment concentration/TOC) (see Section 8.3.)

FIGURE 4-1. Aquatic pollutant fate and exposure pathways. Adapted from Fava *et al.*, 1984.

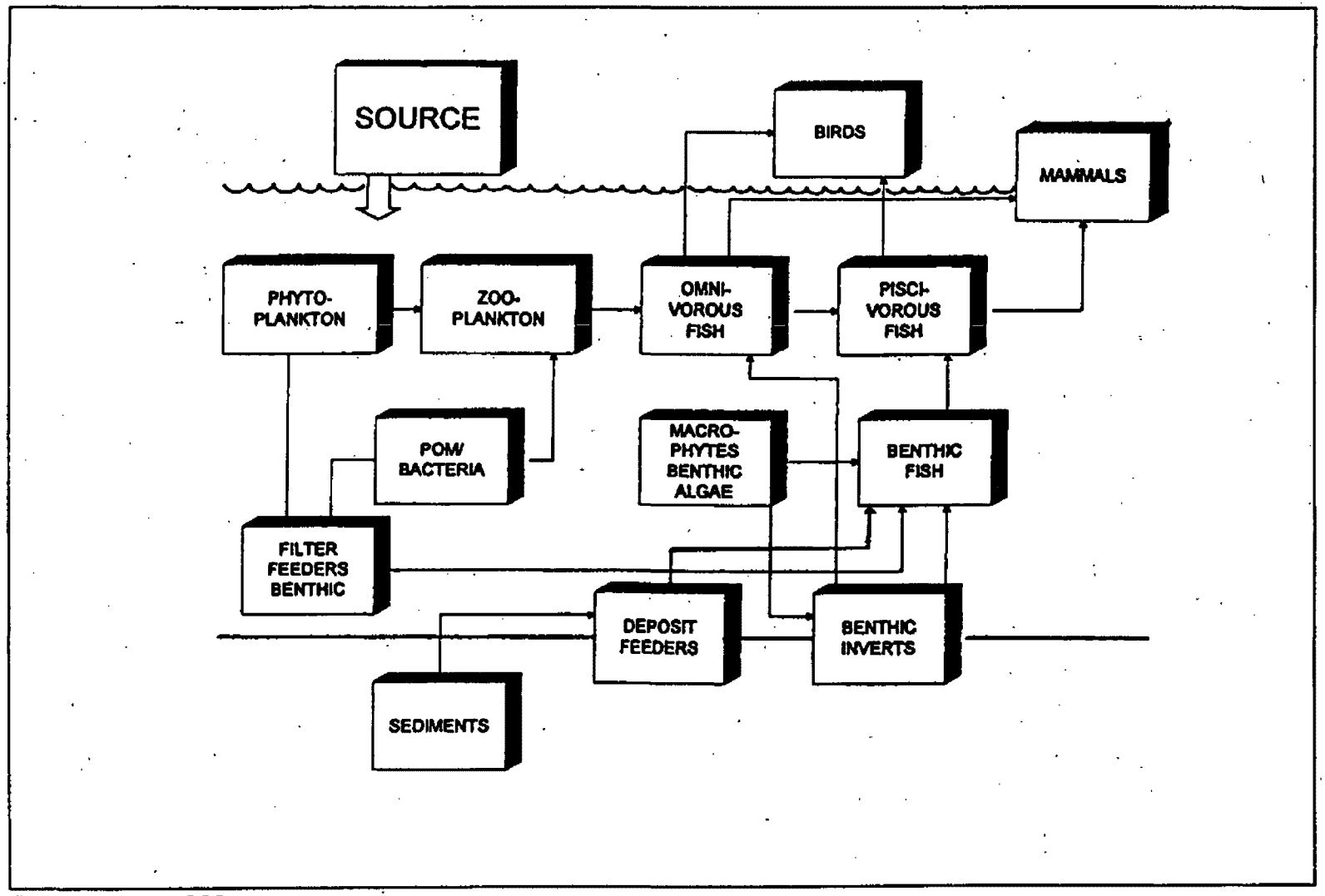
Figure 4-1: Aquatic Pollutant Exposure Pathways and Fates



Adapted from Fava et al., 1984

FIGURE 4-2. Generic food web. Adapted from IEHR, 1993.

FIGURE 4-2: GENERIC FOOD WEB



From IEHR, 1993

5.0 CRITERIA AND TOXICITY ASSESSMENT

This section discusses various standards, criteria, and other information regarding toxicity which may be used to assess ecological threats and provide the basis for establishing cleanup levels at the United Heckathorn site.

The first are standards which have been adopted or promulgated under State or Federal environmental laws. CERCLA (Comprehensive Environmental Response, Compensation, and Liability Act) requires that remedial actions at Superfund sites meet all applicable or relevant and appropriate requirements (ARARs) of Federal and State environmental laws, unless a waiver is justified. EPA's ecological assessment guidance (U.S. EPA 1989a) specifically recommends determining whether any criteria are ARARs for a site, stating: "If usable and applicable criteria exist, the assessment should include sampling and monitoring plans to determine the extent to which those criteria are exceeded by environmental concentrations at the site."

Following the discussion of ARARs are discussions of non-promulgated advisories or guidance issued by Federal or State governments. These do not have the status of potential ARARs, and are referred to as To-be-Considered-Material, or TBCs. Finally, additional scientific literature will be discussed under Additional Resources.

After the review of the various standards and other resources, they will be integrated to determine likely sensitive ecological endpoints, and preliminary remediation goals for various media.

5.1. APPLICABLE OR RELEVANT AND APPROPRIATE REQUIREMENTS OF FEDERAL AND STATE LAW.

The discussion below does not address all environmental laws which are potentially applicable or relevant and appropriate to all Superfund issues at the United Heckathorn site, but only those which are relevant to this ecological assessment, or are based upon an ecological measurement endpoint, such as fish and shellfish tissue residues.

ARARS are generally divided into three categories: chemical-specific, location-specific, and action-specific. Chemical-specific ARARs are concentration-based requirements for specific chemicals in various environmental media such as sediments and surface waters. Chemical-specific ARARs that apply to the water column, fish and shellfish residues are identified below. Location-specific ARARs are those which apply only in specific locations. The beneficial uses of central San Francisco Bay are location-specific ARARs. Action-specific ARARs generally affect the choice of alternatives for remediation and are, therefore, not discussed here.

5.1.1. California Enclosed Bays and Estuaries Plan and San Francisco Bay Regional Basin Plan

The State of California has adopted water quality objectives for toxic pollutants pursuant to the requirements of Section 303 of the Clean Water Act. The State's Enclosed Bays and Estuaries Plan, adopted on April 11, 1991 (State Water Resources Control Board Resolution No. 91-33; Calif. St. Wat. Res. Bd., 1991b), contains the water quality objectives for DDT and dieldrin in surface waters summarized in Table 5-1. These chemical-specific objectives apply to all surface waters of San Francisco Bay, including all channels and canals in Richmond Harbor. Although the human health objectives are not ecologically relevant, they are listed because they are based upon achieving certain contaminant levels in fish and shellfish.

The values of the objectives in Table 5-1 are some two to three orders-of-magnitude below the levels which can be measured using standard analytic methods. Although research laboratories can achieve these levels, routine demonstration of compliance is problematic. However, all of the objectives above are based upon achieving specific DDT and dieldrin residue values in fish tissues, most of which are measurable using routine analytic methods. Thus, fish tissue residues may serve as surrogate measurement endpoints to approximate compliance with the water quality objectives in the event that acceptable direct water column measurements are unobtainable. The State's water quality objectives were based directly upon EPA's Ambient Water Quality Criteria for DDT and dieldrin, the derivation of which are discussed in sections 5.1.2. and 5.1.3.

In addition to the chemical-specific objectives above, the Enclosed Bays and Estuaries Plan (*ibid.*) contains the following narrative objective:

> "The concentration of toxic pollutants in the water column, sediments, or biota shall not adversely affect beneficial uses."

The chemical-specific water quality objectives discussed above were also adopted into the Water Quality Control Plan for the San Francisco Basin by the San Francisco Bay Regional Water Quality Control Board (SFBRWQCB, 1992). In addition, the Basin Plan lists the beneficial uses of Central San Francisco Bay, which includes the waters at the site. These are a location-specific ARAR. In order to meet the requirements of the narrative water quality standard above, the remedial action chosen for the site must achieve pollutant concentrations in the water column, sediments and biota which do not adversely affect the beneficial uses listed in Table 5-2, including fishing and shellfish harvesting, estuarine habitat, fish spawning, and preservation of rare and endangered species.

The water quality objectives and beneficial uses discussed above will be retained for consideration as potential remediation goals in surface waters.

5.1.2. EPA Ambient Water Quality Criteria for DDT

Although EPA's Ambient Water Quality Criteria are guidance values and have not been promulgated under Federal law, they were given special status under the Superfund Amendments and Reauthorization Act of 1986 (SARA). Section 121 of CERCLA as amended by SARA requires that remedial actions meet Water Quality Criteria established under Section 304 or 303 of the Clean Water Act, where relevant and appropriate.

Section 304 of the Clean Water Act requires EPA to publish criteria for water quality that accurately reflect the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare, including plankton, fish, shellfish, wildlife, and plant life, which may be expected from the presence of pollutants in any body of water based on their whole water concentration. EPA's Ambient Water Quality Criteria documents contain exhaustive reviews of scientific literature regarding fate and transport of pollutants, acute and chronic toxicity to aquatic life, including insects, plants, fish and shellfish, bioconcentration in aquatic organisms, and toxic effects on wildlife. These are followed by a review of mammalian toxicity and human health effects, beginning with a pharmacokinetic discussion of absorption, distribution, metabolism and excretion, and followed by a discussion of acute, subacute and chronic toxic effects. The Water Quality Criteria Documents are an exceptionally useful resource for the performance of ecological assessments and the criteria are often the basis of other standards and advisories.

The Ambient Water Quality Criteria document for DDT was published in October, 1980 (EPA 440/5-80-038; U.S. EPA, 1980a). The criteria for the protection of saltwater aquatic life and human health are listed in Table 5-3. California's Water Quality Objectives for DDT listed in Section 5.1. were based directly on these values. The human health values have been updated since the original criteria publications in 1980 to reflect revised carcinogenic potency values from EPA's Integrated Risk Information System (IRIS) database (see Final Rule, 40 CFR Part 131, 57 FR 60848, December 22, 1992). Where the criterion was calculated to achieve a specific residue in fish tissue, the residue value is listed below the criterion in parentheses.

EPA's Ambient Water Quality Criteria for the protection of saltwater aquatic life are, for most pollutants, based upon toxic effects data for water column organisms. However, for DDT and its metabolites, which bioaccumulate to high levels and may cause toxicity to organisms at higher trophic levels, it was determined that more restrictive criteria were necessary to achieve protective tissue residue levels than to prevent toxicity to water column organisms and the organisms feeding on them.

In the criteria analysis of the toxicity of DDT and its metabolites to aquatic organisms, suitable acute toxicity data were available for eleven species of saltwater fishes and six saltwater invertebrates, including a mollusk, a crab, and four species of shrimp. The most sensitive organism was the brown

shrimp, Penaeus aztecus, with a 96-hour LC_{50} of 0.14 µg/l. The saltwater Final Acute Value derived from these data was 0.13 µg/l (130 ng/l). In tests on the metabolites DDD (TDE) and DDE, values were available for three and one species, respectively. These data were all higher than the Final Acute Value for DDT, and were insufficient to derive separate Final Acute Values for the metabolites.

No chronic toxicity data were available for any saltwater species, therefore, no Final Chronic Value for DDT or metabolites was determined. However, Final Acute-Chronic Ratios determined by EPA for other chlorinated pesticides are typically in the range of 2 (the lowest value allowable under the criteria development guidance) to 10. This suggests that chronic effects from DDT to water column organisms would be unlikely below a range of 0.065 to 0.013 µg/l (65 - 13 ng/l).

Limited information on the sensitivity of saltwater aquatic plant species, including algae and rooted vascular plants, indicated that plants are much less sensitive to DDT than are fish or invertebrates. DDT at a concentration of 10 µg/l reduced photosynthesis in saltwater diatoms, green algae, and dinoflagellates.

Field and laboratory-generated bioconcentration factors for DDT were available for over fifty species, including three saltwater invertebrates and nine saltwater fish. Bioconcentration factors for saltwater species ranged from 1,200 to 76,300, and were similar in both field and laboratory tests. The geometric mean of lipid-normalized bioconcentration factors for both freshwater and saltwater aquatic life was determined to be 17,870. The mean lipid concentration for saltwater species was 16%. [Note: The criteria documents calculate lipid-normalized BCFs by dividing by the percent lipid (*e.g.*, 8) rather than the decimal equivalent (e.g., 0.08).]

Maximum permissible DDT tissue concentrations were reviewed from ten studies of toxic effects in birds, such as egg shell thinning and duckling survival, and seven regarding toxic effects in fish, such as reduced sac fry, fry, and fingerling survival. Toxic impacts to fish, including sensitive life stages, were found to occur at higher DDT tissue concentrations (3 to 11.36 mg/kg) than those which caused toxic impacts to birds (0.5 to 3.0 mg/kg). The lowest maximum permissible tissue concentration was 0.15 mg/kg (150 µg/kg) in fish, which was determined to be associated with reduced reproduction among endangered California brown pelicans. This value was used, in conjunction with the geometric mean bioconcentration factor and the percent lipid content (8%) of the major prey of the pelicans, the Northern anchovy, to derive the Final EPA Marine Chronic Water Quality Criteria of 0.001 µg/l (1 ng/l). However, the criteria document states that because the 0.15 mg/l fish tissue residue was associated with reduced reproduction in pelicans, the final criterion of 0.001 µg/l may not be low enough to protect this sensitive species.

The relative sensitivities of the receptors considered in the development of EPA's criteria for DDT are summarized in Table 5-4. It is evident that because of bioaccumulation the most

sensitive ecological receptors to DDT in the water column are fish-eating birds. In addition, if a DDT water column concentration of 1 ng/l or a surrogate residue level in fish below 150 µg/kg is maintained to protect birds, the water column concentration will be at least an order-of-magnitude below the level likely to cause chronic toxicity to sensitive marine organisms, two orders-of-magnitude below the level likely to affect sensitive life-stages of fish, two to three orders-of-magnitude below the level likely to cause acute impacts in fish and shellfish, and five orders-of-magnitude below the level likely to cause adverse effects in aquatic plants. If a water column concentration of 0.6 ng/l or a surrogate fish tissue residue level of 32 µg/kg is maintained to achieve a 10^{-6} excess cancer risk level and the State Water Quality Standard for the protection of human health, all water column ecological receptors likely will be protected.

The EPA Ambient Water Quality Criteria and tissue residues discussed above will be retained for consideration as potential remediation goals in surface waters and in fish and shellfish tissues.

5.1.3. EPA Ambient Water Quality Criteria for Dieldrin

The Ambient Water Quality Criteria document for dieldrin was published in 1980 (U.S. EPA, 1980b). The criteria for the protection of saltwater aquatic life and human health are listed in Table 5-5. California's Water Quality Objectives for dieldrin listed in Section 5.1.1. were based directly on these values. The human health values have been updated since the original criteria publications in 1980 to reflect revised carcinogenic potency values from EPA's IRIS database (see Proposed Rule, 40 CFR Part 131, 56 FR 58420). Where the criterion was calculated to achieve a specific residue in fish tissue, the residue value is listed below the criterion in parentheses.

In the criterion analysis of the toxicity of dieldrin to aquatic organisms, suitable acute toxicity data were available for thirteen species of saltwater fishes and seven saltwater invertebrates, including a mollusk, a crab and five species of shrimp. The most sensitive organism was the pink shrimp, Penaeus duorarum, with a 96-hour LC_{50} of 0.7 µg/l. The saltwater Final Acute Value derived from these data was 0.71 µg/l (710 ng/l).

The only chronic study on a saltwater species found depressed survival in mysid shrimp after 28 days at a geometric mean dieldrin concentration of 0.73 µg/l. The Final Acute-Chronic ratio for dieldrin was 8.5, resulting in a Final Chronic Value for dieldrin of 0.084 µg/l (84 ng/l).

In the proposed Sediment Quality Criteria for dieldrin, released in November, 1991 (U.S. EPA, 1991a; see Section 5.2.3.), the Final Acute Value was revised to 0.6594 µg/l, the Final Acute-Chronic Ratio to 5.748, and the Final Chronic Value to 0.1147 µg/l.

Information on the sensitivity of saltwater aquatic plant species, including algae and phytoplankton, indicated that they

are much less sensitive to dieldrin than are fish or invertebrates. Productivity and growth rates were reduced at concentrations of approximately 1,000 µg/l.

Bioconcentration factors for dieldrin in saltwater fish and shellfish ranged from 400 to 8,000 in the Eastern oyster. The geometric mean of lipid-normalized bioconcentration factors was determined to be 1,557. The mean lipid concentration used for saltwater species was 16%. [Note: The criteria documents calculate lipid-normalized BCFs by dividing by the percent lipid (*e.g.*, 8) rather than the decimal equivalent (*e.g.*, 0.08).]

The final marine chronic criterion of 0.0019 µg/l (1.9 ng/l) was derived to achieve the FDA Action level of 0.3 mg/kg in fish oil. The criterion was obtained by dividing 0.3 by the bioconcentration factor of 1,557, and by 100, the percent lipid content of fish oil. The FDA Action level in whole fish should be achieved at a water column concentration of 0.012 µg/l (12 ng/l), obtained by dividing 0.3 mg/kg by 1,557, and 16, the mean lipid concentration of saltwater fish.

The relative sensitivities of the receptors considered in the development of EPA's criteria for dieldrin are summarized in Table 5-6. From the data in Table 5-6, it is apparent that if a dieldrin water column concentration of 1.9 ng/l or a surrogate residue level in fish below 0.3 mg/kg is maintained to achieve the FDA Action level in fish tissue, the water column concentration will be an order-of-magnitude below the level likely to cause chronic toxicity to fish and shellfish, and five orders-of-magnitude below the level likely to cause adverse effects in aquatic plants. If a water column concentration of 0.14 ng/l or a surrogate fish tissue residue level of 0.0007 mg/kg is maintained to achieve a 10^{-6} excess human cancer risk level, all water column ecological receptors considered in the development of the criteria will be protected. It is important to note, however, that toxic effects to marine birds were not considered in the development of the criteria.

The EPA Ambient Water Quality Criteria and tissue residues discussed above will be retained for consideration as potential remediation goals in surface waters and in fish and shellfish tissues.

5.1.4. Endangered Species Act

The goal of the Endangered Species Act of 1973, 16 USC §1531 *et seq.*, is the conservation of species of fish, wildlife and plants that are threatened with extinction. Protection of endangered species is required on an individual rather than population basis. Compliance with the Act at Superfund sites requires the identification of any threatened or endangered species, or its critical habitat, which will be affected by a proposed remedial action.

The U.S. Fish and Wildlife Service, which is the Federal trustee for the protection of migratory birds, provided the following list of endangered species and species of special concern which are known to nest in Central or Northern San Fran-

cisco Bay, and/or are likely to feed regularly in the immediate vicinity of Richmond Harbor (Table 5-7).

The Fish and Wildlife Service believes that it is appropriate to assume that some resident or migratory birds may consume up to 100% of their diet from Richmond Harbor for significant periods of time. In addition, the Fish and Wildlife Service supports the use of a known sensitive species, such as brown pelicans, for evaluating the potential risk to which a wildlife community may be exposed from a known toxic contaminant. However, the Fish and Wildlife Service stressed that the tissue residue basis of EPA's Ambient Water Quality Criteria for DDT (0.15 mg/kg in prey) discussed in section 5.1.2., still resulted in reproductive levels in pelicans which were 10 to 30% below the levels needed to maintain a stable population. Thus, achieving this level may not be sufficient to protect sensitive species which may be present in the harbor.

5.1.5 California Endangered Species Act

The goal of the California Endangered Species Act (California Fish and Game Code §2050) is to conserve, protect, restore and enhance any endangered or threatened species and its habitat. Among the birds likely to nest or feed in the area (Table 4-1), most of which are listed by the State as endangered or threatened are also federally listed (Table 5-7). The one exception is the California Black Rail, a State threatened species.

The California Department of Fish and Game also submitted the names of two potential plant species listed as rare that have distributions in the general area of Richmond. These are Mason's lilaeopsis, a minute, turf-forming perennial plant in the carrot family, and Soft bird's-beak, a sparingly-branched, semi-parasitic herbaceous annual plant in the figwort family. The known distribution of Mason's lilaeopsis, which is found on saturated clay soils regularly inundated by waves and tidal action, appears to be limited to the Bay delta. Soft bird's-beak occurs in the coastal salt marshes and brackish marshes of northern San Francisco and Suisun Bays.

5.2. TO-BE-CONSIDERED MATERIALS

To-be-considered materials are non-promulgated advisories or guidance issued by Federal or State agencies. Superfund remedies are not required to meet these criteria, but they may, nonetheless, be used as the basis for establishing cleanup levels in the absence of applicable or sufficiently protective ARARs.

5.2.1. Food and Drug Administration (FDA) Action Levels

Like the State's human health water quality objectives discussed above, the Federal FDA action levels are not ecologically relevant, but are discussed here because they are based upon tissue residues in fish and/or shellfish. The FDA action levels (Table 5-8) are guidance used to determine whether contaminated seafood and seafood products should be removed from the

marketplace.

The FDA action levels will be retained for consideration as potential remediation goals in fish and shellfish tissues.

5.2.2. National Academy of Sciences Water Quality Criteria

The National Academy of Sciences and National Academy of Engineering published recommendations in 1972 for DDT and dieldrin residues in composites of 25 or more whole fish of any species within the same size range as those consumed by any bird or mammal in the marine environment (Nat. Acad. Sci., 1972). The recommendations are listed in Table 5-9.

The criteria document cites studies demonstrating DDE-induced shell thinning in mallards, American kestrels, Japanese quail and ring doves, and an inverse relationship between shell thickness and concentrations of DDE in eggs of wild populations of herring gulls, double-crested cormorants, great blue herons, white pelicans, brown pelicans, and peregrine falcons. The document concludes that a wet-weight tissue range of 0.1 to 0.5 mg/kg (100 to 500 ppb) is "evidently higher than one which would permit successful reproduction of several fish-eating and raptorial birds."

The criteria for dieldrin is based upon hazard to fish-eating birds such as the bald eagle, common egret, and peregrine falcon, "which may accumulate lethal amounts from fish or birds which have not themselves been harmed." The document states that dieldrin and closely related pesticides are substantially more toxic to wildlife than are other chlorinated hydrocarbon pesticides, so that more conservative recommendations are, therefore, necessary.

The National Academy of Sciences tissue residue recommendations above will be retained for consideration as potential remediation goals for the protection of marine birds, including endangered species, species of special concern, and for the protection of marine mammals.

5.2.3. EPA Proposed Sediment Quality Criteria for Dieldrin

EPA released proposed sediment quality criteria for dieldrin for the protection of benthic organisms in November, 1991 (U.S. EPA, 1991a). The criteria are based upon the equilibrium partitioning (EqP) approach using the water quality criterion Final Chronic Value and organic carbon partitioning coefficients to estimate the concentration of sediment contaminants which will not cause adverse effects to benthic organisms. The EqP approach is intended to achieve the water quality criteria in sediment interstitial water, and assumes that benthic (infaunal and epibenthic) species will have similar sensitivities as species tested to derive the water quality criteria.

The Final Chronic Value was calculated from a data set updated from the one used in the 1980 Ambient Water Quality Criteria document discussed in Section 5.1.2. The Final Chronic

Value for saltwater organisms was 0.1147 µg/l in the water column, which is the ratio of the Final Acute Value (0.6594 µg/l) and the Final Acute-Chronic Ratio (5.748).

The proposed criteria document states that, except possibly where a locally important species is very sensitive or sediment organic carbon is < 0.2%, benthic organisms should be acceptably protected in saltwater sediments containing < 17 ug dieldrin/g organic carbon, which is equivalent to 170 ppb at 1% organic carbon. Included in the criteria statement are a lower confidence limit of 7.7 µg/g OC (77 ppb @ 1% TOC), "which might be interpreted as a concentration below which impacts on benthic species would be unlikely," and an upper confidence limit of 36 µg/g OC (360 ppb @ 1% TOC), "which might be interpreted as a concentration above which benthic impacts would be highly likely."

In May, 1988, EPA released Interim Sediment Criteria Values for Nonpolar Organic Contaminants (SCD #17; U.S. EPA, 1988), which included values for dieldrin. The document contained EqP-calculated criteria based on both the 1980 Final Chronic Value and Final Residue Value (FRV) for dieldrin, but recommended that those based upon the FRV be used. The interim criteria for dieldrin based upon the FRV was 0.130 µg dieldrin/g organic carbon, which is more than two orders-of-magnitude below the proposed criteria. The FRV is the marine chronic value for dieldrin listed in Table 5-5. Since this value is not based upon a toxic endpoint but rather upon achieving the FDA human health Action Level in fish oil, the ecological significance of a sediment EqP calculation based upon it is unclear. As discussed above, the sediment criteria for the protection of benthic organisms ultimately proposed by EPA is based upon a Final Chronic Value and the assumption that benthic organisms will have similar sensitivities as those used to develop the water quality criteria. The Agency does not currently propose using residue-derived water quality criteria as a basis for sediment criteria intended to protect benthic organisms. This issue will be discussed further in the examination of other guidance.

The proposed sediment quality criteria for dieldrin will be retained for consideration as a potential remediation goal for the protection of benthic organisms.

5.2.4. EPA Interim Sediment Quality Criteria for DDT

In May, 1988, EPA released Interim Sediment Criteria Values for Nonpolar Organic Contaminants (SCD #17; U.S. EPA, 1988), which included an interim sediment quality criteria value of 0.828 ug DDT/g organic carbon, which is equivalent to 8.28 ppb at 1% sediment organic carbon. The value was based upon an EqP calculation using the Ambient Water Quality Criteria Final Residue Value for DDT. The FRV is the marine chronic value for DDT listed in Table 5-3. Since this value is based upon achieving a specific tissue residue in fish eaten by brown pelicans, it is not appropriate to perform EqP calculations using it to derive sediment criteria to protect benthic organisms. As discussed above, the sediment criteria for the protection of benthic organ-

isms ultimately proposed by EPA for dieldrin is based upon a Final Chronic Value, rather than the Final Residue Value. The Agency does not currently propose using residue-derived water quality criteria as a basis for sediment quality criteria. Therefore, the interim sediment quality criteria for DDT has not been retained for consideration as a potential remediation goal.

5.2.5. National Oceanic and Atmospheric Administration (NOAA) Effects Ranges

NOAA Technical Memorandum NOS OMA 52 (NOAA, 1991), entitled The Potential for Biological Effects of Sediment-sorbed Contaminants Tested in the National Status and Trends Program, contains an evaluation of data from a wide variety of sources and methods which have been used to establish effects-based sediment levels. Data were evaluated for numerous pollutants including DDT and dieldrin. For each pollutant, the lower 10th percentile in the data was identified as an Effects Range-Low (ER-L) and the median was identified as an Effects Range-Median (ER-M). NOAA states that these values, listed in Table 5-10, are not to be construed as NOAA standards or criteria.

NOAA reported confidence levels for the effects ranges calculated for each pollutant, based in part on the degree of clustering of results obtained by different methods and authors. Differences in results were not unexpected. As discussed by NOAA, data were assembled "from more than one approach and often from different methods used in any one approach. They included data from studies that involved species with different contaminant sensitivities; therefore, they are less likely to be equivalent and comparable." Other contributions to uncertainty include the results of co-occurrence analyses which suffer from numerous assumptions associated with matching biological and chemical data collected in the field, including the assumption that the chemicals that are quantified are those responsible for the effects observed.

Despite the uncertainties, the data for some pollutants, such as cadmium, are tightly clustered, and NOAA's reported confidence in the effects ranges is very high. However, for DDT, its metabolites, and dieldrin, the data on effects levels range from over 3 to over 7 orders-of-magnitude, and NOAA's confidence in the effects ranges calculated from the data is generally low.

The lack of data clustering for DDT and dieldrin may be due in part to the fact that the data set is not carbon-normalized. In addition, it should be noted that the lower values of the data sets are dominated by the results of EqP calculations performed using EPA's marine chronic criteria. Use of these data results in the calculation of particularly low ER-Ls. As stated in the discussions of EPA's interim sediment quality criteria above, the marine chronic criteria for both DDT and dieldrin are residue-based, and the ecological significance of EqP calculations using them is unclear. Use of these values in EqP calculations may result in criteria which are orders-of-magnitude below the levels necessary to protect benthic organisms. EPA is currently pro-

posing EqP-derived sediment quality criteria based upon toxicity test results rather than residue values.

Because of the low confidence reported by NOAA for the effects ranges calculated for DDT, its metabolites, and dieldrin, and the additional concerns discussed above, these values have not been retained for further consideration as potential remediation goals for sediments.

5.3. ADDITIONAL RESOURCES

Additional resources include information other than State and Federal standard or advisories for acceptable levels in contaminated media which may be used to derive remediation goals.

5.3.1. EPA Integrated Risk Information System

EPA's Integrated Risk Information System (IRIS) contains data which are used primarily in human health risk assessments. However, the chronic toxicity data also have been used in assessments of mammalian risk, and will be considered here in regard to marine mammals.

The most sensitive marine mammal in San Francisco Bay is probably the harbor seal, which is a permanent, breeding resident. Although field studies, including studies in San Francisco Bay (Risebrough *et al.*, 1979), have reported the presence of DDT and other pesticides in harbor seals, it is necessary to associate contaminant levels and effects. Studies done in the Wadden Sea reported both the DDT concentration in fish and reproductive failure in seals; however, it was concluded that the adverse effects were caused by PCB contamination rather than DDT (Reijnders, 1986).

IRIS contains reference doses for lifetime oral consumption of DDT and dieldrin which were established based upon the results of long-term feeding studies in rats. EPA's confidence in these values is medium to low. The occurrence of liver lesions was found to be the critical effect of both pollutants. Other effects, including adverse effects on reproduction, have been found to occur at higher doses.

The no-observed adverse effect levels (NOAEL) and low-observed adverse effect levels (LOAEL) for DDT and dieldrin doses in rats are presented in Table 5-11. When attempting to convert toxicity data between species, it is necessary to adjust doses to account for differences in the ratio of food consumption and body weight. However, since consumption rates for both rats and harbor seals are approximately 5% of their body weight per day (IRIS and Dr. Diane Kopec, personal communication), estimations for harbor seals can be made using the values above. An estimation of a protective level in prey can be obtained by dividing the NOAEL by a species sensitivity factor of 10 to reflect the uncertainty in extrapolating toxicity data from the rat to another mammal. This would result in a fish residue value of 100 ppb for DDT and 10 ppb for dieldrin. The confidence in this value would be low.

The values estimated above are higher than the National

Academy of Sciences recommendations for the protection of marine mammals and fish-eating birds discussed in Section 5.2.2., which were based on toxicity to birds and were retained for consideration as potential remediation goals. In addition, while it is known that some birds consume enough of their diet from Richmond Harbor to significantly accumulate DDT (Ohlendorf *et al.*, 1991), such an assumption may not be reasonable for the harbor seal, a much larger and far-ranging predator. Although harbor seals haul out on Brooks Island and are occasionally seen in Richmond Harbor channels, including the Lauritzen, their main breeding and haul-out grounds are in the South Bay.

Based on the limited information above, it is unlikely that harbor seals are the most sensitive ecological receptor to the pesticide contamination in Richmond Harbor, and no additional remediation goals have been established to protect marine mammals.

5.4. PRELIMINARY REMEDIATION GOALS

Preliminary ecological remediation goals were determined for concentrations of DDT (total, including metabolites) and dieldrin in surface waters, fish and shellfish tissues and sediments (Table 5-12). Also listed in Table 5-12 are the CA/EPA and FDA human-health levels.

A number of observations and conclusions were made in determining these preliminary goals. First, the most sensitive ecological receptors to DDT and dieldrin in the water column are likely to be fish-eating birds. If water column or surrogate fish and shellfish tissue residue levels are attained that are protective of sensitive birds, the water column concentrations will likely be orders-of-magnitude below the levels necessary to protect sensitive marine organisms in Richmond Harbor. Second, State of California and EPA human-health water quality standards and criteria for the protection of human health, and the tissue residue levels upon which they were based, are more stringent than the levels necessary to protect the most sensitive ecological receptors considered during the development of the criteria. Finally, a preliminary remediation goal for the protection of benthic organisms was determined for dieldrin in sediments. These levels are not directly comparable with the water column remediation goals, and it is not known if achieving this goal will result in the protection of water column organisms or aquatic birds. No preliminary remediation goal was determined for DDT in sediment.

5.4.1. Field Study Needs

Based on the preliminary remediation goals listed above, it is clear that field measurement of chemical concentrations in the water column, fish and shellfish tissues, and sediments are necessary to determine if the goals are exceeded in the various channels in Richmond Harbor.

The following additional types of field and laboratory studies, routinely performed on contaminated sediments, were also

selected. Guidance on methods for the assessment of contaminated sediment is contained in Remediation of Contaminated Sediments (U.S. EPA, 1991c).

- Sediment bioaccumulation tests. Since the ultimate goal of this ecological risk assessment is the determination of a cleanup criteria for sediment, it will be necessary through the field studies to establish the relationship between sediment concentrations and the resulting concentrations in the water column or in fish and shellfish tissues. Although tissue residues in field-collected benthos will be analyzed, it is uncertain whether sufficient tissues from comparable organisms will be collected at each sampling location. Bioaccumulation tests in the laboratory provide a controlled analysis of the relationship between sediment concentration and tissue residues and assure that sufficient tissue of the same species will be available for analysis.

- Sediment interstitial water analyses, benthic community analyses, and sediment toxicity tests. Sediment interstitial water analyses will allow direct comparison of pore water concentrations with water concentrations protective of sensitive marine organisms. Both benthic community analyses and sediment toxicity tests are standard measures which will be used to determine a DDT level protective of benthic organisms, and to determine whether the preliminary remediation goal for dieldrin is appropriate to protect benthic organisms.

One of the goals of ecological assessments is to collect sufficient information to support decisions regarding remediation, without collecting more information than is necessary. The studies listed above require only about one week of field work but will provide sufficient information to determine compliance with all applicable criteria and to characterize risks to benthic and water column organisms and higher predators including fish-eating birds.

There are almost a limitless number of other studies which could be performed, including detailed studies of any of the approximately thirty species of fish and eighty species of birds potentially affected by contamination at the site. However, studies of more mobile species, particularly migratory species, would require much more effort and would be subject to inherently higher uncertainty regarding pollutant sources and effects than the studies of sessile and relatively non-mobile organisms chosen here. Furthermore, studies of higher organisms, especially birds, are not necessary because criteria are available for their protection (EPA's Ambient Water Quality Criteria and California's Water Quality Objectives) which are based upon achieving much more easily measurable contaminant concentrations in fish and the water column.

Some chemical manufacturers who have been identified as

potentially liable for the site remediation have suggested performing field surveys to attempt to quantify site consumption rates of potentially impacted birds. The goal of these studies would be to justify the allowance of contaminant levels in fish which are higher than criteria levels, or to demonstrate that consumption rates are insufficient to present risk. However, even a field survey which did not involve tagging or tracking would require significantly greater and longer-term effort than the studies selected. Bird studies at some hazardous waste sites have spanned years. A study at this site would have to be particularly rigorous because of the large number of species potentially impacted, the presence of endangered species feeding in the most contaminated areas, and the fact that based on historic fish-tissue data, consumption rates below 1% may cause adverse impacts. In addition, the meaning of any measured consumption rate would be confounded by the unknown contribution of pesticides from the rest of the bird's diet and by the unknown impact of the extremely elevated contaminant levels in site channels on factors such as avoidance and the availability of prey. Finally, the data would not affect decision-making because EPA's Water Quality Criteria and the State's Water Quality Objectives, which are intended to be protective regardless of consumption rate, are ARARs and must be achieved by the selected Superfund remedy anyway.

In conclusion, for a very moderate field effort, the studies chosen for this ecological assessment provide sufficient information to characterize risk to sensitive ecological receptors, develop protective cleanup levels, and support decisions regarding site remediation.

5.4.2. Summary of Receptors and Methods to Predict End-Points

The field and laboratory studies described above generate an assortment of different types of data relating to various environmental media and ecological receptors. Table 5-13 summarizes the receptors studied and their primary exposure route, potential end-points, methods to predict sediment concentrations related to the end-points, and the potential effects of either other sediment pollutants or physical disturbance on the particular end-point. Obviously, such a table is an overview and the reader is referred to the appropriate sections for discussions of the methods, assumptions, and measures of uncertainty.

TABLE 5-1. State of California water quality objectives.

	Saltwater Aquatic Life Daily Average (ng/l)	Human Health 30-day Average (pg/l)
DDT*	1.0	600
dieldrin	1.9	140

* The sum of the p,p' and o,p' isomers of DDT, DDD (TDE), and DDE.

pg = picogram

TABLE 5-2. Beneficial uses of central San Francisco Bay.

Industrial Service Supply

Industrial Process Supply

Navigation

Water Contact Recreation

Non-contact Water Recreation

Commercial and Sport Fishing

Wildlife Habitat

Preservation of Rare and Endangered Species

Fish Migration

Fish Spawning

Shellfish Harvesting

Estuarine Habitat

TABLE 5-3. EPA ambient water quality criteria for DDT (µg/l).

	Saltwater Aquatic Life		
	Acute	Chronic	Human Health
DDT*	0.13	0.001	0.00059
(residue value µg/kg wet)		(150)[a]	(32)[b]

* DDT and metabolites.

a The 0.001 µg/l criterion is based upon achieving a whole body tissue wet weight residue of 150 ppb in fish in order to protect the California brown pelican, a sensitive piscivorous species. The criterion was obtained by dividing 150 by 17,870, the geometric mean of lipid-normalized bioconcentration factors, and by 8, the percent lipid content of anchovies - the major prey of brown pelicans.

b The 0.00059 µg/l criterion was based upon achieving a 10^{-6} excess cancer risk level for the protection of human health, corresponding to a wet weight residue of 32 ppb in fish and shellfish. The criterion may be obtained by dividing 32 by 53,600; the average bioconcentration factor in aquatic organisms. The updated IRIS carcinogenic potency factor used is 0.34 (kg-day/mg). The criterion assumes total fish and shellfish consumption rate of 6.5 g/day, of which 0.8 g/day is shellfish. If shellfish only from the contaminated area are consumed, the corresponding residue value is 260 ppb.

TABLE 5-4. Summary of saltwater criteria effects data for DDT.

Organism and Effect	Water Conc. (µg/l)	Fish Tissue (µg/kg, wet wt.)
Plants (reduced photosynthesis)	10	
Acute toxicity to fish and invertebrates	89 - 0.14	
Reduced survival in fish fry	0.21 - 0.10[a]	6250 - 3000
Chronic toxicity to sensitive invertebrates	0.065 - 0.013[b]	
Reduced reproduction in birds	0.001	150
Human health 10^{-6} level	0.00059	32

a This range is not presented in the criteria document, but was calculated from the reported fish tissue values for sac fry, fry and fingerling survival, the geometric mean of the lipid-normalized bioconcentration factor of 17,870, and the mean lipid content of 16% for saltwater fish.

b Not presented in the criteria document, but calculated from the criteria Final Acute Value (0.13 µg/l) and the typical range of Acute-to-Chronic ratios for chlorinated pesticides (2-10).

TABLE 5-5. EPA ambient water quality criteria for dieldrin (µg/l).

	Saltwater Aquatic Life		
	Acute	Chronic	Human Health
dieldrin	0.71	0.0019	0.00014
(residue value, µg/kg wet)		(300)[a]	(0.7)[b]

a The 0.0019 µg/l criterion was based upon achieving a residue of 300 ppb - the FDA action level for dieldrin - in fish oil. The criterion was obtained by dividing 300 by 1,557, the geometric mean of the lipid-normalized bioconcentration factors, and by 100, the percent lipid content of fish oil.

b The 0.00014 µg/l criterion was based upon achieving a 10^{-6} excess cancer risk level for the protection of human health, corresponding to a residue of 0.7 ppb in fish and shellfish. The criterion may be obtained by dividing 0.7 by 4,670, the average bioconcentration factor in aquatic organisms. The updated IRIS carcinogenic potency factor used is 16 (kg-day/mg). The criterion assumes total fish and shellfish consumption rate of 6.5 g/day, of which 0.8 g/day is shellfish. If shellfish only from the contaminated area are consumed, the corresponding residue value is 5.7 ppb.

TABLE 5-6. Summary of saltwater criteria effects data for dieldrin.

Organism and Effect	Water Conc. (µg/l)	Fish Tissue (µg/kg)
Plants (reduced productivity)	1,000	
Acute toxicity to fish and invertebrates	0.66	
Chronic toxicity to fish and invertebrates	0.11	
FDA Action level in fish tissue	0.012	300
FDA Action level in fish oil	0.0019	300[1]
Human health 10^{-6} level (~88% fish, ~12% shellfish)	0.00014	0.7
Shellfish only	0.00014	5.7

1 = Concentration in fish oil.

TABLE 5-7. Endangered species and species of special concern.

Common Name	Nests in SF or San Pablo Bays	Feeds In/Around Richmond Harbor	Prey
Endangered Species			
Brown Pelican[1]		Y	fish
Bald Eagle			omni
Peregrine Falcon	Y	Y	bird
Clapper Rail	Y		invert
Least Tern	Y	Y	fish
Special Concern			
Osprey			fish
Burrowing Owl	Y		invert
Short-eared Owl	Y		mammal
Snowy Plover	Y		invert
Long-billed Curlew			invert

KEY:
fish = consumes primarily fish
mammal = consumes primarily terrestrial mammals
bird = consumes primarily birds
omni = diet usually omnivorous/scavenger
invert = consumes primarily small to medium sized invertebrates

1 = Observed feeding in Richmond Inner Harbor (see Section 4.2.2. and Appendix 4-1.

TABLE 5-8. Food and drug administration action levels.

	(µg/kg wet)	Food Type
DDT (total)	5000[a]	fish
dieldrin	300[b]	fish, shellfish and fish oil

a = U.S. FDA Guideline 7420.08, 1978.

b = U.S. FDA Guidelines 7420.08, 1978 (fish and shellfish) and 7426.04, 1977 (fish oil).

TABLE 5-9. National Academy of Sciences saltwater action levels.

Residue Values in Whole Fish	
DDT (and metabolites)	50 μg/kg (ppb, wet)
dieldrin	5 μg/kg (ppb, wet)

TABLE 5-10. National Oceanic and Atmospheric Administration effects-ranges in sediment.

Pollutant	Effects Range (ppb) Low	Median	Confidence Level
DDT (total)	3.0	350	moderate
p,p'-DDT	1.0	7.0	low
p,p'-DDE	2.0	15.0	moderate/low
p,p'-DDD	2.0	20.0	low
Dieldrin	0.02	8.0	low

TABLE 5-11. IRIS chronic sublethal doses.

Pollutant	NOAEL (mg/kg)	LOAEL (mg/kg)
DDT	1	5
Dieldrin	0.1	1.0

NOAEL = No Observed Adverse Effect Level.

LOAEL = Lowest Observed Adverse Effect Level.

TABLE 5-12. Preliminary remediation goals for DDT and dieldrin.

	Water (µg/l)	Sediment (µg/g OC)	Fish Tissue (µg/kg, wet)
DDT and Metabolites			
CA/EPA Marine Chronic	0.001[a]		150
NAS Action Level			50
CA/EPA Human Health (based on ~88% fish, ~12% shellfish)	0.0006[a]		32
FDA Action Level			5,000
Dieldrin			
CA/EPA Marine Chronic	0.0019[a]		300[b]
NAS Action Level			5
EPA Sediment Criteria		7.7 - 36	
CA/EPA Human Health (based on ~88% fish, ~12% shellfish)	0.00014[a]		0.7
FDA Action Level			300

a = The value listed is an applicable or relevant and appropriate requirement of a Federal or State law (ARAR).

b = Concentration in fish oil.

TABLE 5-13. Media and receptors studied, end-points, methods to predict sediment concentration to achieve end-points, and effects of other stressors.

MEDIUM OR RECEPTOR	TROPHIC LEVEL	POTENTIAL END-POINTS	PRIMARY EXPOSURE ROUTE	METHOD TO PREDICT	EFFECTS OTHER POLLUTANTS	EFFECTS PHYSICAL DISTURBANCE AND TRANSPORT
SEDIMENT CONC.	NA[1]	COMPARISON OTHER SITES, SQC, AET[2], ERL-M[2]	HISTORIC DISCHARGES, TRANSPORT FROM MORE CONTAMINATED SITES	BACKCALCULATE FROM EFFECTS/EXPOSURE MODELS	DEPENDS UPON END-POINT, NONE ON SQC	DEPENDS UPON END-POINT
OVERLYING WATER CONC.	NA[3]	ACUTE AND CHRONIC WQC, CHRONIC EFFECT VALUES	INPUTS FROM LOCAL SEDIMENT, TRANSPORT FROM MORE CONTAMINATED AREAS	SITE-SPECIFIC WATER-SEDIMENT RATIOS (WSR)	NONE	TRANSPORT FROM MORE CONTAMINATED SITE ELEVATES WSR
INTERSTITIAL WATER CONC.	NA	ACUTE AND CHRONIC WQC, CHRONIC EFFECT VALUES	PARTITIONING WITH SORBED POLLUTANTS	SITE-SPECIFIC K_{oc}	NONE	HIGH INPUT MAY RESULT IN NON-EQUILIBRIUM
ACUTE AMPHIPOD BIOASSAY	2	COMPARISON TO REFERENCE AND CONTROL, HISTORICAL RESULTS	SEDIMENT[5]	REGRESSION WITH SED. OR IW CONC., TOXIC UNIT APPROACH	INCR. APPARENT TOXICITY OF DDT/DIELDRIN	NONE
LABORATORY MULINIA GROWTH TEST	2 (FILTER-FEEDER)	COMPARISON TO REFERENCE AND CONTROL	SEDIMENT[5]	REGRESSION WITH SED. OR IW CONC.	INCR. APPARENT TOXICITY OF DDT/DIELDRIN	NONE
BENTHIC COMMUNITY STRUCTURE	MOSTLY 2	"CONTROL" & "CHANGED" VALUES FOR INFAUNAL INDEX, RELATIVE CHANGES COMPARED TO REFERENCE SITE OR CONTROL CONDITIONS	SEDIMENT[5], SUSPENDED PARTICULATES FOR FILTER-FEEDERS	REGRESSIONS OF INFAUNAL INDEX, AMPHIPOD ABUN., AND 14 OTHER MEASURES OF COMMUNITY STRUCTURE WITH SED. OR IW CONC.	INCR. APPARENT TOXICITY OF DDT/DIELDRIN	INCREASE APPARENT TOXICITY OF DDT/DIELDRIN
RESIDUES IN FIELD INFAUNA	MOSTLY 2 (FILTER-FEEDERS)	CHRONIC WQC RESIDUES, NAS RESIDUES, CHRONIC EFFECT VALUES, [FDA, CANCER RISK][4]	SEDIMENT[5], OVERLYING WATER AND SUSPENDED PARTICULATES WITH FILTER-FEEDERS	EqP BIOACCUMULATION MODEL, REGRESSIONS WITH SED. OR IW CONC.	NONE	NONE
RESIDUES IN LABORATORY-EXPOSED MACOMA	2 (DEPOSIT-FEEDER)	CHRONIC WQC RESIDUES, NAS RESIDUES, CHRONIC EFFECT VALUES, [FDA, CANCER RISK][4]	SEDIMENT[5]	EqP BIOACCUMULATION MODEL, REGRESSIONS WITH SED. OR IW CONC	NONE	NONE
RESIDUES IN FIELD-COLLECTED MUSSELS	2	CHRONIC WQC RESIDUES, NAS RESIDUES, CHRONIC EFFECT VALUES, [FDA, CANCER RISK][4]	WATER AND SUSPENDED PARTICULATES	SITE-SPECIFIC BCF BASED ON WHOLE WATER	NONE	NONE
RESIDUES IN PELAGIC FISHES	MOSTLY 3.25	CHRONIC WQC RESIDUES, NAS RESIDUES, CHRONIC EFFECTS VALUES, [FDA, CANCER RISK][4]	WATER, POTENTIALLY INCLUDING SUSPENDED PARTICULATES	SITE-SPECIFIC BCF_{uw}	NONE	NONE
RESIDUES IN DEMERSAL FISHES	3.25	CHRONIC WQC RESIDUES, NAS RESIDUES, CHRONIC EFFECTS VALUES, [FDA, CANCER RISK][4]	1. SEDIMENT OR 2. BENTHIC PREY	1. EqP MODEL OR 2. TROPHIC TRANSPORT MODEL	NONE	NONE
EXPOSURE OF FISH-EATING BIRDS[6]	4-5	CHRONIC WQC RESIDUES, NAS RESIDUES	FISH, INVERTEBRATES	PREDICT RESIDUES IN PREY (FISH), ASSUME CONSUMPTION RATE	NONE	NONE

1 = SEDIMENT DETRITUS AND ASSOCIATED MICROFLORA CAN BE CONSIDERED TROPHIC LEVEL 1 FOR DEPOSIT FEEDERS
2 = ERL-M AND AET FOR DDT WERE NOT USED AS FINAL END POINTS
3 = PHYTOPLANKTON AND SUSPENDED DETRITUS CAN BE CONSIDERED TROPHIC LEVEL 1 FOR FILTER FEEDERS
4 = HUMAN HEALTH END-POINTS ARE FOR REFERENCE AND ARE GIVEN IN "[ ]".
5 = SEDIMENT DOES NOT DISTINGUISH BETWEEN UPTAKE FROM SOLID PHASES AND INTERSTITIAL WATER.
6 = EXPOSURE OF FISH-EATING BIRDS IS PREDICTED RATHER THAN MEASURED.

6.0 GENERAL STUDY METHODS

6.1. INTRODUCTION

This section covers the general sampling methods, including the sampling numbering system, an overview of the analytical techniques, and an overview of the samples taken. A more detailed discussion of the methods can be found in the United Heckathorn Superfund Site Study (U.S. EPA, 1991b), the study plan and QA/QC document that was produced by the ERL-N, Newport laboratory for this study. Modifications to the methods described in U.S. EPA (1991b) are identified in the appropriate sections of this document and, as necessary, detailed descriptions of the modified methods are given in an appendix. The locations of the sampling sites are given in Figures 6-1 and 6-2.

Chain-of-custody procedures were followed during all field collections and laboratory work at the EPA, ERL-N laboratory in Newport, Oregon, and other participating laboratories. The chain-of-custody procedures are summarized in Appendix 6-1.

6.2. STATION LOCATIONS AND OVERVIEW OF FIELD SAMPLING

Sample site locations were chosen considering results from previous investigations (Long *et al.*, 1988; Levine-Fricke, 1990; Brown *et al.*, 1990), and the goal to develop predictive relationships. The criteria used included:

1. A wide range in sediment chemistry concentrations in approximately a geometric series.

2. Sediment concentrations that bracket the interim EPA marine sediment quality criteria for dieldrin (170 µg/kg, dry weight, at 1% TOC).

3. Sediment concentrations that bracket the interim EPA marine sediment quality criteria for DDT (8.3 µg/kg, dry weight, at 1% TOC).

4. Avoid confounding effects, such as sites near combined sewer outfalls.

5. Sites with as nearly similar sediment grain size as possible.

6. Avoid sampling recently dredged areas or areas subjected to ship disturbance (*i.e.*, middle of channel).

7. At least 10 feet of water to allow boat access.

Using the available data, we chose nine primary stations (Stations 1-9) - four in Lauritzen Channel, two in Santa Fe Channel, and three in Richmond Inner Harbor Channel (Figure 6-1). The Richmond Inner Harbor Channel station at the mouth of the

Harbor was designated the local reference site in the field study while local Oregon sites were used as controls in the laboratory toxicity and bioaccumulation tests. An additional 11 sites designated as "secondary" stations (Stations 009, 13-20, 22, 23) were located between primary sites in the inner area and extended into San Francisco Bay, past Point Richmond along the entrance channel.

The standard procedure was to take five replicate sediment grabs at each of the primary stations. Sediment for chemical analysis and biological testing was then collected from each grab with cores. Eight replicate grabs were taken at one station, Station 1, to obtain greater statistical power at a highly contaminated site. To measure within-grab variability, triplicate cores for chemical analysis were taken from one of the grabs at Stations 1, 5, and 7. To determine the effects of sediment storage and mixing, pollutant concentrations in bioassay sediment were compared to the field sediment samples that were processed immediately. A single grab was taken at each secondary site for chemical analysis. Additional grabs were taken at several of the stations to collect infauna for tissue analysis. All the benthic grab sampling was conducted in October, 1991.

The study was designed so that samples for sediment chemistry, sediment bioaccumulation testing, sediment toxicity testing, and benthic community analysis were all taken from the same grabs. The standard procedure at the primary stations was to take separate sediment cores from each grab for analytical chemistry (sediment chemistry core; bulk sediment and interstitial water from same core), grain size analysis, a bioaccumulation test beaker (laboratory sediment bioaccumulation core), and a toxicity test beaker (sediment toxicity test core)(Figure 6-3). This approach allows a direct correspondence between chemical concentrations and biological responses on a 0.1 m^2 scale. In order to reduce the number of samples that were analyzed, only three of the five (eight at Station 1) Sediment Toxicity Test core samples were chemically analyzed. The remaining two (five at Station 1) Sediment Toxicity Test core samples and all the Laboratory Sediment Bioaccumulation Test core samples were used as exposure sediments and were not analyzed for pollutants. The appropriateness of using the pollutant concentrations from the minimally disturbed Sediment Chemistry core samples for the acute bioassay and bioaccumulation tests which used stored and mixed sediments was evaluated by splitting the stored and mixed Sediment Toxicity Test core samples into an analyzed T_0 bioassay storage sediment and a not analyzed acute bioassay exposure sediment. The results from replicated Sediment Chemistry cores were compared to the stored and mixed T_0 bioassay storage sediment to evaluate if storage and mixing produced effects that exceeded the within-grab variability of the cores.

Fish and benthic megafauna (i.e., larger invertebrates caught in trawls such as crabs and shrimp) were collected by trawling in October, 1991 and February, 1992. Locations for trawling were assigned over Station 8 in Richmond Harbor Channel,

over Stations 5 and 6 in Santa Fe Channel, and from approximately Station 1 to 2 in Lauritzen Channel. Locations for the October and February trawls are given in Figure 6-1 and 6-2, respectively. To increase the number of individuals, the infaunal species (*e.g.*, clams) caught in the February trawls were included in the analysis of field-collected infaunal tissue residues.

Detailed station descriptions are given in Appendix 6-2. As noted in the trip report (Specht, 1991, Section III), some stations were changed or dropped from the original plan (U.S. EPA, 1991b) due to conditions encountered in the field.

6.3. SAMPLE NUMBER SYSTEM

6.3.1. Field Sample Numbers

The sample numbering system is derived from the sample numbering protocol devised by the Pacific Ecosystems Branch, and, as amended for this project, is detailed in Food Web Research Note #167 (9/3/91) (internal planning document for the Bioaccumulation/Stratozone Team, Newport, OR), excerpted here.

The sample code uses 8 characters (plus the 5 character analytical request number, described below). The first character is a letter which identifies the major operating unit where the project originates; in this case, the lead unit is the Bioaccumulation Team, and the letter is **S**. The second character is the year of origin of the particular sample. The project started in 1991, so the second unit is **1**. The third character is the project designator, sequentially assigned. Any sample associated with the October, 1991 sampling for this project can be identified by this third unit, which is **C**, designating Superfund/-Lauritzen Channel/United Heckathorn. The third character for the February, 1992 sampling is designated as **A**.

The fourth character is the task identifier. These are letters assigned within the project sampling plan, and are as listed below, starting with **A**, continuing through **K**.

Tasks:

A Alternate use assigned to designate sediment samples from the Sediment Chemistry cores.

B Sediment Chemistry (field sampling jars labeled B) Sediment samples labeled A; interstitial water samples labeled B.

C Laboratory Sediment Bioaccumulation Tests.

D Tissue Residues in Field-Collected Infauna.

E Sediment Toxicity Tests (field sampling jars labeled E) Sediment samples from T_0 bioassay storage sediments labeled E; interstitial water samples labeled B.

F Benthic Community Analysis.

G Benthic Recruitment (later dropped from study).

H Tissue Residues in Field-Collected Fish and Mussels.

I AVS (Acid-Volatile Sulfide) analysis.

J Grain size (particulates).

K Bioassay.

The fifth character designates the sample type:

S for sediment;

F for fecal sample;

T for tissue;

I for interstitial water;

O for overlying water;

X for standards;

B for blanks;

C for control samples.

The sixth through eighth character constitute the sample number, **001-999**. These numbers are assigned sequentially, or by prearranged series, according the decision of the sampler and/or project coordinator. In this series, the first two digits indicate station number (01-23), and the last indicates grab number from that station (1-8). One exception to this is the station designated 009, from which one replicate series of samples was completed before abandoning in favor of locating a more promising control station.

The February, 1992 sample numbers are designated S2A__xxx, indicating Bioaccumulation Team, 1992, Superfund/Lauritzen Channel/United Heckathorn Project, with the tasks, types and numbers assigned as described above.

To summarize, the project sample code looks like this, **S1CDT023**, from which one may deduce that this is a Bioaccumulation Team project sample, taken in 1991, the specific project is the Superfund/Lauritzen Channel/United Heckathorn Project, the task is Tissue Residues in Field-Collected Infauna, tissue sample, and it is from the third grab of Station 2. Further qualifications, characterization or information about a sample are contained as notes in the sample list.

The location of specific samples on the site maps (Figures

6-1, 6-2) are indicated in the October, 1991 samples by the 6th and 7th digits of the sample code. The "F" numbers for the water, mussel, and trawl samples in February, 1992 (Figure 6-2) are converted to specific sample numbers in Appendix 6-3.

6.3.2. Analytical Request Numbers

When the samples were processed in the analytical laboratory, they were assigned a request number and a code designating if a subsample or aliquot was taken and the type of subsample and whether a sample was reinjected. The complete codes are given in the electronic spreadsheet data files containing the raw data and a modified version in Appendix 8-1. The average values of subsamples or reinjections were used in the present analysis, and the use of the analytical request number is only necessary to determine whether a value consists of the average (in Appendix 8-1) or to determine the individual subsample/reinject value (in electronic spreadsheet files).

The request number is found in the first 3 digits in the individual sample number (*e.g.*, 408). The next two digits indicate whether a second or third subsample was taken. For all sediment samples, the first digit is a "1" indicating the subsample extract aliquot used for this study with the second digit indicating whether the results of a second or third subsample were reported; "1 0" indicates that no other subsample was taken, "1 1" either indicates that the first extract was compromised and a second subsample was processed in its place or that it was the first of three subsamples ("1 1", "1 2", and "1 3") used to evaluate analytical variability. When a single, entire tissue sample or subsample was processed, the designation was "0 0". If subsamples were taken to assess variability, three subsamples were taken and they were designated as "0 1", "0 2" and "0 3" or "1 0", "2 0" and "3 0". In all cases, the interstitial water sample was subsampled to form two fractions, with "01" identifying the total interstitial concentration and "02" indicating the bound concentration. The free fraction was determined by the difference between the total and bound fractions.

With the tabulated values in Appendix 8-1, samples for which reinjections of an extract were run are identified with a "R" (for Reinject) after the sample number. The "ave" indicates that the reported value is the average of the three injections of the sample extract during a GC/MS run. The individual values of each injection are given in the electronic spreadsheet data files. When the sample name contains "ave" with no trailing-R, the value is from averaging subsamples from a medium (*e.g.*, sediment) and not reinjections. Again, the individual values are given in the electronic spreadsheet data files.

As an example, 408 10 S1CASXXX designates a sediment sample that it is from request 408. The 1 of the 10 is the same for all sediment samples. The number occupying the 0-position in this example indicates that a single within-laboratory subsample was planned for this sample and that it was successfully extracted.

Any other number in the 0-position indicates that triplicate subsamples were planned for this sample or that the first and/or second subsamples were compromised in some way requiring additional subsamples to be taken. In tissue samples, a 00 indicates that a single subsample (or the entire sample) was successfully extracted; 10,20,30 or 01,02,03 indicate that triplicate subsamples were planned and extracted successfully. 408 12 S1CASXXX would designate the second of three subsamples if 11 and 13 were also taken, if not, 10 subsample was compromised. For interstitial water samples 407 01 S1CBIXXX and 407 02 S1CBIXXX, the codes would designate the total pollutant and bound pollutant extracts, respectively.

6.4. FIELD SAMPLING

The detailed field sampling protocol in Section II of the Trip Report (Specht, 1991) is excerpted and amended in Appendix 6-4 for the exact sequence of steps followed in sampling sediment, interstitial water, infauna, and related physical measurements (Specht, 1991, Section II, p. 3). The remainder of this Section summarizes the procedures.

The October, 1991 sampling was accomplished from the R/V Shana Rae, a commercially chartered 42' research vessel owned and operated by Capt. Jim Christman of Monterey Canyon Research, 14 Mason St., Santa Cruz, CA 95060, (408)423-4864. A small inflatable boat (Zodiac®) was used in the collection of samples away from the main operation or adjacent to the shore (mussel and overlying water samples; observation; transfer of off-loaded samples to shore for transport to UPS shipping point each mid-afternoon). February, 1992 sampling was accomplished from a chartered 24' Boston Whaler® operated by Mr. John Brezina, P. O. Box 25, Dillon Beach, CA 94929, (707) 878-2853.

The length of the anchor line was about 50'(~15 m), the average anchor scope was about 1:4, the average depth on station about 30'(~10 m), the average swing at anchor was estimated to be about 20' (~6 m). Using the above approximations, we estimate that the average distance between grabs to be not more than 2 m, and probably on the order of ~1 m.

6.4.1. Test Sediment Collection

The October, 1991 sediment samples (S1CBSxxx) were collected by a modified 0.1 m^2 van Veen® grab (Kahlsico® #214WA265, all stainless steel construction). The grab was operated with a hydraulic winch and boom off the deck of the R/V Shana Rae while at anchor. In general, the procedure involved lowering the grab to within ~1 m of the bottom, stopping briefly, and then releasing the grab to impact the sediment surface. The slack was then taken up until taut and the grab was firmly closed. The sampling depth was recorded from a snatch-block meter wheel (in feet), verified by fathom markings on the cable, and the grab lifted through the water column and set down on the grab stand on deck. The flaps were opened for inspection, and subsampling proceeded if the contents were judged sufficient (*i.e.*, at least 10 cm

depth (those few samples not quite reaching 10 cm are noted in the field log; see Appendices 6-2 and 6-4), relatively even fill).

For the primary stations, subsampling the grab consisted of the following operations:

1) insert E_H probe and thermometer to ~1 cm depth and read mv at 30 seconds, temperature when stable; push probe and thermometer to ~5 cm and read as previously;

2) insert appropriate cores for given station, *i.e.*, 4.1 cm id glass corer for sediment chemistry, 3.6 cm id plastic corer for grain size analysis, 2 each 7.6 cm id glass corers, 1 each for 28-day Macoma nasuta sediment bioaccumulation and amphipod sediment toxicity tests, 3 each 8.0 cm id plastic corers for benthic community analysis, and a modified plastic 10 cc "Plastipak" syringe for AVS sample, all to at least 10 cm depth.

3) The remainder of the grab contents were sieved to collect infauna for tissue residue analysis.

4) In addition, several separate grabs were taken to fill two 32 oz (~2 l) glass jars for long-term "kinetic" M. nasuta bioaccumulation bioassay tests.

At the secondary stations, subsampling the grab consisted of the following operations:

1) E_H and temperature readings.

2) Coring for sediment chemistry and grain size.

3) Sieving the remainder for collection of infauna for tissue residue analysis.

The AVS sample was immediately capped, bagged, labeled and frozen on dry ice. Cores for sediment chemistry, sediment toxicity, bioaccumulation tests, grain size and kinetic bioassay were placed in appropriate containers and immediately placed on "gel" ice in ice chests. The benthic community analysis samples were immediately sieved in running seawater, picked and placed in prelabeled jars of buffered seawater formalin. The sieved infauna from the residua of the grab were placed in white polypropylene 10 qt buckets of fresh seawater, aerators added, covered and allowed to purge for 24 hours (temperature was periodically checked, and gel ice packets added as needed to control temperature). After purging, the infaunal sample was bagged and frozen.

6.4.2. Control Sediment and Bioassay Organism Collection

Control sediment (S1CES001-003) for sediment toxicity

testing was obtained at Ona Beach, about 7 miles south of the Newport, Oregon lab, at the time of collection of the bioassay organisms (week of October 6-11, 1991). Samples of surficial sediment were dug from the north bank of Beaver Creek at Ona Beach with a washed shovel, and transported to the laboratory in clean plastic buckets. The sediment was washed through a 1.0 mm stainless steel sieve, salinity adjusted to 28‰, and placed in a covered polypropylene bucket in 4°C storage in the dark until used.

Control sediment (S1CKSxxx) and clams (Macoma nasuta) for the bioaccumulation test were obtained from Idaho Flat, Yaquina Bay, adjacent to the laboratory, on October 8, 1991. The sediment was collected by skimming the top ~4 cm with a washed shovel into washed polypropylene buckets, transported to the lab, passed through a 1 mm stainless steel sieve, and stored in 1 gallon glass jars (Teflon-lined cap) at 4°C in the dark. The jars were rolled for 30 minutes on a rolling mill at 4°C twice daily to assure that the sediment remained aerobic until used. A subsample of this sediment was included in the sediment-toxicity test chemistry request as sample numbers S1CES005-007.

The clams were dug at the same location, sieved out at the site on coarse (≥ 4 mm) Vexar® screens, collected in washed buckets with ambient seawater, transported to the laboratory for sorting, cleaning, weighing, measuring and numbering. Clams were held in control sediment in ambient filtered running seawater under a 12:12 day/night light cycle until used.

6.4.3. Benthic Community Sampling

Samples for benthic community analysis (S1CFSxxx) were obtained by vertically inserting three 8.0 cm i.d. cores (surface area of each core ~0.005 m^2) to at least 10 cm depth into each grab sample. The cores were then lifted out by hand and the top 10 cm of the core was extruded through the top of the core using a rubber plunger with a graduated stem. The three core samples from a grab were combined in the field resulting in a 0.015 m^2 x 10 cm deep sample. Sample contents were passed through stacked 1.0 and 0.5 mm mesh stainless steel sieves while being gently washed with ambient seawater. The material retained on each sieve was removed with spoons and forceps, placed in separate pre-labeled sample jars, and preserved in a buffered solution of 10% formalin in seawater. The individual jars were returned to fitted wooden packing cases which were sealed with chain-of-custody seals, off-loaded on arrival at the dock to the truck, and kept locked up through transport to Newport, where they were transferred to the Sample Custodian.

6.4.4. Collection of Infauna, Fish, and Mussels for Tissue Residue Analysis

Most of the infaunal species for tissue residue analysis (S1CDTxxx) were collected by sieving the remaining sediment in grabs with ambient seawater through a #5 (4 mm) sieve. Organisms were placed in 10 quart, white polypropylene pails

containing aerated ambient seawater and sieved Yaquina Bay control sediment and then allowed ~24 hours to purge their gut contents. These pails were loosely covered, placed on the deck out of the sun, and the temperature periodically checked. Upon arrival at the dock for the night, the pails were covered and chain-of-custody seals applied. The vessel was attended throughout the night by the skipper. The samples were examined the following morning, and maintained until the 24 hour purging period had elapsed. The animals were then rinsed of sediment and wrapped in baked-out aluminum foil, placed in a pre-labeled Bitran® Saranex double-lock closure bag and frozen on dry ice in a cooler on deck. The coolers were resupplied with dry ice as needed to maintain temperature. Those containers were transferred to a truck that was locked when unattended, maintained with dry ice while under transport to Newport, transferred to laboratory freezers, and accepted by the Sample Custodian.

To increase the number of samples, incidental infauna captured during the trawls were collected for tissue analysis. The trawl infauna were treated the same as the grab infauna except that the trawl infauna were not purged (see Appendix 6-2).

Fish and epibenthic invertebrate ("megafauna") samples taken in October, 1991 (S1CHTxxx) and February, 1992 (S2AHTxxx) were obtained by trawling with a 26' headrope otter trawl over a station or from crab traps deployed at specific locations (Figures 6-1, 6-2). Most trawls were taken in a seaward direction. Specific details are covered in Section IV of the Trip Report (Specht, 1991) and Dr. David Young's field notes (1992a) from February, 1992.

Mussel samples collected in October, 1991 (S1CHT5xx) (Figure 6-1) and February, 1992 (S2AHT5xx) (Figure 6-2) were obtained at three stations: Lauritzen Channel ferry rudder, Santa Fe Boathouse float and Richmond Harbor Channel Buoy red nun #16 anchor chain. Specimens were obtained from the intertidal zone, ranging from points about 0.2 meters above to about 0.3 meters below the water surface, except the sample from the Lauritzen Channel from the rope NE of the ferry, which was from approximately mid-depth (sample # "F-1", S2ADT-20X; ~2 m depth), and the Ferry rudder (sample "F-25", S2ADT-53X; 1.5-4 m depth). Sampling details are covered in the Trip Report (Specht, 1991), Sections V and VI, and Dr. David Young's field notes (1992a) for the February trip, pp. 410008-410042.

6.4.5. Field Measurements

Temperature was measured at 1 and 5 cm depth in the grab contents with a glass mercury-filled field thermometer accurate to ± 0.5°C; E_H measurements were taken by inserting a Beckman E_H probe to 1 and 5 cm. All readings were allowed to equilibrate for ~30 seconds. The thermometer and probe were rinsed with deionized water between grabs. The temperature of the overlying water was taken before "cracking" the grab (and allowing the overlying water to slowly drain off, avoiding disturbance or erosion of the surface). Salinity measurements were made on an

intermittent basis with an AO® refractometer accurate to ~1°/.., and pH with a portable Radiometer® pH meter and combination probe calibrated on site each day with appropriate buffers.

6.4.6. Water Sample Collection and Transport Methods

Overlying water samples were collected in October, 1991 (S1CHOxxx) and February, 1992 (S2AHOxxx) at low, mid and high tide stages within ~1 m of the mussel sampling sites, and ~0.3 m below the surface, from the inflatable dinghy. The exact procedure is related in detail in the transcription of Dr. David Young's October Field Notes (1991b) in the Trip Report (Specht, 1991), Section VI, pages 3-4. Briefly, a ~500 ml sample was taken in a precleaned glass bottle, placed in an ice chest with refrigerated gel-ice, returned to the vessel, chain-of-custody seals applied, and off-loaded with the rest of the samples for transport by UPS to Newport each afternoon.

The February, 1992 samples were similarly collected, but from a 24' Boston Whaler operated by Mr. Brezina (see Dr. David Young's field notes (1992a, 1992b)). Samples were similarly cooled with gel-ice in a cooler on deck. The coolers were stored in the locked trunk of an automobile or in a locked motel room during subsequent field work. The samples were periodically resupplied with gel-ice, transported for air shipment to the Oakland Airport at the conclusion of the field work, and transferred to the Sample Custodian at Newport for cold storage and distribution. For further detail, see Dr. Young's notebook (1992b), p. 402018.

6.5. SEDIMENT STORAGE AND MIXING

Sediment was returned to the Newport laboratory in plastic ice chests, cooled by refrigerated gel-ice packets. Upon acceptance by the Sample Custodian, these pre-numbered sample containers were stored in the dark in a refrigerated (4°C) walk-in ventilated cold room (ASTM, 1991) within laboratory S-118, which has controlled access.

The Sediment Chemistry core samples were processed within 24-36 hours of collection and were not mixed before processing (see Sections 6.6.1. and 6.6.3.). The IW samples were extracted and the sediment solids were frozen within four to six hours of separation. The sediment solids were extracted between one-half and five months later.

The Sediment Toxicity Test and Laboratory Sediment Bioaccumulation Test core samples were stored at 4°C for four to seven days and one to two weeks, respectively, before they were mixed gently (ASTM, 1991; U.S. EPA, 1991b) with a clean spatula, large debris (*e.g.*, shell) picked out, and portioned out for use in bioaccumulation or toxicity tests. Bulk sediment (S1CKSxxx) for the Macoma nasuta kinetic test was rolled twice a day, until used, to avoid development of anoxic conditions (see Section 8.3.2.).

To determine the effects of sediment storage and mixing,

three randomly selected Sediment Toxicity Test samples from each station (sum 27) were mixed, and dispensed into two beakers, one for chemical analysis and one for the amphipod toxicity bioassay. The beaker contents used for chemical analysis are referred to as the "T_0 bioassay storage sediment" and was stored and mixed exactly like the other bioassay sediments except that no amphipods were added to the beakers. Later the same day (day 0), the beakers were sampled for interstitial water (Section 6.6.1.), bulk sediment (Section 6.6.3.), and AVS (Section 6.6.8.).

Because of the need to reduce the number of samples, only three of the five (eight at Station 1) Sediment Toxicity Test core samples were analyzed. For these three samples, there is a one-to-one correspondence between measured sediment concentration values and toxicity; however, for the remaining two (five at Station 1) toxicity tests there is not such a correspondence. Subsequently, all the toxicity and bioaccumulation results from a grab were compared to the concentrations determined in the Sediment Chemistry core sample(s) from the same grab because only some of the Sediment Toxicity Test core samples and none of the Laboratory Sediment Bioaccumulation Test core sample were analyzed. The assumption was that the additional storage time and mixing experienced by the Sediment Toxicity Test and Laboratory Sediment Bioaccumulation Test sediments did not significantly effect any of the parameters. A storage/mixing effect was tested by comparing each T_0 bioassay storage sediment concentration with the minimally-disturbed within-grab Sediment Chemistry core sample mean or its representative single value (in the unreplicated grabs).

6.6. ANALYTICAL TECHNIQUES

Detailed analytical methods can be found in Task A, Appendices A and B, of the United Heckathorn Superfund Site Study (U.S. EPA, 1991b). Any changes to these methods are detailed in the appendices of this document.

The pertinent appendices are Appendix 6-5 for the cleanup to tissue and sediment samples, Appendix 6-6 for the determination of total and bound pollutants in water, Appendix 6-7 for the homogenization of tissue samples, and Appendix 6-8 for GC/MS procedures.

Various chemistry methods are summarized below, while an overview of the procedure is as follows. The analytical procedures focused on seven compounds - 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, 4,4'-DDT, and dieldrin. Sediment samples were centrifuged and the interstitial water (IW) was subsampled for salinity, dissolved organic carbon (DOC), and split into two total dissolved (DOC-bound and free) pollutant subsamples. These various interstitial water fractions are considered "dissolved" because samples were centrifuged to remove particles >0.45 μm. The free and DOC-bound pollutant fractions were separated in an IW subsample by application of the subsample to a C-18 reverse phase column (RPC). The total and DOC-bound

samples were liquid-liquid (L-L) extracted. The sediment pellets resulting from centrifugation were frozen for one-half to five months before they were thawed, then thoroughly mixed and subsampled for total organic carbon (TOC), percent solids, and pollutant analysis by sonication with acetonitrile and C-18 RPC cleanup.

The reason that state-of-the-science methods are used for IW analysis is because the traditional water methods (EPA 625 and SW-846/8270) require 1 liter of sample. At most, a total of ~50 ml of interstitial water is available from minimally disturbed sediment, and analysis by GC/MS in the scan mode results in methods-detection limits orders of magnitude higher than needed for this study. The sediment/tissue extraction and quantification method used in this study (Ozretich and Schroeder, 1986) has been shown to be equivalent to other published methods, and it is similar (sonication) to the sediment extraction method SW-846/3550. Our quality control acceptance criteria result in more accurate results because we internally correct for procedural losses (SW-846 accepts, without correction, recoveries of ~20% to ~140% of surrogate compounds added to water and sediment samples); our procedures insure degradation of DDT to less than 20% as the SW-846 requirement, but, in addition, we make adjustments to the DDD concentrations that make them more accurate given this unavoidable dechlorination. Some or all of these procedures have been used successfully in Swartz *et al.* (1989, 1990, 1991), Ferraro *et al.* (1990) and DeWitt *et al.* (1992).

Carbon and pollutant concentrations in the overlying water are considered "total." Because the overlying samples were not filtered or centrifuged the carbon and pollutant concentrations include particulate-bound, DOM-bound, and free fractions. Water column samples were subsampled for salinity and TOC determinations and the remainders were L-L extracted for pollutant analysis within 24 to 36 hours of sampling.

Pollutant quantitation was by capillary chromatography and selected ion monitoring using response factors relative to surrogate internal standards added prior to sample extraction. Procedural blanks, recovery of compounds from spiked matrices and reference matrices and solutions were used to evaluate performance. Pollutant identity was confirmed by retention time and use of an ion pair (the ratio of the major ion fragment to a second fragment). If the ion pair was within 20% of the expected ratio, the compound was considered confirmed. If the ion pair deviated by more than 20% from expected, a second ion pair was determined using a different secondary fragment. If this ion pair was within 20% of expected, the compound was considered verified. Values confirmed using the second ion pair were marked "NC" in the Appendices and computed at their stated values. There is a high confidence in these values. Samples in which neither of the ion pairs were within ± 20% of expected were not considered verified ("NC3" in Appendices). Although the NC3 values were reported and the computations used their stated value, these non-confirmed values should be considered as upper

limits as some or all of the compound could be a non-target compound (see Section 6.6.6.).

Values that initially exceeded the highest standard concentration were diluted, reanalyzed, and the reanalyzed value reported. Values that were lower than the lowest standard but above the detection limit are reported at their stated values. Computations treated values that were less than detection limits, <DL, as zero except with some calculated values, such as accumulation factors (see Section 8.3.), in which case values below detection limits were treated as missing values. Using zero for values less than the detection limit tends to underestimate the pollutant concentrations at the lowest concentrations. This approach was used rather than using the detection limit or half the detection limit as a balance to the use of the NC3 values, which could tend to overestimate the actual concentration. Recognizing that the number of significant digits for values have to be determined in the context of their use, we report values with more significant digits than may be necessary.

6.6.1. Interstitial Water (IW)

Processing of field sediment and interstitial water samples commenced within 24 to 36 hours of sampling. These samples were shipped with refrigerant and received cool. The entire contents of the field-filled Sediment Chemistry sample jar were transferred with minimal disturbance (*i.e.*, not homogenized) into a centrifuge bottle. Sediment samples from the Sediment Toxicity Test core samples (T_0 bioassay storage sediment) were homogenized before centrifugation, as described in Section 6.5. Sediment was centrifuged at 5,000 rpm (~3,200 x g) for 90 min at 4°C.

Water overlying the sediment pellet was aspirated following procedures found in Appendix A, section VI of U.S. EPA (1991b), and a 12 mL aliquant was L-L extracted (A-VII, U.S. EPA, 1991b). This extract constituted the total dissolved IW pollutant fraction (free and bound). An additional 12 mL was passed through a cartridge containing a C-18 solid phase (A-VIII, U.S. EPA, 1991b, as amended; Appendix 6-6) to retain the freely dissolved compounds and allow the DOM-associated (bound to dissolved organic matter) pollutant fraction to pass through. This bound fraction was also L-L extracted. The free IW pollutant concentration was determined by subtracting the bound fraction from the total fraction.

Additional subsamples of interstitial water were taken for salinity (B-VI, U.S. EPA, 1991b), and acidified for the determination of dissolved organic carbon (DOC)(B-III, U.S. EPA, 1991b). DOM was quantified as DOC.

6.6.2. Water Column Water

Unfiltered, 0.5-liter samples were shipped with refrigerant and received cool. Subsamples for TOC and salinity were taken before pollutants were extracted (within 24-36 hours of sampling) using a gentle, over-night L-L procedure with 12 mL of solvent (A-IX, U.S. EPA, 1991b).

6.6.3. Sediment Processing

Sediment samples (*i.e.*, pellet from the centrifugation in Section 6.6.1.) were frozen after sampling IW, and extracted within 1/2 to 5 months. The sediment pellet from centrifugation was thawed and subsampled for pollutant, percent solids, and total organic carbon analyses. The pollutant extract was obtained by desiccating 2-g samples with sodium sulfate, sonication in acetonitrile, and cleanup by passage through a C-18 solid phase cartridge (A-IV, U.S. EPA, 1991b). Percent solids were determined by oven-drying at 105° C (A-V, U.S. EPA, 1991b) and total organic carbon (TOC) was determined on acidified samples (B-I, U.S. EPA, 1991b) by high temperature combustion (B-II, U.S. EPA, 1991b). In addition to the C-18 cleanup, the sediment extracts were passed through a column of silica gel (Appendix 6-5).

6.6.4. Tissue Processing

Whole animals or dissected portions were frozen in liquid nitrogen and pulverized with mortar and pestle (Appendix 6-7). Subsamples were distributed for pollutant, percent solids, and lipid determination. Pollutant extracts were obtained following desiccation with sodium sulfate, sonication in acetonitrile, and cleanup by passage through C-18 and aminopropyl solid phase cartridges (A-XI, U.S. EPA, 1991b). In addition to the C-18 cleanup, the tissue extracts were passed through a column of silica gel (Appendix 6-5). Percent solids of most tissues was determined following freeze-drying, although some were oven-dried because of insufficient tissue mass (A-XII, U.S. EPA, 1991b). Lipid content was determined on freeze-dried subsamples using a micro-volume technique that uses methanol-chloroform as the solvents (B-V, U.S. EPA, 1991b).

6.6.5. Pollutant Quantitation

Samples for pollutant determination were amended with surrogate internal standards in methanol prior to extraction (A-IV and A-XI, U.S. EPA, 1991b) and the targeted compounds were quantified relative to these surrogates (*i.e.*, any procedural losses sustained by the surrogate are assumed to be sustained by the targets resulting in recovery-corrected target concentrations). Loss of the surrogates was determined by the addition of a recovery standard at the completion of the procedure when the extract was at its final volume. The mass of surrogate compounds, deuterated 4,4'DDE (DDE-d8) and ^{13}C-heptachlor epoxide, added to each sample was equivalent to the approximate historical median concentrations of 4,4'DDD and dieldrin, respectively, found within Lauritzen, Santa Fe, and Richmond Inner Harbor Channels.

Target compounds in the extracts were quantified using high performance gas chromatography with mass spectrometer detection, GC/MS (Appendix 6-8). At least three mass fragments from each target compound were monitored. Natural log transformations of the non-linear detector response to standards, ranging over three

orders-of-magnitude, were used to quantify the targeted compounds.

Degradation of DDT to DDD and DDE was monitored during each GC/MS run. Degradation was below 10% in all cases with the exception of the quantitation of the "residual" tissue samples (*i.e.*, the carcass remaining after removing the flesh) from fish and Crangon where the degradation was 21%. In all cases, DDD was the dominant degradation product (70%-80%); only the areas of 2,4'-DDD and 4,4'-DDD (not DDE) were corrected for the likely peak area contribution from their respective DDTs before response factors of standards and sample concentrations were determined. Not correcting for DDE formation from DDT would result in an increase in the sum DDTs of less than 2%-3%, but could result in substantial overestimates of the true DDE concentrations. Normal standardization procedures for DDT accommodate its degradation and no additional procedures are necessary. Dry weight-normalized concentrations were computed from the initial wet weight concentrations using percent solids.

6.6.6. Quality Control

Spiked matrix and procedural blank samples were processed with each set of samples, as were appropriate reference materials when available. National Institute of Standards and Technology (NIST)-originated solutions of the target compounds were quantified during each GC/MS run to evaluate accuracy based on independently produced standard solutions.

Detection Limit Estimates: During each GC/MS run, an estimate of detection limit was made using a matrix blank sample to obtain a concentration equivalent of a peak height 3-times the noise in the chromatogram of the blank near or at the retention time of the analyte (Appendix 6-8). These estimates are found in Table 6-1. Detection limits were in the low ppb or high ppt range for sediments and tissues and in the low ppt range for overlying and interstitial water. Detection limits for dieldrin in the sediments from Station 4 were less than the other three Lauritzen stations because a different extract was analyzed to provide sufficiently low detection limits to quantify dieldrin at this site, which was lower than at the three other Lauritzen stations.

Procedural Blanks: Concentrations in procedural blanks were negligible for the sediment and water TOC determinations. Except for procedural blanks above the detection limit for 4,4'-DDE in interstitial water (request 407) at approximately 0.8 ng/L and tissue (request 466) at 5 µg/kg (dry), the blanks consistently contained no other targets above the detection limits. No procedural blank correction was made for any compound because often (especially IW samples from request 407), samples would have values of 4,4'-DDE below the detection limit.

Quantitation Accuracy: At least once during each GC/MS run, the NIST standard solution containing the seven compounds of interest was quantified using standards with the targets originating from different sources. The data quality objective (DQO)

for quantitation was an accuracy of ±15% of the expected concentrations. 4,4'-DDD averaged 82 ± 2% SE (n=14) of the reported value; the other five members of the DDT family averaged 91 ± 3% SE (n=70) ranging from 87 to 94%. Dieldrin averaged 97 ± 2% SE (n=14) of the reported value. The consistent transcendence of the DQO by 4,4'-DDD in our standards suggested that an error had been made in its preparation. Therefore, the tabulated values for 4,4'-DDD are the result of multiplying the original values by 1.2 to correct for this initial inaccuracy.

Surrogate Recovery: Surrogate internal standard compounds were added to each sample. DDE-d8 was used as the surrogate compound for all analytes as the labeled heptachlor epoxide suffered from inconsistent matrix interferences. The average recovery of this surrogate, across all matrices, was approximately 90%, with an occasional excursion from 40 to 110%. This indicates that the extraction and processing steps only occasionally resulted in significant losses of analytes. Any losses were internally corrected, as the analytes were quantified relative to the surrogate compound.

Spike and Reference Material Recovery: Matrices were amended (spiked) with masses of analytes and processed as samples to assess recovery as were Standard Reference Materials (SRM). The results of the recovery determinations are found in Table 6-2.

Analytical Codes: Accompanying the concentration data for the individual samples (Appendices 8-1, 8-2, 8-4, 8-5, 8-6, 8-7, 8-8, 8-9) is a code column that contains clarifying information where applicable:

L Below lowest calibration standard (accuracy is less certain)

NC Ratio of second ion to first quantifying ion exceeded ± 20% of expected ratio from standards, ratio of third to first ion **within** ± 20% (compound verified)

NC3 Ratio of second **and** third ion to quantifying ion exceeded ± 20% of expected ratio (compound not verified, upper limit concentration)

NPA No peak area

<DL Area below detection limit (zero used in calculations)

CO Significant carryover (value not reliable and not used in calculations)

I Interfering peak (quantitation less reliable)

Fails Fails relative retention time (not detected, zero assigned to concentration)

DRA Diluted and reanalyzed (extracts whose concentrations initially exceeded the highest calibration standard)

CRA Concentrated and reanalyzed (Station 4 dieldrin only)

Other acronyms used in the tables are:

LC Lauritzen Channel

RCH Richmond Inner Harbor Channel

SFCH Santa Fe Channel

SFCB Santa Fe Channel Boathouse

SFCP Santa Fe Boathouse Crabpot

Concentration Units:

Tissue	ppb, μg/kg (wet and dry)
Sediment	ppb, μg/kg (dry only)
Carbon normalized sediment	ppm, μg/g OC (dry only)
Interstitial water	ppt, ng/L
Overlying water	ppt, ng/L
Lipids	decimal fraction, g/g (dry)
Sediment TOC	decimal fraction, g/g (dry)
Water DOC and TOC	mg C/L

6.6.7. GC/MS Scan and SIM Analysis

Three sediment extracts were selected to obtain a semi-quantitative evaluation of what other compounds are present in the study area. The extracts of samples S1CES021, S1CES063 and S1CES095 were chosen for analysis from Lauritzen Channel, Santa Fe, and Richmond Harbor Channels, respectively. Station 2 in Lauritzen was chosen as representative of the high DDT/dieldrin contaminated sites. Station 6 in Santa Fe was chosen as a "worst case" estimate of PAHs and petroleum products based on the observation of oil during sampling. Station 9 in Richmond Harbor was chosen because it was the local reference site in this study. The extracts were reduced in volume to ~50 μL to enhance the signal strength and then scanned by GC/MS to detect compounds yielding fragments in the spectrometer with mass to charge ratios (atomic mass unit (amu)/z) in 0.1 amu/z increments between 50 and 600. From the scanning procedure, a total abundance chromatogram was obtained that contained the abundance of each fractional mass to charge ratio. Using mass fragment libraries of known compounds and matching algorithms, qualitative information was ob-

tained regarding the detected compounds. Semi-quantitative information was obtained using the internal standards that had been put into the samples prior to extraction in conjunction with standard solutions that were scanned in the same manner.

The strategy was to scan standard solutions containing polynuclear aromatic hydrocarbons (PAHs), Aroclor 1254 (common PCBs), and uncommon PCBs and deuterated PAHs that were added with DDE-d8 to each sample of this study as surrogate internal standards. Using a post-run, selected-ion monitoring (SIM) program, two or three characteristic mass fragments of the compounds in standard solutions were looked for in the total abundance chromatogram of the extracts and quantified using retention times, ion ratios, and response factors generated from the standard solutions. Several compounds that were quantified in this mode had no discernible peaks in the total abundance chromatograms. The relative response of each selected PCB congener in the Aroclor 1254 mixture was used to compute an initial Aroclor 1254 (only) concentration. The mass percent of each congener in Aroclor 1254 (Schulz, Petrick, and Duinker, 1989) was used to compute the individual congener concentrations which were subsequently used to quantify final Aroclor concentrations.

Total abundance chromatograms generated in the scan mode were visually inspected and peaks rising above the background noise were subjected to the Hewlett Packard library matching program. If the many mass fragments found in a peak matched the fragmentation patterns of known compounds, a compound name, Chemical Abstract Service (CAS) number, and probability of match were produced. When a peak was found that had no library match, the abundance and mass of the most abundant fragment was recorded, along with a possible compound classification.

For those compounds not included in standard solutions, estimates of concentrations were based on a response factor of 1.0 relative to the nearest internal standard, utilizing the abundance (peak height) of the dominant fragment in the compound's and internal standard's peak. To demonstrate the uncertainty that could be expected in the concentrations of those compounds for which we had no standards, both extracted-ion (SIM) and total abundance peak (scan) results are reported for the compounds for which we had standards. Differences found between SIM and scan concentrations can be due to: arbitrarily assigning a response factor to 1 to the scan calculations, differences between the quant ion used for the SIM quantitations and the ion used for the abundance-based scan calculations, and variations in the ratio of peak areas to base peak abundances for the various targets, especially where the peaks are broadened due to column overloading. The SIM-based concentrations are the more accurate values. All results from the mass fragment scanning analyses are found in Appendices 8-3A, 8-3B, and 8-3C, with column heading definitions found in Table 6-3.

6.6.8. AVS and Simultaneously Extracted Metals

Acid volatile sulfide (AVS) and simultaneously extracted

metals (SEM) were determined on one of the T_0 bioassay storage sediment beakers from each of the 9 stations on day 0 of the toxicity bioassay. The overlying water in these beakers was removed and a sediment sample was withdrawn from the beaker into an open-barrel 10 cc plastic syringe. Parafilm® was secured over the open end of the syringe and the sample frozen. The sample was shipped frozen to the EPA laboratory in Narragansett, Rhode Island for analysis of AVS and SEM.

AVS was determined by converting the solid phase sulfide to hydrogen sulfide(H_2S) using cold 1 M HCl. The released H_2S was trapped in sulfide anti-oxidant buffer and the sulfide measured using a sulfide-specific electrode. The SEM were determined by inductively coupled plasma spectrometry from a filtered sample of the sediment/acid solution after the AVS was released (DiToro et. al., 1990). The SEM included 6 metals expected to be AVS-reactive (Cu, Cd, Ni, Pb, Ag and Zn).

TABLE 6-1. Estimates of detection limits.

	2,4'E	4,4'E	2,4'D	dieldrin	4,4'D	2,4'T	4,4'T
INTERSTITIAL WATER (ng/L):							
Req. 407: Field-collected	0.4	0.5	0.8	2.3	1.2	2.0	2.5
Req. 419: T_0 Bioassay Sediment	2	2	2	20	3	5	4
WATER COLUMN WATER (ng/L):							
Req. 401 and 452:	0.06	0.06	0.09	1.0	0.1	0.2	0.2
SEDIMENT (µg/kg, dry):							
Req. 408: Field-collected							
Lauritzen Channel							
Stations 1-3	20	44	62	100	44	38	136
Station 4	"	"	"	20	"	"	"
Everywhere else	0.2	0.2	0.4	0.8	0.8	1	1.8
Req. 420: T_0 Bioassay Exposure							
Lauritzen Channel	8	9	14	53	16	30	52
Everywhere else	0.2	0.4	0.4	1	0.4	0.8	0.6
TISSUE (µg/kg, dry):							
Req. 400:							
Field mussels	1	2	4	10	4	4	8
Req. 477/478:							
Large Field epifauna	1/3	2/2	2/5	17/15	4/14	5/8	8/13
Req. 475/476:							
Field infauna	1/4	2/2	2/4	17/23	4/26	5/14	8/24
Req. 466:							
Laboratory Macoma	0.6	1.2	3.0	18	6	6	18

Detection Limits = dry weight normalized using wet to dry ratios of 2:1 and 6:1 for sediment and tissue, respectively

Req. # = request number assigned to groups of samples processed together (first three-digits of each sample number)

Req. #/Req. # = different detection limits for different analytical runs (higher detection limits for higher concentration samples)

TABLE 6-2. Spiked matrices and standard reference material (SRM) recovery.

	2,4'E	4,4'E	2,4'D	dieldrin	4,4'D	2,4'T	4,4'T	
INTERSTITIAL WATER SPIKE (ng/L):								
Req. 407: expected	NA	200	NA	200	1,000	NA	1,000	
% of expected	NA	83	NA	119	114	NA	106	
WATER COLUMN WATER SPIKE (ng/L):								
Req. 401: expected	NA	20	NA	20	100	NA	100	
% recovery	NA	87	NA	108	99	NA	94	
Req. 452: expected	NA	20	NA	20	100	NA	100	
% of expected	NA	97	NA	94	97	NA	99	
SEDIMENT SPIKE (µg/kg, wet):								
Req. 408: expected	NA	250	NA	250	250	NA	250	
% of expected	NA	116	NA	103	115	NA	105	
Req. 408: expected	NA	250	NA	250	250	NA	250	
% of expected	NA	104	NA	67	137	NA	151	
Req. 420: expected	NA	238	NA	238	238	NA	238	
% of expected	NA	102	NA	90	105	NA	103	
STANDARD REFERENCE MATERIAL 1941 (µg/kg, dry): (Chesapeake Bay sediment, F. Scott Key Bridge, MD)								
Req. 408: expected	NR	9	NR	1	10	NR	1*	
% of expected	NR	204	NR	0	217	NR	3,100	(all NC3)
Req. 420: expected	NR	9	NR	1	10	NR	1*	
% of expected	NR	174	NR	0	302	NR	1,500	(2 of 4 NC3, 2 <DL)
TISSUE SPIKES (µg/kg, wet):								
Req. 400: expected	480	480	480	95	480	480	480	
% of expected	98	98	98	91	91	70	90	
Req. 466: expected	493	493	493	98	493	493	493	
% of expected	99	96	91	86	85	89	90	
Req. 475: expected	1,000	1,000	1,000	200	1,000	1,000	1,000	
% of expected	105	97	94	95	96	111	103	
Req. 477: expected	500	500	500	100	500	500	500	
% of expected	113	111	99	113	94	100	84	
STANDARD REFERENCE MATERIAL 1974 (µg/kg, dry): (*Mytilus edulis* from Dorchester Bay, Boston Harbor,MA)								
Req. 400: expected	6	48	20	8*	68	4*	3*	
% of expected	167	126	81	0	66	0	411	(1 of 2 <DL)
Req. 466: expected	6	48	20	8*	68	4*	3*	
% of expected	164	91	62	0	46	0	0	
Req. 475/477: expected	6	48	20	8*	68	4*	3*	
% of expected	53	94	100	0	60	95	0	

req. # = request number assigned to groups of samples processed together (first three-digits of each sample number)

NA = not spiked into matrix

NR = not reported in SRM

* = <DL, below detection limit

NC3 = no second ion confirmation of compound

TABLE 6-3. Interpretation of column headings of scan data tables.

***COMPOUNDS** - Surrogate internal or recovery standards deliberately added to the sample and extracts to aid in quantitation.

COMPOUND NAME - Identified Chemical Name.

CAS # - The Chemical Abstracts Service register number for the Target Name.

SCAN PROB. - The scan library search Probability-Of-Match number.

BASE PEAK ION - The mass of largest abundance in the spectra.

BASE PK ABUND SCAN - The abundance of the most abundant ion in the mass spectra at the total ion peak maximum.

nG/G CONC. SIM - Nanograms per gram concentration of sample on a dry weight basis using an internal standard calibration based on the quantitative base peak ion in the ID_file. For targets not found in the ID_file, no values are reported.

nG/G CONC. SCAN - Nanograms per gram concentration of sample on a dry weight basis using an internal standard based response factor equal to one for all targets assigned to the internal standard.

SIM Q VALUE - The value reported in the quant output file for qualitative ion ratio statistical fit where 100 is a perfect match between expected qualitative ion ratios and measured ratios for the sample. The algorithm has not been openly published by Hewlett-Packard. Other characters are substituted here if the integration has been redone manually (M), if no peak really exists (NP), the ratio is manually determined but still is beyond ± 20% of the expected ratio (FQ), the secondary ion manual integration is not easily determined (NS) and a Flat-Topped peak was present indicating saturation of the detector because of overloading of the column and detector with sample (FT).

FIGURE 6-1. Map of October 1991 grab, water, mussel and trawl samples.

Samples at stations are designated by:

1): a solid circular dot, denoting a primary station at which replicate grabs were taken

2): a solid triangle, a secondary station

3): a solid square, where tissue("T") (Mytilus gallo-provincialis) and overlying water ("OW") samples were taken

4): trawl samples are indicated on Figure 6-2.

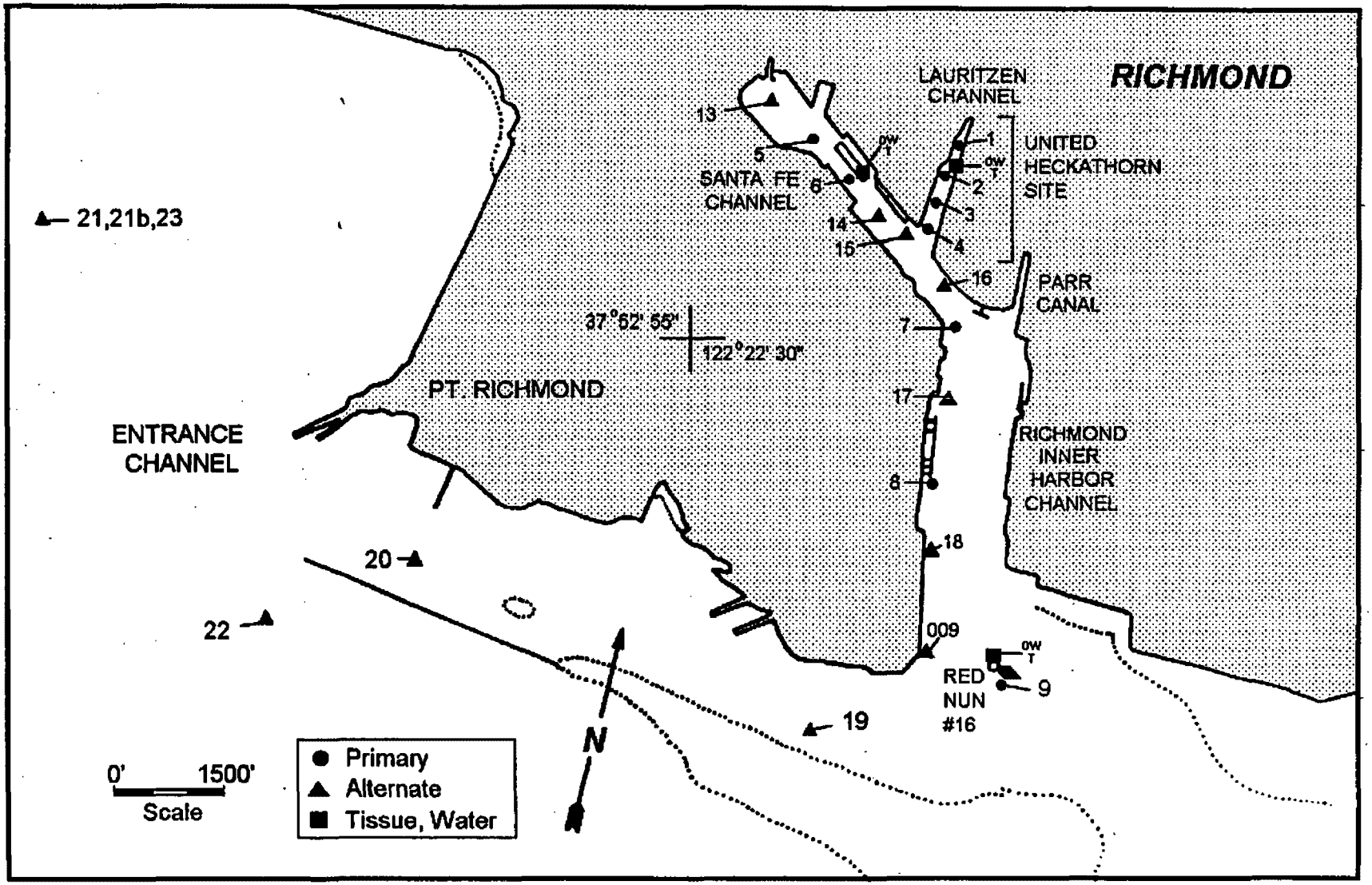

RICHMOND
LAURITZEN CHANNEL
UNITED HECKATHORN SITE
SANTA FE CHANNEL
PARR CANAL
PT. RICHMOND
ENTRANCE CHANNEL
RICHMOND INNER HARBOR CHANNEL
RED NUN #16
21,21b,23
37°52' 55"
122°22' 30"
1
2
3
4
5
6
7
8
009
9
13
14
15
16
17
18
19
20
22
ow
T
N
0'
1500'
Scale
Primary
Alternate
Tissue, Water

FIGURE 6-2. Map of February, 1992 water, trawl, crab pot and mussel samples and October, 1991 trawl lines.

Station numbers in open circles indicate a primary station over which a trawl was conducted. Trawl lines are indicated by arrows denoting direction; length approximates distance covered.

"F" series sample numbers indicate the sequential acquisition of tissue samples from trawls, traps, and harvesting mussels (Mytilus galloprovincialis) from ropes, buoys or floating docks. S2AHO-xxx numbers indicate locations of overlying water samples (same location as M. galloprovincialis samples).

October, 1992 trawl lines are indicated by an arrow with a circled number, denoting trawl path, direction and station number; Sample series -200 cover Lauritzen Channel trawl samples, -300 cover Santa Fe Channel trawl samples and -400 covers Richmond Harbor Channel trawl samples.

Some February trawl lines directly overlay October trawl sites. February trawl lines are indicated by "F-" sample numbers at that site flagged to the line.

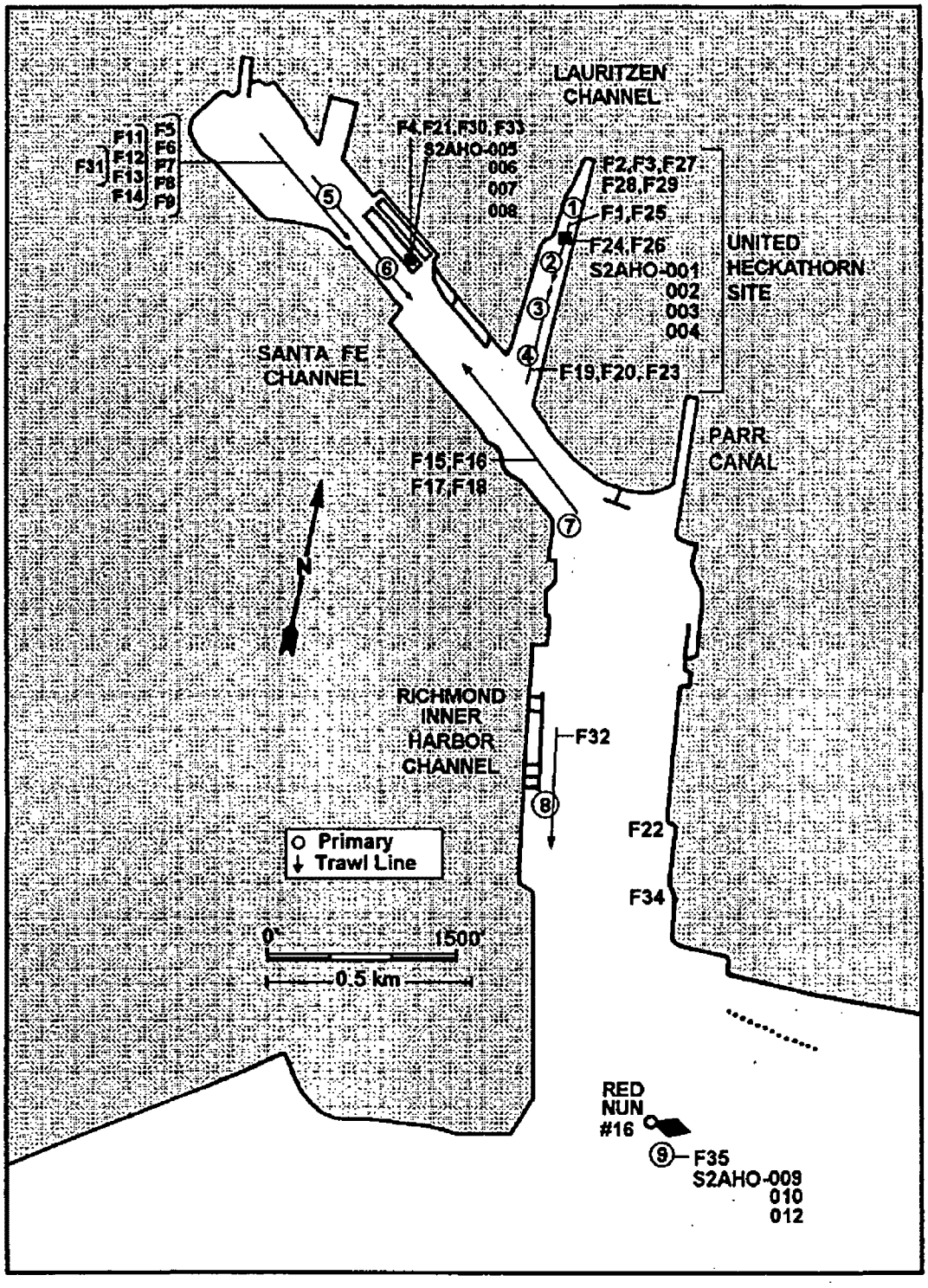
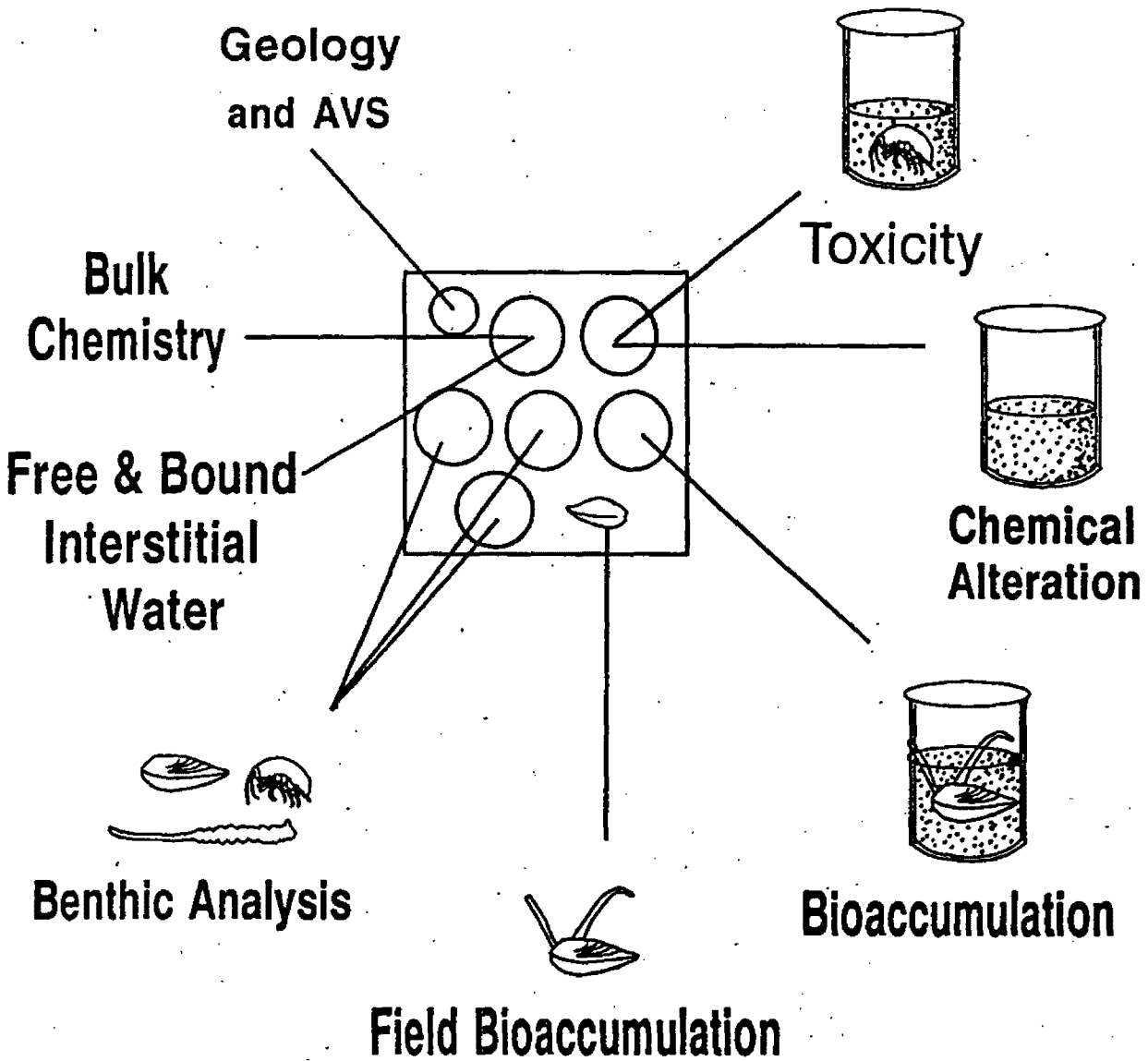

LAURITZEN
CHANNEL
F11
F12
F31
F13
F14
F5
F6
F7
F8
F9
F4,F21,F30,F33
S2AHO-005
006
007
008
F2,F3,F27
F28,F29
F1,F25
F24,F26
S2AHO-001
002
003
004
UNITED
HECKATHORN
SITE
SANTA FE
CHANNEL
F19,F20,F23
PARR
CANAL
F15,F16
F17,F18
N
RICHMOND
INNER
HARBOR
CHANNEL
F32
F22
F34
Primary
Trawl Line
0'
1500'
0.5 km
RED
NUN
#16
F35
S2AHO-009
010
012

FIGURE 6-3. Schematic of the sampling from individual 0.1 M^2 grabs. The "chemical alteration" is the T_0 bioassay storage sediment analyzed to determine the effects of storage and mixing on pollutant concentrations. Core sizes are not to scale; spatial arrangement of cores may vary with sample.

United Heckathorn Superfund Site Study

Sediment Allocation from 1/10 m^2 Grab



7.0 PHYSICAL CHARACTERISTICS OF STUDY SITES

7.1. INTRODUCTION

This section summarizes the grain size analysis of the sites, as well as other physical characteristics observed during sampling.

7.2. METHODS

7.2.1. Sampling Methods

Sediment for grain size was sampled by inserting a 3.6 cm i.d. plastic corer tube vertically into the center of the grab upon opening on the deck stand. Once all sampler corers were in place, and temperature and E_h measurements were complete, the corers were removed by gloved hand. Ten centimeters of the core was extruded using a fitted, indexed plunger into a pre-labeled glass container and sliced off with a clean Teflon-coated spatula. The sample was immediately stored in an ice chest with pre-cooled gel-ice. Cooled samples were transported to the Newport laboratory by truck, and kept in 4°C temperature-controlled rooms under chain-of-custody protocol until analysis.

7.2.2. Grain Size Analysis

Initially, eleven samples from the nine primary stations were analyzed for grain size distribution, with one sample from each of Stations 1-8 and one sample from three of eight grabs from the reference Station 9. Within-grab replicates were analyzed at Stations 1, 2 and 7. Individual grabs within stations were randomly chosen for analysis. TOC was determined on every grab at a station. Complete grain size analysis was performed for the eleven grabs according to standard sieve-pipette methods of sedimentary petrology (Galehouse, 1971; Folk, 1980). Weights of sediment size fractions from phi (Φ) classes -1 to 4 (sand fraction) represent the net weight of sediment collected on graded sieves. The values for Φ 5 through Φ 8 (silt-clay or "mud" fraction) were calculated by the pipette method described in the "Laboratory Methods" section of the SEDANAL Users Manual (Coleman and Wheeler, 1986, which is based on Folk, 1980 and Galehouse, 1971). The balance of samples were later analyzed by an abbreviated method based on Buchanan (1984) and Guy (1969) to yield only sand, silt and clay fraction information. These samples had been stored at 4°C in their original sample containers under Chain-of-Custody procedure, were released for analysis August 18, 1993; the analysis was completed by September 21, 1993.

7.2.3. General Physical Characteristics

Sediment temperature and salinity of sediment surface water were measured as previously described (Section 6.4.). The color, texture and odor of the sediment were described as accurately as possible under field conditions.

7.3. RESULTS/DISCUSSION

Appendix 7-1 lists the detailed field observations and measurements, including temperature, salinity, depth of sediment in grabs, depth of overlying water, sediment color and odor, and E_H. Temperature of the sediment and overlying and surface water was relatively constant, at ~18-19°C, salinity was constant at ~30 /..; the E_H data were generally inconclusive.

The study area appeared relatively quiet with respect to currents, though sediment resuspension due to propwash of large ships was observed at some locations, notably Stations 8 and 009, and to a lesser degree Stations 5, 6 and 7. Sloping surfaces with underlying gravel or rock on the edge of the channels interfered with proper operation of the grab, and a number of grabs were rejected. Overlying water depth varied from ~6 to ~12 m, except at station 9, which averaged ~3 m.

The grain size analysis is presented in Table 7-1. In general, sample sites could be classed as silt to clay, occasionally underlain by gravel, rock, industrial waste or fill, trash and the occasional sunken barge, broken off piling, dock lumber and other assorted debris. Based on the full sedimentological analysis at each station, Stations 1-8 were classed as fine to very fine silt to clay with a mean grain size of 4.8 ± 1.7μm and a median of 3.4 ± 0.4μm.

The variation in size classes among the sites (expressed as the mean % and standard deviation, $\bar{x}$ ± sd), was relatively small; gravel (based on selected grabs only) at 0.8 ± 1.4%, sand at 11.6 ± 6.3%, silt at 44.8 ± 13.5% and clay at 43.2 ± 18.5%. Stations 1-5 had virtually no gravel, Station 5 had little sand and more clay, Stations 2, 3, 6 and 7 tended toward more sand and less clay, but overall, the stations were fairly homogeneous.

Station 9, at the mouth of Richmond Harbor, is classed as a "poorly sorted" medium silt with a mean grain size of 18.9 ± 3.0μm and a median of 20.0 ± 7.8μm. The variance among grabs at this site was greater than at the other stations, with 4.0 ± 1.2% gravel, 15.9 ± 4.1% sand, 63.6 ± 15.6% silt and 18.1 ± 13.8% clay. This station was on the edge of the channel, shoaling quickly to the east from the channel to the west. The station was abandoned after grab 3 of the first day due to decreasing depth from the outgoing tide. Station 9 grabs 4-8 were obtained the following day on a higher tide stage, and in slightly deeper water, approximately 10-15 m south of the original location, partly due to anchor swing differences due to opposite current direction. It is not surprising, in retrospect, that a wider variety of sediment types could be encountered in this small area, which is the most exposed to tidal currents of all the primary stations.

The replicate grain size analyses (selected grabs at Stations 1, 2, 4, 7 and 8) were reasonably close. There were occasional "outliers" (*i.e.*, the worst case, grab 7-1, the sand fraction is <40% of the 5% difference in replicate sample analysis that is considered an acceptable measure of precision). Besides being a measure of the precision of the method, this

similarity indicates that grain size does not vary appreciably on a microscale (0.1 m^2).

The mean percent TOC at each of the primary stations is presented in Table 7-1. Standard errors for the TOC measurements, values for within grab subcores, and TOC for the secondary stations can be found in Table 8-1. The TOCs of the sites were moderate, varying from about 0.9% to 3%. The highest TOCs were at Station 1 in Lauritzen Channel and Station 6 in Santa Fe Channel. The stations with the higher TOCs did not have a higher percent of silt-clays compared to other stations.

TABLE 7-1. Sediment grain size analysis summary.

STN #	GRAB #	MEAN SIZE µM	% GRAVEL	% SAND	% SILT	% CLAY	% TOC	SAMPLE # - REPLICATE #	PHYSICAL DESCRIPTION (*)
1	1			13.6	81.4	5.0	1.90	S1CJS011[1]	
1	2			9.6	80.0	10.3	2.35	S1CJS012[1]	
1	3			5.4	70.6	24.1	2.42	S1CJS013[1]	
1	4	4.5	0.0	4.3	29.3	66.4	2.41	S1CJS014-R1	VERY FINE SILT, POORLY SORTED
1	4	4.7	0.0	4.2	30.3	65.5		S1CJS014-R2	VERY FINE SILT, POORLY SORTED
1	5			4.3	53.0	42.7	2.72	S1CJS015[1]	
1	6			2.2	80.0	17.8	2.63	S1CJS016[1]	
1	7			2.8	34.8	62.4	2.47	S1CJS017[1]	
1	8			19.2	74.8	6.0	2.18	S1CJS018[1]	
1	all		$\bar{x}$ =	7.7	59.3	33.4	2.39	S1CJS01$\bar{x}$ [2]	
			sd =	6.0	22.6	26.1	0.26		
			n =	8	8	8	8		
2	1			17.3	40.8	41.9	1.76	S1CJS021[1]	
2	2			15.3	34.4	50.3	1.85	S1CJS022[1]	
2	3	4.7	0.0	6.1	30.6	63.2	2.05	S1CJS023-R1	VERY FINE SILT, POORLY SORTED
2	3	5.2	0.3	6.5	30.5	62.7		S1CJS023-R2	VERY FINE SILT, POORLY SORTED
2	4			27.6	35.4	37.0	1.19	S1CJS024[1]	
2	5			7.1	59.6	33.4	2.08	S1CJS025[1]	
2	all		$\bar{x}$ =	14.7	40.1	45.1	1.79	S1CJS02$\bar{x}$	
			sd =	8.7	11.5	11.8	0.36		
			n =	5	5	5	5		
3	1			17.0	61.2	21.8	1.79	S1CJS031[1]	
3	2			6.8	34.7	58.5	1.91	S1CJS032[1]	
3	3	10.3	0.8	19.2	31.7	48.3	1.66	S1CJS033	FINE SILT, VERY POORLY SORTED
3	4			19.5	33.3	47.2	1.65	S1CJS034[1]	
3	5			7.3	34.9	57.9	1.65	S1CJS035[1]	
3	all		$\bar{x}$ =	13.7	39.2	46.7	1.73	S1CJS03$\bar{x}$	
			sd =	6.4	12.4	14.9	0.12		
			n =	5	5	5	5		

(cont'd)

TABLE 7-1 (Cont'd). Sediment grain size analysis summary.

STN #	GRAB #	MEAN SIZE µM	% GRAVEL	% SAND	% SILT	% CLAY	% TOC	SAMPLE # - REPLICATE #	PHYSICAL DESCRIPTION (*)
4	1			24.1	26.9	49.1	1.31	S1CJS041[1]	
4	2			5.7	25.5	68.8	1.44	S1CJS042[1]-R1	
4	2			6.1	33.2	60.7		S1CJS042[1]-R2	
4	3	5.2	0.0	6.1	35.6	58.3	1.42	S1CJS043	VERY FINE SILT, POORLY SORTED
4	4			7.7	26.5	65.8	1.58	S1CJS044[1]	
4	5			8.9	28.6	62.5	1.55	S1CJS045[1]	
4	all		$\bar{x}$ =	10.5	29.4	60.1	1.46	S1CJS04$\bar{x}$	
			sd =	7.7	3.7	6.8	0.11		
			n =	5	5	5	5		
5	1			0.5	26.3	73.3	1.37	S1CJS051[1]	
5	2			0.6	31.1	68.4	1.48	S1CJS052[1]	
5	3			0.2	29.1	70.7	1.49	S1CJS053[1]	
5	4	3.7	0.0	0.3	28.0	71.7	1.47	S1CJS054	CLAY, MODERATELY SORTED
5	5			0.3	27.3	72.4	1.52	S1CJS055[1]	
5	all		$\bar{x}$ =	0.4	28.4	71.3	1.47	S1CJS05$\bar{x}$	
			sd =	0.2	1.8	1.9	0.06		
			n =	5	5	5	5		
6	1	6.5	4.5	9.7	29.4	56.4	2.05	S1CJS061	VERY FINE SILT, VERY POORLY SORTED
6	2			3.9	87.0	9.2	2.66	S1CJS062[1]	
6	3			24.2	65.6	10.2	3.97	S1CJS063[1]	
6	4			28.4	61.6	10.1	3.31	S1CJS064[1]	
6	5			34.2	57.2	8.6	0.29	S1CJS065[1]	
6	all		$\bar{x}$ =	20.1	60.2	18.9	2.46	S1CJS06$\bar{x}$	
			sd =	12.8	20.7	21.0	1.41		
			n =	5	5	5	5		
7	1			11.4	76.5	12.1	1.06	S1CJS071[1]-R1	
7	1			13.6	62.3	24.1		S1CJS071[1]-R2	
7	2			16.1	37.9	46.1	1.12	S1CJS072[1]	
7	3			24.5	35.1	40.4	1.00	S1CJS073[1]	
7	4			17.4	60.5	22.2	1.15	S1CJS074[1]	

(Cont'd)

TABLE 7-1 (Cont'd). Sediment grain size analysis summary.

STN #	GRAB #	MEAN SIZE µM	% GRAVEL	% SAND	% SILT	% CLAY	% TOC	SAMPLE # - REPLICATE #	PHYSICAL DESCRIPTION (*)
7	5	7.2	0.9	12.1	34.9	52.1	1.12	S1CJS075-R1	VERY FINE SILT, VERY POORLY SORTED
7	5	7.8	1.9	12.5	34.8	50.8		S1CJS075-R2	FINE SILT, VERY POORLY SORTED
7	all		$\bar{x}$ =	16.6	47.5	35.5	1.09	S1CJS07$\bar{x}$	
			sd =	5.0	16.2	14.6	0.06		
			n =	5	5	5	5		
8	1			2.5	37.9	59.6	1.21	S1CJS081[1]	
8	2			1.2	42.5	56.3	1.27	S1CJS082[1]	
8	3			11.2	36.9	51.9	1.13	S1CJS083[1]	
8	4	3.2	0.9	5.4	21.5	72.3	1.17	S1CJS084	CLAY, POORLY SORTED
8	5			5.6	37.2	57.2	1.12	S1CJS085[1]-R1	
8	5			6.3	36.2	57.5		S1CJS085[1]-R2	
8	all		$\bar{x}$ =	5.2	35.1	59.5	1.18	S1CJS08$\bar{x}$	
			sd =	3.9	8.0	7.7	0.06		
			n =	5	5	5	5		
9	1	16.3	3.5	10.7	80.7	5.2	0.81	S1CJS091	MEDIUM SILT, POORLY SORTED
9	2			16.4	56.5	27.1	0.85	S1CJS092[1]	
9	3	22.2	5.4	13.2	80.2	1.1	0.86	S1CJS093	MEDIUM SILT, POORLY SORTED
9	4	18.3	3.2	17.7	50.9	28.1	0.94	S1CJS094	MEDIUM SILT, VERY POORLY SORTED
9	5			21.3	49.6	29.2	0.88	S1CJS095[1]	
9	all		$\bar{x}$ =	15.9	63.6	18.1	0.87	S1CJS09$\bar{x}$	
			sd =	4.1	15.6	13.8	0.05		
			n =	5	5	5	5		
22	1			10.9	61.4	27.7	0.76	S1CJS122[1]	
21B	1			6.9	48.0	45.2	1.02	S1CJS121B[1]	

* = Descriptions from Folk (1980).
1 = Abbreviated procedure, based on Buchanan (1984) and Guy (1969), used to analyze samples stored at 4°C until August, 1993; "sand" category includes all larger classes.
2 = Means of replicates represent an "n" of 1 in the station mean.

8.0 EXPOSURE ASSESSMENT

This section addresses the exposure assessment conducted at the Lauritzen/Santa Fe/Richmond sites. It covers the measurement of DDT, its metabolites, and dieldrin in the key physical media: bulk sediment, interstitial water, and overlying water. Additionally, it covers the concentrations of these compounds in the tissues of fish and invertebrates collected in trawls, mussels collected on hard surfaces, laboratory-exposed infauna, and field-collected infauna. These measurements are used to document the existing extent of contamination, to quantify how exposure varies among species, and to develop models to predict the extent of exposure at different remediation levels.

8.1. SEDIMENT AND INTERSTITIAL WATER

8.1.1. Introduction

The ecological significance of exposure to pollutants can be evaluated from two perspectives - does the compound exceed concentrations that result in acute or chronic toxicity or, alternatively, result in levels of bioaccumulation within prey species deleterious to higher trophic levels? Prediction of these effects requires estimates of the pollution concentrations in bulk sediment, carbon normalized sediment, interstitial water, and overlying water. The first three media are of importance to organisms directly exposed to the contaminated sediment; whereas overlying water concentrations are important to epibenthic benthos (*e.g.*, mussels), demersal and pelagic fish, and potentially to sediment-dwelling organisms that ventilate overlying water. Ingestion of contaminated prey also can be an important uptake route with high K_{ow} compounds such as DDT.

This section presents the results for DDT, DDT metabolites, and dieldrin in the three sediment-related environmental media. Concentrations in the overlying water are discussed in Section 8.2. while the importance of contaminated benthic prey to fish is discussed in Section 9.4. To quantify the differences in the exposure between field and laboratory tested infaunal organisms, the comparison of bulk sediment and interstitial water concentrations in the T_0 bioassay storage sediment and field sediments is presented. Although the environmental chemical analysis focused on DDT and dieldrin, metals and other organic pollutants were analyzed in bulk sediment from one station in each of the three channels. This analysis is augmented by data from additional studies of sediment contaminations in Lauritzen, Santa Fe, and Richmond Channels specifically and in San Francisco Bay in general.

8.1.2. Partitioning Theory

According to equilibrium partitioning (EqP) theory, within a sediment the pollutant is found in three fractions at equilibrium; these fractions include: the freely dissolved (free), the particle-associated, and the dissolved organic matter (DOM)-

associated fractions. The concentration of free pollutant is inversely proportional to the carbon-normalized equilibrium partition coefficient of the larger carbon reservoir, the sedimentary organic carbon, K_{oc} (Equation 8-1) (DiToro *et al.* 1991). However, the total (free and DOM-associated) dissolved interstitial water concentration is controlled by the K_{oc} of the much smaller carbon reservoir, the DOM.

The following illustration of apparent equilibrium within sediment:

sediment-associated <-----> free <-> DOM-associated

where the free plus DOM-associated fractions constitute the typically-determined total pollutant fraction. This equilibrium can be represented by the apparent equilibrium partition coefficient, K'_{oc} and determined quantities:

K'_{oc} = (sediment conc./sediment OC)/free IW conc. Eq. 8-1

The units of K'_{oc} are L/kgOC, and are demonstrated from the following calculation of K_{oc} for compound Z:

(([μgZ/g(wet)]*g(wet)/g(dry))*[g(dry)/gOC]*1,000gOC/kgOC)*L/μgZ(free)

OR

sediment conc. * 1/% solids * 1/% carbon * conversion * 1/free conc.

It has been repeatedly shown that the K_{oc} of a compound is related to the compound's laboratory-determined octanol-water partition coefficient, K_{ow}. DiToro *et al.* (1991) suggested the following relationship:

$$\log K_{oc} = 0.983 * \log K_{ow} + 0.00028 \qquad \text{Eq. 8-2}$$

Calculation of log K_{oc} from K_{ow} generated using the slow stirring method (DeBruijn, *et al.*, 1989, values in Table 4-2) yield the following $\log K_{oc}$ values (mean, S.E. and n) for dieldrin, 4,4'-DDD, 4,4'-DDT and 4,4'-DDE: (5.309,.003,9), (6.112,.01,9), (6.797,.01,9), and (6.838,004,9), respectively. Reported $\log K_{ow}$s of the 2,4' and 4,4' isomers of DDD and DDE, using different methods (O'Brien ,1974) were found to be essentially indistinguishable (6.0 and 5.8, respectively), however, these values are substantially different from the values of DeBruijn, *et al.* No K_{ow} values could be found for 2,4'-DDT.

Interstitial water concentrations are often predicted from bulk sediment concentrations by using K_{oc} and assuming equilibrium partitioning (see DiToro *et al.*, 1991). In this study, however, interstitial water fraction concentrations were measured directly for three main reasons. First, the freely dissolved form of pollutants has been shown to be the bioavailable form. Using several polynuclear aromatic and chlorinated compounds,

Landrum *et al.* (1985) and Black and McCarthy (1988) demonstrated that the rate of compound uptake by the amphipod Pontoporeia hoyi and the uptake efficiency of trout gills, respectively, was directly proportional to the compounds' freely dissolved concentrations. Second, if the sediment is not at equilibrium, or if different types of carbon have different partitioning behaviors, the relationships between measured interstitial water pollutant concentrations and the biological responses could be better predictors of effects than of assumed equilibrium. For example, differential partitioning between natural and anthropogenic carbon (*e.g.*, petroleum) has been suggested (S. Karickhoff, personal communication, in Lake *et al.*, 1990). Third, measurement of each interstitial water fraction allows calculation of a site-specific K_{oc} which could be used in predicting the environmental fate of a contaminant. For example, the K_{oc} values could be used in any remediation project to predict the concentration in interstitial water versus bulk sediment as a function of changes in organic content.

With these relationships, the distribution of a sediment pollutant among phases (K'_{oc}) can be obtained explicitly by determining free interstitial water concentrations and the carbon-normalized concentrations in the bulk sediment. Implicitly, the free concentration can be calculated via Equations 8-1 and 8-2 from a compound's K_{ow} and bulk sediment parameters or correlatively by comparing K_{ow}-derived K_{oc}s with apparent equilibrium partition coefficients, K'_{oc}s. The fidelity of the free concentration values would impact the utility of the carbon-normalized sediment concentrations approach to exposure as it is based upon the assumptions of equilibrium and an environment-independent K_{oc} for each compound. If the calculated and determined exposures were significantly different, deviations from the assumptions would have to be considered. For example, if K'_{oc} for a compound was significantly correlated to TOC then freely dissolved concentrations could not be accurately calculated from sediment parameters using K_{ow}. Similarly, if, at one site, one compound (of several being monitored) shows a K'_{oc} that is significantly different from other sites while the other compounds show no such differences, non-equilibrium conditions should be considered.

8.1.3. Methods

Sediments and interstitial water were collected and analyzed as described in Section 6.6. The variances of the transformed bulk sediment concentrations ($\log_{10}$ or $\log_{10}$-1) from the nine replicated stations were tested for homogeneity by F_{max} (Sokal and Rohlf, 1981) and differences among mean concentrations were determined by multiple comparisons tests (experimentwise $\alpha = 0.05$) using the Games and Howell method (Sokal and Rohlf, 1981) when variances were heterogeneous.

To determine the effects of sediment storage and mixing on concentrations and phase distributions, one randomly selected bioassay core from three of five (eight from Station 1) grabs

from each station were separated into two beakers, one designated the T_0 bioassay storage sediment, the other, the biological exposure sediment. If a storage and mixing effect were present, the T_0 bioassay sediment would have to be significantly different from the mean field core concentration of a grab with multiple field chemistry cores, or the T_0 bioassay sediment from one core of a grab would have to be significantly different from the undisturbed field sediment from a different core of the same grab. The first test of significance would use a mean core value and *t*-test; the second test of significance would compare the difference between the T_0 bioassay storage sediment and field sediment to a critical value of SE times twice the $t_{.05[df]}$-value.

$$|C_T - C_f| > SE * 2 * 4.303 \qquad \text{Eq. 8-3}$$

where:

C_f and C_T are the field and T_0 bioassay storage sediment concentrations

SE = $(CV * C_f)/\sqrt{n}$ calculated from the average CV of the grabs with replicate field chemistry cores using C_f as the best estimate of the core average for each grab without replicate field chemistry cores

4.303 is the $t_{.05[2]}$-value with n = 3 (triplicate cores per subsampled grabs).

The critical value for sediment ΣDDT with a mean CV of 20% is 1.0 C_f, *e.g.*, if $|(C_T - C_f)/C_f|$ exceeds 1.0 the concentrations are significantly different and may have come about from differential treatment following sampling (*i.e.*, storage and mixing).

The apparent equilibrium partition coefficients, $\log K'_{oc}$, were calculated using equation 8-1 with free being the difference between total and bound concentrations. Mean apparent equilibrium partition coefficients from the Lauritzen Channel stations were compared to the mean calculated from all the other stations, and both means were compared to K_{ow}-derived K_{oc}s from equation 8-2. Homogeneity of variances and differences of means were tested using F_{max} and t-tests, respectively.

8.1.4. Results

Average Station Results: Appendix 8-1 lists the individual sample values for pollutant concentrations in bulk sediment, total interstitial water, and bound interstitial water, as well as sediment TOC and interstitial water DOC. Table 8-1 presents the average station concentrations (and standard errors) of ΣDDT for free and total interstitial water concentrations, bulk sediment concentrations on a dry weight basis, sediment concentrations on a carbon normalized basis, average sediment TOC, and $\log K'_{oc}$ for each station and $\log K'_{oc}$ for each station. The mean $\log K'_{oc}$s for a station were calculated from the $\log K'_{oc}$s

calculated from the sediment and IW concentrations in each grab. When more than one core was taken from a grab, the logK'_{oc} and other constituents from each core were averaged to represent the grab. Table 8-2 presents the same data for dieldrin. Table 8-3 presents the average bulk sediment concentrations for all seven compounds at each of the stations.

The means and standard errors of ΣDDT for the primary Stations 1 through 9 are plotted in Figure 8-1. The variances of the untransformed and $\log_{10}$-transformed, dry weight and organic carbon-normalized station means in the multiple comparisons were heterogeneous by F_{max}. Statistical differences among the nine primary stations in ΣDDT and dieldrin sediment concentrations using the Games and Howell method are illustrated in Figure 8-2.

Distribution of Analytes Between Phases: Percentages of ΣDDT found in each of the metabolites and the ratio of dieldrin to ΣDDT in the sediment and total dissolved phases for three grabs from the Lauritzen Channel stations are found in Table 8-4A (sediment) and 8-4B (interstitial water). The 4,4' isomers of DDD and DDT dominate the bulk sediment averaging 50% and 37%, respectively. In the interstitial water this ratio was not maintained; 4,4'-DDD and 4,4'-DDT averaged 72% and 7% of the ΣDDT with other metabolites making up a greater percentage than in the sediment. The distribution of these compounds in upland samples adjacent to Lauritzen Channel (Levine-Fricke, 1990) are presented in Table 8-4C. The distribution of 2,4' and 4,4' isomers of DDT (13% and 80% of the total, respectively) are very similar to the distributions reported for technical DDT formulations (Gunther and Gunther, 1971). Virtually all 2,4'-DDT, and 40% to 70% of 4,4'-DDT has apparently been metabolized in the Lauritzen Channel sediment to their respective isomeric DDDs, with little difference between the sediment and upland distributions of 4,4'-DDE. In these samples, the majority (rarely less than 80%) of the total IW concentration of any compound consisted of the freely dissolved form. The partitioning between the freely dissolved and the DOM-associated fractions were proportional to the K_{ow}s of the compounds, and the K'_{oc}s for DOM were much smaller than the K'_{oc}s of the sediment carbon. This is consistent with the results of Landrum *et al.* (1984), Morehead *et al.* (1986), and Servos and Muir (1990) with freshwater, and unpublished results from this laboratory using seawater.

Apparent Partition Coefficients: Mean DOC and log of average K'_{oc} values for all stations are found in Table 8-5A. The average K'_{oc}s from the Lauritzen Channel and elsewhere in the study area are grouped and compared to K_{oc}s computed from K_{ow}s (Table 8-5B). Samples from the Lauritzen Channel have the highest concentrations and the computed K'_{oc}s should be the most reliable because they have the fewest unconfirmed values (NC3). The K'_{oc}s for both DDT isomers from Station 2, grab 5 were not included in the station means or comparisons as they exceeded the mean values by two orders of magnitude and were determined to be outliers by the method of Dixon (1950).

The $\log_{10}$-transformed K'_{oc}s of 4,4'-DDT and dieldrin from

the two locations, Lauritzen Channel and the other sites, had homogeneous variances (untransformed values did not by F_{max}); the mean K'_{oc}s means were significantly different and not combined. The variances of the untransformed and transformed K'_{oc}s of 4,4'-DDE and 4,4'-DDD from the two locations were significantly different; their means were compared using a t-test with variance-weighted critical value, t_{α}, and average S.E. (Sokal and Rohlf, 1981) and were found to be not significantly different.

The grand mean K'_{oc}s of both locations were also compared to the K_{ow}-based K_{oc}s. The variances of the K_{ow}-based K_{oc}s (DeBruijn *et al.*, 1989) were significantly smaller than the K'_{oc}s. They were so much smaller than those of the K'_{oc}s that the t-test was, simply, a comparison of an untransformed K'_{oc} mean to a single, K_{ow}-based, untransformed K_{oc} value with no variance. Although the mean dieldrin and 4,4'-DDT K'_{oc}s from the Lauritzen samples were not significantly different from the K_{ow}-derived value ($P>0.05$); they were larger by 0.15 log units (1.4-times) and 0.30 log units (2.0-times), respectively. Differences between the mean K'_{oc} values of the 4,4' isomers of DDD and DDE from Lauritzen Channel and the K_{ow}-derived K_{oc} values were significant ($P<0.05$ and $P<0.01$, respectively). The K'_{oc}s were lower: -0.10 log units (0.8-times) and -0.68 log units (0.2-times) for 4,4'-DDD and 4,4'-DDE, respectively.

Similar to the K_{ow} values reported by O'Brien (1974), there were no significant differences between the apparent partition coefficients of the isomers of DDE and DDD in the Lauritzen samples. The maximum difference between the K_{ow}-derived values and K'_{oc} for these two metabolites was 0.11 log units. Although the isomers of DDT were also not significantly different, the 2,4' isomer was 0.34 log units lower.

Comparison of field vs. T_0 bioassay storage sediments: The standardized differences [(bioassay-field)/field] in interstitial water and bulk sediment concentrations and TOC in the T_0 bioassay storage sediment and corresponding field sediments are given in Table 8-6A with the field concentration-normalized critical values, $(C_T-C_f)/C_f$ from Eq 8-3. The individual values for the T_0 bioassay storage sediment are given in Appendix 8-2.

Measurements of within-grab variability were used to differentiate between spatial heterogeneity and storage/mixing effects. Within-grab variability was determined at Stations 1, 5, and 7 (Table 8-6B). Because of the randomization of bioassay cores for subsampling, the mean field sediment and T_0 bioassay storage sediment samples can be compared directly within single grabs from only Stations 1 and 5. The mean CVs from the three core-replicated stations (Table 8-6B) were used to calculate the critical values from Eq. 8-3 for all 27 comparisons.

Comparing the bulk sediment concentrations in the single T_0 bioassay storage sediments to the mean of the corresponding replicated field values resulted in two significant difference (0.05-level, two-tailed test). The ΣDDTs and dieldrin in the field samples from Station 1, grab 3, and Station 5, grab 5, respectively, were greater.. In the interstitial water from grab

3 at Station 1, 4,4'-DDE was significantly greater in the T_0 bioassay storage sediment. For interstitial water from grab 1 at Station 5, three compounds (2,4'DDE, 4,4'DDD and dieldrin) were significantly greater in the T_0 bioassay storage sediments than the replicated field cores.

Of the hundreds of paired comparisons in Table 8-6A, only the components of T_0 bioassay storage sediment S1CES011 consistently surpassed the normalized critical values. This critical value approach may have been too conservative in that it identified only two of the six compound/matrix/station combinations that were found to be significantly different (using a Student's t-test) from the core mean for that grab.

Sediment Scans: The individual values for the estimated sediment dry weight concentrations of probable compounds found in the GC/MS scan mode are given in Appendices 8-3A, 8-3B, and 8-3C, with the codes to the data in Table 6-3. The compounds are ordered by decreasing concentration, first in the SIM quantified column followed by the SCAN-quantified column. Concentrations determined in the scan mode using extracted ions averaged 100 ± 10%, S.E. (n=12) of the values determined previously in the SIM mode. Using this method of quantitation on the extract of NTIS reference material SRM 1941, (organics in marine sediment) resulted in concentrations generally within ± 50% of the reported concentrations for PAHs, PCB congeners, and organochlorine pesticides (Appendix 8-3D).

The PCB congener concentrations are summarized in Table 8-7. The PCB concentrations at Station 6 are 2-6 times higher than Station 2, with low concentrations of only three congeners determined at Station 9. Table 8-8 summarizes the Aroclor 1254 and 1260 concentrations. Table 8-9 summarizes the fifteen most abundant compounds from the SIM runs, exclusive of PCBs, DDTs, and dieldrin, in each of the three samples.

AVS and Simultaneously Extracted Metals: The AVS and five metals (silver was not detected and therefore not reported) are presented in Table 8-10. The SEM/AVS ratio ranged from 0.05 for Lauritzen Channel Station 1 (grab 1) to 0.49 for Richmond Harbor Station 9 (grab 2). Most other samples had a SEM/AVS ratio of about 0.1.

8.1.5. Discussion

Sediment and Interstitial Water Concentrations: The among-grab variability was greatest at Station 1 for both dieldrin and ΣDDT with coefficients of variations (CV) exceeding 100% for dieldrin and near 60% for ΣDDT (Tables 8-1 and 8-2). The within-grab (Table 8-6B) and core, and among-grab and station variabilities within this study area indicate a spatially heterogeneous system. It was in anticipation of such heterogeneity that the chemical and biological samples were taken from the same grab. In general, Lauritzen Stations 1, 2 and 3 tended to form a high contaminant group, Stations 4-7 a mid-level contaminant group, and Stations 8 and 9 a low-level contaminant group (Figures 8-1 and 8-2). The interstitial water concentrations show a similar

spatial pattern from Lauritzen to Richmond Harbor.

The bulk, surficial sediment concentrations of ΣDDT and dieldrin found in this study generally agree with those found in previous studies in Richmond Inner Harbor Channel (Table 8-11A and 8-11B) with the highest concentrations found in Lauritzen Channel and generally decreasing concentrations through Santa Fe and Richmond channels to the entrance of Richmond Harbor. Concentrations of total 4,4'-DDD in the interstitial water of Stations 14, 15, and 16 averaged 40 ± 13% of the values reported by Brown *et al.* (1990) from composited upper (feet in length) cores from three different sections of Santa Fe Channel that included our station locations. Given the sampling and analytical differences (Brown *et al.* made no corrections of DDD from DDT degradation within instruments), and the horizontal and likely vertical heterogeneity of the area (White *et al.* 1993), the agreement is good. Brown *et al.* (1990) did not detect 4,4'-DDT, its other dechlorination products, or dieldrin in the IW of sediment from these Santa Fe locations. The elutriate IW from composites of the top one foot of sediment from White *et al.* (1993) were from stations over a wider area than that of Brown *et al.* and comparisons to individual samples from this study would be problematic.

The concentrations of ΣDDT in the sediments of Lauritzen Channel Stations 1-3 (Table 8-3) are higher than the highest average concentration (3,540 ng/g dry, Palos Verde, CA) reported in the National Status and Trends Program (NOAA, 1991). Lauritzen Channel Station 4 and Santa Fe Channel Station 6 exceed the second highest value reported by NOAA; all other stations in Santa Fe Channel (5, 13-16) exceed the fifth highest reported concentration of 75 ng/g dry (Santa Monica Bay, CA). The concentrations from Richmond Inner Harbor Channel (Stations 7-9, 17-23) are similar to the range of values reported in the NOAA (1991) and the Long and Markel (1992) summaries of the basins of San Francisco Bay. From this comparison, it is apparent that Lauritzen Channel has some of the highest sediment concentrations of DDT ever reported. Concentrations in Santa Fe are also high in comparison to other sites in the country, while values in Richmond are about equal to the regional values.

The distribution of ΣDDT (Figure 8-2) shows Station 6 in Santa Fe Channel to be different from the other Santa Fe stations and indistinguishable from Station 4 in Lauritzen Channel with a ΣDDT gradient away from Station 6 toward the mouth of Lauritzen Channel (Appendix 8-1). However, TOC was higher at this station, and organic carbon normalization of the ΣDDT reduces the differences, and any evidence of a gradient away from Station 6 in the Santa Fe Channel vanishes as this station becomes indistinguishable from all stations around it (Stations 5, 14, 15, 16 and 7).

Comparison to Criteria: The measured sediment and interstitial water concentrations can be compared to a number of water quality and sediment standards, which can be used as guides to the potential effects of contamination. The acute Water Quality Criteria (WQC, 130 ng/L) for ΣDDT is surpassed in the intersti-

tial water at four stations (1, 2, 3 and 6). Since this value is based on toxicity, it is an appropriate measure of potential effects on the benthos.

The chronic DDT WQC (1 ng/L) is surpassed in the interstitial water of virtually every station in the study. The chronic value is a bioaccumulation criterion based on consumption of contaminated fish by brown pelicans. As discussed in Section 5.2.4., the application of this bioaccumulation-based criterion to interstitial water is not directly related to the protection of the benthos. Based on the chronic/acute ratio, sensitive invertebrates may experience chronic effects at about 13-65 ng/L ΣDDT (Table 5-4). Total interstitial water concentrations exceed the upper range by at least 7-fold at four of the primary stations (1, 2, 3 and 6). At four of the remaining five primary stations, the lower range is exceeded by about 1- to 9-fold.

For dieldrin, the acute WQC (710 ng/L) is not exceeded in any of the interstitial water samples; whereas, the chronic WQC (1.9 ng/L) is exceeded or equaled at five stations (1-4 and 6). As with DDT, the acute value is based on toxic effects and should be applicable to the benthos while the chronic value is a bioaccumulation-based criterion and is not appropriate for predicting effects on the benthos. Assuming acute-to-chronic ratios of 2 to 10, concentrations of dieldrin of 71-355 ng/L could result in chronic effects in sensitive benthic invertebrates (Table 5-5). The upper range is not exceeded at any stations, while the lower range is exceeded by up to 3-fold at three stations (Stations 1-3). The same three stations also exceed the interim sediment quality criterion for dieldrin (17 μg/g OC).

These results strongly suggest that the present levels of ΣDDT sediment contamination are sufficient to result in acute effects on the benthic infauna in Lauritzen Channel and possibly in localized portions of the Santa Fe. The transcendence of the lower range of the calculated chronic effects level for ΣDDT suggests that the most sensitive invertebrates may be affected in the Santa Fe and much of the Richmond Harbor Channels. That ΣDDT exceeds the calculated chronic effects levels to a greater extent than dieldrin indicates that ΣDDT has the dominant impact on the benthos. Effects on the benthos are further discussed in Sections 9.1., 9.2. and 9.3.

The violation of the bioaccumulation-based criteria suggests that unacceptable levels of tissue residues could result in the prey of birds feeding directly on the benthos (*e.g.*, seagulls). Additionally, the uptake of DDT and dieldrin by fish from contaminated sediment and/or prey and the subsequent consumption of the contaminated fish by fish-eating birds could result in adverse effects on higher trophic levels. The uptake of sediment-associated pollutants by fish is discussed in Section 8.4. and the trophic transport of pollutants in Section 9.4.

Distribution of DDT Metabolites and Dieldrin: The distributions of 2,4' and 4,4' isomers of DDT in the upland samples (13% and 80% of the total, respectively) (Table 8-4C) are very similar to the distributions reported for technical DDT formulations

(Gunther and Gunther, 1971). In comparison, 2,4'-DDT formed a trivial percentage in the sediment at most of the stations in this study and the percentage of 4,4'-DDT was 40 to 70% less than in the upland samples, with a concomitant increase in their respective isomeric DDDs. ΣDDT in the upland samples was approximately 100-times higher than dieldrin.

The differences between the distributions of DDT and its metabolites in the upland samples and sediment, and the sediment and interstitial water are suggestive that the original material entering the marine environment has been metabolically altered, and that the interstitial water concentrations of these compounds are not being controlled solely by their relative mass ratios (Table 8-4). For example, 4,4'-DDT in the interstitial water averaged 7.4% of the sum of DDTs, 40% in the sediment and 80% in the upland samples. Within the Lauritzen Channel sediments, virtually all the 2,4'-DDT has been dechlorinated to 2,4'-DDD; whereas only 40 to 70% of the 4,4-DDT has been converted. This may be related to the approximate five-fold higher solubility of the 2,4'-DDT isomer (26 vs. 5.5 ppb, Weil *et al.*, 1974). This higher solubility would mobilize this isomer into the interstitial water where dechlorination would proceed.

Dechlorination of 4,4'-DDT proceeded to the greatest extent (80%) with the highest percentage of products as 4,4'-DDE (38%) at Station 6 in the Santa Fe Channel (Table 8-4A). Dechlorination through a co-metabolic process apparently is the dominant biotransformation mechanism of DDT with DDD thought to be the dominant endproduct under anaerobic conditions and DDE under aerobic conditions (Johnsen, 1976). The oiled sediment of Station 6 apparently enhances the formation of DDE but not to the exclusion of DDD as this compound remains the dominant dechlorination product even at this station (62%).

Apparent Partitioning Coefficients: The defensibility of setting a cleanup level to a single sediment concentration that is protective of organisms based on a water (interstitial) concentration is inherently dependent upon the known and invariant distribution of the pollutant between the bulk sediment solid and liquid phases which is expressed as the K_{oc} of the compound. What are the methods that can be used to know the K_{oc}s of the compounds in this study, how variable is each, and are they comparable? Do our findings suggest that a site-specific K_{oc} would be more useful in subsequent studies of these sites?

The analysis of the K_{oc}s focuses on the Lauritzen data because the variability of K'_{oc}s doubled in samples from outside Lauritzen Channel, reflecting analytical variability inherent in interstitial water concentrations near the detection limit. Also, many more of the calculations of K'_{oc} from outside Lauritzen Channel were based on NC3 (compound unconfirmed) and L (concentration below lowest standard) values. For these reasons, analysis of the Lauritzen samples should result in the most accurate K'_{oc}s and, as such, constitute the most appropriate comparison of K'_{oc} to those derived from K_{ow}.

In Lauritzen Channel, K_{ow}s were able to predict K'_{oc}s with

varying degrees of success. The K_{ow}-predicted interstitial water concentrations of dieldrin and 4,4'-DDD averaged +40% and -20% of the measured concentrations, respectively. With more variation in the dieldrin K_{oc}s, its +40% deviation was not statistically significant (Table 8-5B); whereas, the -20% of 4,4'-DDD was significant. Neither difference is toxicologically significant because the uncertainty of LC_{50}s is on this order. For these two compounds, K_{ow}s seem to be good predictors of freely-dissolved concentrations.

However, the 0.30 log unit difference between the K'_{oc} and K_{ow}-based K_{oc} of 4,4'-DDT, though not significant, would result in a biologically significant K_{ow} predicted concentration twice the mean value determined in the Lauritzen Channel sediments. The difference between the K'_{oc} and K_{ow}-based K_{oc} of 4,4'-DDE is statistically different and would result in a biologically significant under-prediction of 80%, (*i.e.*, one fifth of the mean determined concentrations).

What are some possible explanations for these discrepancies? Are the K_{ow}s wrong for some compounds and not others? Is there an analytical bias affecting some compounds and not others? Analytical bias will be addressed first.

Uniform K'_{oc}s for a compound should be found in a study area if the compound had equilibrated with the sediment substrate it had been transported to or was created within. In the case of those compounds thought to be metabolically formed following deposition of their chemical precursors, the isomers of DDD and DDE have K'_{oc}s with standard errors of their means averaging 12%. In comparison, their DDT progenitors and dieldrin had S.E.s that were 2 to 5 times more variable. The finding that DDT metabolites had reasonably uniform K'_{oc}s but the parent compound did not, suggests that the initially deposited pesticides are not yet equilibrated with the sediment of Lauritzen Channel. For example, 8 of the 15 individual logK'_{oc}s for 4,4'-DDT from Lauritzen Channel were greater than 6.90, the K_{ow}-derived K_{oc} for 4,4'-DDT, and tended to be associated with the highest sediment concentrations (6 of 8 from Station 1). If some of the DDT was still present as the crystalline material (the likely form upon entry into the marine system from the upland source), all DDT present would be extracted in the analyses but only the outer surface of the crystalline material would be in equilibrium with the interstitial water. Because K'_{oc} is based on the extractable masses of pollutant an artificially high value would be computed for a non-equilibrium sediment.

An additional indication of the non-equilibrium of 4,4'-DDT can be found in the observation that the three lowest accumulation factors (AFs), a measure of sediment bioavailability, occur in the three sediments with the highest 4,4'-DDT concentrations (Table 8-16). The other compounds did not display this pattern. The lower 4,4'-DDT bioavailability at the highest sediment concentrations suggests that a portion of the 4,4'-DDT is in a less bioavailable state, such as the crystalline phase. In addition, the differences in acute toxicity between the Palos Verde Shelf

core sediments and the highest ΣDDT sediments of this study (Figure 9-5) may, in part, be attributable to non-equilibrium of the Lauritzen Channel sediments.

Losses of compounds from interstitial water through sorption to sampling equipment surfaces could contribute to higher K'_{oc}s. However, losses of compounds spiked into reference interstitial water at relatively high concentrations were negligible (Table 6-2). DDE formed in the chromatography instruments from the degradation of DDT (self-correcting in standardization) also could have contributed to the deviations in K'_{oc}s. However, DDE formed in sediment samples would result in higher K'_{oc}s, not the observed lower values. "Created" DDE in interstitial water samples would yield values in the direction observed, but would unlikely increase the free concentration by the ~5-fold needed to account for the 0.68 log unit lower K'_{oc} of 4,4'-DDE. Therefore, potential analytical artifacts, sorption and creation of DDE are not reasonable explanations for these deviations from K_{ow} predictions.

Errors in determining the K_{ow} values could also contribute to the difference between the observed and predicted K_{oc}s. For 4,4'-DDT, the range of reported K_{ow} values is at least two orders-of-magnitude, but less for the other compounds. Clearly K_{ow} values must be critically evaluated, especially earlier determinations for low solubility compounds. The technology of determining K_{ow} has improved in recent years and the slow-stirring method used by DeBruijn, *et al.* (1989) results in reproducible, minimally biased values for low solubility compounds. However, the similar K_{ow}s of the 4,4' isomers of DDT and DDE found by DeBruijn *et al.* are in contrast to the solubility (Weil *et al.* 1974) and vapor pressure data of Spencer (1975) which show DDE to have significantly higher values for both solubility (14 vs. 6 ppb) and vapor pressure (65 vs 7.3 x 10^{-7} torr); generally inversely related to K_{ow} (Isnard and Lambert, 1989). The significantly lower K'_{oc} of 4,4'-DDE than the K_{ow}-based K_{oc} value is in the direction that the relative solubility and vapor pressure data would suggest DDE should lie relative to DDT. Therefore, close evaluation of K_{ow}s using relational physical chemical information must be made before they can be considered for <u>a priori</u> predictions of interstitial water concentrations.

For two of the four compounds for which state-of-the-art determined K_{ow}s were available, K_{ow} was an adequate predictor of measured freely-dissolved interstitial water concentrations in equilibrated sediments; for 4,4'-DDE and 4,4'-DDT it was not. Although K'_{oc} and K_{ow}-derived K_{oc} for 4,4'-DDT are indistinguishable, the K_{ow}-derived K_{oc} may more accurately predict IW concentrations in equilibrated sediments. Biologically-significant overestimated IW concentrations of 4,4'-DDT could result from calculations based on concentrations from non-equilibrated sediment using either K_{oc} value (Equations 8-1 and 8-2). Until the discrepancy for 4,4'-DDE can be explained, continued field determinations of equilibrium partition coefficients of this compound (and other compounds) should be made. These findings

suggest that K_{ow} is not necessarily an accurate predictor of interstitial water concentrations for all chemically similar compounds, even when the best current methods have been used to determine K_{ow}.

The WQC for DDT consists of a summation of the concentrations of each isomeric form of DDT and its dechlorination products. In this study the dominant forms of DDT that contribute to the IW are 4,4'-DDT and DDD. Each would contribute to the IW in proportion to its K_{oc} and its contribution to the makeup of the bulk sediment (Table 8-4A). Using the K_{ow}-derived K_{oc} of these two compounds (Table 8-5B), the combinations of relative bulk sediment composition, total sediment concentrations, and organic carbon content that result in IW concentrations exceeding the WQC of 130 ng/L (combined 4,4'-DDT and DDD concentrations) are plotted in Figure 8-3. One observation from this figure suggests that as DDT is dechlorinated to DDD, a sediment initially with IW concentrations below the WQC could exceed the criterion in time.

Sediment Mixing: Previous unpublished data on homogenized and non-homogenized sediments suggest that the interstitial water components are elevated after mixing. Only the components of a single T_0 bioassay storage sediment sample (grab 1 of Station 1) were consistently different from its corresponding field chemistry sample, *i.e.*, differences surpassed a critical value (Table 8-6A). This suggests that any changes in sample composition due to storage and mixing were no greater than the differences expected from two cores taken from the same grab. Using concentrations determined in the field chemistry samples for the toxicity and bioaccumulation experiments places an uncertainty on these values that are the same as the within grab uncertainty (Table 8-6B) and not the generally lower subsampling uncertainty (Tables 8-1 and 8-2) of determining the concentrations in the same sediments used for the biological exposures.

Compounds Other Than DDT and Dieldrin: According to current theory on metal bioavailability (DiToro *et al.* 1990), when acid volatile sulfide (AVS) is present in excess to simultaneously extracted metals (SEM) (both expressed on a molar basis), the metals are bound in a non-bioavailable phase. A ratio of SEM/AVS <1 indicates that metals would not be at high enough concentrations in the pore water to cause toxicity. As all the ratios in this study were substantially less than 1.0, none of the toxic effects can be attributed to the measured metals.

Not unexpectedly, the pollutant scans identified many organic compounds at the sites in addition to ΣDDT and dieldrin. The compounds are dominated by combustion products and petroleum hydrocarbons, with many individual compounds present at 1,000-20,000 ppb. Both Stations 2 (Lauritzen Channel) and 6 (Santa Fe Channel) had many more compounds identified than did extracts from Station 9 (Richmond Harbor). Although Stations 2 and 6 have similar numbers of identifiable compounds, the two stations are different in the kinds and concentrations of compounds present. Station 2 contains μg/g (ppm) concentrations of high molecular

weight PAHs and a complex mixture of industrial compounds in the 50 to 900 ppb range. Station 6 is dominated by 1-20 ppm concentrations of alkylated and non-alkylated PAHs, including several thiophene compounds, all of which are indicative of petroleum hydrocarbon products (Lake *et al.* 1979). The fifteen highest concentration compounds, all of which are PAHs, averaged 65-times higher in the Station 6 extract than Station 2 (Table 8-9). These results are consistent with oil sheens observed at Station 6 during sampling and later in the laboratory. It should be noted this station was chosen for the scan analysis because the presence of the oil sheen suggested high PAH concentrations. Therefore, concentrations of PAHs are most likely higher at Station 6 than at the other Santa Fe stations.

PAH concentrations two to three times higher than found in this study have been reported for Lauritzen Channel (Levine-Fricke, 1990) and similar concentrations have been reported for Richmond Harbor Channel from composited long cores (McPherson *et al.* 1989; Brown *et al.* 1990; Pinza, *et al.* 1992). The high PAH concentrations from Santa Fe that we found were not reported by EVS (McPherson *et al.* 1989) in composites from the end of this channel. Our Santa Fe Channel station, Station 6, was one that was sampled in the Levine-Fricke study, but unfortunately PAH determinations were not done on samples from outside of Lauritzen Channel. In a recent draft report of White *et al.* (1993), the homogenized top 1 foot of sediment from SF-25 (very near our Station 6) was described as "very oily in appearance," and contained a total PAH concentration exceeding 24,000 µg/kg (dry).

The sum of 15 PAH concentrations for Station 6 (149,000 µg/kg, Table 8-9) exceeds the average of the peripheral areas of San Francisco Bay by an order-of-magnitude (Long and Markel 1992). Islais Creek had the highest reported value of 132,000 µg/kg. The sum-PAH concentrations of the Lauritzen Channel sample (8,900 µg/kg) were similar to the moderately contaminated peripheral areas of San Francisco Bay; whereas, the sum-PAH concentrations from the Richmond Inner Harbor Channel sample (1,700 µg/kg) were similar to the average of all the sampled basins of San Francisco Bay (Long and Markel, 1992). The sum-PAH concentration from Station 6 exceeded the highest average value from the National Status and Trends Program (Hud/Rar.Est.NY) by a factor of 3.9 (NOAA, 1991). The sum-PAH concentrations from Station 2 exceeded approximately 75% of the reported values; whereas the Station 9 values exceeded approximately 25% of the reported values (NOAA, 1991).

The PCB congeners at the study site were assumed to have originated from Aroclor 1254 and Aroclor 1260. Preliminary estimates of Aroclor 1242 based on tetrachlorobiphenyl congeners were not possible because only two tetrachloro-congeners were found in each sample and neither was characteristic of Aroclor 1242 (Schulz, Petrick and Duinker, 1989). Four of the five heptachlorobiphenyl congeners are present in Aroclor 1260 at greater than ten times higher percentages than in Aroclor 1254. To estimate more precise and unbiased Aroclor concentrations, the

four highest mass-percent heptachlorobiphenyl congeners were assumed to have come entirely from Aroclor 1260. This gave a maximum concentration for Aroclor 1260 and in the presence of an equal concentration of Aroclor 1254 would be an overestimate of the true concentration by 10% at most. With the concentration of Aroclor 1260 fixed, its contribution to the mass of non-heptachlor congeners was determined by mass percent (Schulz, Petrik and Duinker 1989), subtracted, and the concentration of Aroclor 1254, free of Aroclor 1260 interference, was calculated. The average estimated concentrations of Aroclor 1260 and Aroclor 1254 based on different heptachloro- and pentachloro- congeners are given in Table 8-8, with Aroclor 1254 about twice as abundant as 1260.

Aroclor concentrations were about five times as high at the Santa Fe site than in Lauritzen (Table 8-8). Sediment TOC was higher at Santa Fe than Lauritzen (3.97 vs 1.76%, respectively) and organic carbon normalization significantly reduces the differences between the Lauritzen and Santa Fe Channel Aroclor concentrations. However, organic normalization does not explain the differences in PAHs between these two sites.

White *et al.* (1993) reported Aroclor 1254 (110 to 470 ppb) in the upper two to six feet of recent sediments in Lauritzen Channel, no other Aroclor was found above detection limits. In the Santa Fe Channel Aroclor 1254 concentrations averaging ~150 ppb, and ranging from 5 to 110 ppb have been reported from composited cores by McPherson *et al.* (1989), and White *et al.* (1993), respectively. Aroclor 1254 concentrations averaging ~50 ppb for Richmond Harbor Channel have been reported by McPherson *et al.* (1989), and Pinza *et al.* (1992). Aroclor 1260 at ~30 ppb was reported in Richmond Harbor Channel core composites (McPherson *et al.*, 1989).

The Aroclor 1254 concentration (~600 ppb dry) found at Station 6 in Santa Fe Channel is similar to the concentrations (600 ppb to 1800 ppb) that were reported for surface sediment from the Palos Verde shelf (Ferraro *et al.* 1990) but exceeded the 110 ppb value reported from 0-4 foot deep composites from a close station (White *et al.* 1993). From a station close to our Station 2 in Lauritzen Channel, White *et al.* (1993) reported an Aroclor 1254 concentration of 201 ppb compared to our value of 118 ppb. The sum of 20 PCB congeners (Table 8-7) for Stations 2 and 6 are similar to the average (290 ± 250 ppb, mean ± S.D.) of the peripheral areas of San Francisco Bay, and the Richmond Inner Harbor Channel concentrations are similar to the average (45 ± 35 ppb, mean ± S.D.) of the basins within San Francisco Bay (Long and Markel 1992). In comparison to the nationwide, National Status and Trends, concentrations (NOAA, 1991), the Lauritzen and Santa Fe Channel PCB values represent a moderate level of contamination by these compounds and the Richmond Inner Harbor Channel result represents a low level of contamination.

TABLE 8-1. Station results for ΣDDT for interstitial water and sediment concentrations.

STATION	SEDIMENT TOTAL ORGANIC CARBON AVE	S.E.	n	SUM DDT FREE IW ng/L AVE	S.E.	n	SUM DDT TOTAL IW ng/L AVE	S.E.	n	SUM DDT SEDIMENT µg/kg (dry) AVE	S.E.	n	SUM DDT SEDIMENT µg/gOC AVE	S.E.	n	SUM DDT log'Koc AVE	S.E.	n
1	2.38%	0.09%	8	4487.5	2469.4	8	3258.7	1123.9	8	77737	17628	8	3500	959	8	6.07	0.10	8
2	1.78%	0.16%	5	1952.7	576.4	5	2115.7	588.2	5	47829	9613	5	2712	474	5	6.17	0.18	5
3	1.73%	0.05%	5	889.5	223.6	5	977.5	229.8	5	26005	5853	5	1524	367	5	6.25	0.11	5
4	1.46%	0.05%	5	99.6	36.1	3	122.6	14.9	5	2744	109	5	189	11.5	5	6.36	0.15	3
13	1.55%	NA	1	20.7	NA	1	26.3	NA	1	556	NA	1	36.0	NA	1	6.24	NA	1
5	1.49%	0.01%	5	32.3	38.6	4	58.0	30.0	4	420	24.2	5	28.3	1.73	5	5.96	0.20	3
6	2.98%	0.32%	5	381.0	90.0	5	463.0	104.6	5	2345	578	5	90.2	33.7	5	5.37	0.12	5
14	1.51%	NA	1	19.8	NA	1	27.1	NA	1	730	NA	1	48.2	NA	1	6.39	NA	1
15	1.46%	NA	1	19.7	NA	1	21.8	NA	1	522	NA	1	35.9	NA	1	6.26	NA	1
16	1.35%	NA	1	NA			16.6	NA	1	696	NA	1	51.6	NA	1	NA		
7	1.08%	0.03%	5	23.0	4.36	3	28.4	3.0	4	368	46.0	5	34.8	5.33	5	6.22	0.10	3
17	1.34%	NA	1	NA			NA			132	NA	1	9.81	NA	1	NA		
8	1.18%	0.03%	5	9.21	4.29	2	6.75	3.0	4	82.5	4.5	5	6.99	0.31	5	5.93	0.23	2
18	1.25%	NA	1	32.5	NA	1	42.1	NA	1	38.4	NA	1	3.08	NA	1	4.98	NA	1
009	1.22%	NA	1	NA			15.3	NA	1	24.0	NA	1	1.98	NA	1	NA		
9	0.87%	0.02%	5	5.44	3.11	3	21.5	16.4	5	11.6	1.4	5	1.34	0.18	5	5.20	0.10	2
19	1.14%	NA	1	0.03	NA	1	5.55	NA	1	16.6	NA	1	1.47	NA	1	7.69	NA	1
20	1.10%	NA	1	NA			NI			14.6	NA	1	1.33	NA	1	NA		
22	0.76%	NA	1	NA			CO			11.2	NA	1	1.47	NA	1	NA		
21	1.05%	NA	1	NA			CO			18.5	NA	1	1.76	NA	1	NA		
23	1.02%	NA	1	NA			CO			14.6	NA	1	1.43	NA	1	NA		

DETECTION LIMIT: meaningless for ΣDDT

AVERAGES OF INDIVIDUAL CORES FROM SINGLE GRAB SAMPLES:

STAT.	GRAB	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n
1	3	2.4%	0.02%	3	1261.1	403.2	3	1360.5	415.6	3	33251	1989	3	1383	92	3	6.04	0.11	2
5	1	1.5%	0.05%	3	6.1	1.4	3	147.6	133.0	3	408	63	3	28	5	3	6.05	0.64	3
7	1	1.0%	0.04%	3	24.9	3.6	3	32.3	3.9	3	512	64	3	51	5	3	6.29	NA	1

AVERAGES OF SUBSAMPLES FROM SINGLE CORE SAMPLES:

STAT.	GRAB	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n
1	1	1.9%	0.06%	3	NA			NA			167422	43685	3	8692	2260	3	6.21	0.10	3
5	1	1.4%	0.06%	3	NA			NA			528	70	3	39	6	3	6.70	0.07	3
7	1	1.1%	0.03%	3	NA			NA			487	34	3	46	3	3	6.37	0.03	3

S.E. = standard error of mean
n = number of grabs, cores, or subsamples per calculation
NA = meaningless calculation (e.g., a standard deviation with n=1, negative free concentration)
NI = no interstitial water, sample lost
CO = value lost because of carry over

TABLE 8-2. Station results for dieldrin for interstitial water and sediment concentrations.

STATION	SEDIMENT TOTAL ORGANIC CARBON AVE	S.E.	n	DIELDRIN FREE IW ng/L AVE	S.E.	n	DIELDRIN TOTAL IW ng/L AVE	S.E.	n	DIELDRIN SEDIMENT µg/kg (dry) AVE	S.E.	n	DIELDRIN SEDIMENT µg/gOC AVE	S.E.	n	DIELDRIN log'Koc AVE	S.E.	n
1	2.38%	0.09%	8	219.3	89.5	8	231.4	89.7	8	748.0	299.7	8	35.2	16.1	8	5.19	0.05	7
2	1.78%	0.16%	5	167.8	24.7	5	167.8	24.7	5	527.9	158.6	5	28.7	8.2	5	5.20	0.05	5
3	1.73%	0.05%	5	52.5	14.1	5	101.6	19.1	5	441.8	119.7	5	25.8	7.3	5	5.73	0.19	5
4	1.46%	0.05%	5	16.4	1.1	5	16.9	1.2	5	35.7	1.07	5	2.46	0.1	5	5.18	0.02	5
13	1.55%	NA	1	0.00	NA		0.00	NA		16.9	NA	1	1.09	NA	1	NA		
5	1.49%	0.01%	5	0.02	0.02	2	1.10	0.90	4	5.54	1.16	5	0.374	0.079	5	6.18	0.72	2
6	2.98%	0.32%	5	1.77	1.77	5	1.77	1.77	5	78.4	12.8	5	2.59	0.232	5	5.49	NA	1
14	1.51%	NA	1	0.00	NA		0.00	NA		16.9	NA	1	1.12	NA	1	NA		
15	1.46%	NA	1	0.00	NA		0.00	NA		9.35	NA	1	0.642	NA	1	NA		
16	1.35%	NA	1	0.00	NA		0.00	NA		11.1	NA	1	0.824	NA	1	NA		
7	1.08%	0.03%	5	0.63	0.34	5	0.63	0.34	5	9.46	1.46	5	0.889	0.147	5	6.02	0.34	3
17	1.34%	NA	1	0.00	NA		0.00	NA		2.97	NA	1	0.221	NA	1	NA		
8	1.18%	0.03%	5	0.00	0.00	4	0.00	0.00	5	1.73	0.46	5	0.150	0.040	5	NA		
18	1.25%	NA	1	0.00	NA		0.00	NA		0.00	NA		0.000	NA		NA		
009	1.22%	NA	1	0.00	NA		0.00	NA		0.00	NA		0.000	NA		NA		
9	0.87%	0.02%	5	0.00	0.00	2	0.00	0.00	5	0.63	0.63	5	0.072	0.072	5	NA		
19	1.14%	NA	1	0.00	NA		0.00	NA		2.35	NA	1	0.207	NA		NA		
20	1.10%	NA	1	NA	NA		NI			1.91	NA	1	0.173	NA	1	NA		
22	0.76%	NA	1	0.00	NA		0.00	NA		0.00	NA		0.000	NA		NA		
21	1.05%	NA	1	NA	NA		CO			0.00	NA		0.000	NA		NA		
23	1.02%	NA	1	NA	NA		CO			3.44	NA	1	0.338	NA	1	NA		
DETECTION LIMIT:				~0.0			2.3			100/20(Lauritzen stations: 1-3/4) 0.8 (all other stations)								

AVERAGES OF INDIVIDUAL CORES FROM SINGLE GRAB SAMPLES:

STAT.	GRAB	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n
1	3	2.4%	0.02%	3	52.0	NA	1	49.6	2.4	3	182.5	14.8	3	7.6	0.7	3	5.19	NA	1
5	1	1.5%	0.05%	3	1.9	NA	1	0.6	0.6	3	8.0	0.2	3	0.5	0.02	3	5.46	NA	1
7	1	1.0%	0.04%	3	1.3	0.7	3	1.3	0.7	3	9.5	1.0	3	1.0	0.1	3	5.49	NA	1

AVERAGES OF SUBSAMPLES FROM SINGLE CORE SAMPLES:

STAT.	GRAB	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n	AVE	S.E.	n
1	1	1.9%	0.06%	3	NA			NA			2679	79	3	142	8	3	5.36	0.02	3
5	1	1.4%	0.06%	3	NA			NA			8	2	3	0.6	0.2	3	NA		
7	1	1.1%	0.03%	3	NA			NA			11	1	3	1.1	0.1	3	5.85	0.05	3

S.E. = standard error of mean
n = number of grabs, cores, or subsamples per calculation
NA = meaningless calculation (e.g., standard deviation with n=1)
NI = no interstitial water, sample lost
CO = value lost because of carry over

TABLE 8-3. Station results for average bulk sediment concentrations (μg/kg, dry).

	TOC		2,4'-DDE		4,4'-DDE		2,4'-DDD		4,4'-DDD		2,4'-DDT		4,4'-DDT		SUM DDT		Dieldrin	
station	gOC/gdry	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.	mean	S.E.
1	2.38%	0.09%	105.4	18.5	1536.8	294.6	6344.9	1984	32726.9	7829	2546.1	1233	34476.7	7885	77736.9	17628	748.0	299.7
2	1.78%	0.16%	40.5	10.6	863.3	100.2	2555.1	299.7	15713.1	2003	1094.4	752	27562.2	8676	47828.5	9613	527.9	158.6
3	1.73%	0.05%	28.8	12.2	643.3	91.8	2485.9	607.2	15308.8	4034	149.4	86.7	7388.7	1358	26004.8	5853	441.8	119.7
4	1.46%	0.05%	13.5	13.5	109.4	15.5	217.9	23.6	1522.8	104.2	0.0	0.0	880.8	96.4	2744.4	109.3	35.7	1.1
13	1.55%		4.2		48.3		61.8		375.5		0.0		66.2		556.0		16.9	
5	1.49%	0.01%	2.7	0.2	46.3	4.6	40.2	1.6	241.3	13.0	5.2	4.2	84.0	17.8	419.7	24.2	5.5	1.2
6	2.98%	0.32%	34.2	4.6	581.1	104.3	238.3	33.6	953.1	183.1	65.9	51.7	472.2	275.8	2344.9	577.9	78.4	12.8
14	1.51%		3.1		53.0		78.8		506.4		0.0		88.2		729.6		16.9	
15	1.46%		2.2		30.7		51.4		341.5		4.6		92.2		522.5		9.4	
16	1.35%		2.6		39.7		48.0		342.2		3.8		259.4		695.7		11.1	
7	1.08%	0.03%	2.2	0.2	27.4	1.7	35.0	3.6	219.5	23.2	1.7	1.7	82.5	25.8	368.3	46.0	9.5	1.5
17	1.34%		0.6		9.6		11.4		72.0		0.0		38.0		131.6		3.0	
8	1.18%	0.03%	0.6	0.1	9.5	0.9	8.2	0.4	46.2	0.7	0.4	0.4	17.6	4.2	82.5	4.5	1.7	0.5
18	1.25%		0.6		4.7		3.7		24.4		0.0		5.1		38.4		0.0	
009	1.22%		0.3		3.7		2.8		14.0		0.0		3.4		24.0		0.0	
9	0.87%	0.02%	0.1	0.1	2.0	0.3	1.3	0.2	5.3	0.8	0.0	0.0	3.0	0.4	11.6	1.4	0.6	0.6
19	1.14%		0.4		3.0		1.6		6.8		1.5		3.5		16.6		2.4	
20	1.10%		0.4		2.6		1.4		6.1		0.0		4.2		14.6		1.9	
22	0.76%		0.3		1.2		1.6		4.9		0.0		3.2		11.2		0.0	
21	1.05%		0.3		2.8		2.3		9.7		0.0		3.4		18.5		0.0	
23	1.02%		0.4		2.5		1.5		5.0		0.0		5.2		14.6		3.4	

S.E. = standard error of mean

TABLE 8-4A. Phase distributions of metabolites and dieldrin in Lauritzen Channel.

PERCENT OF COMPOUND RELATIVE TO ΣDDT
BULK SEDIMENT

STATION	GRAB	DIELDRIN	24E	44E	24D	44D	24T	44T
1	1	1.6%	0.1%	1.9%	10.3%	41.0%	0.6%	46.1%
1	4	0.5%	0.2%	1.8%	4.9%	29.1%	16.6%	47.4%
1	5	0.8%	0.2%	2.8%	7.3%	43.4%	3.5%	42.8%
2	1	1.5%	0.2%	2.5%	9.6%	54.0%	0.5%	33.3%
2	2	2.2%	0.1%	1.8%	6.1%	37.5%	1.6%	53.0%
2	3	1.3%	0.1%	2.2%	8.2%	53.1%	0.8%	35.6%
3	1	2.2%	0.2%	2.3%	11.4%	62.1%	0.2%	23.8%
3	2	1.4%	0.0%	2.8%	9.2%	58.3%	0.4%	29.4%
3	3	1.7%	0.3%	2.8%	9.3%	59.9%	0.0%	27.8%
4	1	1.2%	2.3%	5.8%	10.6%	58.2%	0.0%	23.0%
4	2	1.2%	0.0%	3.4%	6.8%	50.0%	0.0%	39.8%
4	3	1.3%	0.0%	3.9%	7.6%	47.2%	0.0%	41.3%
	Mean:	1.4%	0.3%	2.8%	8.4%	49.5%	2.0%	36.9%

STATION AVERAGES
PERCENT OF COMPOUND RELATIVE TO Σ(4,4'-DDT + 4,4'-DDD + 4,4'-DDE)

STATION	44E	44D	44T
1	2.4%	47.3%	50.3%
2	2.1%	40.6%	57.3%
3	2.9%	64.1%	33.0%
4	4.4%	60.6%	35.1%
5	12.5%	65.5%	22.0%
6	30.7%	49.5%	19.8%
7	8.7%	68.2%	23.2%
8	13.0%	64.1%	23.0%
9	19.6%	51.3%	29.2%

24E= 2,4'-DDE; 44E= 4,4'-DDE; 24D= 2,4'-DDD; 44D= 4,4'-DDD; 24T= 2,4'-DDT; 44T= 4,4'-DDT

TABLE 8-4B. Phase distributions of metabolites and dieldrin in Lauritzen Channel.

PERCENT OF COMPOUND RELATIVE TO ΣDDT
TOTAL IW

STATION	GRAB	DIELDRIN	24E	44E	24D	44D	24T	44T
1	1	11.2%	0.15%	1.8%	17.5%	73.8%	1.7%	5.0%
1	4	6.0%	0.2%	2.4%	14.8%	79.7%	0.5%	2.4%
1	5	4.7%	0.1%	2.1%	16.5%	72.8%	0.0%	8.4%
2	1	4.0%	0.1%	1.6%	12.0%	61.2%	5.8%	19.4%
2	2	13.4%	0.3%	3.7%	14.8%	78.6%	0.6%	2.0%
2	3	9.7%	0.2%	3.3%	14.0%	70.6%	2.6%	9.3%
3	1	7.6%	0.2%	3.7%	13.4%	66.7%	0.0%	16.0%
3	2	18.1%	0.3%	3.1%	16.5%	77.1%	0.0%	3.0%
3	3	12.4%	0.4%	3.4%	15.0%	81.2%	0.0%	0.0%
4	1	16.2%	0.0%	4.7%	14.6%	69.6%	0.0%	11.2%
4	2	10.9%	4.1%	6.4%	15.5%	62.1%	6.1%	5.8%
4	3	12.0%	0.0%	4.6%	14.8%	74.7%	0.0%	5.9%
	Mean:	10.5%	0.5%	3.4%	14.9%	72.3%	1.4%	7.4%

24E= 2,4'-DDE; 44E= 4,4'-DDE; 24D= 2,4'-DDD; 44D= 4,4'-DDD; 24T= 2,4'-DDT; 44T= 4,4'-DDT

TABLE 8-4C. Phase distributions of metabolites and dieldrin in Lauritzen Channel.

PERCENT OF COMPOUND RELATIVE TO ΣDDT
UPLAND SAMPLES FROM LEVINE-FRICKE STUDY*

		ΣDDT						
SAMPLE	DIELDRIN	24E	44E	24D	44D	24T	44T	µg/g (dry)
LF-14-4	0.7%	NR	2%	NR	7%	14%	78%	12,100
E-4D	0.5%	NR	2%	NR	5%	12%	81%	17,119
cream 11/27	0.8%	NR	1%	NR	5%	10%	84%	65,049
T-1AS	1.6%	NR	2%	NR	8%	14%	76%	59,790
MEAN:	0.9%		1.8%		6.3%	12.5%	79.8%	

24E = 2,4'-DDE; 44E = 4,4'-DDE; 24D = 2,4'-DDD; 44D = 4,4'-DDD; 24T = 2,4'-DDT; 44T = 4,4'-DDT
* = Levine-Fricke, 1990
NR = Not reported
LF-14-4 = Upland sample
E-4D = Sample from embankment next to Lauritzen Channel
cream = "Cream-colored" sample taken near E-4D
T-1AS = Transect on embankment

TABLE 8-5A. Apparent partition coefficients (K'_{oc}) for dieldrin, DDT and metabolites[1].

	DOC mgOC/L		2,4'-DDE K'_{oc}		4,4'-DDE K'_{oc}		2,4'-DDD K'_{oc}		4,4'-DDD K'_{oc}		2,4'-DDT K'_{oc}		4,4'-DDT K'_{oc}		Dieldrin K'oc	
STATION	MEAN	S.E.	LOG AVE	SE/AVE	LOG AVE	SE/AVE	LOG AVE	SE/AVE	LOG AVE	SE/AVE	LOG AVE	SE/AVE	LOG AVE	SE/AVE	LOG AVE	SE/AVE
1	8.68	1.5	6.13	17%	6.14	16%	5.80	15%	5.90	8%	6.99	86%	7.14	26%	5.20	9%
2	6.98	0.6	5.79	23%	6.09	34%	5.77	15%	5.86	17%	6.14	86%	7.15	83%	5.21	13%
3	7.92	1.4	5.99	8%	6.26	15%	6.07	16%	6.16	21%	4.84	NA	6.30	23%	5.88	36%
4	9.08	0.4	NA		6.10	54%	5.95	18%	6.17	17%	NA		7.18	49%	5.18	5%
13	5.70		5.78		6.21		5.94		6.10		NA		NA		NA	
5	10.4	2.3	5.35	NA	6.16	16%	6.10	37%	6.29	22%	NA		6.42	64%	6.61	93%
6	15.2	3.4	5.58	60%	5.40	23%	5.73	48%	5.27	20%	NA		5.25	47%	5.49	NA
14	7.60		NA		NA		6.09		6.22		NA		NA		NA	
15	8.90		NA		NA		5.98		6.15		NA		NA		NA	
16	9.00		5.74		6.87		NA		NA		NA		NA		NA	
7	30.3	9.9	5.67	NA	6.24	43%	6.26	18%	5.17	34%	NA		6.56	60%	6.30	66%
17	8.00		NA		NA		5.96		NA		NA		NA		NA	
8	11.3	1.2	4.90	NA	4.99	NA	6.12	6%	5.87	NA	NA		5.34	NA	NA	
18	16.6		NA		6.15		6.19		6.76		NA		4.10		NA	
009	17.9		NA		5.40		5.21		5.29		NA		NA		NA	
9	12.3	1.7	NA		4.95	NA	4.79	NA	4.99	NA	NA		4.58	NA	NA	
19	8.80		NA		NA		5.77		NA		NA		5.49		NA	
20	NI		NA		NA		NA		NA		NA		NA		NA	
22	5.30		NA		NA		NA		NA		NA		NA		NA	
21	6.60		NA		NA		NA		NA		NA		NA		NA	
23	5.60		NA		NA		NA		NA		NA		NA		NA	

1 = Grab subsamples were averaged for best estimate of grab concentration; all grabs were averaged for each station except Station 2, grab 5 for both DDTs.
S.E. = Standard error of mean
SE/AVE = Standard error of the mean of the untransformed values divided by the average and multiplied by 100 to convert to percent.
NA = Meaningless calculation (e.g., SD with n=1, missing value)
NI = No interstitial water, sample lost

TABLE 8-5B. Apparent partition coefficients for dieldrin, DDT and metabolites.

Using all determinations of K'_{oc} (including each grab core at Stations 1, 5, and 7, but excluding outliers 2,4'-and 4,4'-DDT from Station 2 grab 5):

	2,4'-DDE	4,4'-DDE	2,4'-DDD	4,4'-DDD	2,4'-DDT	4,4'-DDT	Dieldrin
Lauritzen Channel: log ave (SE/ave,n):	6.07 (14%,14)	6.16 (11%,22)	5.90 (10%,24)	6.01 (11%,22)	6.76 (79%,11)	7.10 (24%,15)	5.46 (26%,23)
All other stations:		6.13 (26%,21)		6.12 (21%,22)		6.15 (47%,13)	6.33 (52%,7)
All values combined:		6.15 (13%,43)		6.07 (13%,44)		LDE	LDE
K_{ow}-based K_{oc}s[a]:log transformed values: average (S.E.,n):		6.838 (.004,9)		6.112(.010,9)		6.797 (.010,9)	5.309 (.003,9)

Are there significant differences between the apparent organic carbon-normalized partition coefficient (K'_{oc}) and K_{oc} calculated from Kow?

	4,4'-DDE	4,4'-DDD	4,4'-DDT	Dieldrin
Lauritzen Channel values:	***	*	NS	NS
All other stations:	***	NS	***	NS
all values:	***	NS	LDE	LDE

Are there significant differences between the 2,4' and 4,4' isomers?

Lauritzen Channel values, only:	NS	NS	NS

SE/ave = standard error of the mean of the untransformed values divided by the average
LDE = Lauritzen Channel mean significantly different from mean of all other stations, so, not appropriate to combine.
S.E. = standard error of mean
NS = not significant at the 0.05-level
*, **, *** = significant at the 0.05, 0.01, 0.001-levels, respectively
a = from DeBruijn, *et al.*, 1989

TABLE 8-6A. Sediment and total interstitial concentrations in T_0 bioassay storage sediment compared to field sediment.

STANDARDIZED DIFFERENCE BETWEEN T_0 BIOASSAY AND FIELD VALUES
(BIOASSAY-FIELD)/FIELD

Sediment:	TOC	2,4'E	4,4'E	2,4'D	dieldrin	4,4'D	2,4'T	4,4'T	ΣDDTs
|Critical value/Cf| :	0.3	0.8	0.4	0.6	0.6	0.5	5.0	2.4	1.0
420 10 S1CES011	0.61	1.10	0.91	1.29	1.37	1.96	18.05	0.98	1.51
420 10 S1CES013	0.00	-0.15	-0.06	-0.14	0.08	-0.16	-0.76	-0.47	-0.31
420 10 S1CES015	-0.01	0.20	-0.04	-0.27	0.43	-0.06	-0.26	-0.13	-0.11
420 12 S1CES021	0.04	0.39	0.33	-0.18	0.24	0.01	21.40	1.79	0.70
420 10 S1CES022	0.15	0.73	0.37	0.08	-0.18	0.47	0.71	1.09	0.78
420 10 S1CES023	0.08	0.20	0.05	-0.19	0.35	0.01	1.85	0.13	0.05
420 10 S1CES032	-0.06	ERR	-0.07	-0.32	0.32	-0.37	29.70	0.38	-0.02
420 10 S1CES033	0.02	0.23	0.28	0.44	0.71	0.47	ERR	1.02	0.68
420 10 S1CES034	0.02	ERR	-0.26	-0.05	0.55	0.08	0.92	-0.35	-0.12
420 ave S1CES041	0.14	-1.00	-0.40	-0.35	ERR	-0.38	ERR	0.17	-0.27
420 ave S1CES042	0.16	ERR	0.22	0.01	ERR	-0.28	ERR	-0.48	-0.30
420 10 S1CES044	0.03	ERR	0.45	0.05	ERR	-0.13	ERR	1.03	0.24
420 10 S1CES051	-0.03	-0.13	-0.17	-0.33	-0.48	-0.24	-0.91	-0.61	-0.34
420 10 S1CES054	-0.01	-0.39	-0.21	-0.07	3.10	-0.09	0.36	-0.78	-0.32
420 12 S1CES055	-0.03	0.15	0.01	-0.21	-0.09	-0.11	ERR	-0.56	-0.17
420 10 S1CES062	-0.04	0.18	-0.04	0.26	-0.22	-0.10	-0.10	-0.74	-0.11
420 10 S1CES063	-0.25	0.16	1.66	0.07	-0.31	1.15	-0.72	-0.74	0.50
420 10 S1CES065	0.11	0.06	0.36	0.27	0.16	0.05	ERR	-0.55	0.15
420 10 S1CES073	0.06	0.01	-0.05	-0.02	-0.33	0.10	ERR	1.19	0.22
420 10 S1CES074	0.05	-0.13	-0.11	-0.24	-0.22	-0.22	ERR	-0.35	-0.23
420 10 S1CES075	0.11	-0.10	-0.15	0.10	-0.56	0.05	ERR	-0.53	-0.09
420 10 S1CES081	-0.02	-0.16	-0.14	-0.18	-1.00	-0.31	-0.05	-0.63	-0.31
420 10 S1CES082	0.01	-1.00	-0.43	-0.35	ERR	-0.43	ERR	-0.77	-0.51
420 10 S1CES084	0.03	ERR	-0.16	-0.33	-1.00	-0.39	ERR	-0.98	-0.56
420 10 S1CES091	0.13	-1.00	-0.04	0.10	ERR	0.10	ERR	-1.00	-0.17
420 10 S1CES092	-0.04	ERR	-0.01	0.37	ERR	0.00	ERR	-1.00	-0.25
420 10 S1CES095	0.05	ERR	-0.51	-0.04	-1.00	-0.20	ERR	-0.32	-0.30
Interstitial water:	DOC	2,4'E	4,4'E	2,4'D	dieldrin	4,4'D	2,4'T	4,4'T	ΣDDTs
|Critical value/Cf| :	0.8	4.6	2.4	1.9	4.6	1.3	5.0	5.7	3.7
419 0 1 S1CBI011	0.11	65.42	49.87	36.41	7.61	31.56	23.69	22.07	32.19
419 0 1 S1CBI013	0.31	1.30	2.29	1.75	0.21	2.23	5.32	4.27	2.44
419 0 1 S1CBI015	0.46	1.99	1.32	1.77	4.93	1.97	ERR	0.53	1.84
419 0 1 S1CBI021	-0.11	-0.41	-0.14	-0.38	0.01	-0.42	-0.88	-0.91	-0.53
419 0 1 S1CBI022	0.31	0.18	0.39	0.94	0.22	0.43	4.25	4.61	0.61
419 0 1 S1CBI023	0.24	0.10	-0.10	0.06	0.09	-0.02	-0.30	-0.26	-0.04
419 0 1 S1CBI032	2.90	-0.38	-0.08	0.05	-0.15	0.16	ERR	-0.67	0.11
419 0 1 S1CBI033	0.15	0.02	0.43	1.33	0.59	1.31	ERR	ERR	1.32
419 0 1 S1CBI034	-0.23	0.00	0.53	0.35	0.60	0.24	-0.89	-0.87	-0.06
419 0 1 S1CBI041	-0.26	ERR	0.28	0.18	-0.10	0.19	ERR	-0.57	0.11
419 0 1 S1CBI042	-0.16	-1.00	-0.51	-0.41	-0.38	-0.25	-1.00	-1.00	-0.41
419 0 1 S1CBI044	0.16	-1.00	-0.27	-0.09	-0.29	0.30	-1.00	-0.18	-0.05
419 ave S1CBI051	0.05	6.26	0.65	-0.10	9.03	0.94	ERR	-0.97	-0.80
419 0 1 S1CBI054	-0.17	ERR	0.66	0.38	ERR	0.63	ERR	-1.00	-0.24
419 0 1 S1CBI055	0.16	ERR	ERR	ERR	ERR	ERR	ERR	ERR	ERR
419 0 1 S1CBI062	0.41	ERR	0.63	0.44	ERR	0.07	ERR	1.26	0.56
419 0 1 S1CBI063	-0.19	ERR	4.26	4.70	ERR	1.76	ERR	ERR	2.76
419 0 1 S1CBI065	-0.69	6.97	3.26	2.96	-1.00	2.39	ERR	-0.84	1.87
419 0 1 S1CBI073	-0.74	ERR	-1.00	0.65	-1.00	0.13	ERR	1.23	0.28
419 0 1 S1CBI074	-0.76	ERR	ERR	0.44	ERR	ERR	ERR	ERR	NA
419 0 1 S1CBI075	-0.71	ERR	0.50	0.35	-1.00	0.64	ERR	-1.00	-0.25
419 0 1 S1CBI081	-0.23	ERR	ERR	-1.00	ERR	-0.13	ERR	8.47	1.77
419 0 1 S1CBI082	-0.07	ERR	ERR	ERR	ERR	-1.00	ERR	ERR	1.00
419 0 1 S1CBI084	-0.19	ERR	ERR	-0.09	ERR	ERR	ERR	ERR	NA
419 0 1 S1CBI091	-0.54	ERR	ERR	ERR	ERR	ERR	ERR	ERR	ERR
419 0 1 S1CBI092	-0.47	ERR	ERR	ERR	ERR	ERR	ERR	ERR	ERR
419 0 1 S1CBI095	0.89	-1.00	-1.00	-1.00	ERR	-1.00	-1.00	ERR	-0.79

ERR =comparisons where values were compromised in some way, or the field value was zero

TABLE 8-6B. Within-grab variability, coefficients of variation (times 100) from replicate cores.

SEDIMENT (dry weight-basis):

STATION-GRAB	n	TOC	2,4'E	4,4'E	2,4'D	dieldrin	4,4'D	2,4'T	4,4'T	ΣDDTs
1-3	3	1	24	3	10	14	6	68	27	10
5-1	3	6	18	16	14	4	14	138	57	27
7-1	3	8	8	7	13	17	8	116	62	22
MEAN:	3	5	17	8	12	12	9	107	49	20
Std. Dev:		4	8	7	2	7	4	36	19	8

TOTAL INTERSTITIAL WATER:

STATION-GRAB	n	DOC	2,4'E	4,4'E	2,4'D	dieldrin	4,4'D	2,4'T	4,4'T	ΣDDTs
1-3	2	7	17	16	36	7	39	106	80	44
5-1	3	10	88	71	53	173	19	NA	171	157
7-1	3	28	173	57	25	94	18	NA	87	21
MEAN:	3	15	93	48	38	91	25	106	113	74
Std. Dev:		11	78	29	14	83	12	NA	51	73

TABLE 8-7. PCB congeners in sediment collected from Lauritzen Channel (Station 2), Santa Fe Channel (Station 6), and Richmond Inner Harbor Channel (Station 9). PCBs are listed by congener number with values in µg/kg (dry).

PCB Congener #	Lauritzen	Santa Fe	Richmond
118	9	62	<DL
153	15	60	3
52	17	39	<DL
138	9	31	1
99	5	29	<DL
110	7	22	<DL
149	6	20	2
180	5	20	<DL
44	<DL	18	<DL
105	6	15	<DL
174	3	11	<DL
136	2	11	<DL
141	3	9	<DL
183	1	6	<DL
176	1	6	<DL
170+190	2	4	<DL
66	1	<DL	<DL
101	12	<DL	<DL
95	8	<DL	<DL
Total	112	363	6

TABLE 8-8. Aroclor 1260 and Aroclor 1254 sediment concentrations (ng/g, dry).

	Aroclor 1260 Mean ± S.D.	n	Aroclor 1254 Mean ± S.D.	n
Lauritzen Channel:				
S1CES021	60 ± 16 ng/g	4	118 ± 21 ng/g	6
Santa Fe Channel:				
S1CES063	252 ± 117 ng/g	4	607 ± 310 ng/g	4
Richmond Inner Harbor Channel:				
S1CES095	<DL, ~50 ng/g	0	<DL, ~50 ng/g	0

S.D. = standard deviation
n = Number of estimated concentrations of Aroclor 1260 or 1254 from different heptochloro- or pentachloro- congeners.

TABLE 8-9. Fifteen most abundant organic pollutants identified in SIM run (excluding PCBs, DDT derivatives and dieldrin) in sediment collected from Lauritzen Channel (Station 2), Santa Fe Channel (Station 6) and Richmond Inner Harbor Channel (Station 9). Values are in μg/kg (dry).

Compound	Lauritzen	Santa Fe	Richmond
Phenanthrene	290	19,555	41
Fluoranthene	728	17,648	117
Pyrene	1,235	16,548	158
Benzo(a)Anthracene	440	14,851	62
Naphthalene	154	14,539	24
Fluorene	79	10,967	6
Acenaphthene	31	10,165	7
Chrysene	1,016	10,073	72
1-methyl-naphthalene	77	9,660	8
Benzo(b)Fluoranthene	1,700	7,448	84
2-methyl-naphthalene	190	6,917	13
Benzo(a)Pyrene	1,274	5,740	108
Anthracene	328	4,701	10
Benzo(e)Pyrene	1,015	4,134	70
1-methyl-phenanthrene	45	3,473	7

TABLE 8-10. Simultaneously-extracted metals (SEM) - AVS analyses.

SAMPLE	STATION	AVS, μMOLE/G	CU, μG/G	ZN, μG/G	PB, μG/G	NI, μG/G	CD, μG/G	TOT. SEM μMOLE/G	SEM/AVS
S1CES011	1	135.3	68.1	326.5	92.8	29.4	1.1	7.02	0.05
S1CES023	2	27.8	60.4	173.5	65.9	28.2	0.2	4.40	0.16
S1CES033	3	44.0	51.6	141.7	52.8	34.7	0.2	3.83	0.09
S1CES044	4	22.1	30.3	101.9	37.7	22.7	0.0	2.60	0.12
S1CES054	5	20.2	33.4	90.0	35.2	17.9	0.0	2.38	0.12
S1CES065	6	112.9	47.6	298.5	110.6	25.3	1.4	6.29	0.06
S1CES075	7	29.9	38.9	118.6	41.7	27.9	0.1	3.10	0.10
S1CES082	8	11.0	29.5	89.9	30.2	26.7	0.0	2.44	0.22
S1CES092	9	2.3	10.1	49.9	11.2	8.0	0.0	1.11	0.48

All sediment concentrations in dry weight.

TABLE 8-11A. Comparisons of ΣDDT sediment concentrations in San Francisco Bay.
MEAN VALUES (RANGES) - μG/KG DRY WEIGHT

SURVEY	LAURITZEN CHANNEL	SANTA FE CHANNEL	RICHMOND INNER HARBOR	PERIPHERAL; BASIN	NO. SAMPLES
ANATEC, 1981				(5-1960)	31
Harding Lawson Assoc. 1986b	260,700 (900-765,000)				13
NOAA-NS&T, 1984-6				(0.3-46)	33
Spies, 1986				(3-97)	22
U.S. Navy, 1986				(3-300)	10
U.S. EPA, 1986				(1-91)	11
E.V.S., 1989 [Deep composites]		285 (203-366)	6 (0-19)	(0-109)	14
Levine & Fricke, 1990 [Surficial samples]	11,210 (170-49,200)	370 (0-2310)			38
White et. al., 1993	24,400 (500-72,400)	260 (30-540)	22 (3-45)	(9-10)	29
Present Study	38,579[1] (2744-77,737)	878[2] (420-2345)	206[3] (12-363)	(11-19)[4]	55

PERIPHERAL: harbors, ship channels, marinas, industrial waterways around the perimeter of the estuary (not including Lauritzen Channel, Santa Fe Channel, and Richmond Harbor Channel)

BASIN: San Pablo, Central, and South Bays (generally, less contaminated than the periphery of the estuary)

1 = Stations 1-4
2 = Stations 5-6, 13-16
3 = Stations 7-9, 009, 17-18
4 = Stations 19-23, Richmond Harbor Entrance Channel (Potrero Reach)

TABLE 8-11B. Comparisons of dieldrin sediment concentrations in San Francisco Bay.

MEAN VALUES (RANGES) - μG/KG DRY WEIGHT

SURVEY	LAURITZEN CHANNEL	SANTA FE CHANNEL	RICHMOND INNER HARBOR	PERIPHERAL; BASIN	NO. SAMPLES
E.V.S., 1989 [Deep composites]		72 (67-76)	0.8 (0-2)	(0-1.5)	14
Levine & Fricke, 1990 [Surficial samples]	3,460 (12-48,600)	32 (0-216)			38
White, *et al.*, 1993	254 (30-880)	11 (1-33)	0.6 (0.2-1)	(0.2-0.5)	29
Present Study	438[1] (36-748)	23[2] (6-78)	2[3] (0-9)	(0-3)[4]	55

PERIPHERAL: harbors, ship channels, marinas, industrial waterways around the perimeter of the estuary (not including Lauritzen Channel, Santa Fe Channel, and Richmond Harbor Channel)

BASIN: San Pablo, Central, and South Bays (generally, less contaminated than the periphery of the estuary)

1 = Stations 1-4
2 = Stations 5-6, 13-16
3 = Stations 7-9, 009, 17-18
4 = Stations 19-23, Richmond Harbor Entrance Channel

FIGURE 8-1. Mean ΣDDT (dry weight normalized) with standard errors versus station number.

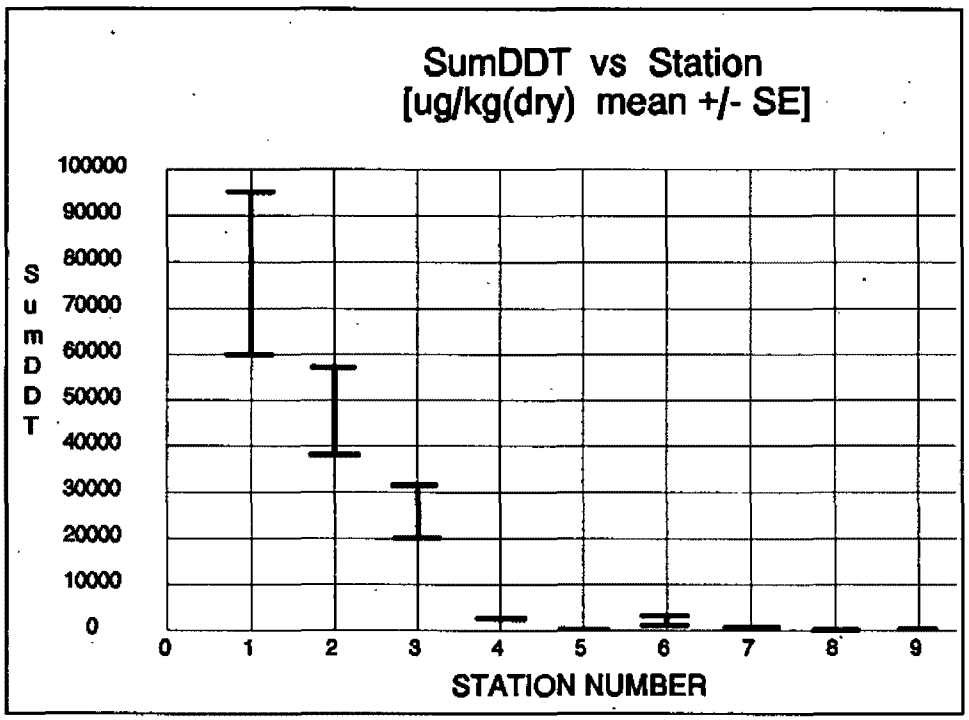


FIGURE 8-2. Station comparisons of bulk sediment for ΣDDT and dieldrin.

Station numbers are arranged in ascending order from lowest to highest mean pollutant concentration on a dry-weight and dry-weight-organic carbon normalized basis. Multiple comparisons of log_{10}-transformed data were by the Games and Howell method because the variances were heterogeneous. Underlines and vertical bars indicate which stations are not significantly different ($P<0.05$).

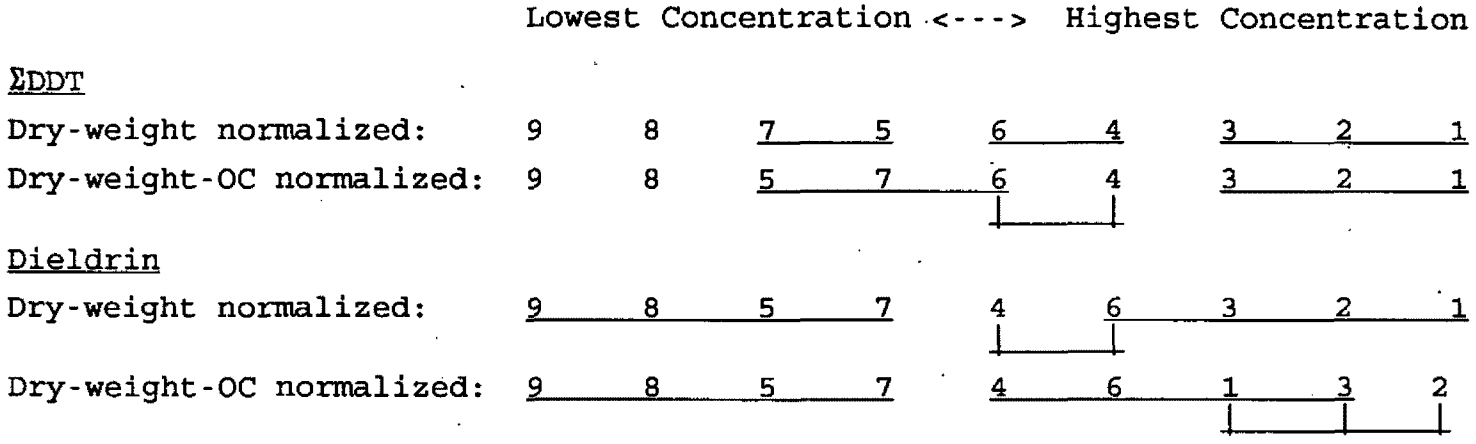
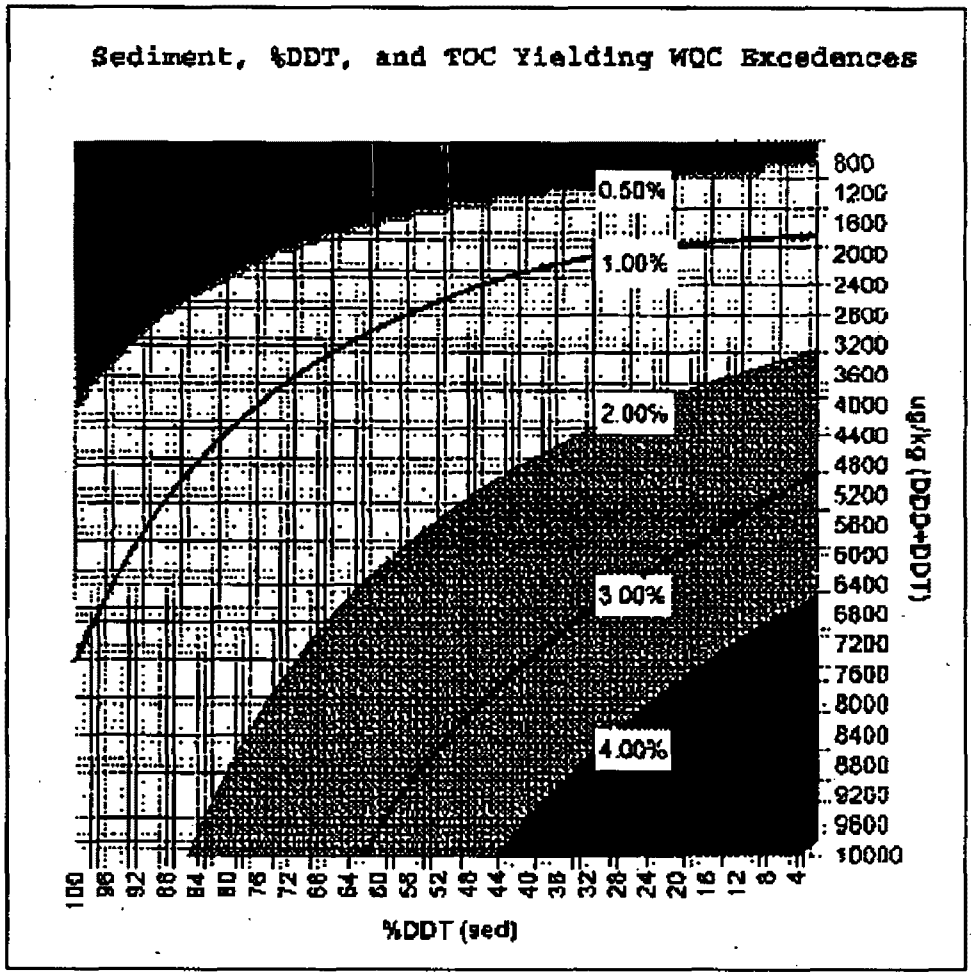


FIGURE 8-3. Combinations of TOC, %DDT and sediment concentrations consisting of 4,4'-DDD and 4,4'-DDT that result in interstitial water (IW) concentrations exceeding the WQC for ΣDDT of 130 ng/L (combinations to the right and below the labeled TOC percentages yield IW concentrations in excess of the WQC).



8.2. OVERLYING WATER CONCENTRATIONS

8.2.1. Introduction

Overlying water can be a pollutant uptake route for pelagic organisms, such as fish, and for epibenthic invertebrates living off the sediment bottom, such as byssal mussels. This section presents concentrations of ΣDDT and dieldrin measured in whole water samples collected in the three channels. These values are used to assess the present exposure of pelagic fishes and mussels and are compared to the ambient Water Quality Criteria for ΣDDT and dieldrin. Additionally, the water and sediment concentrations are used to derive water-sediment ratios that can be used to predict the sediment concentrations required to achieve various overlying water concentrations.

8.2.2. Methods

Because the intertidal byssal mussel Mytilus spp. is so widely used as an indicator organism for pollutant bioaccumulation from water, whole water samples were collected at the same locations and water depths from which the Mytilus galloprovincialis samples were obtained. The water and mussel collection sites are illustrated in Figures 6-1 and 6-2. In Lauritzen Channel, the site was the metal rudder on the northern end of an abandoned ferryboat hull resting on the bottom of the eastern bank of the channel. In Santa Fe Channel, the site was a slip near the southeast end of a floating boathouse situated near the northeast bank of the channel. In Richmond Inner Harbor Channel, the site was a metal channel buoy (No. 16) situated at the mouth (southern end) of the channel.

Water (and mussel) samples were collected during both the primary (October, 1991) and secondary (February, 1992) surveys. During the first survey, a single water sample was collected from each of the three sites on three successive days (Oct. 7 - 9), such that the set of three samples for a site covered as large a range of tidal heights as possible. The mean of these values constitute a 72 hour average water concentration. The sampling of the three channels was repeated during the second survey; however, to obtain estimates of precision, a set of triplicate water samples was collected at each of the three sites within a one-hour period on Feb. 7, 1992 at an intermediate tidal height (4.4-4.6 ft.).

Owing to the trace (parts-per-trillion) levels of the toxic organic pollutants anticipated in the water samples, as little processing of the samples as possible was conducted. As described in Section 6.4.6., water samples were collected in pre-cleaned 500 ml glass Wheaton bottles from a depth of about 0.3 meter, filling completely and recapping the bottles before withdrawal above the water surface. At the end of each day of water sampling, the cooler containing that day's water samples was sent via air freight to the Newport laboratory. The samples arrived the following morning for immediate processing of the unfiltered whole water samples. During the second survey, at

each of the three sites an additional pre-cleaned sample bottle (selected randomly) was opened, held upside down just above the water surface for approximately ten seconds, and then recapped and bagged. These three "empty bottle travel blanks" were processed to obtain information on possible contamination of the water sample bottles, during transport to and from the collection sites, and as a result of the uncapping and recapping process.

8.2.3. Results

Appendix 8-4 lists the values for the individual water samples while Table 8-12 summarizes the data for both surveys, the estimated analytical detection limits, and laboratory and field procedural blank values. Blanks usually were near or below the detection limits; therefore, none of the water concentration values have been corrected for blanks. All concentrations below the detection limit have been treated as zero values for the summary statistics, as discussed in Section 6.6. The tidal ranges for the October and February sampling periods are shown in Table 8-12.

The results listed in Table 8-12 show that the dieldrin concentrations measured in the triplicate samples of near-surface water collected in February, 1992 from the Santa Fe Channel site (1.7, 1.7, and 1.7 ng/L) were very precise, even though their average was less than a factor of two above the detection limit of the method (1.0 ng/L). A similar result is seen for the corresponding ΣDDT concentrations from the Richmond Channel site (1.5, 1.5, and 1.3 ng/L). For the five cases yielding measurable concentrations, the relative standard error values are all less than ± 15 percent. There is no reason to conclude that this level of precision is not approximately representative of what would have been achieved by replicate sampling during the first survey.

Application of the Student's t-test to the results in Table 8-12 showed that for both ΣDDT or dieldrin there were no significant differences ($p<0.05$) between the measurable concentrations obtained for the October, 1991 and the February, 1992 concentrations from a given channel site. Thus, the six sample values for ΣDDT or dieldrin at a site were averaged to obtain an estimate of the typical near-surface whole water concentration during the study period.

Standard ANOVA procedures were conducted to test the null hypothesis that there were no significant differences between the two survey average concentrations of ΣDDT and or dieldrin in the whole water samples from the three channel sites. This H_o was rejected ($p<0.001$) for both pollutants. Application of the Multiple Range Test showed that, in both cases, the Lauritzen Channel concentration was significantly different ($p<0.05$) from the concentrations measured at the other two channels, and that the ΣDDT concentrations in the Santa Fe and Richmond Inner Harbor Channel samples were significantly different ($p<.05$).

8.2.4. Discussion

Applicability of Whole Water Samples: The unfiltered, whole water values measured here include freely dissolved, DOM-associated, and particulate-associated pollutants. These values are compared to the WQC value for ΣDDT of 1.0 ng/L (the Saltwater Final Residue Value) which is based on bioaccumulation rather than aquatic toxicity as the critical path. Even though such whole water samples are consistent with EPA's conservative policy of applying water quality criteria on a whole-effluent toxicity basis ("Technical Support Document for Water Quality-based Toxics Control", EPA Office of Water Report No. EPA-440/4-85-032, September 1985), it is worth examining how these whole water concentrations potentially affect the bioconcentration factors (BCF) derived in this study. (As discussed in Section 8.4., the bioconcentration factors in this study are referred to as BCF_{uw} to point out that they were derived from unfiltered water.)

The relationship used to derive the WQC value utilizes the geometric mean (17,870) of available (percent lipid) normalized BCF values (24 field-generated data points for 22 freshwater fish and invertebrate species, and 18 laboratory-generated data points for 16 fish and invertebrate species). The mean of the three channel averages for the (wet wt.) BCF_{uw} values for ΣDDT, obtained from the mussel whole soft tissue and whole surface water samples from the three channels, is 46,000 (Section 8.4.). Since the mussels average about 1.3 percent lipid, the corresponding normalized BCF_{uw} value is about 35,000. Thus, an agreement within a factor of 2 is obtained between the field-derived normalized BCF_{uw} values in this study and the normalized BCF value of about 18,000 used to derive the chronic WQC for ΣDDT.

A similar situation exists for dieldrin. The WQC of 1.9 ng/L (the Saltwater Final Residue Value) is calculated using a geometric mean (1,557) of two normalized BCF values (1,160 for freshwater mussel (Lampsilis siliquoidea) and 2,300 for spot (Leiostomus xanthurus)). In comparison, the mean of the three channel average for the mussel (wet wt.) BCF_{uw} value for dieldrin is 8,200, and the corresponding normalized BCF_{uw} is about 6,300 (Section 8.3.). In this case, there is agreement within a factor of 4.

For neither pollutant are the normalized BCF_{uw} values obtained from this field study markedly lower than the reference normalized BCF values, which would be expected if the whole water concentrations reported here were biased high owing to dominance of a particulate-associated phase of ΣDDT and dieldrin concentrations that was not bioavailable. Further, as discussed in Section 8.4., similar agreement was obtained between the field-derived normalized BCF_{uw} values for ΣDDT and for dieldrin in fishes obtained in this study, and those used to derive the saltwater WQC values. The similarity in the field and laboratory BCFs probably was due, to a large part, to the low suspended load of the water, as indicated by the lack of any noticeable turbidity.

Violations of Water Quality Criteria:

A major objective of the water sampling program was to document the exposure of the water-column organisms to the target pollutants. In view of the large gradients in distributions of ΣDDT in the survey area observed previous to this study, it seemed probable that large spatial gradients could lead to significant temporal variation in near-surface water concentrations as a result of tidal transport. Thus, in the first (multi-day survey), the three collections per site were spread over as large a period (three days) and tidal height range (~ 2 to 6 ft.) as was practical.

The results listed in Table 8-12 show that much greater variability in near-surface water concentrations was obtained in the first survey than in the second. The major difference in these two surveys was that the first incorporated a broad tidal height variation while the second (where the triplicate sets all were collected at one intermediate tidal height) did not. Assuming that the high precision obtained from the second survey is representative of the single sample results obtained over varying tidal height in the first survey, it is reasonable to conclude that the temporal differences observed in the first survey are at least partly a result of tidal transport, and therefore, averaging over a broad range of tide and time provided the most representative average concentrations for ΣDDT and dieldrin. Thus, by incorporating data from receiving water samples collected on four separate days at different tidal stages, we have obtained average values for pollutant whole water concentrations that should be applicable to the EPA chronic Water Quality Criteria for ΣDDT (1.0 ng/L) and for dieldrin (1.9 ng/L) as a 24-hour average (U.S. EPA 1980a; U.S. EPA 1980b).

The average value for ΣDDT (1.0 ± 0.3 ng/L) obtained at the mouth of Richmond Inner Harbor Channel is similar to the detection limits for the DDT compounds, and only three times the overall mean (0.3 ± 0.2 ng/L) of the three average blank values. Thus, it is possible only to state that the whole water concentration of ΣDDT at that site is similar to the chronic WQC value of 1.0 ng/L. However, these data do indicate that there was an approximate 50-fold gradient in the average concentration of ΣDDT in the whole water samples obtained along the transect joining the three sites. The average values obtained for the Santa Fe Channel (8.6 ± 1.3 ng/L) and Lauritzen Channel (50 ± 6 ng/L) sites both substantially exceed the chronic WQC value. None of the sites exceed the acute DDT WQC value of 130 ng/L.

Dieldrin was not detected at the Richmond Inner Harbor Channel site, and the average concentration (1.8 ± 0.4 ng/L) obtained for the Santa Fe Channel site was less than twice the detection limit (1.0 ng/L). Thus, again it is possible only to state that the whole water concentration of dieldrin at that site is similar to the WQC chronic value of 1.9 ng/L. At the minimum, however, there was an approximate 20-fold gradient in the average concentration of dieldrin in the whole water samples obtained along the transect. The average value obtained for the Lauritzen

Channel site (18 ± 3 ng/L) substantially exceeds the WQC chronic value of 1.9 ng/L.

The water samples presented here are based on near-surface water samples and the design did not address possible vertical stratification in water concentrations. If such stratification existed, it seems likely that the near-surface water samples collected represented minimum rather than maximum concentrations (or at least the lower rather than the higher portion of a concentration range). For example, water closer to the bottom might have contained somewhat higher concentrations owing to a diffusion gradient from the surficial sediment-overlying water interface, or owing to a contribution from a nepheloid layer containing higher concentrations of particle-associated pollutants. If so, epibenthic invertebrates and demersal fishes would have experienced higher, not lower, whole water concentrations. Thus, the present data may underestimate the water exposure of bottom organisms to a certain extent.

Water-Sediment Ratios: One approach to predicting overlying water concentrations is to derive empirical water-sediment ratios (WSR) from the water and sediment values for each of the sites. The sediment concentration required to achieve any particular overlying whole water concentration can then be predicted from:

$$C_s = C_w/WSR \qquad \text{Eq. 8-3}$$

where

C_s = sediment concentration (μg/kg dry)

C_w = target whole water concentrations (μg/l)

WSR = water-sediment ratio (kg/l)

This approach assumes that there is a relatively predictable relationship between water and sediment concentrations at each site but does not assume equilibrium partitioning. The assumption of a relatively constant ratio is supported by the similarity in ΣDDT and dieldrin water concentrations taken over three stages of the tidal cycle on a single day (February sampling) and by the similar mean values of ΣDDT and dieldrin from samples taken four months apart. This consistency over several temporal scales indicates that local WSRs are sufficiently constant to predict long-term or "average" surface water concentrations. However, higher water concentrations, and hence higher WSRs, would be expected after storms or other events that resuspend sediments or transport water with a substantially higher concentration into a channel. Additionally, the WSRs are based on surface water concentrations, and bottom water may have somewhat different, and presumably higher, WSRs.

The WSRs for ΣDDT are given in Table 8-13 and dieldrin in Table 8-14. These values are based on the average water concentration (Table 8-12) and average sediment concentration (see

Table 8-31) for each of the channels. The range in WSRs based on the maximum and minimum water concentrations from both sampling periods are given as an indication of the potential range in the values. The higher WSRs for dieldrin presumably reflect its higher solubility compared to DDT and its metabolites (Table 4-2). The use of the WSRs to calculate the sediment concentrations to achieve various water quality values is discussed in Section 10.

The WSRs can also generate insight into the movement of water-borne contaminants. The WSR for ΣDDT for Lauritzen Channel is about 10-fold lower than that for Santa Fe and Richmond Inner Harbor Channels, which are very similar. The WSR for dieldrin is also lower in the Lauritzen than in the Santa Fe. A possible explanation is that the more highly contaminated Lauritzen water is tidally mixed with the Santa Fe and Richmond Harbor water, resulting in a water concentration and, hence, WSR values in these two channels higher than would be expected from exchange with the local sediment.

The 4+ foot tidal range that occurred during the October, 1991 sampling is a likely mechanism for mixing of Lauritzen water with the other sites. In the four cases in the October survey where average concentrations were more than twice the detection limit of the method, the highest concentration of ΣDDT or dieldrin was obtained in the water sample collected at the lowest of the three tidal stages at a given site. This observation is consistent with the hypothesis that the Lauritzen Channel bottom sediments were the principal source of the observed contamination to the overlying water, and that this contamination was transported from Lauritzen Channel to the Santa Fe Channel and Richmond Channel sites as evidenced by higher concentrations there under ebb tide conditions.

TABLE 8-12A. ΣDDT and dieldrin concentrations (ng/l) in individual whole (unfiltered) water samples collected[a,b] during October 7-9, 1991 and February 7, 1992.

	October, 1991					February, 1992					Oct.+ Feb.	
Site	day	hr	ft.	ΣDDT	Dieldrin	day	hr	ft.	ΣDDT	Dieldrin	ΣDDT	Dieldrin
											$\bar{x}$ conc.	
Lauritzen Channel	10/07	1303	4.9	22	10	2/07	1248	4.4	54	16		
	10/08	1045	6.0	60	19	2/07	1250	4.4	50	15		
	10/09	0630	2.1	68	34	2/07	1252	4.4	47	13		
			$\bar{x}$ =	50	21				50	15	50	18
			SEM =	15	7				2	1	6	3
Santa Fe Channel	10/07	1228	5.5	5.1	3.1	2/07	1303	4.5	9.8	1.7		
	10/08	1030	5.7	9.3	2.8	2/07	1305	4.5	6.6	1.7		
	10/09	0618	2.0	14	<1.0	2/07	1307	4.5	6.7	1.7		
			$\bar{x}$ =	9.4	2.0				7.7	1.7	8.6	1.8
			SEM =	2.5	1.0				1.1	0.0	1.3	0.4
Richmond Harbor Channel	10/07	1416	2.9	0.3	<1.0	2/07	1328	4.6	1.5	<1.0		
	10/08	1118	6.1	<DL	<1.0	2/07	1330	4.6	1.5	<1.0		
	10/09	0650	2.2	1.5	<1.0	2/07	1332	4.6	1.3	<1.0		
			$\bar{x}$ =	0.6	<1.0				1.4	<1.0	1.0	<1.0
			SEM =	0.5	---				0.1	---	0.3	---
	Analytical Blank			0.7	<1.0				0.1	<1.0		
	Field Blank			---	---				0.2	<1.0		
	Detection Limits			0.06-0.2[c]	1.0				0.06-0.2	1.0		

a = Time of collection is Pacific Standard Time.
b = Surface water elevation (ft.) above Mean Lower Low Water datum for Richmond Inner Harbor (U.S. Dept. Comm., NOAA, 1990, 1991).
c = Range of detection limits for DDT and metabolites (see Table 6-1).

TABLE 8-12B. Average percent composition of ΣDDT measured in overlying water samples.

SITE	2,4'DDE	4,4'DDE	2,4'DDD	4,4'DDD	2,4'DDT	4,4'DDT
October, 1991 Samples						
Lauritzen Channel						
Average	0.3%	3.8%	15.7%	58.4%	6.9%	14.9%
SE	0.2%	0.2%	0.7%	3.0%	0.8%	2.2%
Santa Fe Channel						
Average	3.1%	6.3%	15.3%	60.8%	5.8%	8.8%
SE	3.1%	1.5%	0.8%	11.9%	4.1%	3.6%
Richmond Channel[1]						
Average	0.0%	3.7%	26.6%	61.8%	0.0%	7.8%
SE	0.0%	3.7%	6.8%	4.7%	0.0%	7.8%
February, 1992 Samples						
Lauritzen Channel						
Average	0.4%	3.5%	17.1%	51.4%	9.0%	18.7%
SE	0.0%	0.1%	0.1%	0.7%	0.1%	0.7%
Santa Fe Channel						
Average	0.0%	3.9%	25.1%	54.8%	5.9%	10.3%
SE	0.0%	0.4%	6.0%	4.5%	0.6%	0.8%
Richmond Channel						
Average	0.0%	5.4%	31.5%	53.0%	0.0%	9.8%
SE	0.0%	0.4%	8.3%	3.5%	0.0%	9.8%

N = 3 for all samples, except the October Richmond Channel sample; see (1) below

1 N = 2 as all DDT isomers were below detection limits.

TABLE 8-13. Water-sediment ratios for ΣDDT.

CHANNEL	WATER CONC. (µG/L)	SEDIMENT CONC. (µG/KG)	WATER-SEDIMENT RATIO (KG/L)	MIN. WATER-SEDIMENT RATIO[1]	MAX. WATER-SEDIMENT RATIO[2]
LAURITZEN	0.05	50,500	0.99×10^{-6}	0.4×10^{-6}	1.4×10^{-6}
SANTA FE	0.0086	1,010	8.5×10^{-6}	5.1×10^{-6}	13.9×10^{-6}
RICHMOND	0.001	84	11.9×10^{-6}	3.6×10^{-6}	17.9×10^{-6}

1 = Minimum WSR calculated from the lowest water concentration in Table 8-12. Values below detection limit not included.

2 = Maximum WSR calculated from the highest water concentration in Table 8-12.

TABLE 8-14. Water-sediment ratios for dieldrin.

CHANNEL	WATER CONC. (µG/L)	SEDIMENT CONC. (µG/KG)	WATER-SEDIMENT RATIO (KG/L)	MIN. WATER-SEDIMENT RATIO[1]	MAX. WATER-SEDIMENT RATIO[2]
LAURITZEN	0.018	570	31.6×10^{-6}	11.9×10^{-6}	59.6×10^{-6}
SANTA FE	0.0018	29	62.1×10^{-6}	58.6×10^{-6}	107×10^{-6}
RICHMOND	NA	1.6	NA	NA	NA

1 = Minimum WSR calculated from the lowest water concentration in Table 8-12. Values below detection limit not included.

2 = Maximum WSR calculated from the highest water concentration in Table 8-12.

NA = Not applicable because all values below detection limit.

8.3. INFAUNAL BIOACCUMULATION

8.3.1. Introduction

Accumulation of sediment pollutants by infaunal organisms can result in deleterious effects on the benthos while predation on contaminated infauna can serve as a route by which pollutants enter the food web. To quantify the potential for these effects, one goal of the bioaccumulation study was to assess the present extent of bioaccumulation of DDT, its metabolites, and dieldrin in infauna. The other goal was to derive values for sediment bioaccumulation models, which would allow prediction of infaunal tissue residues at different sediment pollutant concentrations.

One method used to assess bioaccumulation was 28-day bioaccumulation tests with Macoma nasuta. Besides assuring sufficient biomass for analytical analysis, laboratory tests with a single species allow direct among-site comparisons (Lee, 1992; Boese and Lee, 1992). As a check on whether 28 days was sufficient to approach steady-state tissue residues, a long-term kinetic study using sediment from one site was conducted. The extent of bioaccumulation also was assessed by measuring tissue residues in field-collected infauna.

The primary sediment bioaccumulation model used in this study is the equilibrium partitioning (EqP) bioaccumulation model (Rubinstein *et al.*, 1987; Lake *et al.*, 1987, 1990; Ferraro *et al.*, 1990, 1991; Lee, 1992; Landrum *et al.*, 1992; Boese and Lee, 1992). This model was chosen, in part, because it directly accounts for differences in sediment TOC and because it does not require extensive toxicokinetic data. A key parameter in the model is the "accumulation factor" (AF), the measure of pollutant partitioning between sediment carbon and tissue lipids. The approach taken was to derive AFs from the laboratory and field infauna for each of the compounds. These site-specific AFs could then be used to predict tissue residues in infaunal organisms at different pollutant and/or TOC concentrations.

8.3.2. Methods: 28-Day Bioaccumulation Test Sediments:

A standard 28-day sediment bioaccumulation test (Lee *et al.*, 1989, 1993; ASTM, in progress) was conducted using sediment (approximately 100g) from each for the first five grabs at Stations 2-9 and the first eight grabs at Station 1. The sediment was transported to Newport and stored at 4°C for approximately 1 to 2 weeks before being used in the bioaccumulation test. Control sediment was surface sediment (top 5 cm) collected from the test organism collection site in Yaquina Bay (Idaho Point mud flat), Newport, Oregon.

Test Organisms: As a surface-deposit feeder, the tellinid bivalve M. nasuta is an appropriate organism to measure sediment bioavailability and it is frequently used in the assessment of dredge materials (*e.g.*, U.S. EPA/ACE, 1991). One to two weeks before the experiment, M. nasuta (25 to 32 mm length) were collected from the Idaho Point Mud Flat, Yaquina Bay, Newport, Oregon. Clams were maintained in sediment from their collection

site in fiberglass water tables which were supplied with flow-through seawater (12-17°C, 25-32 ‰). Twenty-four hours before the start of the test, clams were removed from the holding facility, verified as to species, measured (anterior to posterior valve length), blotted dry, weighed, and a sequential number written on one valve with an indelible marking pen.

Experiment Initiation and Maintenance: Twenty-four hours before the start of the test, sediment from each grab was homogenized by stirring, and then 100 g portions individually placed into numbered, 250 ml beakers which were then submerged into an 8 liter aquarium. Each aquarium was supplied with flow-through filtered seawater (1 μm) at approximately 5 to 10 turnovers per day (25-32 ‰) via gravity feed. Individual aquaria contained 5 beakers (*i.e.*, the experimental units) all of which were collected from the same station so as to reduce possible cross contamination. (A discussion of experimental units, pseudoreplication, and the importance of avoiding cross-contamination as they relate to bioaccumulation tests is given in Lee *et al.* (1989), Lee *et al.* (1993), and ASTM (in progress)). An additional 5 control beakers containing approximately 100g of sediment collected from Yaquina Bay were prepared and placed in a separate aquarium. The purpose of this negative control was to test for system contamination and the viability of the test organisms. All exposure aquaria were contained within a vented biological exposure cabinet that had a controlled lighting cycle (12L:12D).

After the beakers containing the sediment had been submerged for 24 hours, numbered clams were randomly assigned to treatment groups with one clam placed on the sediment surface in each beaker. Five clams that had been assigned to the T_0 control group were sampled immediately prior to the exposure, composited into a single sample and prepared for pollutant and lipid analysis (Figure 8-4).

During the 28-day exposure period, water temperature was checked daily (Mon - Fri) and turnover rate of each of the aquaria was determined. Beakers were checked daily (Mon - Fri) for obvious mortalities, and to determine if clams remained buried during the test.

Sampling: At the end of 28 days, clams were removed from the beakers and allowed to purge their gut contents by placing individual clams in 250 ml beakers containing 100g of Yaquina Bay control sediment for 24 hours. Purge beakers were segregated by treatment in aquaria (8 liter) supplied with seawater at the same temperature, salinity, and flow rates used in the 28-day test. After purging, clams were weighed, sized, shucked, and the pallial fluid discarded. Soft-tissue from each individual clam was then frozen in liquid N_2 and ground into a fine powder using a mortar and pestle which was prechilled and filled with liquid N_2. The frozen powdered tissue was then separated into two weighed portions, placed in sealed vials, and stored (-80°C) until analyzed for lipids and pollutants (Figure 8-4).

Data Analysis: Values below detection limits were used as

zero (0) in calculating means and standard errors. 28-day tissue pollutant concentrations were corrected to steady-state tissue residues using factors determined in the long-term kinetic exposure described below. Mean tissue residue values were compared among stations (ANOVA) and then compared individually to the mean tissue residues in the control and reference sediment treatments (t-test). The dry-weight residues for M. nasuta reported here can be converted to wet-weights using a dry/wet ratio of 0.147 (Specht and Lee, manuscript).

The accumulation factors were calculated from the EqP bioaccumulation model:

$$C_t/L = AF * C_s/TOC \qquad \text{Eq. 8-4}$$

or:

$$AF = (C_t/L)/(C_s/TOC) \qquad \text{Eq. 8-5}$$

Where:

C_t = Tissue pollutant conc. (µg/kg dry-tissue), values corrected to steady-state.

L = Tissue lipid conc. (g lipid/g dry-tissue; decimal fraction).

C_s = Sediment pollutant conc. (µg/kg dry-sediment)

TOC = Total Organic Carbon content of sediment (g carbon/g dry-sediment; decimal fraction).

AF = Accumulation Factor (g OC/g lipid).

AFs were calculated for each compound using the individual tissue residue and sediment concentrations from each individual grab. No AF was calculated if either the residue or the sediment concentration was below the detection limit. AFs were compared among stations (ANOVA) and regressed against TOC and sediment pollutant concentration in order to determine if these factors contributed to AF variation. Note that AFs are also referred to as the "biota sediment accumulation factor" (BSAF).

8.3.3. Methods: Long-Term Kinetic Experiment

Sediments:

Sediment used in the kinetic exposure was collected from Station 1. Because of the large sediment requirement, the sediment was collected from several separate grabs taken after those used to collect sediment for the 28-day test or chemical analysis. This sediment was homogenized by stirring and placed into 1 gallon glass jars. These were stored at 4°C and rolled twice daily for 30 minute intervals on a rolling mill to assure that they remained aerobic. The control sediment used in this

experiment was the same as that used for the 28-day test.

Test Organisms: Same as 28-test.

Experiment Initiation and Maintenance: As in the 28-day test, test and control sediments (approximately 100g) were placed in 250 ml beakers in 8 liter aquaria with test and control sediments segregated into separate aquaria. The experiment was concurrent with the 28-day test. Test and control sediments were changed in all remaining beakers following the 28-day and 60-day sampling periods. Additionally, two gram portions of the appropriate sediment type was added to each beaker three times a week. Sediments were added/replaced so as to maintain a more constant pollutant concentration and to assure that the available food in the surface layer fed on by M. nasuta was not depleted. This replenishment sediment had been stored (4°C) since collection in one gallon glass jars and rolled twice daily for 30 minute intervals on a rolling mill to maintain aerobic conditions.

Sampling: Five clams exposed to the test sediment were sampled following 1, 2, 3, 6, 10, 28, 42, 60, and 90 days of exposure, while five control clams were sampled following 28, 60, and 90 days of exposure. Clams exposed for 28-90 days were purged in control sediment for 24 hours before homogenization. However, clams exposed for 1-10 days were not purged but immediately homogenized. It was decided not to purge these clams because depuration during purging could reduce the initially low tissue residues below detection limits. Sample preparation for pollutant and lipid analysis was identical to that used in the 28-day test (Figure 8-4) with the exception that the five clams from each sampling interval were combined into a single composite sample before splitting into the lipid and pollutant analytical portions.

Data Analysis: Tissue residues for each pollutant were plotted against exposure time and the time to steady-state visually evaluated. When the slope of the plot approached zero, the steady-state tissue residue was estimated from the mean of the tissue residues at that and all subsequent sampling periods. In cases where the uptake curve did not become asymptotic, the steady-state tissue residue was estimated from the highest value attained at either day 60 or day 90. The steady-state values were used to determine a steady-state correction factor for 28-day tissue residues at all stations.

8.3.4. Methods: Field Infauna Organisms Collection and Sample Preparation:

Infauna collected from grabs (October, 1991 sampling) were purged for 24 hours by placing them in buckets (one bucket for each station) which contained Yaquina Bay control sediment (see Section 6.4.4.). This sediment was covered with water from the collection site and aerated with a bubbler. After 24 hours, the organisms were removed from purge buckets and frozen for shipment back to Newport. The incidental infauna caught in the trawl samples in the February sampling period were not purged. In Newport, bivalves were thawed and then prepared for tissue

pollutant and lipid analysis using the same method as in the 28-day test (Figure 8-4). For Pachycerianthus fimbriatus, the entire organism was homogenized and similarly prepared for pollutant and lipid analysis. Individual organisms were analyzed when possible; however, in several cases it was necessary to composite individuals of the same species from the same site in order to obtain sufficient tissue for analysis.

Data Analysis: Both the infauna collected in grabs and trawls were used to increase the number of replicates. AFs were calculated for infauna captured in grabs using the average station sediment concentrations (Table 8-1 and 8-2). Average station sediment concentrations were used because some of the infauna were collected from additional biological grabs from which sediment concentrations were not analyzed. The average sediment concentrations for the trawl stations were used to calculate AFs in individuals captured in trawls (Table 8-30 and 8-31). As in the 28-day analysis, values below detection limits were set to zero (0) in calculating means and standard errors but were excluded in calculating AFs. To determine if there were differences in uptake by feeding type, the AFs for the filter-feeding bivalves collected from the Lauritzen Channel, Tapes japonica and Musculus senhousia, were compared (t-test) to the AFs derived from Macoma nasuta, a deposit feeder, also collected from the Lauritzen Channel.

8.3.5. Results/Discussion:

Macoma nasuta - Survival and Growth: Measures of live wet weight and shell length from day 0 of the 28-day and 210-day kinetic (90 day uptake/120 depuration phase which is not discussed here) laboratory exposures show that M. nasuta grew slowly, if at all. Because the season for rapid growth for this clam in Oregon is normally May to July, when phytoplankton sedimenting from the water column are abundant, lack of growth from mid-October through early May is not unexpected. The indication of growth in the 4 remaining individuals at day 210 could, in fact, have resulted from increased phytoplankton concentrations from mid-April to mid-May in the flow-through aquaria. Although not much growth took place, mortality was low (~4%) and within suggested guidelines (Lee *et al.*, 1989, 1993). The low mortality and maintenance of weight indicates that the sediments constituted an adequate substrate and diet for these clams. The lack of substantial growth obviated the need to incorporate growth dilution into the bioaccumulation calculations.

Long-Term Kinetic Exposure: The individual residue values from the kinetic exposure are listed in Appendix 8-5. Ideally, a 28-day laboratory bioaccumulation test estimates tissue residues within 80% of the steady-state value (Lee *et al.*, 1989, 1993). The plot of ΣDDT versus time (Figure 8-5) is typical of the results for dieldrin and DDT metabolites. It is apparent from these plots that the time to steady-state usually required more than 42 days of exposure. Thus, with two exceptions, steady-

state tissue residues were estimated as the average of the tissue pollutant concentrations at days 60 and 90. The two exceptions were 2,4'-DDT and 4,4'-DDT. For 2,4'-DDT, the 90-day residue was below detection limits, suggesting analytical or sampling error. However, there was no apparent difference between the day 42 and 60 values; therefore, the estimate of steady-state was calculated as the mean of those two values. For 4,4'-DDT, the tissue residue appeared not to attain steady-state by 90 days of exposure. Therefore, the best estimate of steady-state is the highest tissue residue attained in the experiment (*i.e.*, day 90). As these two compounds constituted less than 2% of the ΣDDT in tissues, any error associated with estimating their steady-state tissue residues would introduce negligible error in estimating the ΣDDT steady-state tissue residue.

Comparison of 28-day composite residues to the steady-state tissue residues (Table 8-15) indicates that the 28-day exposure generally attained only about 1/3 to 1/2 of the apparent steady-state values. Although the kinetic study was conducted only with sediment from Station 1, the percentage of the steady-state residues attained should be applicable to the other stations based on the first-order kinetic bioaccumulation model:

$$C_t(t) = \frac{k_s C_s}{k_2} (1 - e^{-k_2 t}) \qquad \text{Eq. 8-6}$$

Where:

$C_t(t)$ = pollutant concentration in tissue at time t

C_s = pollutant concentration in sediment.

k_s = sediment uptake rate coefficient ($g\text{-sed} \cdot g\text{-tissue}^{-1} \cdot day^{-1}$)

k_2 = elimination rate constant (day^{-1})

t = time (days)

Pollutant uptake of many dissolved and sediment-associated pollutants are accurately described by this first-order kinetic model (*e.g.*, Spacie and Hamelink, 1982; Davies and Dobbs, 1984; Landrum *et al.*, 1983, 1992) and plots of the present data are generally consistent with the model. This model predicts the time to steady-state is a function of the elimination rate constant, k_2, and independent of the pollutant concentration (see Lee *et al.*, 1989, Landrum *et al.* 1992). Therefore, the correction factors derived from Station 1 (Table 8-15) are applicable to the other stations.

28-Day Test Tissue Residues: The individual (uncorrected) residue values from the 28-day exposures are listed in Appendix 8-6. Table 8-16 presents the mean 28-day laboratory tissue

residues for M. nasuta that have been corrected to steady-state for each of the pollutants and stations. Table 8-16 also presents the individual residue values for the eight M. nasuta captured in grabs at the same stations.

The corrected laboratory results compare favorably to tissue residue in Macoma collected at the same stations. Sixty-three percent of the dieldrin and DDT derivative tissue residues in the field M. nasuta fall within the 95% confidence intervals of the corrected 28-day tissue residues (Table 8-16). Of the sixteen comparisons for ΣDDT and dieldrin, only one field value deviated by more than 3-fold from the laboratory residue. These results indicate that the corrected 28-day tissue residues can predict field tissue residues in the same species within 2- to 3-fold.

Table 8-16 also compares 28-day tissue residues from test sediments to Yaquina Bay control (Station 0) and Richmond Inner Harbor Channel Station 9, which is used as a local reference sediment. For ΣDDT, all residues from the test stations were significantly different than control and reference residues (t-test, $p<0.05$). For individual DDT derivatives and dieldrin, test organisms exposed to Lauritzen Channel sediment and, to a lesser extent, those exposed to Santa Fe Channel sediment tended to have tissue residues that were significantly different than control and reference M. nasuta. Exceptions are 2,4'-DDT and 2,4'-DDE, where tissue residues were near detection limits, and dieldrin and 2,4'-DDD in clams exposed to sediment from Station 1, due to high variability.

Mean 28-day tissue residues are displayed in homologous groups by station (Figure 8-6) using a multiple range test (Scheffe Method). For most compounds, residues in Lauritzen Channel Stations 1-3 were similar and higher than the other stations. Tissue residues in clams exposed to Stations 5-7 sediments were similar but lower than those from Lauritzen Channel. Residues at Station 4 at the mouth of Lauritzen Channel were intermediate in concentration. This is suggestive of a graded response from the suspected source of the pollutant in Lauritzen Channel to other areas in the study site.

Regression Bioaccumulation Models: Regression models of the natural log of the 28-day tissue residues in M. nasuta versus the natural log of the bulk sediment concentration, organic-normalized sediment concentration, and total interstitial water concentration were determined for ΣDDT and dieldrin (Table 8-17). All interstitial water samples <DL were excluded from the regressions.

For both ΣDDT and dieldrin, all three measures of sediment contamination were significantly related to tissue residues and explained at least 70% of the variation in residues. In previous studies, normalizing to TOC has reduced the variability in the relationship between tissue residues and sediment pollutant concentration (*e.g.*, Rubinstein *et al.*, 1983). In the present study, normalizing sediment pollutant concentrations to TOC only marginally improved the regressions. However, TOC did not vary greatly among sites in the present study compared to studies

where TOC was found to be important.

The ΣDDT total interstitial water concentration regression model was not as good as those with the bulk or carbon normalized sediment concentrations. For dieldrin, however, the interstitial water explained more of the variation than the bulk sediment or organic normalized concentrations. The poorer regression with ΣDDT interstitial water concentrations could be a result of ingested sediment rather than interstitial water being the dominant uptake route for DDT, as was found for hexachlorobenzene uptake into M. nasuta (Boese *et al.*, 1990). Alternatively, the poorer fit may be due to the greater variation inherent in interstitial water measurements at the part per trillion (ppt) level compared to sediment measurements at ppb or ppm levels.

Based on the r^2 values, the regression model using carbon-normalized concentrations is the best predictor of ΣDDT residues; whereas, the regression using the interstitial water concentration is the best predictor of dieldrin tissue residues. Both of these regressions explain approximately 90% of the variation, allowing precise predictions of the residues of both compounds. It is important to note that these equations are based on 28-day tissue residues, so that the predicted ΣDDT residue would have to be multiplied by 2.9 and the predicted dieldrin residue by 1.7 to estimate the steady-state tissue residues (see Table 8-15). It is also important to note that these equations are based on M. nasuta, and they may not accurately predict residues in other species, especially filter feeders, as discussed below.

EqP Bioaccumulation Model: Table 8-18 presents the AFs for each of the stations and pollutants calculated from the corrected 28-day tissue residues. As expected, AFs showed much less among-station variation than did the tissue residues. Significant differences among stations were noted only for dieldrin, 2,4'-DDT, and 4,4'-DDT (ANOVA, $p<.05$). The differences noted for 2,4'-DDT may be discounted due to the limited data set. Likewise the differences noted for dieldrin are likely due to a single AF value. For 4,4'-DDT, Stations 8 and 9 had significantly greater AFs than all the other sites, which were statistically indistinguishable (multi-range test, Scheffe method). These higher AF values occurred in the two stations with the lowest TOC and pollutant concentrations among the primary stations, which is consistent with previous reports of higher AFs in low TOC and/or low pollutant concentration sediments (Rubinstein *et al.*, 1987; McElroy and Means, 1988; Ferraro *et al.*, 1990; Lake *et al.*, 1990).

AF values (Table 8-18) compare favorably to values (corrected to steady-state) reported by White *et al.*, (1993) for M. nasuta exposed to sediment composites from the same study area. In general, White's AF values for 4,4 DDT, 4,4 DDD, 4,4 DDE, and dieldrin fall within the upper part of the range of values for stations in Lauritzen Channel and the Santa Fe Channel. White's mean values for dieldrin in the Santa Fe Channel and for ΣDDT in Lauritzen and Santa Fe Channels are within 2 to 3-fold of our overall mean value (AF = 0.9 and 1.51, respectively). White's

mean dieldrin AF for Lauritzen Channel is 4-fold higher (AF = 5.76) than ours. These AF differences are likely due to the larger tissue bioaccumulations for ΣDDT and dieldrin that occurred in White's study compared to ours (see below discussion heading: Comparison of DDT and dieldrin Residues to other Studies). However, this reasoning cannot explain White's AFs for Richmond Harbor (mean AF for ΣDDT and dieldrin = 48.4 and 40.5), which are the highest AF values ever reported.

Although mean AFs were similar across stations, there could be as much as an order-of-magnitude variation within individuals exposed to the same sediment. These results suggest that biological variation (*e.g.*, feeding rate) rather than site conditions (*e.g.*, grain size) have the major effect on AFs. Much of this biological variation can be "averaged out" by pooling the samples from all the sites. Thus, 95% confidence limits for the pooled AFs for ΣDDT and dieldrin have less than a 2-fold range (Table 8-18). Therefore, the EqP bioaccumulation model using the overall mean AFs in Table 8-18 could be used to predict mean tissue residues of ΣDDT and dieldrin in M. nasuta with an accuracy of about two-fold. Extrapolation of AF values from M. nasuta to other species would result in a less accurate prediction, and as discussed below, could overestimate tissue residues in filter feeders.

Field Infauna Tissue Residues and AFs: The mean ΣDDT and dieldrin tissue residues for field-collected infauna by species within a channel and pooled across species are given in Table 8-19, while Appendix 8-7 lists the individual values. Because the of the need to obtain sufficient biomass for chemical analysis, field tissue-residue values are limited to larger bivalves and Pachycerianthus fimbriatus, a tube-dwelling sea anenome. Although the taxonomic representation was limited, these organisms were the biomass dominants. Mollusks constituted 63.8% of the average biomass at all the stations, while P. fimbriatus constituted over 90% of the biomass at one station.

The residues in the field-collected Macoma and filter-feeders showed the same pattern as the laboratory-exposed Macoma. Tissue residues of both ΣDDT and dieldrin are at least 10-fold higher in Lauritzen Channel than in Santa Fe Channel; while residues are about 2-fold to 10-fold higher in Santa Fe than in the Richmond Harbor Channel.

The mean accumulation factors derived from the field tissue residues are summarized in Table 8-20. Field AFs for M. nasuta were similar to the corrected laboratory Macoma values, and all the site values were within 2-3 fold of the overall laboratory value. The AFs for both ΣDDT and dieldrin were higher for field-collected M. nasuta than the filter-feeding bivalves. In Lauritzen, the M. nasuta AF for ΣDDT were approximately 20-fold and 6-fold higher than those for Tapes japonica and Musculus senhousia, respectively ($p<0.05$, t-test). For dieldrin, these AF differences were also significant ($p<0.05$, t-test). Across all the sites, Macoma nasuta AFs were about 8-fold larger than the filter-feeder average. These results are consistent with other

studies that have found lower AFs in filter-feeding organisms compared to deposit feeders (*e.g.*, Lake *et al.*, 1990).

As a result of these species-specific differences, predators on benthic organisms could have substantially different exposures depending upon their preferred infaunal prey. In general, the AFs for M. nasuta would be the best predictor of residues in the prey of predators feeding on Macoma and polychaetes, many of which ingest sediment. The filter-feeder AF would be the best predictor of residues in the prey of predators feeding on surface bivalves, such as Musculus. The average of all the species would be the best predictor of residues in the prey of generalist predators. The use of these AFs to predict exposure to benthic feeding fishes is discussed in Section 9.4.

The average site AFs for both ΣDDT and dieldrin show a pattern of the lowest values in Lauritzen and the highest values in Richmond (ΣDDT) or Santa Fe (dieldrin <DL in Richmond). This pattern is not due to differences in species or feeding types as M. nasuta, Pachycerianthus fimbriatus and Potamocorbula amurenis all show the same trend. Although the number of individuals is too small to allow a rigorous statistical analysis, the pattern with the three species is suggestive of among-site differences in bioavailability. This trend was not apparent with the laboratory exposed Macoma (Table 8-18); therefore, the cause is not due to differences in pollutant concentration, TOC, or other bulk sediment characteristics.

A possible cause for this pattern is that infauna in the Santa Fe and Richmond Harbor Channels are exposed to highly contaminated resuspended particles, presumably from Lauritzen Channel. Filter feeders or surface-deposit feeders ingesting these particles would receive a higher exposure than predicted from our measurement of sediment concentrations, which integrated the upper few centimeters. Because the bioaccumulation test would not simulate this water column or micro-surface exposure, the laboratory residues would not be expected to demonstrate site differences, as was the case. This hypothesis is supported by the increase in the ratio of Mytilus ΣDDT residues to bulk sediment concentrations, from 0.011 in Lauritzen to 0.017 in Santa Fe Channel to 0.440 in Richmond Harbor Channel. Similarly, the ratio of dieldrin residues to sediment concentrations increases from 0.999 in Lauritzen to 5.960 in Santa Fe to 23.567 in Richmond. These relative increases in pollutant uptake in a water column species indicates that water column exposure relative to sediment exposure increases from Lauritzen to Richmond Harbor Channel, which is consistent with the transport of contaminated particles from Lauritzen Channel.

Predicting Infaunal Tissue Residues: The present study developed two models to predict ΣDDT and dieldrin tissue residues - the regression equations and the site-specific AF values for use in the EqP model. One limitation of the regression models is that they are only available for M. nasuta; while AFs theoretically are applicable to all infauna, and empirical AFs were derived for several of the common infaunal organisms in Richmond

Harbor. Another potential advantage of the EqP model is that it directly accounts for differences in sediment TOC and/or in lipid content of the organism. Conversely, the advantage of the regression equations is that they do not require lipid values or TOC if the bulk sediment regressions are used. We recommend using the EqP model for M. nasuta if both TOC and lipid data are available; otherwise, use the bulk sediment regressions. For other species, we recommend using the EqP model with AF values derived from the species of concern or, if not available, from species of the same feeding type.

Comparison of DDT and Dieldrin Residues to Other Studies: The studies most comparable to the present one are the assessment of bioavailability of sediment pollutants off the Los Angeles County sewage outfall (Ferraro *et al.*, 1990) and at the United Heckathorn site (White *et al.*, 1993).

In the Ferraro study, a 28-day sediment bioaccumulation test was conducted with M. nasuta, though only 4,4'DDD and 4,4'DDE were analyzed. Los Angeles differs from San Francisco Bay in that 4,4' DDE rather than 4,4'DDD is the dominant metabolite, presumably because of different degradation pathways during sewage digestion. The highest total tissue residues of 4,4' DDD and 4,4' DDE in clams exposed to the Southern California sediment was 20,000 µg/kg (dry weight). In comparison, the total residue of the two 4,4'-metabolites was 29,240 µg/kg after 28 days exposure to Station 1 sediment (Table 8-21). Thus, Macoma tissue residues were up to a third higher in the most contaminated area of Lauritzen Channel than off the Los Angeles County sewage discharge, a site known for DDT contamination.

In the White *et al.*, (1993) study, sediment cores were taken from multiple sites in the Lauritzen Channel, Santa Fe Channel, and Richmond Harbor study areas, and a 28-day bioaccumulation test was conducted using M. nasuta. Test sediments used in this 28-day test were composites which combined the upper 1 ft of multiple cores from the study area. Three of these composites (Comp LC, Comp USFC, and Comp 1HC-1) roughly correspond, respectively, to Stations 1-4, 5-6 and 7.

Clams exposed to Lauritzen Channel sediment (Comp LC) in White's study, took up approximately twice the ΣDDT (mean = 35,000 µg/kg dry weight) in a 28-day test compared to the clams in the present study (mean = 16,000 µg/kg dry weight)[1]. Clams exposed to Comp LC bioaccumulated 3 times the dieldrin (mean = 2,100 µg/kg dry weight) than those in the present study (mean = 670 µg/kg dry weight). Uptake of ΣDDT and dieldrin at the other two comparable sites was also greater than in the present study. As in the present study, the dominant DDT isomer taken up by the test organisms was 4,4 DDD, which accounted for 70 to 90% of the ΣDDT tissue residue (ibid). Isomers of 2,4 DDT were not reported

[1]Tissue residue values from the present study that are compared to White's values are 28-day values and have not been corrected to steady-state.

in White *et al.* (1993).

The reason why clam 28-day ΣDDT and dieldrin bioaccumulations are 2-fold greater in White's study than in ours is not readily apparent. Exposure methods were nearly identical and were conducted at approximately the same time of year. Exposure sediment pollutant concentrations and TOC were similar, especially in the Lauritzen Channel test sediments (mean ΣDDT in our study = 50,500 μg/kg dry weight, Comp LC composite = 53,886 μg/kg dry weight).

Comparison of DDT and Dieldrin Residues to Other Compounds: We had intended to indirectly compare DDT and dieldrin tissue residues to other compounds. To do this we measured PAH and PCB sediment concentrations at three selected sites (Stations 2,6, and 9) in the study area (Tables 8-7, 8-8, and 8-9). The M. nasuta tissue residues were then estimated using site specific AF's derived from a 28-day bioaccumulation test conducted by Battelle (Pinza *et al.*, 1992). However, after our study, White *et al.*, (1993) conducted an additional 28-day bioaccumulation test in the study area. In that study, a 28-day M. nasuta bioaccumulation test was conducted using composite sediments from Lauritzen Channel (Comp LC), Santa Fe Channel (Comp USFC) and Richmond Harbor (1HC-1). These sediments are directly comparable to some of the sediments used in the 28-day test in the present study (Table 8-21). M. nasuta tissue residues for PAHs, PCBs, pesticides, and metals were measured. Using these data provides a more direct approach to the assessment of the relative ecological risk associated with ΣDDT and dieldrin.

In White's study, M. nasuta bioaccumulated pesticides in addition to dieldrin. These included endrin, aldrin, chlordane (trans- and cis-), and endosulfan. The greatest bioaccumulations of these other pesticides occurred in clams exposed to Lauritzen Channel sediments, with tissue residues of trans-Chlordane being the largest (700 μg/kg dry weight). Bioaccumulations of these pesticides were also measured in some of the clams exposed to sediments composited from Santa Fe Channel. However, in many of cases the values were at or near instrument detection limits (ibid).

White *et al.* (1993) reported that PAH's and PCB's were bioaccumulated by clams exposed to all of their composite test sediments. The greatest uptake occurred from exposures to sediment from Lauritzen Channel (Comp LC) and in sediments from the upper reaches of the Santa Fe Channel (USFC). Total PAH's for these two sites was approximately 5,500 μg/kg dry weight (Table 8-21). PCBs were bioaccumulated to the greatest extent in clams exposed to Lauritzen Channel sediment, with residues of approximately half that amount measured in clams exposed to sediment composites from the upper and lower reaches of the Santa Fe Channel (ibid).

Comparison of these values to tissue residues of ΣDDT (Table 8-21), indicates that DDT dominates the tissue residues in clams exposed to Lauritzen Channel sediment. However, in Santa Fe Channel and Richmond Harbor, PAH's become the largest contri-

butor to tissue residues. In Santa Fe Channel, this result may be partially explained by the high sediment PAH concentrations found at our Station 6 (Table 8-9), which was included in the Comp USFC sediment used in White's study. As discussed in Section 6.6.7, Station 6 was chosen as a "worst case" for PAH contamination based on the oil observed in the sediment. This observation was confirmed by White *et al.*, (1993) who found that sediment composites from the upper reaches of the Santa Fe Channel (Comp USFC) had a total PAH concentration of 9,000 µg/kg dry weight. This concentration was considerably greater than that of sediment from the lower reaches of the Santa Fe Channel (1,800 µg/kg dry weight). Therefore, the large relative contribution to tissue residues of PAHs in the upper Santa Fe Channel may represent a localized phenomenon.

PCB's increased in relative importance in Santa Fe Channel and Richmond Harbor, but concentrations in clam tissue was always less than for ΣDDT (Table 8-21). The measured tissue residue of Aroclor 1254 in Lauritzen sediment (ibid) was a trivial percentage of the ΣDDT residues measured in M. nasuta (Table 8-21). The measured Aroclor 1254 tissue residue (ibid) was in the same range of PCB residues in measured in M. nasuta (300-530 µg/kg dry weight) exposed to sediment from a reference site in Santa Monica Bay, California (Ferraro *et al.*, 1990). Although PCB residues were relatively more important in the Santa Fe Channel, they were still only a quarter of the Santa Fe ΣDDT residues. Santa Fe Channel and Richmond Harbor PCB residues (Table 8-21) were lower than residues at the Santa Monica reference site, and thus did not appear to constitute a high residue in an absolute sense.

As mentioned previously and in Section 3.1.3., Battelle (Pinza *et al.*, 1992) conducted 28-day sediment bioaccumulation with M. nasuta and the polychaete Nephtys ceasoides using sediment from the Inner Richmond Harbor, San Francisco Bay and offshore reference sites. Pinza's COMP V and COMP VI cover the west and east side of Richmond Harbor Channel, respectively. Our Stations 8, 17, 18, and 009 fall within COMP V. The only compounds other than DDT and dieldrin showing significant ($p<0.1$) increases in tissue residues in comparison to the San Francisco Bay references were Aroclor 1254 and pyrene. When compared to the offshore reference sites, significant uptake of fluoranthene, chrysene, lead, and arsenic were observed in some cases. Though this study showed a few compounds had significantly elevated residues compared to reference sites, the relative increases were not large, especially when compared to DDT metabolites.

Metal data reported by White *et al.*, (1993) were similar to that reported by Pinza *et al.* (1992). Based on the AVS and SEM analysis (Table 8-10), concentrations of bioavailable metals are not sufficient to result in toxicity.

These comparisons indicate that ΣDDT is by far the dominant bioaccumulable compound in Lauritzen Channel. Besides being high in a relative sense, the residues are very high in an absolute sense, with Σ(4,4'-DDD+ 4,4'-DDE) residues at Lauritzen exceeding those found in Macoma exposed to the most contaminated sediment

off the Los Angeles County sewage discharge (Ferraro *et al.*, 1990). Although, Lauritzen Channel dieldrin residues were less than a tenth of the ΣDDT residues, they were considerably greater than the measured PCB residues and about half of the total PAH residues measured by White *et al.* (1993). In the heavily oiled sediments of the upper reaches of the Santa Fe Channel, total PAH residues exceeded ΣDDT residues by several fold. ΣDDT residues were sufficiently low in the Richmond Inner Harbor Channel that PAHs rather than ΣDDT were the dominant residue.

TABLE 8-15. Comparison of 28-day and steady-state pollutant tissue residues in Macoma nasuta from the kinetic exposure with sediment from Station 1.

Tissue residues = µg/kg, dry weight. 28d C_t = tissue concentration at 28 days (composite sample from kinetic exposure), SS C_t = calculated tissue concentration at steady-state, % SS = percent of steady-state attained after 28-days, SS Factor = factor used to correct 28-day tissue residues to steady-state tissue residues.

Compound	28d C_t	SS C_t	% SS	SS Factor
2,4'-DDE	31.4	74.8[a]	42	2.4
4,4'-DDE	1,335.5	2,749.0[a]	49	2.1
2,4'-DDD	2,393.7	4,334.3[a]	55	1.8
4,4'-DDD	27,905.	85,095.[a]	33	3.0
2,4'-DDT	30.0	54.7[b]	55	1.8
4,4'-DDT	150.6	1,634.6[c]	9	10.8
ΣDDT	31,847.	93,384.[a]	34	2.9
Dieldrin	730.4	1,248.4[a]	58	1.7

a = Mean of 60 and 90 day samples used to estimate SS.
b = Mean of 42 and 60 day samples used to estimate SS.
c = 90 day sample used as estimate of SS.

TABLE 8-16. Tissue residues in Macoma nasuta exposed in 28-day test, corrected to steady-state values and residues in field-collected M. nasuta.

COMPOUND: 2,4'-DDE

STATION	CORRECTED STEADY-STATE MEANS	N	95% CONFIDENCE INTERVAL (CORRECTED TO STEADY-STATE)	INDIVIDUAL FIELD VALUE(S)
CONTROL "0"	1.25	4	0.0 - 5.21	-
1	98.20	5	0.0 - 222.77	61.60
2	89.34	5	0.0 - 219.53	95.05
3	90.37[C]	5	0.0 - 180.87	<DL - 61.95
4	16.48[CR]	5	4.50 - 28.45	<DL - 38.86
5	22.84	4	0.0 - 83.65	<DL
6	22.44	4	0.0 - 55.01	-
7	25.43	4	0.0 - 92.29	-
8	3.19	4	0.0 - 9.24	<DL
REFERENCE 9	1.69	3	0.0 - 8.98	-
FIELD	VALUES		WITHIN 95% OF C. I.	6 OF 8

Tissue residues are in µg/kg dry tissue weight.

"N" = the number of clams per treatment in the laboratory exposure.

The 95% confidence limits are for the laboratory residues corrected to steady state.

Superscripts "C" and "R" denote that the laboratory tissue residues at a site were significantly different than the Control or Reference residues (t-test, $p<0.05$).

Field values are individual residues for the eight clams collected in grabs.

<DL = residue in individual field-collected clam less than detection limit (considered equivalent to 0).

TABLE 8-16 (cont'd). Tissue residues in Macoma nasuta exposed in 28-day test, corrected to steady-state values and residues in field-collected M. nasuta.

COMPOUND: 4,4'-DDE

STATION	CORRECTED STEADY-STATE MEANS	N	95% CONFIDENCE INTERVAL (CORRECTED TO STEADY-STATE)	INDIVIDUAL FIELD VALUE(S)
CONTROL "0"	10.31[R]	4	6.54 - 14.08	-
1	2,677.78[CR]	5	1,466.06 - 3,889.51	1,693.96
2	2,865.59[CR]	5	1,445.64 - 4,285.54	3,345.12
3	3,085.79[CR]	5	2,266.39 - 3,905.19	1,737.90 2,662.43
4	574.95[CR]	5	485.23 - 664.66	475.90 1,901.00
5	219.64[CR]	5	149.79 - 289.49	422.12
6	594.84[CR]	4	368.29 - 821.41	-
7	137.61[CR]	4	36.90 - 238.31	-
8	60.21[CR]	4	42.77 - 77.66	150.14
REFERENCE 9	18.46[C]	3	10.76 - 26.18	-
FIELD	VALUES		WITHIN 95% OF C. I.	3 OF 8

Tissue residues are in µg/kg dry tissue weight.

"N" = the number of clams per treatment in the laboratory exposure.

The 95% confidence limits are for the laboratory residues corrected to steady state.

Superscripts "C" and "R" denote that the laboratory tissue residues at a site were significantly different than the Control or Reference residues (t-test, $p<0.05$).

Field values are individual residues for the eight clams collected in grabs.

<DL = residue in individual field-collected clam less than detection limit (considered equivalent to 0).

TABLE 8-16 (cont'd). Tissue residues in Macoma nasuta exposed in 28-day test, corrected to steady-state values and residues in field-collected M. nasuta.

COMPOUND: 2,4'-DDD

STATION	CORRECTED STEADY-STATE MEANS	N	95% CONFIDENCE INTERVAL (CORRECTED TO STEADY-STATE)	INDIVIDUAL FIELD VALUE(S)
CONTROL "0"	0	4	NA	-
1	4,491.66	5	0.0 - 10,020.2	2,661.46
2	5,092.48[C]	5	155.90 - 10,029.0	4,298.73
3	4,733.22[CR]	5	1,074.54 - 8,391.91	<DL 2,825.17
4	428.35[CR]	5	329.87 - 526.83	222.18 810.34
5	104.94	5	0.0 - 242.96	75.82
6	115.74[CR]	4	57.67 - 173.80	-
7	90.52	4	0.0 - 233.27	-
8	9.55	4	0.0 - 27.67	38.54
REFERENCE 9	1.71	3	0.0 - 9.04	-
FIELD	VALUES		WITHIN 95% OF C. I.	4 OF 8

Tissue residues are in µg/kg dry tissue weight.

"N" = the number of clams per treatment in the laboratory exposure.

The 95% confidence limits are for the laboratory residues corrected to steady state.

Superscripts "C" and "R" denote that the laboratory tissue residues at a site were significantly different than the Control or Reference residues (t-test, $p<0.05$).

Field values are individual residues for the eight clams collected in grabs.

<DL = residue in individual field-collected clam less than detection limit (considered equivalent to 0).

TABLE 8-16 (cont'd). Tissue residues in Macoma nasuta exposed in 28-day test, corrected to steady-state values and residues in field-collected M. nasuta.

COMPOUND: 4,4'-DDD

STATION	CORRECTED STEADY-STATE MEANS	N	95% CONFIDENCE INTERVAL (CORRECTED TO STEADY-STATE)	INDIVIDUAL FIELD VALUE(S)
CONTROL "0"	3.60^{R}	4	0.0 - 10.30	-
1	40,508.7^{CR}	5	23,061.0 - 57,956.4	19,408.87
2	44,159.2^{CR}	5	15,212.6 - 73,105.7	34,805.90
3	61,147.8^{CR}	5	11,220.5 - 111,075.0	15,217.10 23,252.72
4	6,641.24^{CR}	5	5,717.23 - 7,565.26	3,207.17 12,770.97
5	1,058.97^{CR}	4	20.53 - 2,097.41	1,041.60
6	1,085.95^{CR}	4	838.56 - 1,333.33	-
7	761.36^{CR}	4	205.57 - 1,317.16	-
8	158.26^{CR}	4	96.67 - 219.84	490.47
REFERENCE 9	32.27^{C}	3	14.39 - 50.16	-
FIELD	VALUES		WITHIN 95% OF C. I.	4 OF 8

Tissue residues are in µg/kg dry tissue weight.

"N" = the number of clams per treatment in the laboratory exposure.

The 95% confidence limits are for the laboratory residues corrected to steady state.

Superscripts "C" and "R" denote that the laboratory tissue residues at a site were significantly different than the Control or Reference residues (t-test, $p<0.05$).

Field values are individual residues for the eight clams collected in grabs.

<DL = residue in individual field-collected clam less than detection limit (considered equivalent to 0).

TABLE 8-16 (cont'd). Tissue residues in Macoma nasuta exposed in 28-day test, corrected to steady-state values and residues in field-collected M. nasuta.

COMPOUND: 2,4'-DDT

STATION	CORRECTED STEADY-STATE MEANS	N	95% CONFIDENCE INTERVAL (CORRECTED TO STEADY-STATE)	INDIVIDUAL FIELD VALUE(S)
CONTROL "0"	0	4	NA	-
1	4.93	5	0.0 - 18.61	60.11
2	20.14	5	0.0 - 69.71	1,639.47
3	36.02	5	0.0 - 88.79	<DL 37.46
4	12.04	5	0.0 - 28.86	<DL <DL
5	14.31	4	0.0 - 59.86	<DL
6	1.67	4	0.0 - 6.99	-
7	15.03	4	0.0 - 51.50	-
8	0	4	NA	<DL
REFERENCE 9	0	3	NA	-
FIELD	VALUES		WITHIN 95% OF C. I.	6 OF 8

Tissue residues are in µg/kg dry tissue weight.

"N" = the number of clams per treatment in the laboratory exposure.

The 95% confidence limits are for the laboratory residues corrected to steady state.

Superscripts "C" and "R" denote that the laboratory tissue residues at a site were significantly different than the Control or Reference residues (t-test, $p<0.05$).

Field values are individual residues for the eight clams collected in grabs.

<DL = residue in individual field-collected clam less than detection limit (considered equivalent to 0).

TABLE 8-16 (cont'd). Tissue residues in Macoma nasuta exposed in 28-day test, corrected to steady-state values and residues in field-collected M. nasuta.

COMPOUND: 4,4'-DDT

STATION	CORRECTED STEADY-STATE MEANS	N	95% CONFIDENCE INTERVAL (CORRECTED TO STEADY-STATE)	INDIVIDUAL FIELD VALUE(S)
CONTROL "0"	31.51	4	0.0 - 131.79	-
1	3,424.63^{CR}	5	2,668.42 - 4,180.83	1,017.39
2	6,654.82^{CR}	5	2,455.66 - 10,854.0	1,483.46
3	5,413.14^{C}	5	124.19 - 10,702.1	595.67 839.14
4	1,328.84^{CR}	5	538.82 - 2,118.85	170.83 223.97
5	259.17	4	0.0 - 557.71	152.67
6	349.96^{C}	4	0.0 - 734.20	-
7	329.22	4	0.0 - 784.17	-
8	124.52	4	0.0 - 356.94	39.76
REFERENCE 9	65.82	3	0.0 - 217.24	-
FIELD	VALUES		WITHIN 95% OF C. I.	6 OF 8

Tissue residues are in µg/kg dry tissue weight.

"N" = the number of clams per treatment in the laboratory exposure.

The 95% confidence limits are for the laboratory residues corrected to steady state.

Superscripts "C" and "R" denote that the laboratory tissue residues at a site were significantly different than the Control or Reference residues (t-test, $p<0.05$).

Field values are individual residues for the eight clams collected in grabs.

<DL = residue in individual field-collected clam less than detection limit (considered equivalent to 0)

TABLE 8-16 (cont'd). Tissue residues in Macoma nasuta exposed in 28-day test, corrected to steady-state values and residues in field-collected M. nasuta.

COMPOUND: ΣDDT

STATION	CORRECTED STEADY-STATE MEANS	N	95% CONFIDENCE INTERVAL (CORRECTED TO STEADY-STATE)	INDIVIDUAL FIELD VALUE(S)
CONTROL "0"	27.68^{R}	4	6.29 - 49.07	-
1	51,945.1^{CR}	5	25,268.9 - 78,621.3	24,903.40
2	56,776.3^{CR}	5	18,767.8 - 94,784.9	45,667.74
3	72,617.4^{CR}	5	16,885.9 - 128,349.0	17,550.67 29,678.87
4	8,300.09^{CR}	5	7,064.33 - 9,535.86	4,076.08 15,745.15
5	1,616.31^{CR}	4	374.18 - 2,858.44	1,692.21
6	2,181.45^{CR}	4	1,591.76 - 2,771.14	-
7	1,215.19^{CR}	4	139.56 - 2,290.83	-
8	288.81^{CR}	4	165.89 - 411.74	718.91
REFERENCE 9	79.17^{C}	3	29.28 - 129.06	-
FIELD	VALUES		WITHIN 95% OF C. I.	4 OF 8

Tissue residues are in μg/kg dry tissue weight.

"N" = the number of clams per treatment in the laboratory exposure.

The 95% confidence limits are for the laboratory residues corrected to steady state.

Superscripts "C" and "R" denote that the laboratory tissue residues at a site were significantly different than the Control or Reference residues (t-test, $p<0.05$).

Field values are individual residues for the eight clams collected in grabs.

<DL = residue in individual field-collected clam less than detection limit (considered equivalent to 0).

TABLE 8-16 (cont'd). Tissue residues in Macoma nasuta exposed in 28-day test, corrected to steady-state values and residues in field-collected M. nasuta.

COMPOUND: DIELDRIN

STATION	CORRECTED STEADY-STATE MEANS	N	95% CONFIDENCE INTERVAL (CORRECTED TO STEADY-STATE)	INDIVIDUAL FIELD VALUE(S)
CONTROL "0"	5.40	4	0.0 - 22.60	-
1	1,354.37	5	0.0 - 2,846.15	822.74
2	1,669.98^{CR}	5	641.60 - 2,698.36	2,550.45
3	1,392.3^{CR}	5	569.84 - 2,214.75	632.14 747.87
4	141.29^{CR}	5	39.74 - 242.85	95.69 311.89
5	32.31	4	0.0 - 75.97	50.56
6	68.92^{C}	4	0.0 - 145.73	-
7	29.20	4	0.0 - 94.11	-
8	0	4	NA	<DL
REFERENCE 9	0	3	NA	-
FIELD	VALUES		WITHIN 95% OF C. I.	7 OF 8

Tissue residues are in µg/kg dry tissue weight.

"N" = the number of clams per treatment in the laboratory exposure.

The 95% confidence limits are for the laboratory residues corrected to steady state.

Superscripts "C" and "R" denote that the laboratory tissue residues at a site were significantly different than the Control or Reference residues (t-test, $p<0.05$).

Field values are individual residues for the eight clams collected in grabs.

<DL = residue in individual field-collected clam less than detection limit (considered equivalent to 0).

TABLE 8-17. Simple linear regression equations of natural log (ln) of ΣDDT and dieldrin tissue residues (TR) in Macoma nasuta exposed for 28-days in a laboratory test versus the natural log of the bulk sediment concentration, organic-normalized sediment concentration and total interstitial water concentration.

Units are as follows: TR[a] = µg/kg dry weight; Bulk sediment = µg/kg dry weight; Carbon Normalized Sediment concentrations = (µg/kg OC); Total Interstitial Water = ng/l.

ΣDDT: BULK SEDIMENT

LN TR = 1.353 + 0.787 LN (X) r^2 = 0.925, p<0.01, n = 39

ΣDDT: CARBON-NORMALIZED SEDIMENT

LN TR = -3.166 + 0.860 LN (X) r^2 = 0.931, p<0.01, n = 39

ΣDDT: TOTAL INTERSTITIAL WATER

LN TR = 2.434 + 0.957 LN (X) r^2 = 0.733, p<0.01, n = 34

DIELDRIN: BULK SEDIMENT

LN TR[b] = -0.354 + 1.089 LN (X) r^2 = 0.729, p<0.01, n = 36

DIELDRIN: CARBON-NORMALIZED SEDIMENT

LN TR[b] = -2.579 + 1.196 LN (X) r^2 = 0.731, p<0.01, n = 36

DIELDRIN: TOTAL INTERSTITIAL WATER

LN TR = 2.988 + 0.726 LN (X) r^2 = 0.892, p<0.01, n = 23

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a = In order to convert predicted 28-day tissue residues to steady-state, multiply the predicted 28-day values by 2.9 for ΣDDT and 1.7 for dieldrin (see Table 8-15).

b = Natural logs of tissue concentrations were determined from (concentrations + 1) to account for 0 (below detection limit) tissue concentrations.

TABLE 8-18. Accumulation factors determined from residues in Macoma nasuta in 28-day bioaccumulation test corrected to steady-state values.

STATION NO.	POLLUTANT							
	2,4'DDE	4,4'DDE	2,4'DDD	4,4'DDD	2,4'DDT	4,4'DDT	ΣDDT	DIELDRIN
#1 Mean	0.46	1.09	0.52	0.94	NA	0.06	0.49	1.23
SE	0.13	0.29	0.11	0.26		0.02	0.14	0.24
Range	0.15-0.92	0.52-2.15	0.30-0.92	0.41-1.89		0.01-0.15	0.23-1.01	0.54-1.85
N	5	5	5	5		5	5	5
#2 Mean	0.81	1.33	0.74	1.09	0.05	0.12	0.58	1.45
SE	0.42	0.29	0.22	0.19		0.06	0.20	0.29
Range	0.18-2.01	0.59-2.34	0.39-1.55	0.65-1.71		0.03-0.35	0.18-1.26	0.79-2.05
N	4	5	5	5	1	5	5	5
#3 Mean	0.68	2.03	0.99	1.98	0.05	0.22	1.35	1.66
SE	0.50	0.56	0.37	0.77		0.09	0.53	0.54
Range	0.25-1.68	0.96-3.94	0.19-2.24	0.58-4.78	0.02-0.08	0.05-0.57	0.44-3.26	0.40-3.22
N	3	5	5	5	2	5	5	5
#4 Mean	0.10	1.58	0.61	1.32	NA	0.31	0.90	1.17
SE		0.13	0.08	0.08		0.04	0.05	0.07
Range		1.22-1.90	0.38-0.84	1.09-1.53		0.16-0.42	0.75-1.01	1.07-1.38
N	1	5	5	5		5	5	4
#5 Mean	3.68	1.60	0.85	1.47	2.26	0.91	1.29	3.49
SE	2.90	0.25	0.36	0.49		0.15	0.40	1.16
Range	0.34-9.46	0.90-2.09	0.31-1.90	0.69-2.87		0.89-1.17	0.61-2.45	1.47-5.50
N	3	4	4	4	1	3	4	3
#6 Mean	0.53	0.74	0.30	0.88	0.06	1.16	0.70	0.68
SE	0.13	0.27	0.07	0.29		0.14	0.20	0.07
Range	0.37-0.79	0.29-1.51	0.11-0.41	0.32-1.70		0.92-1.39	0.19-1.16	0.60-0.82
N	3	4	4	4	1	3	4	3
#7 Mean	2.58	0.92	0.44	0.64	NA	0.84	0.61	0.95
SE	1.92	0.20	0.19	0.15		0.50	0.17	
Range	0.21-6.37	0.59-1.45	0.11-0.96	0.31-1.00	NA	0.24-1.83	0.30-0.96	0.50-1.40
N	3	4	4	4		3	4	2
#8 Mean	2.40	1.59	0.58	0.79	NA	2.18	0.81	NA
SE		0.34		0.16			0.14	
Range	1.21-3.59	1.00-2.44	0.48-0.68	0.53-1.20		1.24-3.13	0.49-1.06	
N	2	4	2	4		2	4	
#9 Mean	NA	2.45	1.13	1.43	NA	5.04	1.59	NA
SE		0.47		0.21			0.21	
Range		1.87-3.38		1.13-1.84		5.03-5.05	1.34-2.01	
N		3	1	3		2	3	
All Stations								
Mean	1.37	1.46	0.65	1.18	0.60	0.81	0.90	1.51
SE	0.45	0.13	0.08	0.13	0.55	0.22	0.10	0.22
Range	0.10-9.46	0.29-3.94	0.11-2.24	0.31-4.78	0.02-2.26	0.01-5.05	0.18-3.26	0.40-5.50
N	24	39	35	39	4	33	39	27
95% C.I.	0.43-2.30	1.20-1.71	0.49-0.82	0.92-1.44	0-2.36	0.36-1.26	0.69-1.10	1.07-1.95

TABLE 8-19. Mean ΣDDT and dieldrin tissue-residues (μg/kg, dry weight) in field-collected infauna. Grab and trawl collections are combined.

<DL = Below detection limit, Comp. = Samples consists of composite, B = Bivalve, C = Cerianthid, FF = Filter feeder, SDF = Surface-deposit feeder, P = Predator

SITE	SPECIES (TAXON/FEEDING TYPE)	N (Ind. or Composite)	ΣDDT	DIELDRIN
LAURITZEN				
	Macoma nasuta (B/SDF-FF)	7 (6 Ind., 1 Comp.)	30182.06	1012.69
	Musculus senhousia (B/FF)	7 (7 Comp.)	13980.70	308.89
	Pachycerianthus fimbriatus (C/P)	3[1] (1 Ind., 2 Comp.)	7117.67[1]	239.85[1]
	Tapes japonica (B/FF)	7 (7 Comp.)	4395.81	227.07
	AVERAGE LAURITZEN	24	15052.62	481.67
SANTA FE				
	Macoma nasuta (B/SDF-FF)	1 (1 Ind.)	1692.21	50.56
	Pachycerinthus fimbriatus (C/P)	3 (3 Ind.)	568.95	15.49
	Potamocorbula amurensis (B/FF)	2 (2 Comp.)	545.93	32.47
	AVERAGE SANTA FE	6	748.48	26.99
RICHMOND				
	Macoma nasuta (B/SDF-FF)	1 (1 Comp.)	718.91	<DL
	Potamocorbula amurensis (B/FF)	1 (1 Comp.)	59.34	<DL
	AVERAGE RICHMOND	2	389.13	<DL
FIELD AVERAGE		32	11454.13	366.31

1 = Excludes sample 475 3 0 S1CDT026 (ΣDDT = 405,822; Dieldrin = 2180)

TABLE 8-20. Mean accumulation factors in field-collected infauna. Grab and trawl collections are combined.

ND = no data, B = Bivalve, C = Cerianthid, FF = Filter feeder, SDF = Surface-deposit feeder, P = Predator

SITE	SPECIES (TAXON/FEEDING TYPE)	N ΣDDT/DIELDRIN	ΣDDT AF	DIELDRIN AF
LAURITZEN				
	Macoma nasuta (B/SDF-FF)	6/6	0.419	0.932
	Musculus senhousia (B/FF)	7/7	0.072	0.135
	Pachycerinthus fimbriatus (C/P)	3[1]/3[1]	0.077	0.222
	Tapes japonica (B/FF)	7/7	0.021	0.084
	MEAN LAURITZEN	23/23	0.148	0.339
SANTA FE				
	Macoma nasuta (B/SDF-FF)	1/1	1.014	2.297
	Pachycerinthus fimbriatus (C/P)	3/1	0.321	0.512
	Potamocorbula amurensis (B/FF)	2/1	0.112	0.461
	MEAN SANTA FE	6/3	0.366	1.092
RICHMOND				
	Macoma nasuta (B/SDF-FF)	1/0	2.50	ND
	Potamocorbula amurensis (B/FF)	1/0	0.823	ND
	MEAN RICHMOND	2/0	1.67	ND
FIELD AVERAGE - FILTER FEEDERS[2]		17/15	0.10	0.13
FIELD AVERAGE - DEPOSIT FEEDERS[3]		8/7	0.75	1.13
FIELD AVERAGE - ALL SPECIES[4]		31/26	0.289	0.426

1 = Sample 475 3 S1CDT026 excluded
2 = Mean of individual values for Potamorcorbula, Musculus and Tapes
3 = Mean of individual values for Macoma
4 = Mean of individual values for all species

TABLE 8-21. Comparisons of DDT tissue residues of Macoma nasuta exposed to sediment in a laboratory bioaccumulation test to residues of other organic compounds.

Values are mean of tissue (μg/kg dry weight) of measured tissue residues in clams exposed for 28-days. Data taken from White *et al.*, (1993). % of ΣDDT = (tissue residue/ΣDDT RESIDUE) * 100.

		COMPOUND				
Present Study Site	**Comparable Sediment[a]**	**Total PAH**	**PCB's[b]**	**Dieldrin**	**Other Pesticides[c]**	**Total DDT[d]**
Lauritzen Sites 1-4	Comp LC	5432	308	2065	843	35214
	% ΣDDT	15.4%	0.9%	5.9%	2.4%	
Santa Fe Site 5-6	Comp USFC	5571	181	34	5.1	726
	% ΣDDT	767%	25.0%	4.7%	0.7%	
Richmond Site 7	Comp 1HC-1	575	91	10.3	1.9	189
	% ΣDDT	305%	48.0%	5.4%	1.0%	

a = Sediment codes are from White *et al.* (1993) and represent composites of multiple cores that were taken near site listed in the present study.
b = PCB values were quantified as Aroclor 1254.
c = Sum of the mean tissue residues for Endrin, Aldrin, trans-Chlordane, cis-Chlordane, and Endosulfan.
d = Sum of 4,4' DDD, 4,4' DDE and 4,4' DDT

FIGURE 8-4. Flow chart for tissue samples.

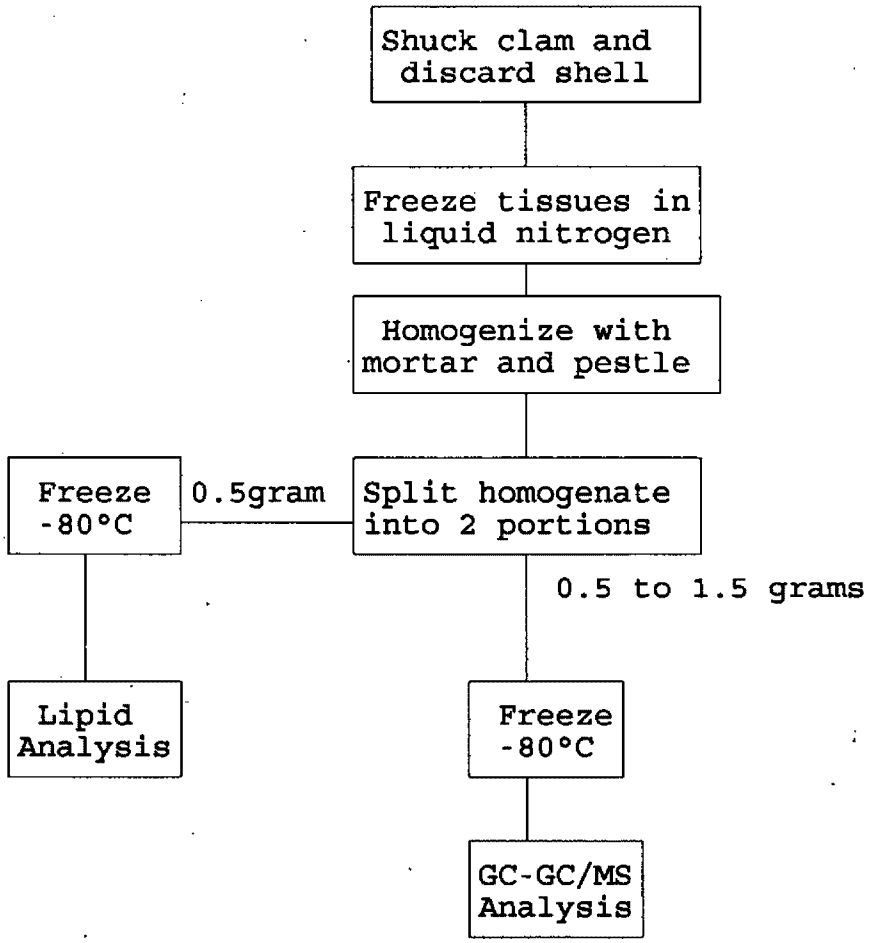

FIGURE 8-5. Macoma nasuta tissue uptake of ΣDDT versus laboratory exposure time to sediment collected from Lauritzen Channel (Station 1).

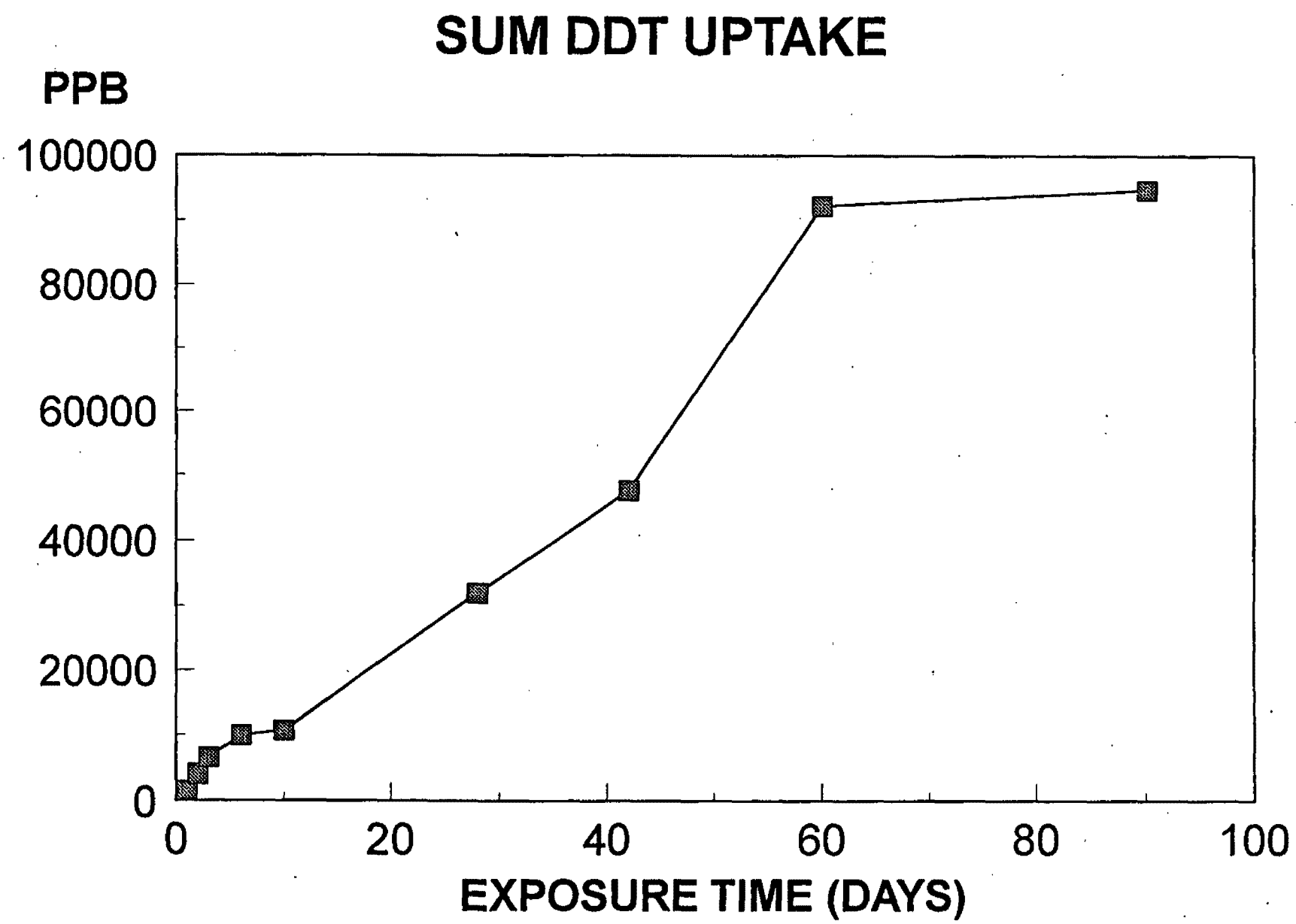
SUM DDT UPTAKE
PPB
100000
80000
60000
40000
20000
0
0
20
40
60
80
100
EXPOSURE TIME (DAYS)

FIGURE 8-6. Homologous tissue residues by station.

Station numbers are arranged in ascending order from lowest to highest mean pollutant concentration. Analysis was done using one-way ANOVA and a multiple range test (Scheffe method) on the natural log of 28-day tissue residues. Underlines indicate which stations are not significantly different.

Lowest Concentration → Highest Concentration

ΣDDT

0 9 8 7 5 6 4 1 2 3

4,4'-DDT

0 9 8 5 7 6 4 1 3 2

2,4'-DDT

0 8 9 6 1 5 2 7 4 3

4,4'-DDD

0 9 8 7 5 6 4 1 2 3

2,4'-DDD

0 9 8 7 5 6 4 2 1 3

4,4'-DDE

0 9 8 7 5 4 6 1 2 3

2,4'-DDE

0 9 8 5 7 4 6 2 3 1

Dieldrin

8 9 0 7 5 6 4 1 3 2

Note: Both stations 8 and 9 below detection limits for dieldrin.

8.4. BYSSAL MUSSELS, MEGAFAUNA, AND FISHES

8.4.1. Introduction

This section presents concentrations of ΣDDT and dieldrin measured in intertidal mussels, demersal fishes, megafauna (large epibenthic invertebrates such as crabs and shrimp), pelagic fishes, and surficial sediments of trawl transects. The objectives of this portion of the study were: (1) To document existing concentrations of ΣDDT and dieldrin in invertebrates and fishes occupying both the benthic zone and the overlying water column of the three channels and (2) To relate these concentrations to water and sediment concentrations in an effort to provide predictive capabilities. Specifically, the results are used to obtain field-derived estimates of bioconcentration factors to predict exposure from water and accumulation factors to predict exposure from sediment. The analysis focuses on ΣDDT rather than the individual metabolites because the Water Quality Criterion is based on the total DDT. This section also lists the mega-invertebrate, fish, bird, and mammal species observed at the sites. These observations help identify the larger invertebrate, fish, and wildlife species potentially at risk at these sites.

8.4.2. Methods

As described in Section 6, specimens of the byssal mediterranean mussel Mytilus galloprovincialis were collected from one site in each of the three channels on October 7, 1991 and on February 7, 1992. [Note: The taxonomy of Mytilus species is in dispute. Currently, electrophoretic and sophisticated morphometric analyses are required to resolve species. The Mytilus have been tentatively identified as M. galloprovincialis or a hybrid between M. galloprovincialis and M. trossulus. Dr. Paul Scott of the Santa Barbara Museum of Natural History has kindly reviewed specimens from our specimen archive, and agrees with that judgement, stating that all could be grouped under M. galloprovincialis for the purposes of this excercise (Scott, pers. comm., Appendix 8-4B; Coan *et al.*, in prep.).] Specimens approximately 4 to 6 cm in length were removed from the substrate within the intertidal zone ranging between points about 0.2 meter above to about 0.3 meter below the water surface. Generally, at least 30 mussels were obtained from each collection. Each specimen was rinsed in ambient water to remove as much sediment as practical, wrapped in pre-cleaned aluminum foil, and then placed in a pre-cleaned glass container with a Teflon-lined cap. The container then was placed in a metal cooler under dry ice, and returned frozen to the laboratory in Newport, Oregon following the Chain-of-Custody protocols. Following the October survey, Mytilus californianus were collected from the rocky intertidal zone of the central Oregon coast near Newport as control specimens.

All mussel specimens from a given site and date were measured, and those falling within the size range 4.0 to 6.0 cm in length were enumerated. Five specimens then were randomly

selected. The whole soft tissue of each mussel (minus byssal threads) was excised in a fume hood using solvent-rinsed glass cutting boards and metal dissection tools. Each sample then was transferred to a pre-cleaned glass container for homogenization and analysis of water content and DDT and dieldrin residues on the individual mussel tissue. An aliquot of each mussel homogenate also was analyzed for lipid concentration.

Bottom trawls were used to collect fishes and epibenthic megafauna. Trawls were conducted along one transect in each of the three study channels (Figures 6-1 and 6-2) during both the October and February surveys. As with the mussels, specimens were rinsed in ambient seawater to remove sediment, wrapped in pre-cleaned aluminum foil, and then placed in pre-cleaned glass containers with Teflon-lined caps (except for the foil-wrapped Cancer crab specimens, which were too large for the glass containers). These containers were placed in metal coolers under dry ice, and returned frozen to the laboratory in Newport, Oregon following the Chain-of-Custody protocols. The taxa were identified and specimens dissected (or selected whole) as described above.

For all specimens judged to be of sufficient size (wet weight exceeding 1 gram), the tissues were separated into two samples - edible tissue and remainder; otherwise, the entire specimen was taken for analysis. For two organisms, the bay shrimp Crangon franciscorum and the shiner surfperch Cymatogaster aggregatus, both fractions were analyzed, and the resultant wet weight concentrations were combined with the corresponding fractions of the whole body weights to obtain whole body wet weight concentrations of ΣDDT and dieldrin. In these cases, lipid content was determined only on the edible tissue portion.

Bioconcentration factors (BCFs) were used to predict uptake from water in pelagic fishes and mussels. Site specific BCF_{uw}s were calculated by:

BCF_{uw} = wet tissue residue/whole unfilt. water conc. Eq. 8-6

These were obtained by first calculating individual mussel BCF_{uw} values relative to the appropriate average water value (based on the six whole water samples collected from the site during the two surveys, Table 8-12), and then determining median and average BCF values for a given survey and site. These values are designated BCF_{uw} to indicate that they are based on unfiltered water concentrations. The difference between a BCF_{uw} and a BCF based on filtered or free water concentrations will depend upon the contribution of the particulate-bound phase of the whole water sample, which in turn depends upon the extent of sorption of the compound, the particulate load of the water, and bioavailability of the sorbed versus the dissolved pollutant. If there is a significant contribution from the particulate phase, the BCF_{uw} will be lower than a BCF based on free water concentrations. As discussed above (Section 8.2.4.), the bioconcentration factors obtained from the mussel tissue and whole water concen-

trations in the study are higher, not lower, than those used to derive the WQC values for ΣDDT and dieldrin. These results indicate that the contribution to a whole water concentration from the particulate phase of the whole water samples was not of primary importance, as it led to no measurable decrease or negative bias in the empirical BCF_{uw} values relative to the WQC literature benchmarks. When comparing the bioconcentration factor values from this study to literature BCFs it is also important to note that the BCF_{uw} values reported here are based on wet tissue residues and, in some cases, on muscle rather than whole body residues.

The equilibrium partitioning bioaccumulation model was used to predict uptake of sediment contaminants in demersal organisms. The accumulation factors (AFs) for demersal organisms were calculated using Equation 8-4 and the estimates of ΣDDT and dieldrin sediment concentrations for a trawl. To obtain estimates of the surficial sediment exposure concentrations for the trawl-caught organisms, the sediment concentrations were averaged for any stations that a trawl covered as well as any stations immediately bracketing the trawls. The stations averaged for each site were: Stations 1, 2, and 3 for the Lauritzen Channel trawls; Stations 13, 5, 6, and 14 for the Santa Fe Channel trawls; and Stations 17, 8, and 18 for the Richmond Inner Harbor Channel trawls.

8.4.3. Results

Table 8-22 lists the species observed during sampling. The invertebrates include both the epifaunal megainvertebrates as well as some larger infaunal invertebrates dug up by the trawls. (The smaller infaunal species sieved from the grab samples are discussed in Section 9.1. and listed in Appendix 9-1 and 9-2.) The Mytilus galloprovincialis were observed on hard surfaces in each of the channels while the fish and large crustacean species were observed in the trawls. The birds, marine mammals, and human activities were observed from ship or shore during the October, 1991 and February, 1992 surveys. Appendix 4-1 lists the birds observed during two brief sampling trips in November and December, 1993 to assess the presence of birds in the more contaminated sections of Richmond Harbor. The feeding types and expected mobility of the fish and megafauna captured in the trawls are listed in Table 8-23. "Resident" is used in this report to indicate species that have a confined range and largely limited to one channel. "Non-resident" refers to species that have a larger range and are expected to swim between the channels and perhaps out of Richmond Harbor into San Francisco Bay.

Appendix 8-8 lists the individual residue and lipid values for all the mussels on a dry weight basis. Median and average (± S.E.) values for the wet weight concentrations of ΣDDT and dieldrin obtained from each site and survey are listed in Table 8-24A, as well as combined 1991-92 values for each site. Average (± S.E.) values for lipid content (as percent of wet weight) and decimal fractions of dry-to-wet weight ratios for each site and survey also are presented. Table 8-24B gives average values for

the percent contribution to the ΣDDT concentrations in the mussel samples by the six DDT residues. Table 8-25 lists corresponding values for the field-determined, whole water bioconcentration factors (BCF_{uw}). Combined survey (1991-92) median and average BCF_{uw} values for a given site also are listed.

Appendix 8-9 lists the individual values for all the megafauna and fishes. Median and average (± S.E.) values for the wet weight concentrations of ΣDDT and dieldrin (and average percent lipid values) are listed in Table 8-26A, with overall average dry/wet weight tissue ratios to allow the conversion into dry weight residues. Average values for the percent contribution to the ΣDDT values for these biota by the six DDT residues are listed in Table 8-26B, while Table 8-26C lists the minimum average concentrations of ΣDDT and dieldrin based on 25-fish samples in each of the channels. For those fishes principally inhabiting the water column, bioconcentration factors were calculated employing the same approach as was with Mytilus BCF_{uw} (Table 8-27).

Table 8-28A lists the median and average (± S.E.) values for the sediment dry weight concentrations of ΣDDT, dieldrin, and TOC for the sediment stations estimated to best represent the trawl transects. Carbon-normalized values for each of the three site transects also are listed. Table 8-28B lists average values for the percent contribution to the ΣDDT values by the six DDT residue in these sediments. Table 8-29 lists the median and average (± S.E.) values for the accumulation factors from the equilibrium partitioning bioaccumulation model (see Equations 8-4 and 8-5). These AF values were obtained by first normalizing a given individual benthic megafauna wet weight concentration to the wet lipid content of that sample, and then dividing this value by the corresponding average TOC normalized sediment concentration for the stations representative of the trawl transect (Table 8-28A). Table 8-32 gives the overall average AFs for several combined taxa (error values for the numerator and denominator were not propagated to the resultant quotient).

Table 8-30 summarizes the average ΣDDT sediment concentrations at the three trawl sites, ΣDDT water concentrations in the three channels, and residues in species primarily exposed to water contaminants (mussels and shiner surfperch) and sediment contaminants (gobies). Table 8-31 summarizes the same information for dieldrin. Estimates of within site and among-season variation for the summarized data are given in their respective tables in Sections 8.2. and 8.4.

8.4.4. Discussion

Species Present and Food Web Relationships: A wide variety of organisms were observed during sampling, including several higher trophic level predators (Table 8-22). The main prey of the fish and megafauna captured in the trawls (Table 8-23) were estimated from the investigator's general knowledge of the species or similar species. The average positions of these prey in the food web were estimated using classical trophic level

concepts (Odum, 1971; Mearns, 1982). Assuming that each prey taxa contributed approximately equally to a given predator's diet, an average position in the food web for that diet was obtained. This number is called the trophic level assignment (TLA) of the predator's diet (Mearns, 1982). The TLA of the predator itself is defined as 1.0 unit higher than the TLA of its diet. The resulting TLA values are similar to previous independent trophic level assignments made by Dr. Alan Mearns for the same or similar species (in Young, 1988), which lends credence to our assignments.

With the exception of the anchoveta, all the fish and megafauna feed partially or primarily on benthic invertebrates. This trophic link establishes a direct mechanism by which sediment contaminants can be transported to higher trophic levels. Several species, such as the flatfish and gobies, also have direct physical contact with sediment, which is another route by which sediment contaminants could accumulate in these higher trophic level organisms. Flux of sediment contaminants into the water column would be the primary method of exposure for anchoveta.

Several of the larger fish, as well as Cancer crabs, feed on other fish or megafauna. In such cases, these higher trophic level predators would be indirectly exposed to sediment associated contaminants even if they did not feed on infaunal invertebrates or have contact with the sediment. Because DDT is known to biomagnify (Woodwell *et al.*, 1967; Risebrough, 1969; Wurster, 1969; Macek *et.al.*, 1979; Mearns and Young, 1980; Young, 1982, 1988; Kucklick *et al.*, 1994), the resultant residues would be greater than for other neutral organic pollutants that do not show this tendency (*e.g.*, PAHs). In the case of non-resident predators, their feeding in contaminated sites and subsequent migration would be a mechanism of transporting ΣDDT and dieldrin into San Francisco Bay proper. Although the actual mass of pollutant moved through this biologically-mediated mechanism may be small compared to physical fluxes, it is important to recognize that the pollutant is in a bioavailable form (Swartz and Lee, 1980).

The fish- and invertebrate-consuming birds and marine mammals observed during the October and February surveys were Brown pelicans, curlew, cormorants, seagulls, and Harbor seals. Brown pelicans, seagulls, and the curlew were all observed feeding. Harbor seals were observed in all three channels, and the one observed diving in Santa Fe may have been feeding. A fisherman was also observed in the Lauritzen Channel, as well as in the adjoining Parr Canal. The two trips in November and December, 1993 also recorded a large number of bird species (Appendix 4-1). These observations confirm that the fish and megafauna in the Lauritzen/Santa Fe/Richmond system are a food source for higher trophic levels, including top predators. Therefore, tissue residues in the prey of these higher trophic level species are an ecologically relevant measure of exposure and the resulting impacts on both birds and marine mammals, as well as of potential human exposure.

Mussel Residues: The Mytilus galloprovincialis soft tissue average wet weight concentrations of ΣDDT and dieldrin (Table 8-24A) for the October survey were substantially (although not significantly) higher than those obtained for the February survey. However, the average lipid content values also were two to three times higher for the October samples. On a lipid-normalized basis, the average survey values of ΣDDT or dieldrin, for all three channels, generally agreed within a factor of about three and were not significantly different. For ΣDDT, the wet weight values span a three order-of-magnitude difference between the average (± S.E.) concentration measured for the Lauritzen Channel specimens (2,900 ± 1,100 ppb) and that obtained for the central Oregon coast control specimens (1.4 ± 0.1 ppb). Corresponding values for dieldrin (97 ± 36 ppb vs. <1.7 ppb) yield a two order-of-magnitude difference, at the least. Average concentrations of the two pesticides in the mussels fell by factors of about 72 and 24, respectively, between the Lauritzen and Richmond Channel sites. These results document the extensive degree of contamination of the Lauritzen Channel ecosystem by these two pollutants.

The combined-survey average concentrations of the residues in the mussels did not exceed the FDA action level for fish for either DDT or dieldrin in any of the channels, although this level for ΣDDT (5,000 μg/kg wet weight) was exceeded by the average for the specimens collected in October 1991 from the Lauritzen Channel. However, on average the residues exceeded the NAS standard for DDT to protect fish-eating birds (50 μg/kg wet) by 58-fold in Lauritzen Channel and 7-fold in Santa Fe Channel, as well as the NAS standard for dieldrin (5 μg/kg, wet) by about 19-fold in Lauritzen and 4-fold in Santa Fe. The mussel dieldrin residue in Richmond Inner Harbor Channel was slightly below the standard. Mussels are consumed by various shorebirds, including oystercatchers, surf scoters, and white-winged scoters (Shaw et. al, 1988), of which the latter two are abundant in the Richmond Harbor area (Table 4-1). Therefore, there is a direct ecological link between DDT and dieldrin contamination in mussels and exposure to shorebirds. Because of the magnitude of the exceedance, mussels from Lauritzen and/or Santa Fe Channels would only have to constitute a relatively small portion of these shorebirds' diets to result in a exposure equal to or greater than that of birds feeding 100% of the time on prey at the NAS standards.

The mean wet weight concentrations of ΣDDT and dieldrin in mussel soft tissue from the Lauritzen Channel 1991-92 specimens are similar to the values reported for a sample of "resident bay mussels" (identified then as Mytilus edulis) collected in March 1986 near this site by the California State Mussel Watch Program (State Water Resources Control Board, 1988)(but see Section 8.4.2.). The respective values for ΣDDT are 2,900 (± 1,100) μg/kg in this study versus 2,831 μg/kg in the Mussel Watch Program. For dieldrin, the corresponding values are 97 (± 36) μg/kg versus 176 μg/kg for this study and Mussel Watch, respectively. Although the database is small, this comparison indicates

that no ecologically significant decrease in ΣDDT or dieldrin contamination in M. galloprovincialis has occurred in Lauritzen Channel over the 5 - 6 year interval.

Mussel Bioconcentration Factors: Neither the average wet weight tissue/whole water bioconcentration factors (BCF_{uw}) for ΣDDT nor for dieldrin are significantly different among the three channels (Table 8-25). The overall mean of the three average BCF_{uw} values for ΣDDT is 46,000 ± 5,800. This value falls within the range of BCF values for DDT (1,200 to 76,300) reported for saltwater species (Section 5.1.2.). The corresponding overall mean BCF_{uw} for dieldrin, 8,200 ± 2,800, is in agreement with the upper end of the range (400 - 8,000) reported for fish or shellfish (Section 5.1.3.).

On a wet weight basis, the overall mean of the six channel-survey average percent lipid values for the mussel soft tissues was 1.3 ± 0.3 percent. This corresponds to average percent lipid-normalized BCF_{uw} values for ΣDDT and dieldrin of about 35,400 and 6,300, respectively. [NOTE: These values are normalized to lipids based on division by the percent lipid (1.3%) rather than on the decimal fraction of lipids (0.013) for consistency with Section 5. If they were normalized on a decimal fraction, the values would be 100-fold greater, and would have the units of g-water/g-lipid.] These results are in reasonable agreement with the values cited in Sections 5.1.2. and 5.1.3. for the geometric mean of percent lipid-normalized BCF values for DDT (17,870) and dieldrin (1,557).

Residues in Benthic Megafauna and Fishes: The steep spatial gradient in pesticide residues observed in the sessile filter-feeding mussel also was observed in the epibenthic gobiid fish (probably Lepidogobius lepidus, the bay goby) and the pelagic (though benthic feeding) shiner surfperch Cymatogaster (Table 8-26A). Both fish species, which are believed to be resident, as discussed below, were collected from all three trawl transects. Average whole body concentrations obtained for these two resident fish species, mussels, surficial sediment, and overlying water in the three sampling areas are summarized for ΣDDT in Table 8-30 and dieldrin in Table 8-31.

The average whole body concentrations for ΣDDT and dieldrin in surfperch in the Lauritzen Channel specimens (7,500 ± 1,500 and 390 ± 90 μg/kg wet wt., respectively) exceed the interstate commerce limits of 5,000 and 300 μg/kg wet wt., respectively, established for seafoods by the U.S. Food and Drug Administration (See Section 5.2.1.). The whole body concentration in the single goby specimen from Lauritzen Channel (5,400 μg/kg wet wt.) also exceeded the FDA action level. Although it is not likely that Lauritzen Channel surfperch and gobies are part of interstate commerce, this finding is relevant in view of our observations that recreational and/or subsistence fishing takes place in Lauritzen Channel, as well as the neighboring Parr Canal.

The shiner surfperch, as a water column species, is a potential prey item for pelicans and other fish-eating birds; thus, the NAS standards are ecologically relevant measures of

potential effects on birds. ΣDDT residues in whole surfperch exceed the NAS standards by orders-of-magnitudes in two of the sites - 150-fold and 18-fold in the Lauritzen and Santa Fe Channels, respectively - and by 2-fold in Richmond Inner Harbor Channel. The level of exceedance for dieldrin is about half that for DDT, with exceedances of 78-fold and 8-fold in Lauritzen and Santa Fe, respectively. The dieldrin standard is not exceeded in the Richmond Inner Harbor Channel.

One limitation of the comparison to the NAS standards is that our comparison is based on fewer than 25 individuals whereas NAS (1972) recommends 25 or more. To address this issue, the whole body fish concentrations from each channel and habitat (*i.e.*, benthic or pelagic) were combined to calculate a corresponding projected lower limit average whole body residue for a group of 25 individual fish specimens, of equivalent composition regarding fish species and size to those sampled. The procedure used is as follows: For a given species, the average whole body concentration was multiplied by the total number of individuals ("i" in Table 8-26A) that contributed to the number of samples (n) to obtain the combined average value. The resultant values for each fish species sampled from the channel and habitat then were summed , and the total was divided by 25 to obtain the projected minimum average whole body concentration for 25 individual fish specimens of equivalent species composition and size. Note that this procedure assumes a zero pollutant concentration for the balance of the fish specimens that would have been needed to obtain a total of 25 individual fish. Therefore, the projected 25-fish average is an absolute minimum residue.

The results (Table 8-26C) show that, for ΣDDT, the projected minimum average whole body concentration for pelagic fish from Lauritzen, Santa Fe, and Richmond Inner Harbor Channels are 1504, 171, and 54 μg/kg weight wet, respectively. Corresponding values for benthic fishes are 216, 246, and 30 μg/kg (note that the first value is based on only one goby specimen that contained 5,400 μg/kg ΣDDT). In five of the six cases, the projected minimum average whole body ΣDDT concentration for 25 fish exceeds the NAS standard of 50 μg/kg. For dieldrin, the projected minimum average whole body concentrations from the three channels are 79, 5, and 1 μg/kg wet weight, while corresponding values for benthic fishes are 8, 7 and 1 μg/kg (again, the first value is based on the concentration of 1 fish divided by 25). In four of these six cases, the projected minimum average whole body concentration exceeds the NAS dieldrin standard of 5 μg/kg. Overall, in nine of twelve cases, the NAS standards were exceeded for the target pesticides even when using an approach that generates the lowest possible average residue for a 25 fish sample.

As was the case for the intertidal mussel tissues, the results for the fish specimens collected from Lauritzen Channel in 1991-92 are consistent with those for fishes sampled there in 1986 (summarized in Section 3.1.2.). The mean ΣDDT value

obtained for muscle tissue of the shiner surfperch (Cymatogaster aggregatus) is 2,000 (± 410) μg/kg wet weight, compared to values ranging up to 1,400 μg/kg wet weight reported for muscle tissue of white croaker (Genyonemus lineatus) collected there in 1986 (Cal. Dept. Fish & Game, 1986). Similarly, the mean whole body ΣDDT concentration for the 1991-92 Lauritzen Channel shiner surfperch specimens, 7,500 (± 1,500) μg/kg wet weight, is similar to the value of 13,600 μg/kg wet weight obtained for a whole shiner surfperch collected there in 1986 (ibid.). Again, there is no evidence that ΣDDT levels in the biota of Lauritzen Channel has changed substantially over this 5-6 year interval.

Bioconcentration Factors for Pelagic Fishes: A BCF is an ecologically meaningful measure only if the individuals are exposed to a relatively constant water exposure of the pollutant for a period sufficient to approach steady-state residues. (The BCF approach also assumes that the water is the predominant source of uptake.) As discussed in Section 8.2., overlying water concentrations appear relatively stable in the three channels. However, non-resident organisms swimming among the sites would be exposed to a varying pollutant concentration. The BCFs of such mobile species should show less correspondence to the local water concentrations and greater variation among individuals and sites.

The average BCF_{uw} values for ΣDDT in muscle tissue of the shiner surfperch, Cymatogaster, show a relatively small within-site variation. In terms of among-site variation, the averages vary by less than 2-fold among the three sites (Table 8-27) and the overall mean of 36,000 (± 5,900) of the three sites has a relative standard error (RSE = S.E./mean) of only 16 percent. Similar precision is obtained for the whole body BCF_{uw} values for ΣDDT, which have an overall mean of 119,000 ± 16,000 and an RSE of only 13 percent. In view of the 50-fold gradient in average water concentration of ΣDDT over the three sites, this among-site similarity is a strong indication of the resident nature of this pelagic species. Additionally, there was no indication of a strong seasonal variation as the October and February muscle tissue values in Santa Fe Channel varied by less than 2-fold. Dieldrin BCF_{uw}s showed the same consistency both between the Lauritzen and Santa Fe Channels and seasonally. The muscle tissue and whole body BCF_{uw} values had overall means of about 8,000 ± 1,200 (RSE = 15 percent) and 22,000 ± 200 (RSE = 1 percent), respectively, and the October and February muscle tissue BCF_{uw} values again agreed within a factor of 2.

The low variation in BCF_{uw}s for both compounds within-sites, among-sites, and seasonally indicates that although the data set was relatively small, these empirically derived BCF_{uw}s should be predictive of residues in similar sized shiner surfperch to within about 2-fold.

Further confidence in this approach is provided by a comparison of the BCF_{uw} values for the shiner surfperch muscle and the Mytilus whole soft tissue samples. The respective overall average (± S.E.) lipid concentrations (wet weight basis) for these two tissue classes, 2.0 ± 0.2 and 1.3 ± 0.3 percent,

are not significantly different, rendering such a comparison meaningful. For ΣDDT, the average BCF_{uw} values for the surfperch and mussel are 36,000 ± 5,900 and 46,000 ± 5,800, respectively. The corresponding values for dieldrin are 8,000 ± 1,200 and 8,200 ± 2,800. The good agreement obtained lends support to the hypothesis that the principal uptake route of ΣDDT and dieldrin to the surfperch is via the water, and the BCF_{uw} approach provides a useful predictive capability.

In contrast to the shiner surfperch, the BCF_{uw} values for whole anchovy, a species expected to move in and out of the channels, showed a wide variation among sites, with average values of 2,000, 78,000, and 17,000, respectively. Although the overall mean of these average BCF_{uw}s (32,000 ± 23,000) is similar to that of the surfperch, the anchovy's large RSE of about 70 percent and lack of correlation with water or sediment concentrations is indicative of this species' highly mobile nature.

The muscle tissue sample of the single white croaker Genyonemus from Richmond Inner Harbor Channel had a BCF_{uw} of 44,000, similar to the three-channel average (36,000) for the shiner surfperch. The two muscle samples of striped bass (Morone) from Lauritzen Channel had an average BCF_{uw} of 6,700 (± 3,600). The fact that this value is well below that of the resident surfperch suggests that the bass had spent the majority of their lives in less contaminated waters. This hypothesis is supported by the relatively low BCF_{uw} for dieldrin (1,400 ± 820) in the bass compared to the surfperch's average for Lauritzen Channel (6,700 ± 1,400) and Santa Fe Channel (9,200 ± 1,200). In general, the wide range of BCF values for the anchovy, white croaker, and striped bass may well reflect the variability caused by the mobility of such fishes.

Accumulation Factors for Demersal Fishes and Megafauna: The equilibrium partitioning bioaccumulation model (see Section 8.3.) can be applied to demersal fishes and epibenthic invertebrates that have regular, direct contact with the sediment. As with the BCF_{uw}, there is an implicit assumption of a relatively constant exposure. Levels of ΣDDT and dieldrin sediment contamination are essentially constant over the life span of a demersal fish or benthic invertebrate. However, as with the water exposure, movement along the contamination gradient would result in a varying exposure.

The field-derived accumulation factors for individual taxa (Table 8-29) are generally less consistent than the BCF_{uw}s. A likely contributing cause is the steeper concentration gradient in sediment than water (see Tables 8-30 and 8-31). With this steeper concentration gradient, small movements (*e.g.*, 10-100 m) by demersal organisms would result in greater exposure differences than equivalent movements by pelagic organisms, so that relating residues to an average channel sediment concentration would introduce greater variation than using a corresponding average water concentration. In addition, with two exceptions only three or fewer samples of each species were analyzed for any site and season.

For these reasons, it is more meaningful to average over the entire region rather than seek resolution among sites. The demersal fish and megafauna were separated into four classes each containing at least four AF values: shrimp (Crangon) muscle, crab muscle, whole juvenile flatfish, and whole gobiid fish (Table 8-32). For six of the eight taxa-compound AF values, the 95 percent confidence interval includes zero, indicating that the variation is too great to allow precise predictions. The shrimp demonstrate the most variable AFs of the four taxa. Only for ΣDDT in crab muscle and whole gobiid fish tissues are the average AF values significantly above zero.

One approach to predicting the general trends of sediment-associated uptake in fish/megafauna is to use the gobies or crabs as "indicators". That is, to use the AFs from taxa that best fit the assumptions of the model (*e.g.*, are not highly mobile and have constant contact with the sediment) and that have reasonably tight confidence limits as indicators of the accumulation of sediment-associated contaminants. Of the two taxa, the gobies are the better candidate indicator because they are less mobile than crabs. As mentioned, the goby ΣDDT AF, 0.63, is significantly above zero and therefore should be a reasonably precise predictor of ΣDDT accumulation in this species. Although the goby dieldrin AF, 1.9, is not significantly different than 0, it is possible to use the mean value as a general indicator of the accumulation trends.

Another approach to generating an overall prediction of general tendencies is to average the three taxa, excluding the gobies. The resultant mean of the three average AF values for ΣDDT is 1.2 ± 0.7, which is not significantly different from the mean value of 0.63 for the goby. For dieldrin, the three-taxa mean is 5.2 ± 3.8, which again does not differ significantly from the goby mean (1.9). Thus, these combined means agree with those of the goby within factors of about 2 and 3, respectively. They also agree, within factors of about 1.5 and 3.5, with the overall values determined for M. nasuta in the laboratory exposures (Table 8-18) and show the same trend with dieldrin having a larger AF than ΣDDT. (It should be noted that AF values generally are based on whole organism concentrations; thus, the use of AF values based on muscle or other tissue concentrations assumes that the pollutant is distributed among organs based solely on lipid content).

These indicator and combined species values can be compared to the theoretical equilibrium value (1.7) predicted for infaunal organisms by McFarland and Clark (1986). The goby ΣDDT value is within a factor of 3 of the predicted value while the dieldrin value approximately equals the calculated value. The combined species AFs agree with the theoretical value within factors of about 1.5 and 3, respectively.

The indicator and combined species AFs for ΣDDT can also be compared to AFs for Dover sole Microstomus pacificus collected over very large sediment gradients of 4,4'-DDE (p,p'-DDE) and PCB 1254 in the Southern California Bight (Young *et al.*, 1991).

Those researchers found that the AF fugacity model can successfully predict tissue concentrations of these high Kow (5.8 - 6.5) neutral organic contaminants. Specifically, the overall average AF values for 4,4'-DDE in muscle and liver tissue of specimens from five highly contaminated stations off Los Angeles were 1.4 ± 0.2 and 1.7 ± 0.4, respectively, compared to the whole body goby means of 1.3 ± 0.3 for 4,4'-DDE and 0.63 ± 0.18 for ΣDDT. The critical factors appears to be the ability to obtain sufficiently large numbers of samples of epibenthic organisms which are relatively non-mobile and which are in contact with the bottom sediment.

The general agreement between the indicator species AFs and the composite species AFs, field-derived AFs in fish/megafauna and laboratory-exposed infauna, the field-derived AFs and the theoretical AF value, and the DDE AF in Southern California and Richmond Harbor fish/megafauna all indicate that the equilibrium partitioning bioaccumulation model has sufficient precision to predict the general trends of accumulation in fish/megafauna with direct sediment contact. The various empirical and theoretical approaches generate AF values clustering around 1.2 for ΣDDT and 1.8 for dieldrin. In general, the most precise predictions should be for the goby, the indicator species, and the least for the shrimp.

Other Chlorinated Hydrocarbons: The results of the California Mussel Watch Program (State Water Resources Control Board, 1988) indicate that in March 1986 mean concentrations (μg/kg wet wt.) of other chlorinated pesticides and polychlorinated biphenyl mixtures (PCB) in intertidal Mytilus in Lauritzen Channel were substantially lower than those for ΣDDT (2,800) and, in most cases, dieldrin (180). Values observed for other chlorinated hydrocarbons were: aldrin (13); Σchlordane (55); trans-nonachlor (12); endrin (15); HCH-alpha (1); HCH-delta (1); heptachlor (2); hexachlorobenzene (1); ΣPCB (140); toxaphene (110). Other target compounds were below 1 μg/kg wet weight or were not detected. These results further confirm the dominant nature of the DDT and dieldrin contamination at this site.

TABLE 8-22. Species observed at the study sites, October, 1991 and February, 1992.

Location \ Species	Lauritzen Channel	Santa Fe Channel	Richmond Harbor Channel
Infaunal and Epifaunal Bivalves	Tapes japonica Macoma nasuta M. balthica Musculus senhousia Theora lubrica Potamocorbula amurensis Mytilus galloprovincialis* Clinocardium nuttalli	Tapes japonica Macoma nasuta Musculus senhousia Theora lubrica Potamocorbula amurensis Mytilus galloprovincialis* Clinocardium nuttalli Pododesmus cepio	Tapes japonica Macoma nasuta Theora lubrica Potamocorbula amurensis Mytilus galloprovincialis*
Crustacea	Cancer antennarius Crangon franciscorum	Crangon franciscorum Pyromaia tuberculatus tuberculatus Paleamon macrodactylus Heptacarpus sp.	Cancer antennarius Crangon franciscorum (~ 8 % C. f. parasitized by Argeia pauperata) Pyromaia tuberculatus tuberculatus Paleamon macrodactylus Heptacarpus sp. Hemigrapsis sp.
Miscellaneous invertebrates	Pachycerianthus fimbriatus Asychis elongata Misc. polychaetes (NOI)	Pachycerianthus fimbriatus Misc. polychaetes (NOI) Tunicates, sponges on wood splinter (NOI)	Asychis elongata Glycinde cf. armigera Misc. polychaetes (NOI) Brittle stars @ Stn 21 (NOI)
Fish	Morone saxitalis Cymatogaster aggregata Engraulis mordax Goby (prob. Lepidogobius lepidus, the bay goby)	Cymatogaster aggregata Engraulis mordax Leptocottus armatus Parophrys vetulus Goby (prob. Lepidogobius lepidus, the bay goby) Shark, juv. (NOI),	Cymatogaster aggregata Engraulis mordax Genyonemus lineatus "Sand sole" (NOI) "Speckled sand dab" (NOI) Goby (prob. Lepidogobius lepidus, the bay goby) Pipefish (NOI)
Birds	Seagulls	Curlew feeding in swash near Stn 7. Cormorant flying through. Seagulls on surface eating @ Stn 6. Brown pelican flying ~Stn 5 -> 6. Brown pelican at rest on buoy ~Stn 7.	Cormorant on water. Brown pelican on water near Stn 7, eating catch.
Marine mammals	Harbor seal near Stn 3->4	Harbor seal diving off Stn 6	Harbor seal off Stn 8
Human activities	Fishing off pier with rod and reel, west shore LC @Stn2.	Marina, leisure sailing, boating	Shipping

* Taxonomy confirmed (Scott, pers. comm.); Mytilus species concepts currently under revision, see Section 8.4.2..
NOI = Not otherwise identified

TABLE 8-23. Estimated diet, trophic levels (TL), and mobility of the fishes and megafauna collected in the trawls.

Trophic level of fish and megafauna based on the trophic level of their estimated diet. Trophic levels of Reference Consumers are independent trophic level estimates for similar fish and megafauna from the Southern California Bight (by Dr. A. Mearns in Young, 1988). Also provided are estimated classifications regarding the "resident" or "non-resident" nature of the species.

Consumer/ Size	Diet Taxa	Diet TL	Diet TL	Consumer TL	Res/ Non-Res	Reference Consumer	Reference Consumer TL
FISH/MEGAFAUNA							
Crangon	Detritus	1.5	2.0	3.0	Res?	Prawn	3.5
Shrimp	Benth. Inv.[1]	2.5					
3-5 cm	Copepods	2.0					
Cancer	Mollusks[2]	2.25	2.5	3.5	Non-Res	Crab	3.5
Crab	Benth. Inv.[1]	2.5					
7-12 cm	Smelt	2.5					
	Flatfish	3.25					
Gobiid	Benth. Inv.[1]	2.5	2.25	3.25	Res	Mudsucker	3.5
Fish	Copepods	2.0					
4-6 cm							
Pleuronec.	Benth. Inv.[1]	2.5	2.25	3.25	Res	Dover Sole	3.5
Engl.Sole	Copepods	2.0					
4 cm							
Platichth.	Polychaetes	2.25	2.25	3.25	Res	Dover Sole	3.5
Starry Fl.	Mollusks[1]	2.25					
9 cm	Clam siph.	2.0					
Citharich.	Crustacea	2.0	2.50	3.5	Res	Sanddab	3.75
Sp.Sanddab	Pelag.Fish	3.25					
8 cm	Tube Orgs.	2.75					
	Gammariids	2.5					
	Mysiids	2.5					
	Polychaetes	2.25					
Genyonemus	Benth.Inv.1	2.5	2.75	3.75	Non-Res	Wh. Croak.	3.7
Wh.Croaker	Gammariids	2.5				Yel.Croak.	3.7
7-11 cm	Crabs	3.5					
	Shrimp	3.0					
Cymatogast.	Copepods	2.0	2.25	3.25	Res	Top Smelt	3.3
Surfperch	Gammariids	2.5				Croakers	3.7
6-9 cm	Foul. Orgs	2.5					

TABLE 8-23 (cont'd.). Estimated diet, trophic levels (TL), and mobility of the fishes and megafauna collected in trawls.

Consumer/ Size	Diet Taxa	Diet TL	Consumer TL		Res/ Non-Res	Reference Consumer	TL
Anchoveta Anchovy 5-7 cm	Copepods	2.0	1.75	2.75	Non-Res	Anchovy	2.8
	Zooplank.	2.0					
	Phytopl.	1.0					
	Larvae	2.0					
Morone Str. Bass 7-10 cm	Mysiids	2.0	2.5	3.5	Non-Res	Topsmelt	3.3
	Shrimp	3.0					
	Small Fish	3.0					
	Amphipods	2.5					
	Sanddabs	3.5					
INFAUNA							
Macoma Bivalve Mollusk	Deposit & Suspension Feeder	1.0	2.0		Res		
Musculus Bivalve Mollusk	Suspension Feeder	1.0	2.0		Res		
Tapes Bivalve Mollusk	Suspension Feeder	1.0	2.0		Res		
Potamocor. Bivalve Mollusk	Suspension Feeder	1.0	2.0		Res		
Pachyceri. Burrowing Anemone	Suspension Feeder/Pred.	2.0	3.0		Res		

1 = Benthic invertebrates include predaceous polychaetes and other trophic level 3 infauna as well as deposit-feeding or filter-feeding invertebrates.

2 = Mollusks includes predaceous snails.

TABLE 8-24A. ΣDDT and dieldrin in Mytilus galloprovincialis soft tissues (μg/kg wet weight)*.

Site	ΣDDT Oct.	ΣDDT Feb.	ΣDDT Combined	Dieldrin Oct.	Dieldrin Feb.	Dieldrin Combined
Lauritzen (Mytilus galloprovincialis)						
Median	2,600	670	1,600	100	23	57
Average	5,100	680	2,900	170	25	97
Std.Err.	1,700	150	1,100	56	3	36
(n)	5	5	10	5	5	10
Santa Fe (M. galloprovincialis)						
Median	480	110	440	32	6	18
Average	520	180	350	30	8	19
Std.Err.	42	67	69	3	2	4
(n)	5	5	10	5	5	10
Richmond (M. galloprovincialis)						
Median	40	36	38	<2	3	3
Average	40	40	40	2	5	4
Std.Err.	12	6	6	1	2	1
(n)	5	5	10	5	5	10
Control (M. californianus) (Oregon Coast)						
Median		1.5			<1.7	
Average		1.4			<1.7	
Std.Err.		0.1			---	
(n)		5			5	
Anal. Blank		<3.8			<1.7	

*Average (± std. error) values of lipid content (as percent of wet weight) for the October vs. February surveys are as follows. Lauritzen Channel: 1.42 ± 0.33 vs. 0.48 ± 0.07; Santa Fe Channel: 2.27 ± 0.10 vs. 0.86 ± 0.17; Richmond Inner Harbor Channel: 1.71 ± 0.27 vs. 0.95 ± 0.17; Oregon Coast Control: 1.05 ± 0.14. Corresponding values for the decimal fraction of the dry-to-wet weight ratio are as follows: Lauritzen Channel, 0.17 ± 0.015 vs 0.089 ± 0.003; Santa Fe Channel, 0.20 ± 0.005 vs. 0.10 ± 0.016; Richmond Inner Harbor Channel, 0.18 ± 0.013 vs. 0.11 ± 0.006; Oregon Coast control, 0.18 ± 0.003. Individual sample values are listed in Appendix 8-8.

TABLE 8-24B. Average percent composition of ΣDDT in mussel tissues.

Site/Survey	2,4'DDE	4,4'DDE	2,4'DDD	4,4'DDD	2,4'DDT	4,4'DDT
Oct. 1991						
Lauritzen						
Average	0.69%	11.25%	12.73%	52.95%	5.74%	16.64%
SE	0.08%	1.50%	1.07%	2.52%	0.65%	2.25%
Santa Fe						
Average	0.97%	14.14%	11.94%	61.87%	1.74%	9.35%
SE	0.04%	0.39%	0.62%	0.43%	0.17%	0.20%
Richmond						
Average	1.99%	25.13%	10.57%	47.05%	0.00%	15.26%
SE	0.24%	1.37%	0.50%	3.51%	0.00%	2.01%
Control						
Average	0.00%	100.00%	0.00%	0.00%	0.00%	0.00%
Feb. 1992						
Lauritzen						
Average	0.95%	10.60%	10.13%	31.95%	17.44%	28.94%
SE	0.07%	0.87%	0.57%	0.89%	0.45%	0.51%
Santa Fe						
Average	0.95%	15.46%	10.65%	47.03%	7.66%	18.25%
SE	0.26%	1.11%	0.37%	0.77%	0.51%	0.95%
Richmond						
Average	1.00%	17.28%	10.15%	40.74%	8.41%	22.42%
SE	0.08%	0.87%	0.45%	1.31%	0.60%	1.22%

Average and standard error (SE) composition of ΣDDT (% contribution by specific residues) measured in soft tissues of Mytilus galloprovincialis. N = 5 for each site/survey.

TABLE 8-25. Whole water bioconcentration factors (BCF_{uw}, wet weight) for ΣDDT and dieldrin in Mytilus galloprovincialis.

Site	ΣDDT			Dieldrin		
	Oct.	Feb.	Combined	Oct.	Feb.	Combined
Lauritzen						
Median	51,000	13,000	32,000	5,600	1,300	3,500
Average	102,000	14,000	58,000	9,600	1,400	5,500
Std.Err.	35,000	3,000	22,000	3,100	180	2,000
(n)	5	5	10	5	5	10
Santa Fe						
Median	56,000	13,000	51,000	18,000	3,600	9,900
Average	61,000	20,000	41,000	17,000	4,200	11,000
Std.Err.	4,900	7,800	8,000	1,800	1,100	2,400
(n)	5	5	10	5	5	10
Richmond						
Median	40,000	28,000	38,000	---	---	---
Average	40,000	33,000	40,000	---	---	---
Std.Err.	12,000	10,000	6,300	---	---	---
(n)	5	5	10	5	5	10
Overall average	68,000	22,000	46,000	13,300	2,800	8,300

BCF_{uw}s based on average whole water concentrations for a given site (Table 8-10).

TABLE 8-26A. ΣDDT and dieldrin in megafauna and fishes (μg/kg wet weight) and average percent lipid (wet weight).

Site	ΣDDT Oct.	ΣDDT Feb.	ΣDDT Combined	Dieldrin Oct.	Dieldrin Feb.	Dieldrin Combined
Lauritzen Channel						
BENTHIC:						
Crangon-Muscle						
Median		76			<2.6	
Average		76			<2.6	
Std.Err.		4			---	
(n/i)*		2/5			2/5	
Ave. Lipid (%)		1.5			1.5	
Crangon-Remainder						
Median		470			21	
Average		470			21	
Std.Err		27			21	
(n/i)*		2/5			2/5	
Crangon-Whole**						
Median		310			12	
Average		310			12	
Std.Err.		4			12	
(n/i)*		2/5			2/5	
Cancer-Muscle	360			11		
(n/i)*	1/1			1/1		
Ave. Lipid (%)	0.94			0.94		
Gobiid-Whole		5400			200	
(n/i)*		1/1			1/1	
Ave. Lipid (%)		2.2			2.2	
PELAGIC:						
Cymatogaster-Muscle						
Median	1900			120		
Average	2000			120		
Std.Err.	410			25		
(n/i)*	5/5			5/5		
Ave. Lipid (%)	2.1			2.1		
Cymatogaster-Remainder						
Median	12,100			500		
Average	10,300			520		
Std.Err	2,000			120		
(n/i)*	5/5			5/5		
Cymatogaster-Whole**						
Median	8300			340		
Average	7500			390		
Std.Err.	1500			90		
(n/i)*	5/5			5/5		
Morone-Muscle						
Median	330			25		
Average	330			25		
Std.Err.	180			15		
(n/i)*	2/2			2/2		
Ave. Lipid (%)	1.3			1.3		
Anchoveta-Whole	98			15		
(n/i)*	1/1			1/1		
Ave. Lipid (%)	1.8			1.8		

TABLE 8-26A (cont'd.). ΣDDT and dieldrin in megafauna and fishes (μg/kg wet weight) and average percent lipid (wet weight).

Site	ΣDDT Oct.	ΣDDT Feb.	ΣDDT Combined	Dieldrin Oct.	Dieldrin Feb.	Dieldrin Combined
Santa Fe Channel						
BENTHIC:						
Crangon-Muscle						
Median		36			<2.6	
Average		35			2	
Std.Err.		6			2	
(n/i)*		3/24			3/24	
Ave. Lipid (%)		1.5			1.5	
Crangon-Remainder						
Median		150			5	
Average		160			4	
Std.Err.		16			2	
(n/i)*		3/24			3/24	
Crangon-Whole**						
Median		99			3	
Average		110			3	
Std.Err.		10			1	
(n/i)*		3/24			3/24	
Gobiid-Whole						
Median	730			28		
Average	680			28		
Std.Err.	57			3		
(n/i)*	5/5			5/5		
Ave. Lipid (%)	2.9			2.9		
Citharichthys-Whole		2200			22	
(n/i)*		1/1			1/1	
Ave. Lipid (%)		1.8			1.8	
Parophrys-Whole		200			<2.6	
(n/i)*		1/2			1/2	
Ave. Lipid (%)		1.9			1.9	
Platichthys-Whole		140			7	
(n/i)*		1/1			1/1	
Ave. Lipid (%)		1.4			1.4	
PELAGIC:						
Cymatogaster-Muscle						
Median	480	270	370	19	14	16
Average			370			16
Std.Err.			72			2
(n/i)*	1/1	1/1	2/2	1/1	1/1	2/2
Ave. lipid (%)	3.1	1.7	2.4	3.1	1.7	2.4
Cymatogaster-Remainder		1300			55	
(n/i)*		1/1			1/1	
Cymatogaster-Whole**		920			40	
(n/i)*		1/1			1/1	
Anchoveta-Whole	670			17		
(n/i)*	1/5			1/5		
Ave. Lipid (%)	1.9			1.9		

TABLE 8-26A(cont'd.). ΣDDT and dieldrin in megafauna and fishes (μg/kg wet weight) and average percent lipid (wet weight).

Site	ΣDDT Oct.	ΣDDT Feb.	ΣDDT Combined	Dieldrin Oct.	Dieldrin Feb.	Dieldrin Combined
Richmond Inner Harbor Channel						
BENTHIC:						
Crangon-Muscle						
Median	2200	5	7	69	3	6
Average		5	730		3	25
Std.Err.		1	590		3	18
(n/i)*	1/10	2/70	3/80	1/10	2/70	3/80
Ave. Lipid (%)	1.6	1.5	1.5	1.6	1.5	1.5
Crangon-Remainder						
Median		41			<2.6	
Average		41			<2.6	
Std.Err.		3			---	
(n/i)*		2/70			2/70	
Crangon-Whole**						
Median		24			2	
Average		24			2	
Std.Err.		3			2	
(n/i)*		2/70			2/70	
Cancer-Muscle						
Median		26			2	
Average		24			3	
Std.Err.		4			1	
(n/i)*		5/5			5/5	
Ave. Lipid (%)		0.81			0.81	
Cancer-Egg mass						
Median		220			2	
Average		220			2	
Std.Err.		5			2	
(n/i)*		2/2			2/2	
Ave. Lipid (%)		7.0			7.0	
Gobiid-Whole***						
Median	160	110	120	<8.3	9	6
Average		110	130		9	6
Std.Err.		15	19		3	3
(n/i)*	1/1	2/3	3/4	1/1	2/3	3/4
Ave. Lipid (%)		1.1			1.1	
Citharichthys-Whole		60			<2.6	
(n/i)*		1/4			1/4	
Ave. Lipid (%)		---			---	

TABLE 8-26A (cont.'d). ΣDDT and dieldrin in megafauna and fishes (μg/kg wet weight) and average percent lipid (wet weight).

Site	ΣDDT Oct.	ΣDDT Feb.	ΣDDT Combined	Dieldrin Oct.	Dieldrin Feb.	Dieldrin Combined
Richmond Inner Harbor Channel						
PELAGIC:						
Cymatogaster-Muscle						
Median	17			<2.6		
Average	24			1		
Std.Err.	9			1		
(n/i)*	5/5			5/5		
Ave. Lipid (%)	1.6			1.6		
Cymatogaster-Remainder						
Median	130			<2.6		
Average	130			<2.6		
Std.Err.	55			---		
(n/i)*	5/5			5/5		
Cymatogaster-Whole**						
Median	91			---		
Average	100			2		
Std.Err.	21			2		
(n/i)*	5/5			5/5		
Genyonemus-Muscle	44			4		
(n/i)*	1/3			1/3		
Ave. Lipid (%)	1.9			1.9		
Anchoveta-Whole	170			4		
(n/i)*	1/5			1/5		
Ave. Lipid (%)	1.8			1.8		

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* Number of samples analyzed (n), and total number of individuals (i) contributing to these samples; (n) is less than (i) if individuals were composited .

** Values calculated by combining weighted concentrations in muscle tissue and "residue", the remaining tissues.

*** The October 1991 values are estimates obtained from muscle tissue measurements and the median ratios of whole body to muscle tissue concentrations for ΣDDT (3.9) and dieldrin (3.2) obtained for Cymatogaster.

Average dry/wet ratios:
- Anchoveta = 0.21
- Cancer muscle = 0.18
- Cancer eggs = 0.31
- Citharichthys = 0.22
- Crangon muscle = 0.25
- Cymatogaster muscle = 0.22
- Genyonemus muscle = 0.21
- Gobiid whole = 0.21
- Morone muscle = 0.20

Table 8-26B. Average percent composition of ΣDDT in fish and megafauna.

October, 1991 Samples Site/Species	2,4'DDE	4,4'DDE	2,4'DDD	4,4'DDD	2,4'DDT	4,4'DDT
Lauritizen Channel						
BENTHIC:						
Cancer antennarius (muscle): n=1						
Average	0.00%	81.59%	2.25%	13.22%	0.00%	2.93%
PELAGIC:						
Cymatogaster aggegata (muscle): n=5						
Average	0.42%	28.62%	9.07%	49.86%	3.29%	8.75%
SE	0.08%	2.44%	1.15%	3.41%	0.86%	1.43%
Morone saxitalis (muscle): n=2						
Average	0.67%	33.54%	7.43%	42.36%	5.24%	10.76%
SE	0.20%	0.33%	0.21%	4.65%	2.65%	1.92%
Anchoveta (whole): n=1						
Average	0.00%	20.13%	9.07%	49.44%	5.87%	15.49%
Santa Fe Channel						
BENTHIC:						
Gobiid fish (whole): n=5						
Average	0.34%	26.27%	1.88%	65.55%	0.34%	5.62%
SE	0.02%	0.72%	0.07%	0.72%	0.05%	0.05%
PELAGIC:						
Cymatogaster aggegata (muscle): n=1						
Average	0.47%	35.60%	6.44%	47.15%	2.17%	8.18%
Anchoveta (whole): n=1						
Average	0.53%	26.38%	6.10%	51.43%	3.98%	11.58%
Richmond Inner Harbor Channel						
BENTHIC:						
Crangon franciscorum (tail muscle): n=1						
Average	20.50%	20.24%	15.44%	17.12%	14.82%	11.88%
Gobiid fish (muscle): n=1						
Average	6.49%	26.78%	5.72%	43.72%	5.76%	11.52%
PELAGIC:						
Cymatogaster aggegata (muscle): n=5						
Mean	1.41%	40.60%	7.61%	43.50%	1.75%	5.14%
SEM	0.99%	4.85%	2.01%	2.13%	1.11%	2.13%
Genyonemus lineatus (muscle): n=1						
Average	1.20%	31.01%	4.96%	54.76%	0.00%	8.07%
Anchoveta (whole): n=1						
Average	1.01%	38.56%	5.20%	42.23%	3.02%	9.98%

Table 8-26B (Cont'd).

February, 1992 Samples Site/Species	2,4'DDE	4,4'DDE	2,4'DDD	4,4'DDD	2,4'DDT	4,4'DDT
Lauritzen Channel						
BENTHIC:						
Crangon franciscorum (tail muscle)[1]: n=2						
Average	0.00%	100.00%	0.00%	0.00%	0.00%	0.00%
SE	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
Gobiid fish (whole):n=1						
Average	0.09%	44.00%	0.58%	45.91%	0.44%	8.98%
Santa Fe Channel						
BENTHIC:						
Crangon franciscorum (tail muscle): n=3						
Average	0.79%	52.27%	3.47%	38.66%	0.00%	4.82%
SE	0.79%	1.79%	0.06%	3.40%	0.00%	2.43%
Citharichthys stigmaeus (whole): n=1						
Average	1.01%	29.14%	3.08%	45.83%	4.17%	16.77%
Parophrys vetulus (whole): n=1						
Average	0.00%	21.48%	7.19%	47.18%	2.73%	21.43%
Platichthys stellatus (whole): n=1						
Average	0.81%	29.83%	4.27%	60.05%	0.96%	4.09%
PELAGIC:						
Cymatogaster aggregata (whole): n=1						
Average	0.36%	29.13%	6.72%	53.90%	2.41%	7.48%
Richmond Inner Harbor Channel						
BENTHIC:						
Crangon franciscorum (tail muscle)[2]: n=2						
Average	0.00%	100.00%	0.00%	0.00%	0.00%	0.00%
SE	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
Cancer antennarius (muscle): n=5						
Mean	0.13%	91.94%	1.18%	6.28%	0.14%	0.34%
SE	0.13%	3.95%	0.49%	3.85%	0.14%	0.34%
Cancer antennarius (embryo mass): n=2						
Average	0.35%	87.67%	1.27%	9.43%	0.46%	0.82%
SE	0.11%	3.47%	0.34%	2.79%	0.07%	0.16%
Gobiid fish (whole): n=2						
Average	0.31%	16.75%	2.85%	72.71%	0.00%	7.38%
SE	0.31%	0.22%	0.07%	1.04%	0.00%	1.49%
Citharichthys stigmaeus (whole): n=1						
Average	1.86%	23.09%	5.63%	46.60%	9.60%	13.21%

1 = Only 4,4'DDE detected, probably owing to very small sample sizes (approximately 0.2 and 0.4 g wet weight).
2 = Only 4,4'DDE detected, probably owing to small sample sizes (approximately 1.6g and 2.4 g), and lower ΣDDT concentrations in Richmond Inner Harbor Channel.

TABLE 8-26C. Minimum average whole-body concentration (μg/kg wet weight) of ΣDDT and dieldrin in pelagic and benthic fishes adjusted to 25 individuals. The subtotal (N) is the actual number of individuals the 25 individual estimate is based on.

I. PELAGIC FISHES

TAXA	N	ΣDDT	PRODUCT	N	DIELDRIN	PRODUCT
Lauritzen Channel						
Cymatogaster	5	7500	37,500	5	390	1,950
Anchoveta	1	98	98	1	15	15
SUBTOTAL	6		37,598	6		1,965
MIN. CONC.	25	1,504		25	79	
Santa Fe Channel						
Cymatogaster	1	920	920	1	40	40
Anchoveta	5	670	3,350	5	17	85
SUBTOTAL	6		4,270	6		125
MIN. CONC.	25	171		25	5	
Richmond Channel						
Cymatogaster	5	100	500	5	2	10
Anchoveta	5	170	850	5	4	20
SUBTOTAL	10		1350	10		30
MIN. CONC.	25	54		25	1	

TABLE 8-26C (Cont'd). Minimum average whole-body concentration (μg/kg wet weight) of ΣDDT and dieldrin in pelagic and benthic fishes based on 25 individuals. The subtotal (N) is the actual number of individuals the 25 individual estimate is based on.

I. BENTHIC FISHES

TAXA	N	ΣDDT	PRODUCT	N	DIELDRIN	PRODUCT
Lauritzen Channel						
Gobiid	1	5,400	5,400	1	200	200
MIN. CONC.	25	216		25	8	
Santa Fe Channel						
Gobiid	5	680	3,400	5	28	140
Citharichthys	1	2,200	2,200	1	22	22
Parophrys	2	200	400	2	0	0
Platichthys	1	140	140	1	7	7
SUBTOTAL	9		6,140	9		169
MIN. CONC.	25	246		25	7	
Richmond Channel						
Gobiid	4	130	520	4	6	24
Citharichthys	4	60	240	4	0	0
SUBTOTAL	8		760	8		24
MIN. CONC.	25	30		25	1	

TABLE 8-27. Bioconcentration factors (BCF_{uw}, wet weight) for ΣDDT and dieldrin in pelagic fishes.

Site	ΣDDT Oct.	ΣDDT Feb.	ΣDDT Combined	Dieldrin Oct.	Dieldrin Feb.	Dieldrin Combined
Lauritzen Channel						
Cymatogaster - Muscle						
Median	38,000			6,400		
Average	40,000			6,700		
Std.Err.	8,200			1,400		
(n)	5			5		
Cymatogaster - Whole						
Median	166,000			19,000		
Average	151,000			22,000		
Std.Err.	30,000			4,900		
(n)	5			5		
Morone - Muscle						
Median	6,700			1,400		
Average	6,700			1,400		
Std.Err.	3,600			820		
(n)	2			2		
Anchoveta - Whole	2,000			850		
Santa Fe Channel						
Cymatogaster - Muscle						
Median	55,000	32,000	43,000	10,400	7,900	9,200
Average			43,000			9,200
Std.Err.			12,000			1,200
(n)	1	1	2			2
Cymatogaster - Whole		107,000			22,000	
Anchoveta - Whole	78,000			9,500		
Richmond Channel						
Cymatogaster - Muscle						
Median	17,000			---		
Average	24,000			---		
Std.Err.	9,100			---		
(n)	5			5		
Cytamogaster - Whole						
Median	91,000			---		
Average	100,000			---		
Std.Err.	21,000			---		
(n)	5			5		
Genyonemus - Muscle	44,000			---		
Anchoveta - Whole	17,000			---		

BCF_{uw}s based average whole water concentrations for a given site (Table 8-10).

TABLE 8-28A. ΣDDT and dieldrin concentrations in trawl transect sediments.

SITE	TOC Fraction	ΣDDT (µg/kg dry wt.)	Dieldrin (µg/kg dry wt.)	ΣDDT (µg/g OC)[1]	Dieldrin (µg/g OC)[1]
Lauritzen (n=3)[2]					
Median	0.0178	47,800	530	2,440	26.9
Average	0.0197	50,500	570	2,570	29.1
Std.Err.	0.0021	15,000	91	760	4.6
Santa Fe (n=4)[2]					
Median	0.0153	640	17	34.2	0.90
Average	0.0188	1,010	29	53.8	1.57
Std.Err.	0.0037	450	17	23.8	0.88
Richmond (n=3)[2]					
Median	0.0125	82	1.7	6.53	0.19
Average	0.0126	84	1.6	6.70	0.12
Std.Err.	0.0005	27	0.9	2.14	0.07

1 = Organic carbon normalized values based on average TOC value for each channel

2 = Number of station means included in transect mean. See text for specific stations included.

TABLE 8-28B. Average percent composition of ΣDDT in trawl transect sediments.

Site	2,4'DDE	4,4'DDE	2,4'DDD	4,4'DDD	2,4'DDT	4,4'DDT
Lauritzen Channel: (n=3) [1]						
Average	0.1%	2.1%	7.7%	44.6%	2.0%	43.5%
SE	0.0%	0.2%	1.2%	7.6%	0.8%	8.4%
Santa Fe Channel: (n=4) [1]						
Average	0.8%	12.9%	10.4%	58.8%	1.0%	16.0%
SE	0.2%	4.0%	0.3%	6.6%	0.7%	2.3%
Richmond Inner Harbor Channel: (n=3) [1]						
Average	0.9%	10.3%	9.4%	58.0%	0.2%	21.2%
SE	0.3%	1.5%	0.4%	2.7%	0.2%	4.5%

1 = Organic carbon normalized values based on average TOC value for each channel.

TABLE 8-29. Accumulation factors for ΣDDT and dieldrin in benthic fishes and megafauna.

Site/Species	ΣDDT Oct.	ΣDDT Feb.	ΣDDT Combined	Dieldrin Oct.	Dieldrin Feb.	Dieldrin Combined
Lauritzen Channel						
Crangon - Muscle						
Median		0.002			---	
Average		0.002			---	
Std.Err.		0.000			---	
(n)		2			2	
Cancer - Muscle	0.01			0.04		
Gobiid - Whole		0.09			0.31	
Santa Fe Channel						
Crangon - Muscle						
Median		0.05			---	
Average		0.04			0.07	
Std.Err.		0.01			0.07	
(n)		3			3	
Gobiid - Whole						
Median	0.44			0.59		
Average	0.43			0.61		
Std.Err.	0.02			0.06		
(n)	5			5		
Citharichthys - Whole		2.2			0.78	
Parophrys - Whole		0.20			---	
Platichthys - Whole		0.19			0.33	
Richmond Channel						
Crangon - Muscle						
Median	21	0.05	0.07	35	1.6	3.1
Average		0.05	6.9		1.6	13
Std.Err.		0.02	6.8		1.6	11
(n)		2	3		2	3
Cancer - Muscle						
Median		0.45			2.2	
Average		0.43			2.4	
Std.Err.		0.07			0.9	
(n)		5			5	
Cancer - Embryo						
Median		1.2			0.34	
Average		1.2			0.34	
Std.Err.		0.9			0.09	
(n)		2			2	
Gobiid - Whole						
Median		1.4			6.1	
Average		1.4			6.1	
Std.Err.		0.2			2.3	
(n)		2			2	
Citharichthys - Whole		0.49			---	

AFs based on the October 1991 average sediment concentrations for the stations best representing the trawl transects (Table 8-28).

TABLE 8-30. Concentration gradients for ΣDDT.

Channel	ΣDDT				
	Water	Sediment[1]	Mussel	Gobiid	Surfperch
Lauritzen	50	50,500	2,900	5,400	7,500
Santa Fe	8.6	1,010	350	680	920
Richmond	1	84	40	130	100

1 = Sediment conc. based on stations covered by trawls as well as any stations immediately bracketing the trawls. See text for specific stations included.

Water = ng/L

Sediment = μg/kg dry

Mussel, gobiid, and surfperch = μg/kg wet (whole body)

TABLE 8-31. Concentration gradients for dieldrin.

Channel	Dieldrin				
	Water	Sediment[1]	Mussel	Gobiid	Surfperch
Lauritzen	18	570	97	200	390
Santa Fe	1.8	29	19	28	40
Richmond	<1	1.6	4	6	2

1 = Sediment conc. based on stations covered by trawls as well as any stations immediately bracketing the trawls. See text for specific stations included.

Water = ng/l

Sediment = μg/kg dry

Mussel, gobiid, and surfperch = μg/kg wet (whole body)

TABLE 8-32. Overall average accumulation factor values (and 95 percent confidence intervals).

Organism	Total DDT				Dieldrin			
	Avg.	95%	CI	n	Avg.	95%	CI	n
Shrimp Muscle	2.6	-3.5	- 8.8	8	12.8	-35	- 61	3
Crab Muscle	0.36	0.13	- 0.60	6	2.4	-0.03	- 4.8	5
Flatfishes	0.77	-0.76	- 2.3	4	0.56	-2.3	- 3.4	2
Avg. 3 taxa[1]	1.2	--	--	3	5.2	--	--	3
Gobiid Fish	0.63	0.21	- 1.1	8	1.9	-0.50	- 4.3	8

1 = Mean of the means of shrimp muscle, crab muscle and flatfish.

9.0 EFFECTS ASSESSMENT

This section covers the ecological effects assessment conducted at the Lauritzen/Santa Fe/Richmond sites. The purposes of the effects assessment were to quantify the nature and extent of any existing ecological impacts, to determine the role of ΣDDT and dieldrin in causing any observed impacts, and to develop models predicting the type and extent of ecological effects as a function of the concentrations of the dominant pollutant stressors. The approaches used in assessing ecological effects were a survey of macrobenthic community structure and composition, two types of sediment bioassays, and modeling the exposure of fish-eating birds to contaminated prey.

9.1. BENTHIC SURVEY

9.1.1. Introduction

The primary purpose of the macrobenthic field study was to determine if a concentration-response relationship exists between ΣDDT and dieldrin concentrations in sediments or interstitial water and macrobenthic community structure and composition at the United Heckathorn Superfund site (U.S. EPA, 1991b). According to Brown *et al.* (1990), Levine-Frick (1990), and Pinza *et al.* (1992), DDT and its metabolites and dieldrin are the primary sediment contaminants at the study site. In this study, ΣDDT and dieldrin sediment concentrations were found to vary by 5 and 4 orders-of-magnitude, respectively (167,421 to 8 μg ΣDDT/kg, dry wt and 2,679 to 0 μg dieldrin/kg, dry wt), among samples (Appendix 8-1). In contrast, metals contamination at the site is probably unimportant since SEM/AVS ratios were < 1 (Table 8-10), and with the exception of Station 6, PAH sediment contamination is low to moderate (Table 8-9). The variability of sediment temperature (18-20°C), salinity (30-31 ‰) and water depth (10-38 feet) (Appendix 7-1) was small and probably not ecologically significant. Sediment grain size (%silt+clay; range: 99.8-71.7%) (Table 7-1) and total organic carbon (TOC; range: 4.0-0.7%) (Appendix 8-1) also did not vary greatly. Nevertheless, because sediment characteristics are known to affect the structure and composition of benthic communities (Sanders, 1958; Bloom *et al.*, 1972), the potential influence of %silt+clay and TOC on two measures of macrobenthic community structure was addressed using hierarchical regression (Section 9.1.2.). Obvious effects of shipping disturbance were dealt with by a priori deletion of suspect data (see Section 9.1.2.). While other factors such as competition and predation may influence benthic community structure and composition, they are difficult or impossible to measure accurately and their consideration was beyond the scope of this study.

In summary, the main factors considered in this study were the degree of ΣDDT and dieldrin sediment contamination. Available data from this and previous studies indicated that these two contaminants were probably important contributing factors if not

the principal causes of any alterations in the macrobenthos. Furthermore, as the contaminants of concern, ΣDDT and dieldrin would be the focus of any remedial actions. Concentration-response relationships between sediment or interstitial water chemistry and measures of macrobenthic community structure and composition found in this study could be used as best predictors of the kind and amount of change in the macrobenthic community structure measures investigated following any reductions in ΣDDT and dieldrin sediment contamination as a result of remediation. No other quantitative studies of both the sediment chemistry and the macrobenthos have been previously conducted in the study area.

9.1.2. Methods

Field sampling methods are described in Section 6.4. Basically, macrobenthic infauna were sampled (sample unit = three replicate 8.0 cm i.d. cores, totaling 0.015 m^2, x 10 cm deep) from each replicate grab sample at primary Stations 1 through 9 and at secondary Stations 22 and 23 (Figures 6-1 and 6-2). Eight replicates were taken at Station 1, five replicates at Stations 2 through 9, and single samples were taken at Stations 22 and 23. Infauna were separated from the sediment using stacked 1.0 and 0.5 mm mesh sieves and were preserved in a buffered solution of 10% formalin in seawater. All macrobenthic analyses in this section are based on data obtained from only the 1.0 mm mesh sieve collections (Appendix 9-1). At the time of this writing, organisms in the 0.5 mm mesh samples have been identified (Appendix 9-2) but the data analysis is incomplete. (Note: Results of statistical analyses completed to date on the 0.5 mm mesh samples are in general agreement with the results reported here for the 1.0 mm mesh samples). Station 8 samples were excluded from the macrobenthic analyses because the sampling crew observed severe physical disturbance of the bottom sediment from the propeller wash of deep-draft ships passing that station (see Section 7.2. and Specht, 1991). The field sampling crew did not observe disturbance of bottom sediments by prop wash at any other station, and since there was an extreme paucity of animals only at Station 8 (Appendix 9-1), there is no evidence of physical disturbance by prop wash being a major factor affecting the benthos at the other stations.

In the laboratory, preserved samples were transferred to 70% ethanol and stained with rose bengal. Animals were sorted with the aid of either a 3-diopter magnifier or a dissecting microscope (6-12x) and each sample was subsequently resorted under a dissecting microscope. Animals found in each sample were separated into major taxonomic groups (Polychaeta, Mollusca, Crustacea, "others") and placed in small containers with Nitex®-screen bottoms which were allowed to drain onto absorbent paper for 5 minutes prior to weighing. Group biomasses (± 0.01 g, wet wt) were determined by subtracting the container weight from the container plus sample weight. The weights of a clump of mussels, Musculus senhousia, in replicate 2 at Station 2, and a large

anthozoan, Pachycerianthus fimbriatus, in replicate 4 at Station 7, were subtracted from the biomass estimates to remove the skewing effect of one or a few exceptionally large individuals. Specimens were identified to the lowest possible taxon, usually species, and counted. EPA's in-house taxonomists, Faith Cole and Waldemar De Ben, performed all identifications and cross-verified specimens in all but two taxa; Leslie Harris (Los Angeles County Natural History Museum) verified the Polychaeta and John Chapman (AScI, Newport, OR) verified the Crustacea.

The infauna data (Appendix 9-1) were used to determine the following measures of macrobenthic community structure and composition per 0.015 m^2 x 10 cm deep sampling unit:

(1) number of species,

(2) numerical abundance of all taxa,

(3) biomass of all taxa (g, wet wt),

(4) Infaunal Index (aka the "Infaunal Trophic Index"). This is a biotic index developed by Word (1978) to characterize macrobenthic community responses to pollution in the Southern California Bight. In this study, we used the six group version of the Infaunal Index (Word, 1990; see Table 9-1 for a list of taxa in each group). The formula is:

$$\text{Infaunal Index} = 100 - \left[20 \left(\sum_{i=1}^{j} (i-1)\, N_i \Big/ \sum_{i=1}^{j} N_i \right)\right] \qquad \text{Eq. 9-1}$$

where N_i = number of individuals in group i, j = 6 specific groups of animals differing in their sensitivity to pollution, and 20 is a scaling factor to obtain index values from 0 (highly polluted) to 100 (unpolluted),

(5) a Dominance index equal to the minimum number, or fraction, of species whose combined abundance was equal to 75% of the individuals (Swartz *et al.*, 1985b, 1986),

(6) Brillouin's (1962) Index,

$$H = (1/N) \left(\log_{10} N! - \sum_{j=1}^{s} \log_{10} n_j!\right) \qquad \text{Eq. 9-2}$$

where N = total number of individuals in s species and n_j = number of individuals in the jth species (j = 1, 2,...s),

(7) the complement of Simpson's (1949) Index:

$$\text{1-Simpson's Index} = 1 - \sum_{j=1}^{s} n_j\,(n_j - 1)/\, N\,(N-1) \qquad \text{Eq. 9-3}$$

(8) McIntosh's (1967) Index:

$$\text{McIntosh's Index} = N - \sqrt{\sum_{j=1}^{s} n_j^2} \;/\; N - \sqrt{N} \qquad \text{Eq. 9-4}$$

(9) number of Amphipoda,

(10) number of Amphipoda excluding Grandidierella japonica,

(11) number of Crustacea,

(12) number of Mollusca,

(13) number of Polychaeta,

(14) Polychaeta biomass (g, wet wt),

(15) Mollusca biomass (g, wet wt),

(16) Crustacea biomass (g, wet wt).

Measures 1 to 3 are commonly used in pollution studies (Pearson and Rosenberg, 1978). The Infaunal Index was a sensitive indicator of oil pollution in Puget Sound (Ferraro *et al.*, 1989) and sewage-industrial pollution in the Southern California Bight (Ferraro *et al.*, in press). Brillouin's Index is the appropriate information theory index for measuring a collection's diversity (Pielou, 1966, 1975). The complement of Simpson's Index is the probability that two individuals chosen at random and independently from a population will belong to different species. McIntosh's Index is a Euclidean distance measure which ranges from 0 when only one species is present to a maximum of 1 when individuals are evenly distributed among species. According to Washington (1984), Simpson's and McIntosh's Index are suitable for pollution studies. Magurran (1988) describes the characteristics, relative merits, and shortcomings of these and other diversity indices for investigating pollution and other environmental effects. Numerical abundance and biomass by major taxa are measures of taxonomic composition. Measure (10) was chosen because G. japonica appears to be a pollution tolerant species at the study site (see Sections 9.2.3. and 9.2.4.) and elsewhere (Swartz, unpublished data).

Our statistical models were simple linear [$Y = f(X)$] and hierarchical multiple linear regression [$Y = f(X_1, X_2, X_3, X_4)$] (Cohen and Cohen, 1975; Tabachnick and Fidell, 1983; Neter *et al.*, 1989). The dependent variables (Ys) were the sixteen measures of community structure identified above. The independent variables (Xs) for the simple linear regression were $\log_{10}$(ΣDDT μg/g OC) and $\log_{10}$(dieldrin+1 μg/g OC). The null hypothesis for the simple linear case was that no significant ($p \leq 0.05$)

functional relationship exists between the independent variables and dependent variables. The simple linear regressions ignore possible effects of other variables on the dependent variable. The independent variables for the hierarchical regression were: X_1 = %silt+clay, X_2 = TOC, X_3 = $\log_{10}$(ΣDDT μg/g OC), and X_4 = $\log_{10}$(dieldrin+1 μg/g OC). The order of entry of the independent variables in the hierarchial regression (X_1 through X_4) was chosen to determine if ΣDDT and dieldrin sediment concentrations account for a significant amount of the variation in the dependent variables beyond that accounted for by sediment %silt+clay and TOC. Log_{10}(ΣDDT μg/g OC) (X_3) was given priority entry into the regression model over $\log_{10}$(dieldrin+1 μg/g OC) (X_4) because bioassay results and toxic unit analyses indicate ΣDDT is the principal ecotoxicological factor at the study site (Section 9.2.4.). The close proximity of sediment chemistry and macrobenthic infauna samples (samples were taken from the same grab and were at most a few cm apart) allows strong inferential linkage between exposure conditions and potential biological effects.

The sediment chemistry variables were log-transformed to meet the assumptions of linear regression. The $\log_{10}(Y+1)$ transformation was used for dieldrin concentrations because zero (<DL) values were encountered in the data set. Linearity, normality, and homoscedasticity were checked by inspection of scatter plots of the predicted values of the dependent variable against residuals (Tabachnick and Fidell, 1983). Data points potentially having an unduly large influence on the simple linear regression analyses (outliers) were identified by their high leverage coefficients ($\underline{h} > 4/\underline{n}$) and large (>3 or <-3) standardized residuals (Sokal and Rohlf, 1981). Observations with |DFFITS| > 1 or |DFBETAS| > 1 were identified as outliers in the hierarchical multiple regression analyses (Neter *et al.*, 1989). Analyses were conducted on the complete data set and after sequentially removing the largest outlier. A regression was considered significant and robust if the probability of significance (p) was ≤ 0.05 after removal of outliers.

We calculated simple linear regressions for four independent variables for both ΣDDT and dieldrin (sediment concentration in μg/kg, dry wt and μg/g OC and interstitial water concentrations in ng/L, total and ng/L, free) to determine the medium and unit of measurement which best predicted the variability in the dependent variables. In summary, we found little or no difference between units of measurement within a medium and sediment concentration was usually a better predictor than the corresponding interstitial water concentrations (see Section 9.1.3.). Based on these results, and because organic compounds, such as DDT and dieldrin, tend to be most closely associated with the organic carbon fraction in sediments (Karickhoff *et al.*, 1979; DiToro *et al.*, 1991), we report results only for ΣDDT and dieldrin sediment concentrations normalized to sediment TOC.

We report the regression equations, coefficients of determination (r^2 = the proportion of the variance in the dependent variable explained by [attributable to] the independent

variable), and the probability of significance of the simple linear regressions. Figures are plots of the dependent variable as a function of the independent variable and include the least-squares linear regression line. Pearson product-moment correlations (r) are reported in each figure. We report only the results of the hierarchical regression analyses for the two measures (Measures 4 and 10, above) which had the highest r^2s in the simple linear regressions on $\log_{10}$(ΣDDT μg/g OC). We report the correlations among the variables in the hierarchical regression model, the unstandardized (B) and standardized (β) regression coefficients, the Y intercept, the squared semipartial correlations (sr^2), and the multiple correlation coefficients (R^2 and R^2 adjusted) of the complete model. All statistical analyses were performed using SAS (1985).

9.1.3. Results

Diversity, as measured by the Dominance, 1-Simpson's, and McIntosh's indices, and the number and biomass of Mollusca significantly increased with increasing $\log_{10}$(ΣDDT μg/g OC) while the Infaunal Index, number of Amphipoda, number of Amphipoda excluding G. japonica, and number and biomass of Crustacea significantly decreased (Table 9-2; Figure 9-1). The regressions of number and biomass of Crustacea on $\log_{10}$(ΣDDT μg/g OC), however, were not significant ($p > 0.05$) after removing an outlier (Station 23). There was no significant regression of number of species, abundance, total biomass, Brillouin's Index, and number and biomass of Polychaeta on $\log_{10}$(ΣDDT μg/g OC) (Table 9-2; Figure 9-1).

The probabilities of significance and r^2s in the simple linear regressions of each of the sixteen dependent variables on $\log_{10}$(ΣDDT μg/g OC) and $\log_{10}$(ΣDDT μg/kg, dry wt) were similar. Sediment TOC was fairly constant among the stations in the study area (Appendix 8-1), and, consequently, there was little difference between regressions calculated using ΣDDT concentrations on a sediment dry weight and TOC-normalized basis. Four fewer measures (Dominance, number and biomass of Crustacea, and number of Mollusca), however, were significantly related to $\log_{10}$(ΣDDT ng/L, total) than to $\log_{10}$(ΣDDT μg/g OC), and five fewer measures (Dominance, number of Amphipoda, Crustacea, and Mollusca, and Crustacea biomass) were significantly related to the $\log_{10}$(ΣDDT ng/L, free) than to $\log_{10}$(ΣDDT μg/g OC). The generally poorer predictive ability of ΣDDT interstitial water compared to ΣDDT sediment concentrations may have been due to higher analytical variability, greater number of below detection limit values for interstitial water chemistry, and smaller sample size due to missing data ($n = 41$ for interstitial water ΣDDT and $n = 45$ for sediment ΣDDT) (Appendix 8-1).

Diversity (Dominance, Brillouin's, 1-Simpson's, and McIntosh's indices), total biomass, and Mollusca biomass significantly increased while the Infaunal Index and number of Amphipoda excluding G. japonica significantly decreased with increasing $\log_{10}$(dieldrin+1 μg/g OC) (Table 9-3; Figure 9-2).

Removal of outliers had no affect on the probabilities of significance of the regressions and little affect on r^2s. There was no significant regression of number of species, abundance, number of Amphipoda, Crustacea, Mollusca, and Polychaeta, and Polychaeta and Crustacea biomass on $\log_{10}$(dieldrin+1 µg/g OC) (Table 9-3; Figure 9-2). As with ΣDDT, the results of simple linear regression analyses were similar for the independent variables $\log_{10}$(dieldrin+1 µg/g OC) and $\log_{10}$(dieldrin+1 µg/kg, dry wt), and $\log_{10}$(dieldrin+1 µg/g OC) was almost always as good or better predictor of the variability in the sixteen dependent variables than either the total or free $\log_{10}$(dieldrin+1 ng/L) concentrations in interstitial water.

Tables 9-4 and 9-5 present the correlations among variables and the results of the hierarchical regression analyses of the Infaunal Index and the $\log_{10}$(number of amphipods excluding G. japonica), respectively, on %silt+clay (X_1), TOC (X_2), $\log_{10}$(ΣDDT µg/g OC) (X_3), and $\log_{10}$(dieldrin+1 µg/g OC) (X_4). The interpretation of hierarchical regression analyses is basically the same after removing outliers, but R^2 increases to 0.71 and B for X_3 changes to -8.551 for the Infaunal Index regression after removing three outliers and R^2 increases to 0.63 and the B for X_3 changes to -0.424 for the $\log_{10}$(number of amphipods excluding G. japonica) regression after removing four outliers.

The pattern of the correlations between the variables in the hierarchical regression models (Tables 9-4 and 9-5) is consistent with the conclusion that TOC and $\log_{10}$(dieldrin+1 µg/g OC) are redundant with respect to, or are a "suppressor" (sensu Cohen and Cohen, 1975) of the effects of, $\log_{10}$(ΣDDT µg/g OC) on both the Infaunal Index and $\log_{10}$(number of amphipods excluding G. japonica). In other words, TOC and $\log_{10}$(dieldrin+1 µg/g OC) at least partly carry the same information as $\log_{10}$(ΣDDT µg/g OC) or they primarily function to suppress some of the variance in the $\log_{10}$(ΣDDT µg/g OC) that is irrelevant to the prediction of the Infaunal Index and $\log_{10}$(number of amphipods excluding G. japonica).

The squared semipartial correlations (sr^2) in the hierarchical regressions are particularly useful as they indicate the amount of variation in Y explained by each X as it enters into the model beyond that explained by Xs already in the model. Log_{10}(ΣDDT µg/g OC), therefore, accounted for a highly significant amount of the variation in both the Infaunal Index (Table 9-4) and $\log_{10}$(number of amphipods excluding G. japonica) (Table 9-5) (sr^2 = 0.36 and 0.40, respectively) after allowing for the effects of sediment %silt+clay and TOC. Log_{10}(dieldrin+1 µg/g OC) is not found to account for a significant amount of the variation in either the Infaunal Index or the $\log_{10}$(number of amphipods excluding G. japonica) beyond that accounted for by %silt+clay, TOC, and $\log_{10}$(ΣDDT µg/g OC). As noted earlier (Section 9.2.1.), $\log_{10}$(ΣDDT µg/g OC) was given assumed causal priority over $\log_{10}$(dieldrin+1 µg/g OC) based on the results of toxic unit analyses (Section 9.2.4). But even if this assumption is incorrect, changing the order of entry of the independent

variables has no effect on the coefficients in the complete regression model though changes may occur in the sr^2s associated with each independent variable as it is entered into the model. Reversing the order of $\log_{10}$(ΣDDT μg/g OC) and $\log_{10}$(dieldrin+1 μg/g OC), therefore, yields the same regressions as in Tables 9-4 and 9-5 but the sr^2s change and both $\log_{10}$(dieldrin+1 μg/g OC) and $\log_{10}$(ΣDDT μg/g OC) are found to account for a significant amount of the variation in the Infaunal Index and $\log_{10}$(number of amphipods excluding G. japonica) after accounting for the effects of the preceding independent variables.

The regression coefficient for $\log_{10}$(ΣDDT μg/g OC) in the Infaunal Index multiple regression (B = -5.936; Table 9-4) indicates that for any given sediment %silt+clay and TOC, on average, each additional $\log_{10}$(ΣDDT μg/g OC) unit change is associated with a decrease of 5.936 units in the Infaunal Index (Cohen and Cohen, 1975). The regression coefficient for $\log_{10}$(ΣDDT μg/g OC) in the $\log_{10}$(number of amphipods excluding G. japonica) multiple regression (B = -0.483; Table 9-5) indicates that for any given sediment %silt+clay and TOC, on average, each additional $\log_{10}$(ΣDDT μg/g OC) unit change is associated with a decrease of 0.483 units in the $\log_{10}$(number of amphipods excluding G. japonica). The similarity in the regression coefficients for $\log_{10}$(ΣDDT μg/g OC) in the simple linear regression (B = -6.564) (Table 9-2) and the multiple linear regression (B = -5.936) (Table 9-4) for the Infaunal Index indicates that changes in the Infaunal Index as a function of $\log_{10}$(ΣDDT μg/g OC) are similar whether or not the effect of %silt+clay and TOC is taken into account. The same conclusion follows for the regression prediction equations for $\log_{10}$(number of amphipods excluding G. japonica). The $\log_{10}$(ΣDDT μg/g OC) regression coefficients in this case are -0.311 for the simple regression model (note: Bs given in Table 9-2 are calculated on untransformed data; the ps and rs of the regressions did not change appreciably for untransformed and log-transformed dependent variables in Tables 9-2 and 9-3) and -0.483 (Table 9-5) for the multiple regression model.

Table 9-6 is a list of infaunal species (≥ 1.0 mm) having abundances ≥ 5 in the total collection, their Infaunal Index group, and numerical abundance by station groupings determined by significant ($p \leq 0.05$) differences in the $\log_{10}$(ΣDDT μg/g OC) among stations determined by Student-Newman-Keuls test. Species most abundant at Stations 1-3 (*e.g.*, Exogone lourei, Tubificidae Groups 1, 2 and 3, Zeuxo normani, Musculus senhousia, and Grandidierella japonica) are tolerant of DDT and dieldrin or are local populations with (evolved) tolerance, or are species with behavioral or other mechanisms which reduce their exposure to DDT and dieldrin in the sediments in which they live. Species most abundant at Stations 4-7 (*e.g.*, Theora lubrica, Pseudopolydora paucibranchiata, Cossura sp., Cirratulid sp. A, and Tharyx spp.) are probably moderately tolerant of DDT and dieldrin, while species which were only collected at Stations 9, 22, or 23 (Cryptomya californica, Phoronis *cf.* pallida, Glycinde polygnatha, Photis brevipes, and Turbellaria spp.) may be

sensitive to DDT and/or dieldrin.

9.1.4. Discussion

Regression analyses do not establish cause-and-effect; concomitant variation does not necessarily imply causation. Regression analyses, however, can be used to identify possible or probable causal relationships, to screen out causal relationships (*i.e.*, if the regression of Y on X is not significant, X is very unlikely to cause Y), to establish useful predictive empirical relationships, and for statistical control (Sokal and Rohlf, 1981). The ability to infer probable causal relationships with regression improves with greater understanding of the scientific or mechanistic interrelationships among the variables.

In this study there is no confusion of potential cause and effect. Toxic effects of the insecticides DDT and dieldrin on many animals are well documented (WHO 1989a, 1989b). There is a priori reason, therefore, to believe that high concentrations of ΣDDT and dieldrin in sediments may affect the biota by reducing or excluding sensitive taxa and, perhaps, increasing tolerant taxa. The converse proposition - that the biota cause, or significantly affect, the high ΣDDT and dieldrin sediment contamination at the study site - is not worthy of consideration.

At a minimum, one can not rule out the possible effect of ΣDDT and dieldrin on those community structure measures for which there were significant regressions in Tables 9-2 through 9-5. The regressions are also the best available predictors of macrobenthic community changes which may occur as a result of remediation of contaminated sites in the study area. As noted earlier (Section 9.1.1.), some environmental variables (*e.g.* temperature and salinity) were essentially constant in the study area, and, therefore, could be ignored; good faith actions were taken to ensure only relevant data were used in generating the regressions; and hierarchical regression was used to statistically control for the potential effects of sediment %silt+clay and TOC on the Infaunal Index and $\log_{10}$(number of amphipods excluding G. japonica) (Tables 9-4 and 9-5). If the regressions reported in this section are used for prediction, extrapolations should not be made beyond the joint region of the observations used in calculating the regressions (Neter *et al.*, 1989).

As sediment concentrations of ΣDDT increased, the Infaunal Index and the number of Amphipoda (especially after excluding the pollution tolerant species, G. japonica) decreased while diversity and the number and biomass of Mollusca increased (Table 9-2; Figure 9-1). The same trends were present for dieldrin but the regressions for number of Amphipoda and Mollusca were not significant (Table 9-3; Figure 9-2). The positive relationship between diversity and ΣDDT and dieldrin sediment concentrations primarily reflects a more even distribution of individuals among species in the more highly contaminated sediments since there was no significant trend in either the number of species or abundance (Figures 9-1,a,b and 9-2,a,b).

In general, crustaceans (especially amphipods) are sensitive

to DDT (Sanders, 1969; WHO, 1989a; Nebeker *et al.*, 1989; and see Section 9.2.). The observed inverse relationship between the number of Amphipoda and ΣDDT sediment concentration (Figure 9-1,i,j) is further evidence of amphipod sensitivity to DDT. DDT and dieldrin are insecticides (WHO, 1989a,b), and some crustaceans may be sensitive to DDT and dieldrin by reason of phylogenetic affinity (both insects and crustaceans are in the phylum Arthropoda). Mollusks, on the other hand, are generally insensitive to DDT (WHO, 1989a). The mollusk Macoma nasuta had high survival in the laboratory even when exposed to the most highly contaminated Lauritzen Channel sediment (Section 8.3.). Though mollusks can persist and may even thrive in areas contaminated with DDT and dieldrin, this does not mean there is no biological effect resulting from molluscan exposures to these contaminants. Bioaccumulated contaminants in mollusks and other tolerant taxa can move up the food chain and affect animals at higher trophic levels (see Section 9.4.).

Infaunal Index values decreased with increasing ΣDDT and dieldrin sediment concentrations (Tables 9-2 and 9-3; Figures 9-1,d and 9-2,d). Infaunal Index values decreased with increasing ΣDDT even after accounting for the possible influence of sediment %silt+clay and TOC (Table 9-4). The Infaunal Index is a measure of the proportion of animals by feeding categories and sensitivity to pollution (Word, 1978, 1980, 1990; Mearns and Word, 1982). High Infaunal Index values occur when the benthic fauna is dominated by pollution-sensitive detrital feeders, while low values occur when the benthic fauna is dominated by pollution-tolerant deposit feeders. Depressed Infaunal Index values in this study to a large extent reflected decreases in the abundance of the sabellid polychaete, Euchone limnicola, the amphipod, Ampelisca abdita, and the phoronid, Phoronis cf. pallida, and increases in the abundance of dorvilleids, oligochaetes, and capitellid polychaetes.

The Infaunal Index was the response variable with the strongest relationship (highest r^2; Tables 9-2 and 9-3) with ΣDDT and dieldrin sediment concentrations. Almost 50% of the variance in the Infaunal Index could be accounted for by the logarithm of the TOC-normalized ΣDDT or dieldrin sediment concentrations. The proportion of the variance uniquely attributable to $\log_{10}$(ΣDDT μg/g OC) after accounting for sediment %silt+clay and TOC was 36% (Table 9-4; 53% after removing outliers) (Note: removal of outliers may increase or decrease sr^2). The proportion of the variance of $\log_{10}$(number of amphipods excluding G. japonica) uniquely attributable to $\log_{10}$(ΣDDT μg/g OC) after accounting for sediment %silt+clay and TOC was 40% (Table 9-5; 45% after removing outliers).

Other physical, biotic, and chemical factors (physical disturbance of the sediment, predation, competition, other sediment contaminants, etc.) may have affected the distribution and abundance of the infaunal biota at the study site and consequently the values of our measures of macrobenthic community structure. However, there is no evidence and no a priori reason to believe

that any of these other factors were more important than ΣDDT and dieldrin sediment contamination or that they varied so closely with ΣDDT and dieldrin sediment concentrations as to render our regressions spurious. The previous statement is especially true for the Infaunal Index and the $\log_{10}$(number of amphipods excluding G. japonica) where, after accounting for sediment %silt+clay and TOC, and the amount of variance explained by $\log_{10}$(ΣDDT μg/g OC) was about 40% (Tables 9-4 and 9-5). As noted in Section 9.1.1., metals contamination does not appear to be a factor (Table 8-10), PAH contamination was highest at Station 6 (Table 8-9), a station only moderately contaminated with ΣDDT and dieldrin, and the only evidence of physical disturbance of the bottom sediment was at Station 8 (and those data were deleted). Our analyses (Tables 9-2 to 9-5) and observations (Table 9-6) of the macrobenthic community, therefore, suggest that ΣDDT and/or dieldrin sediment contamination are probably among the more important if not the most important factors affecting the macrobenthos at contaminated sites in the study area. Sediment bioassays and other ecotoxicological evidence (Section 9.2.4.) strongly suggest that the ΣDDT concentration was the dominant factor causing acute toxicity to sensitive amphipods.

The regression equations in this report (Tables 9-2 to 9-5) allow prediction of changes in macrobenthic community structure and composition at the study site as a function of the logarithm of the TOC-normalized ΣDDT and dieldrin sediment concentrations. Regressions with higher r^2s or R^2s are more likely to provide better predictions. Since the independent variables, $\log_{10}$(ΣDDT μg/g OC) and $\log_{10}$ (dieldrin+1 μg/g OC), formed three easily identifiable groups by stations (Stations 1-3, 4-7, and 9, 22, and 23, representing high, intermediate, and low concentrations of ΣDDT and dieldrin, respectively (Figures 9-1 and 9-2), predictions of macrobenthic changes are probably best made in increments of the independent variable(s) corresponding to these three station-groups.

For example, Infaunal Index estimates obtained using the formula in Table 9-2 and values of 2,732 μg ΣDDT/g OC (the mean for Stations 1-3) and 86 μg ΣDDT/g OC (the mean for Stations 4-7), are 67 and 77, respectively. The regression-based prediction, therefore, is that the Infaunal Index would increase from about 67 to about 77 if ΣDDT was reduced to about 86 μg ΣDDT/g OC at Stations 1-3. Predictions of the Infaunal Index made using the multiple regression model (Table 9-4) and appropriate values of the independent variables do not differ appreciably (± ~1) from those using the simple regression model.

According to Bascom *et al.* (1978), Infaunal Index values of 83-100 indicate "control", of 60-100 indicate "normal", of 30-60 indicate "changed", and of 0-30 indicate "degraded" benthic conditions in the Southern California Bight. If one assumes that these benchmarks are also appropriate for the study area, then remediation may be required at Stations 1-7 to obtain "control" conditions at ΣDDT- and dieldrin-contaminated sites in the study area (Figure 9-1,d). To obtain "normal" conditions, remediation

may be required at Stations 1-3 (Figure 9-1,d).

Judgments on appropriate remedial actions can also be made using only data internal to this study. For example, the lowest Infaunal Index value in a sample from sediments with low concentrations of ΣDDT and dieldrin was 68 (at Station 23), and the lowest Infaunal Index value in a sample from sediments with moderate concentrations of ΣDDT and dieldrin was 62 (replicate 5 at Station 6) (Figure 9-1,d). If either of these Infaunal Index values were taken as the cleanup criterion, then remediation may be required at Stations 1-3. Obviously, this approach could be applied using other criteria (*e.g.*, means or lower confidence limits of the Infaunal Index or other measures of community structure at uncontaminated sites).

TABLE 9-1. Taxa in the six infaunal index groups in the present study.

Lauritzen Channel Infaunal Index Groups

Detrital Feeders

Group 1
Water Column

Streblospio benedicti
Asychis elongata
Amaena occidentalis
Eupolymnia heterobranchia
Euchone limnicola
Macoma cf. *balthica*
Macoma nasuta
Ampelisca abdita
Phoronis cf. *pallida*
Amphipholis squamata
Aoroides columbiae
Caprella mutica
Dulichia rhabdoplastis

Group 2
Sediment/water interface

Exogone lourei
Nephtys caecoides
Nephtys cornuta franciscana
Polydora ligni
Polydora socialis
Pseudopolydora paucibranchiata
Tharyx spp.
Cirratulus cirratus
Cirratulidae sp A
Cirriformia spirabrancha
Cossura sp.
Notomastus tenuis
Mediomastus spp.
Arenicolidae
Nippoleucon hinumensis
Zeuxo normani
Photis brevipes
Boccardia proboscidia
Cirratulidae, unid.
Caulleriella alata
Leptochelia dubia
Sarsiella sp A
Eudorella pacifica
Sipuncula unid.
Theora lubrica

Group 3
Burrowing

Heteromastus filiformis

Deposit Feeders

Group 4
Interface

Eteone lighti
Leitoscoloplos pugettensis

Group 5
Buried

Not present

Group 6
Anaerobic Sediment Interface and buried specialized feeders

Schistomeringos longicornis
Armandia brevis
?*Barantolla* sp.
Capitella spp.
Tubificidae group 1
Tubificidae group 2
Tubificidae group 3

TABLE 9-2. Simple linear regression equations, coefficients of determination (R^2), and significance probability (P) of sixteen measures of community structure (Y) on X = $\log_{10}$(ΣDDT μg/g OC).

Y	=	A ± B(X)	r^2	P
Number Species	=	13.38 - 0.072 X	0.0003	0.9172
Abundance	=	82.22 - 2.606 X	0.0027	0.7366
Total Biomass	=	0.624 + 0.200 X	0.0529	0.1283
Infaunal Index	=	89.49 - 6.564 X	0.4923	0.0001 ***
Dominance	=	2.769 + 0.507 X	0.1049	0.0300 *
Brillouin's Index	=	0.611 + 0.036 X	0.0599	0.1050
1-Simpson's Index	=	0.659 + 0.047 X	0.1796	0.0037 **
McIntosh's Index	=	0.475 + 0.054 X	0.1858	0.0031 **
Number Amphipoda	=	16.09 - 4.758 X	0.1929	0.0025 **
Number Amphipoda (excluding Grandidierella japonica)	=	16.78 - 5.619 X	0.2628	0.0003 ***
Number Crustacea	=	15.96 - 3.944 X	0.1252	0.0171 *
Number Mollusca	=	12.84 + 3.000 X	0.0957	0.0386 *
Number Polychaeta	=	37.26 + 2.001 X	0.0036	0.6963
Polychaeta Biomass	=	0.297 - 0.022 X	0.0067	0.5934
Mollusca Biomass	=	0.049 + 0.297 X	0.1578	0.0069 **
Crustacea Biomass	=	0.074 - 0.018 X	0.1112	0.0252 *

Symbols highlight significant regressions: * = $0.01 < P \leq 0.05$; ** = $0.001 < P \leq 0.01$; *** = $P \leq 0.001$.

TABLE 9-3. Simple linear regression equations, coefficients of determination (R^2), and significance probability (P) of sixteen measures of community structure (Y) on X = $\log_{10}$(dieldrin+1 μg/g OC).

Y	=	A ± B(X)	r^2	P	
Number Species	=	12.49 + 1.012 X	0.0129	0.4580	
Abundance	=	71.55 + 7.035 X	0.0050	0.6447	
Total Biomass	=	0.656 + 0.551 X	0.1040	0.0307	*
Infaunal Index	=	84.09 - 12.08 X	0.4289	0.0001	***
Dominance	=	2.935 + 1.280 X	0.1724	0.0046	**
Brillouin's Index	=	0.620 + 0.095 X	0.1070	0.0283	*
1-Simpson's Index	=	0.691 + 0.097 X	0.1940	0.0025	**
McIntosh's Index	=	0.511 + 0.111 X	0.2036	0.0019	**
Number Amphipoda	=	10.27 - 6.107 X	0.0818	0.0568	
Number Amphipoda (excluding Grandidierella japonica)	=	10.30 - 7.753 X	0.1288	0.0155	*
Number Crustacea	=	10.27 - 3.854 X	0.0308	0.2490	
Number Mollusca	=	16.84 + 3.401 X	0.0317	0.2420	
Number Polychaeta	=	35.11 + 8.948 X	0.0184	0.3739	
Polychaeta Biomass	=	0.262 - 0.017 X	0.0011	0.8316	
Mollusca Biomass	=	0.173 + 0.714 X	0.2346	0.0007	***
Crustacea Biomass	=	0.050 - 0.022 X	0.0401	0.1870	

Symbols highlight significant regressions: * = $0.01 < P \leq 0.05$; ** = $0.001 < P \leq 0.01$; *** = $P \leq 0.001$.

TABLE 9-4. Hierarchical regression of the infaunal index (Y) on %silt+clay (X1), TOC (X2), $\log_{10}$(ΣDDT μg/g OC) (X3), and $\log_{10}$(dieldrin+1 μg/g OC) (X4).

	Correlations (r)						sr^2
	Y	X1	X2	X3	B	β	incremental
X1	-0.06				0.003	0.003	0.00
X2	-0.35*	-0.02			6.155	0.004	0.12**
X3	-0.70***	+0.10	+0.51***		-5.936	-0.634	0.36***
X4	-0.65***	-0.04	+0.45**	+0.92***	-1.399	-0.076	0.00
				Y Intercept =	88.75		
						R^2 =	0.49***
						R^2 adj. =	0.44***

* = P < 0.05; ** = P < 0.01; *** = P < 0.001.

TABLE 9-5. Hierarchical regression of the log(10) number of amphipods excluding Grandidierella japonica (Y) on %silt+clay (X1), TOC (X2), $\log_{10}$(ΣDDT µg/g OC) (X3), and $\log_{10}$(dieldrin+1 µg/g OC) (X4).

	Correlations (r)						
	Y	X1	X2	X3	B	β	sr^2 incremental
X1	-0.15				-0.000	-0.006	0.02
X2	-0.36*	-0.02			1.935	0.028	0.13***
X3	-0.75***	+0.10	+0.51***		-0.483	-1.160	0.40***
X4	-0.61***	-0.04	+0.45**	+0.92***	0.360	0.438	0.03
				Y Intercept =	1.135		
						R^2 =	0.59***
						R^2 adj. =	0.55***

* = P < 0.05; ** = P < 0.01; *** = P < 0.001.

TABLE 9-6. Mean abundance ("n"/0.015 m^2) of species with collection abundance ≥ 5.

SPECIES	TAXA	II Group	STATION (MEAN DDT, µG/G OC) 1 2 3 (2,579)	4 (189.5)	6 (90.2)	7 5 (31.5)	9 (1.34)	22 (1.47)	23 (1.43)
Theora lubrica	bivalve	2	17.7	24.2	15.6	19.5	2.0		1.0
Pseudopolydora paucibranchiata	polychaete	2	14.7	22.0	9.4	5.7	2.8		
Exogone lourei	polychaete	2	8.8	1.8	0.2	3.3	1.4		
Tubificidae Group 2	oligochaete	6	3.2		0.2	0.5			
Cossura sp.	polychaete	2	3.2	26.0	0.6	20.8	2.0		
Tubificidae Group 3	oligochaete	6	2.9		0.4	0.6	0.8		
Zeuxo normani	tanaid	2	2.8	0.8		0.4	0.8		
Musculus senhousia	bivalve	na	2.7	1.6		0.1	0.2		
Grandidierella japonica	amphipod	na	2.6	0.6		0.1	0.2		
Euchone limnicola	polychaete	1	2.6	2.2	5.8	2.7	37.2	5.0	4.0
Schistomeringos longicornis	polychaete	6	2.2	1.6	0.2	0.3	0.4		2.0
Cirriformia spirabrancha	polychaete	2	2.2	1.6	1.8	1.9			
Polydora ligni	polychaete	2	2.1	1.6	1.8	2.0			
Tubificidae Group 1	oligochaete	6	1.7	1.0	0.2	0.6			
Cirratulidae spA	polychaete	2	1.2	1.4	1.0	0.3			
Harmothoe imbricata	polychaete	na	1.2	0.4		0.3	1.4	1.0	
Capitella spp	polychaete	6	1.0	0.8	0.2	0.7	0.2		
Cirratulus cirratus	polychaete	2	0.6						
Tapes japonica	bivalve	na	0.6	0.2		0.2	2.8		1.0
Tharyx spp	polychaete	2	0.6	0.8	0.6	0.6			
Leitoscoloplos pugettensis	polychaete	4	0.5	0.2	0.8	0.9			
Asychis elongata	polychaete	1	0.4				1.0		1.0

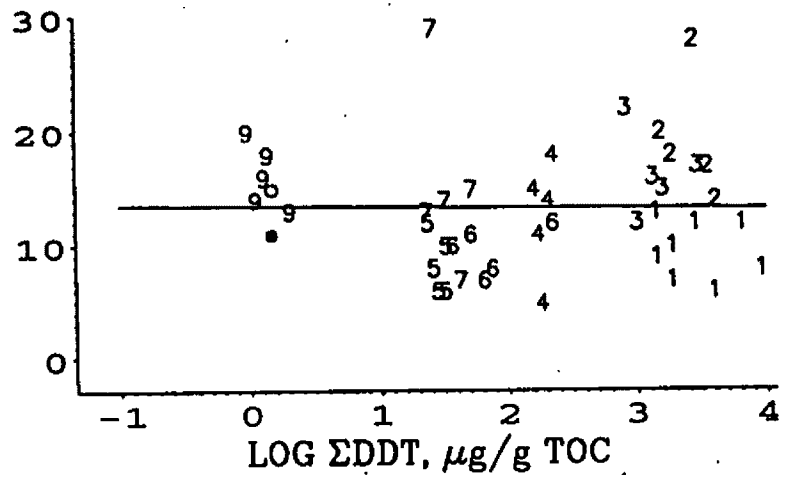
Heteromastus filiformis	polychaete	3	0.3						
Macoma nasuta	bivalve	1	0.2			0.1	0.2		
Mediomastus spp	polychaete	2	0.2	0.4		1.1	0.6	1.0	1.0
Harpacticoid copepod	copepod	nd	0.2			0.4			
Heteronemertea spp.	nemertean	na	0.2	0.2			0.8	2.0	
Philine spA	gastropod	nd	0.2		0.6	0.3			
Potamocorbula amurensis	bivalve	nd	0.1			0.5	0.2		1.0
Corophium heteroceratum	amphipod	nd		0.2	3.4	3.2	1.4	25.0	69.0
Cryptomya californica	bivalve	na					2.0	9.0	3.0
Ampelisca abdita	amphipod	1				0.2	7.0	10.0	
Phoronis cf. pallida	phoronid	1					33.4	1.0	
Glycinde polygnatha	polychaete	na					1.8		
Photis brevipes	amphipod	2						2.0	8.0
Turbellaria spp	turbellarian	na							5.0

na = not applicable, nd = not determined, no entry = not present

FIGURE 9-1. Sixteen measures (a-p) of macrobenthic community structure and composition as a function of $\log_{10}$(ΣDDT μg/g OC).

Regressions plotted as solid lines (see Table 9-1 for regression equations); r = product-moment correlation coefficient.

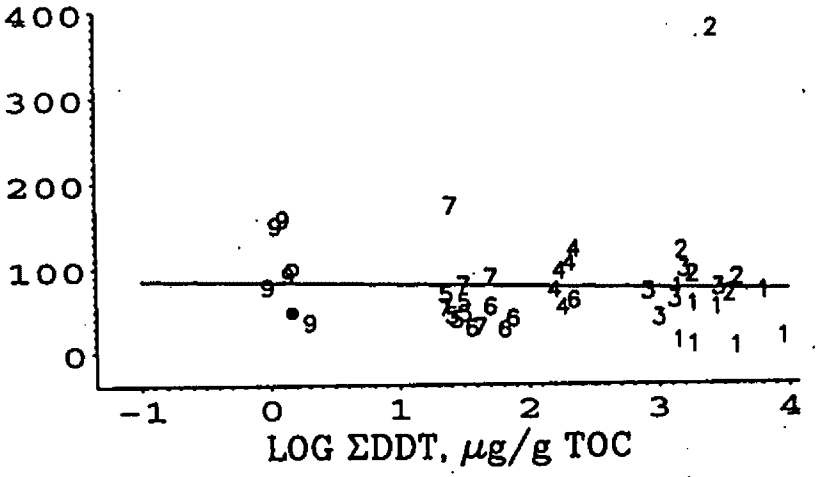
NUMBER SPECIES/0.015 M^2 **(a)** r= 0.02 ns



1–9 = STATIONS 1–9

● = STATION 22 o = STATION 23

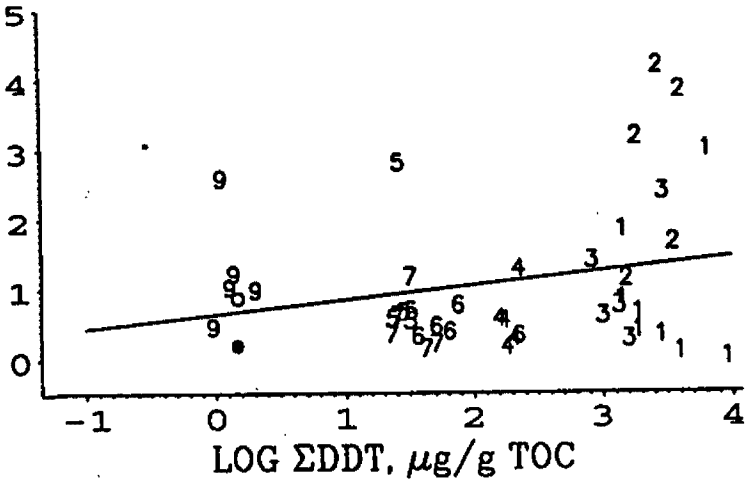
ABUNDANCE/0.015 M^2 **(b)** r = 0.05 ns



1–9 = STATIONS 1–9

● = STATION 22 o = STATION 23

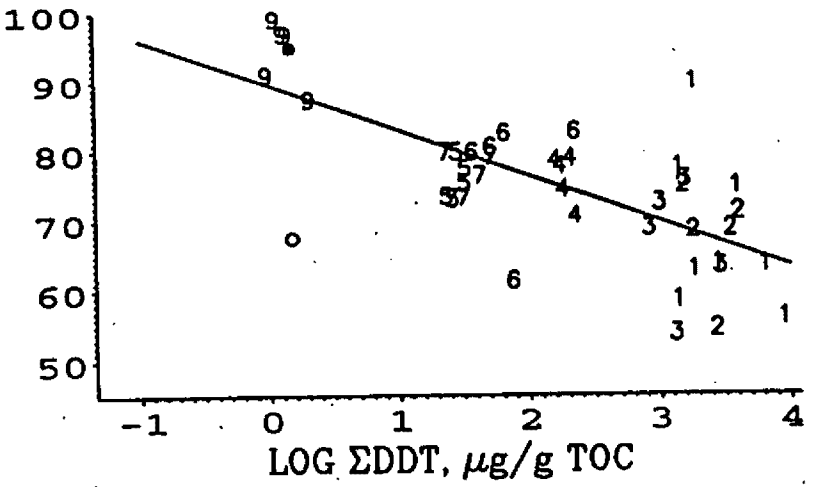
TOTAL BIOMASS, g/0.015 M^2 **(c)** r = 0.23 ns



1–9 = STATIONS 1–9

● = STATION 22 o = STATION 23

INFAUNAL INDEX **(d)** r = 0.70 ***

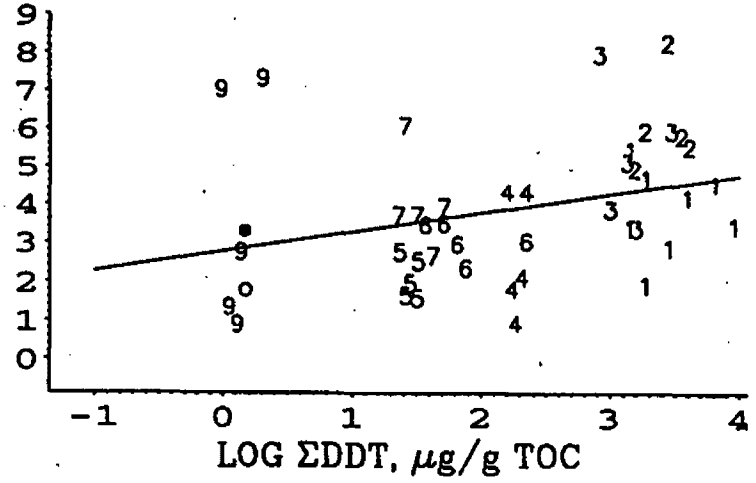


1–9 = STATIONS 1–9

● = STATION 22 o = STATION 23

(e)

DOMINANCE r = 0.32 *

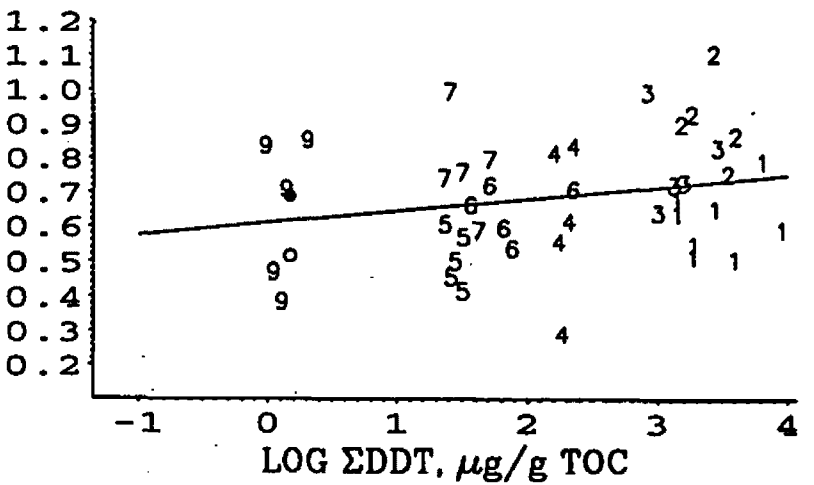


1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

(f)

BRILLOUIN'S INDEX r = 0.24 ns

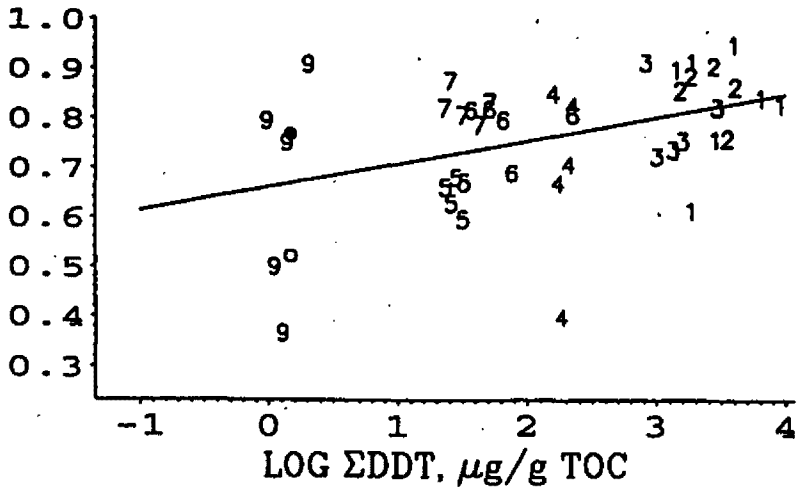


1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

(g)

1–SIMPSON'S INDEX r = 0.42 **

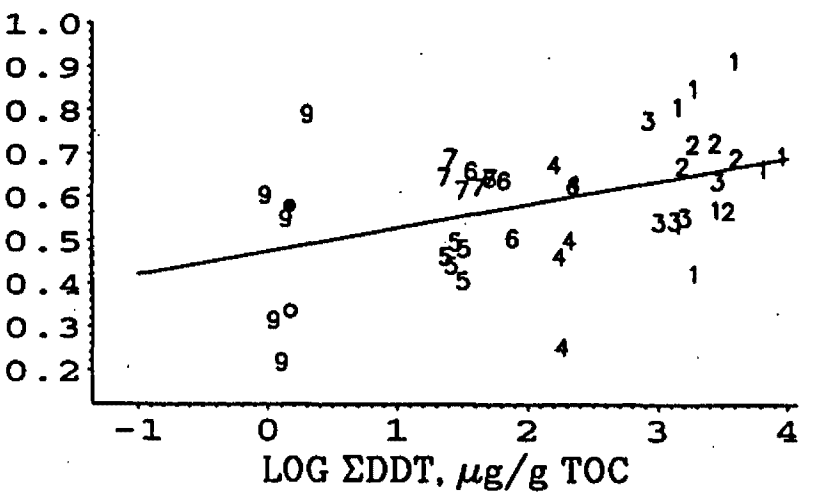


1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

(h)

MCINTOSH'S INDEX r = 0.43 **

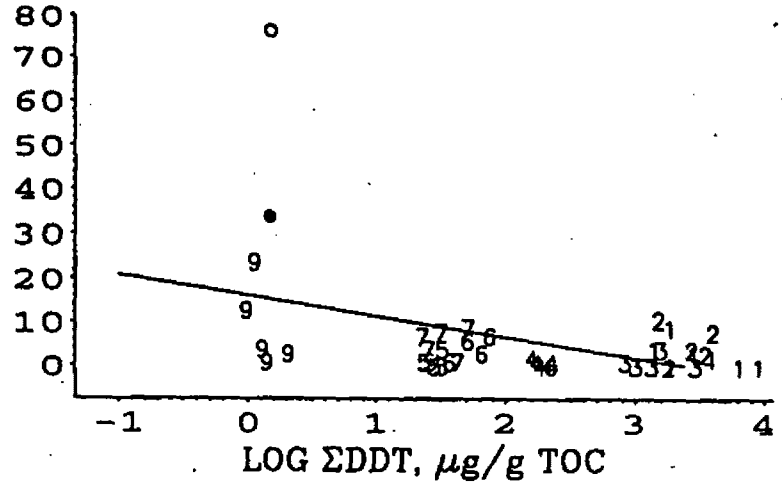


1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

[i]

NUMBER AMPHIPODA/0.015 M^2 r = 0.44 **

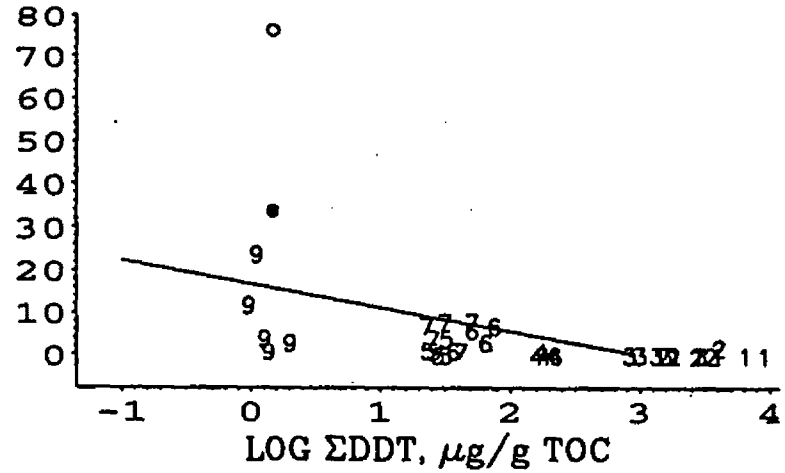


1-9 = STATIONS 1-9

• = STATION 22 o = STATION 23

[j]

NUMBER AMPHIPODA/0.015 M^2 r = 0.51 ***
EXCLUDING GRANDIDIERELLA JAPONICA



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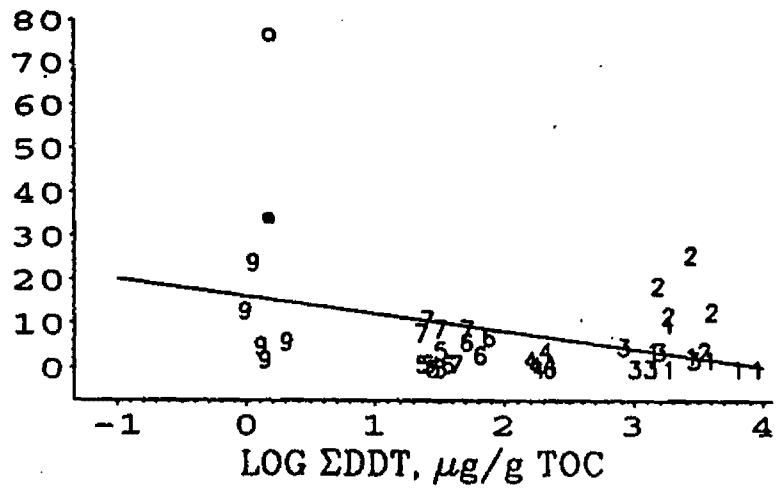
• = STATION 22 o = STATION 23

[k]

NUMBER CRUSTACEA/0.015 M^2 r = 0.35 *

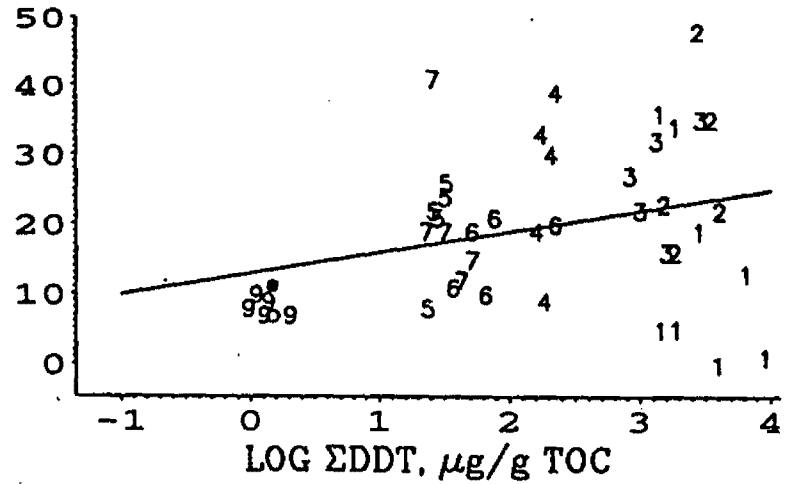


1-9 = STATIONS 1-9

• = STATION 22 o = STATION 23

[l]

NUMBER MOLLUSCA/0.015 M^2 r = 0.31 *

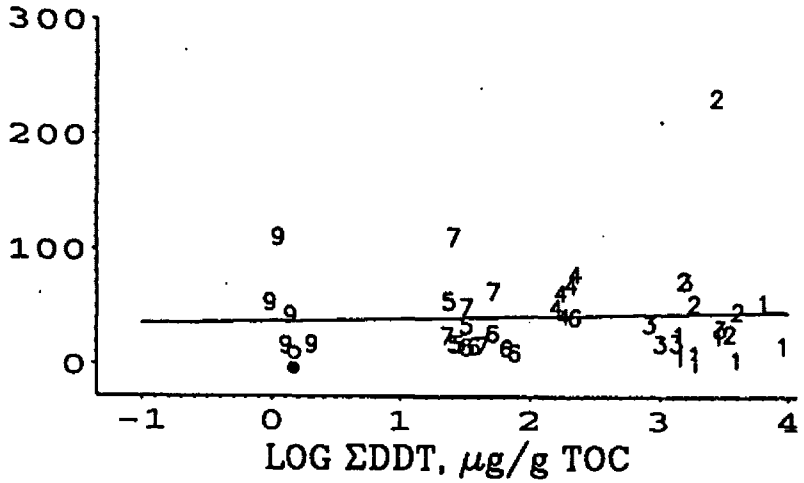


1-9 = STATIONS 1-9

• = STATION 22 o = STATION 23

(m)

NUMBER POLYCHAETA/0.015 M^2 r = 0.06 ns

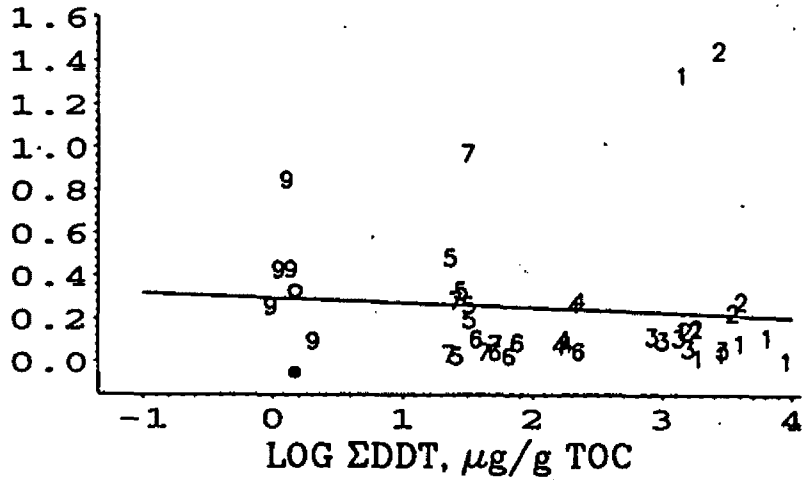


1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

(n)

POLYCHAETA BIOMASS, g/0.015 M^2 r = 0.08 ns

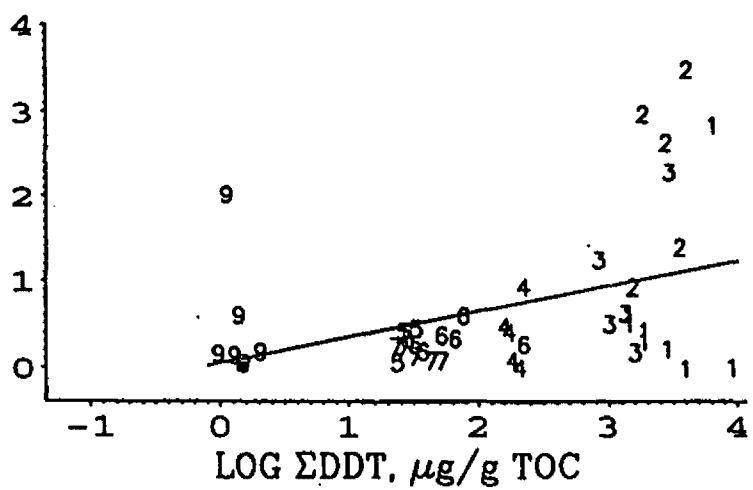


1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

(o)

MOLLUSCA BIOMASS, g/0.015 M^2 r = 0.40 **

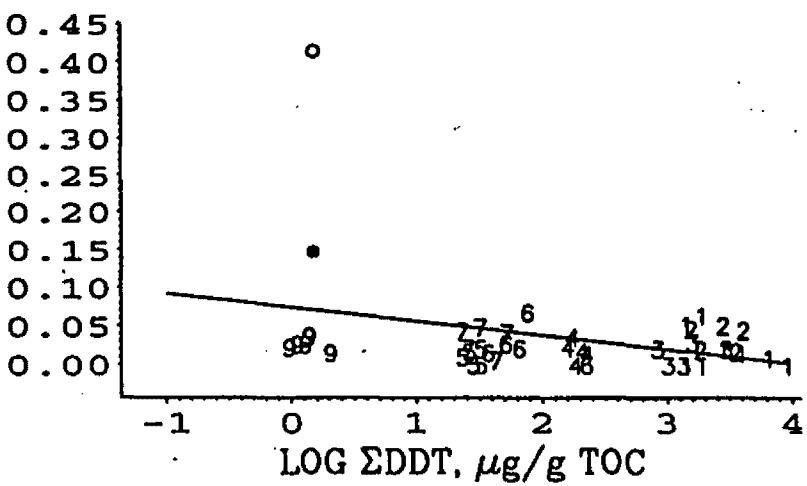


1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

(p)

CRUSTACEA BIOMASS, g/0.015 M^2 r = 0.33 *



1–9 = STATIONS 1–9

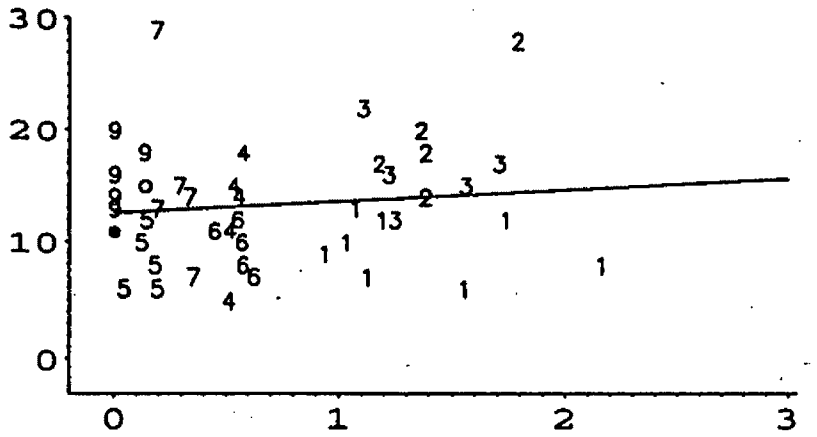
• = STATION 22 o = STATION 23

FIGURE 9-2. Sixteen measures (a-p) of macrobenthic community structure and composition as a function of $\log_{10}$(dieldrin+1 μg/g OC).

Regressions plotted as solid lines (see Table 9-2 for regression equations); r = product-moment correlation coefficient.

(a)

NUMBER SPECIES/0.015 M^2 r= 0.11 ns



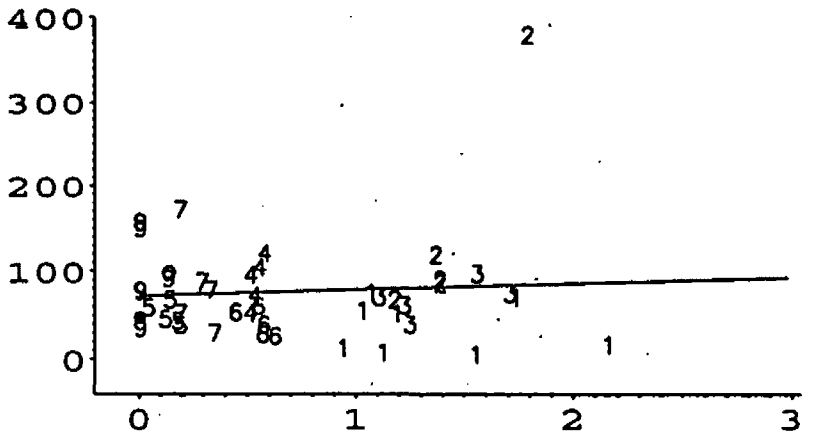
LOG (DIELDRIN + 1), $\mu g/g$ TOC

1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

(b)

ABUNDANCE/0.015 M^2 r = 0.07 ns



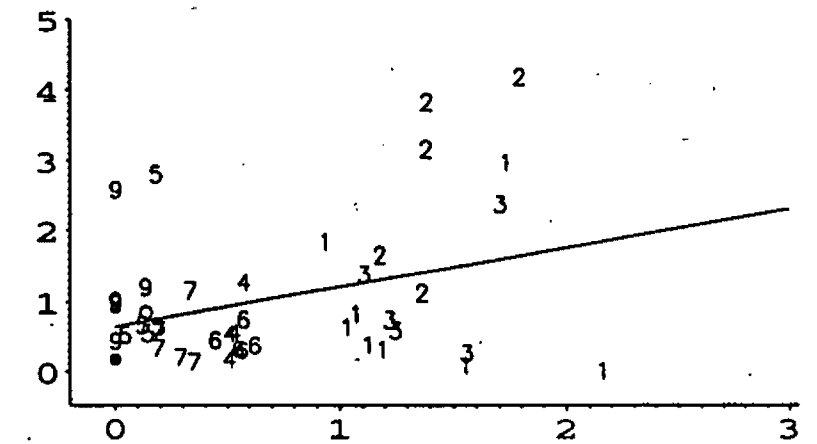
LOG (DIELDRIN + 1), $\mu g/g$ TOC

1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

(c)

TOTAL BIOMASS, g/0.015 M^2 r = 0.32 *



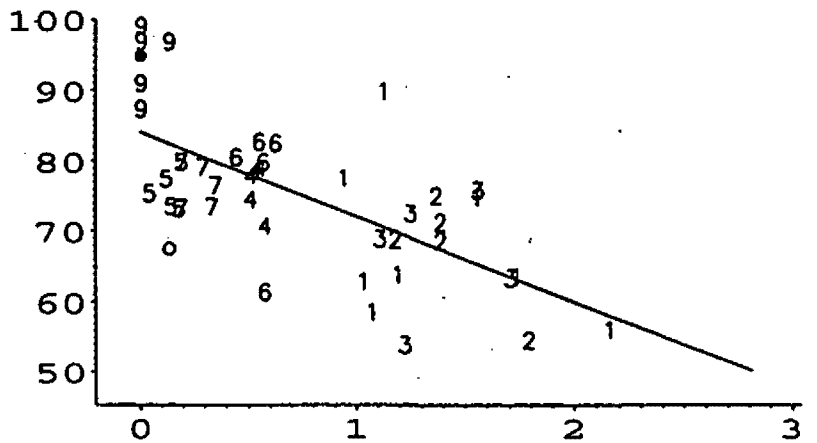
LOG (DIELDRIN + 1), $\mu g/g$ TOC

1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

(d)

INFAUNAL INDEX r = 0.65 ***

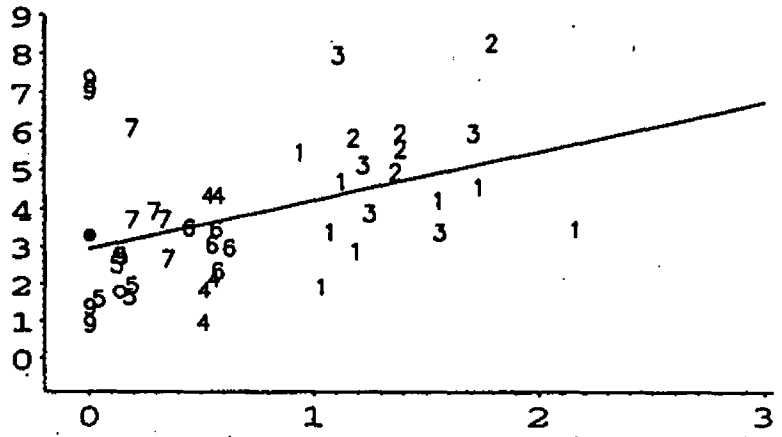


LOG (DIELDRIN + 1), $\mu g/g$ TOC

1–9 = STATIONS 1–9

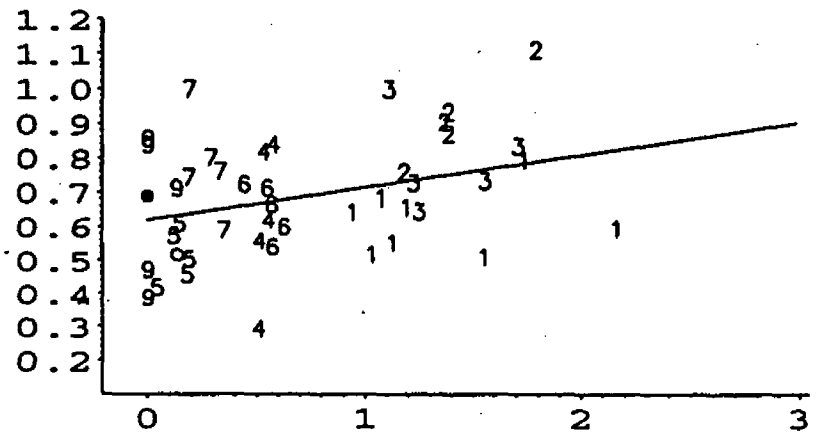
• = STATION 22 o = STATION 23

DOMINANCE [e] r = 0.42 **



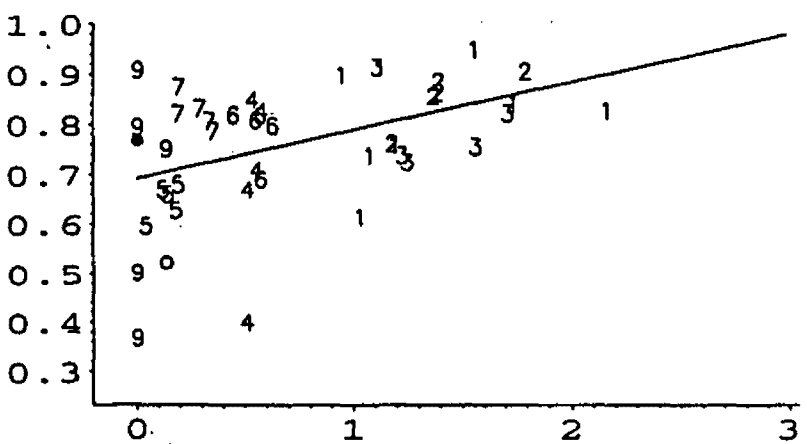
1-9 = STATIONS 1-9
• = STATION 22 o = STATION 23

BRILLOUIN'S INDEX [f] r = 0.33 *



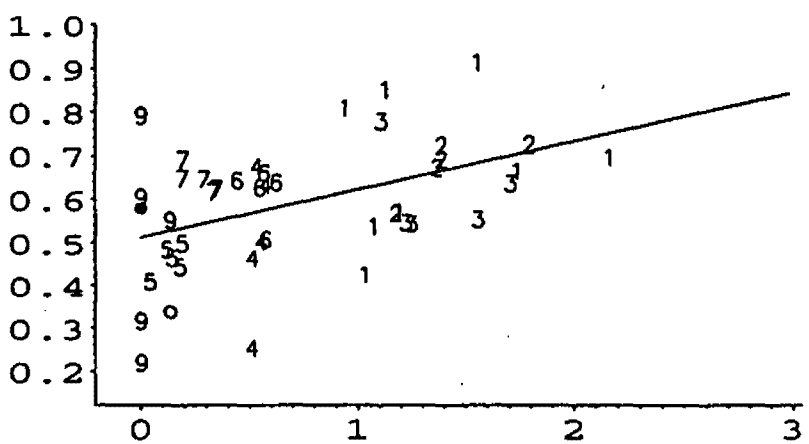
1-9 = STATIONS 1-9
• = STATION 22 o = STATION 23

1-SIMPSON'S INDEX [g] r = 0.44 **



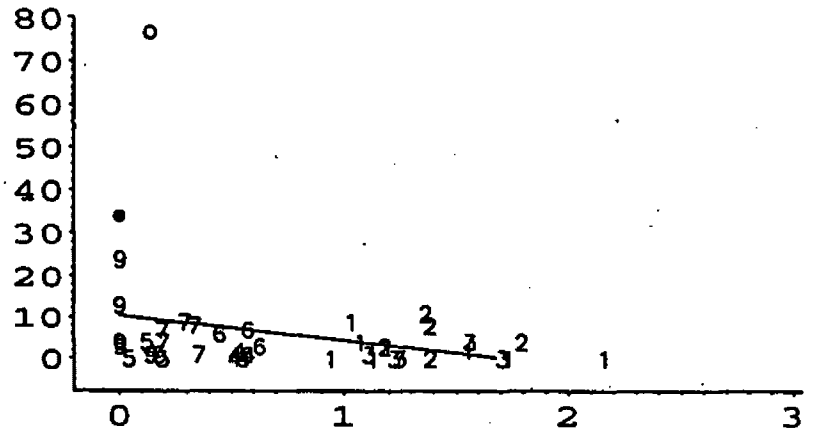
1-9 = STATIONS 1-9
• = STATION 22 o = STATION 23

MCINTOSH'S INDEX [h] r = 0.45 **



1-9 = STATIONS 1-9
• = STATION 22 o = STATION 23

NUMBER AMPHIPODA/0.015 M^2 [i] r = 0.29 ns



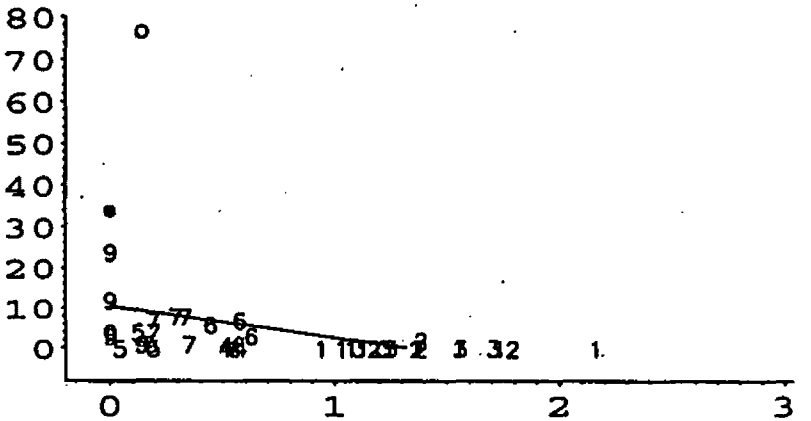
LOG (DIELDRIN + 1), $\mu g/g$ TOC

1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

NUMBER AMPHIPODA/0.015 M^2 [j] r = 0.36 *

EXCLUDING GRANDIDIERELLA JAPONICA

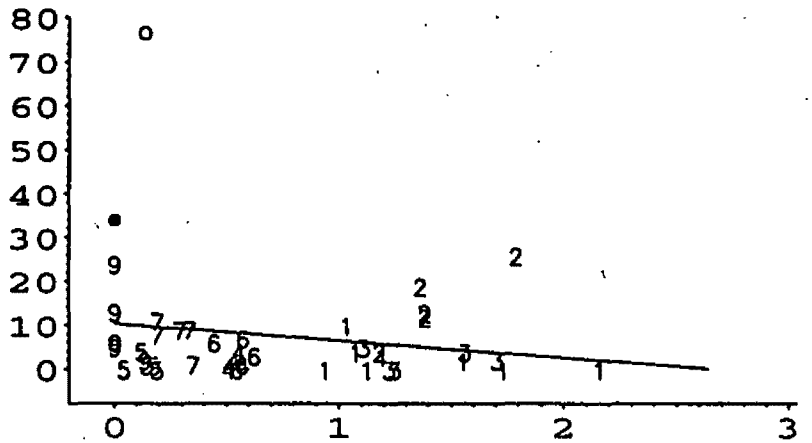


LOG (DIELDRIN + 1), $\mu g/g$ TOC

1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

NUMBER CRUSTACEA/0.015 M^2 [k] r = 0.18 ns

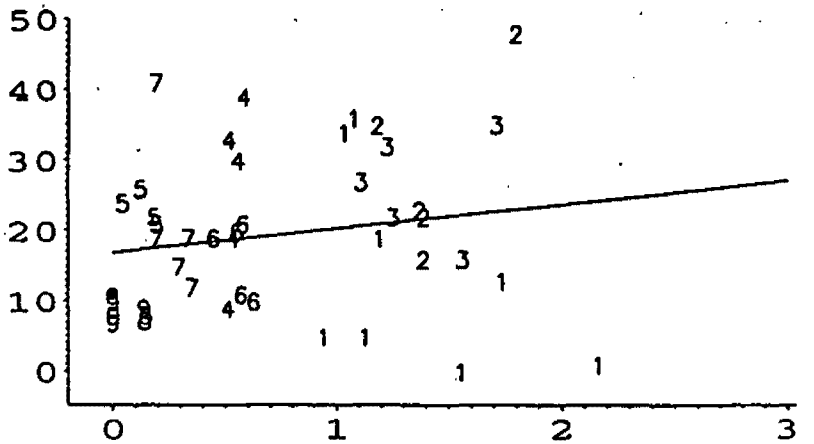


LOG (DIELDRIN + 1), $\mu g/g$ TOC

1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

NUMBER MOLLUSCA/0.015 M^2 [l] r = 0.18 ns

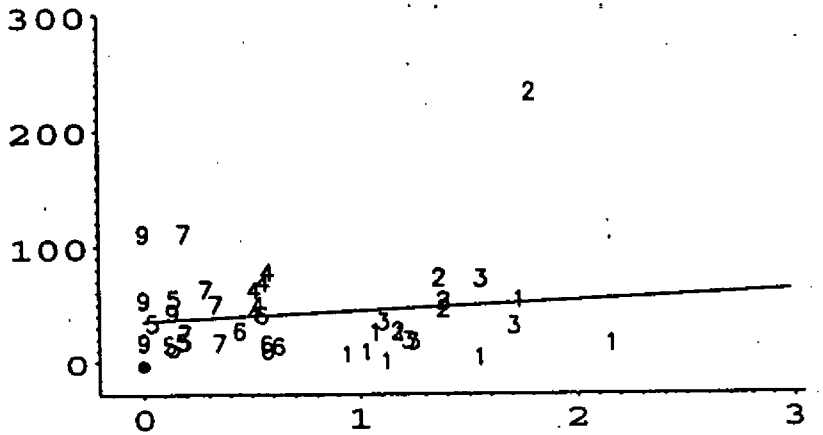


LOG (DIELDRIN + 1), $\mu g/g$ TOC

1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

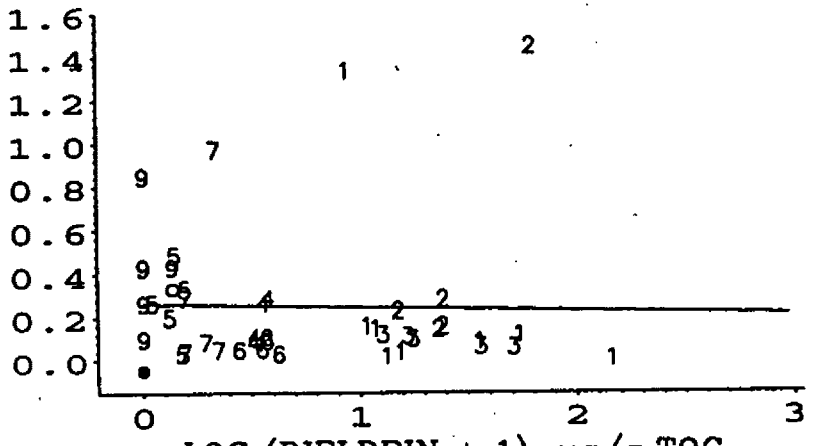
NUMBER POLYCHAETA/0.015 M^2 (m) r = 0.14 ns



1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

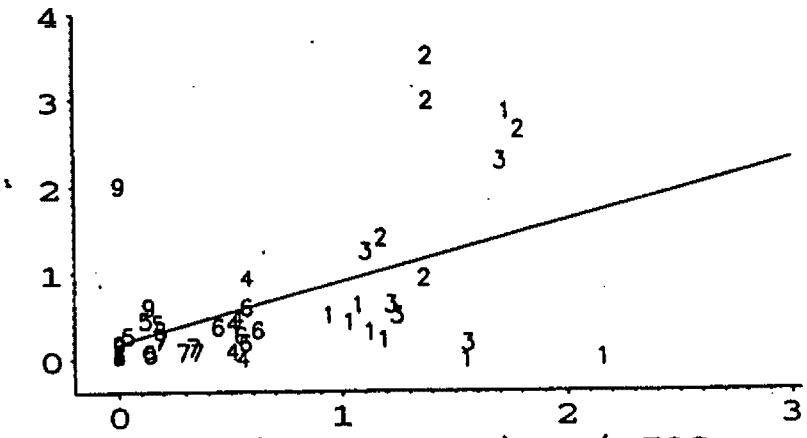
POLYCHAETA BIOMASS, g/0.015 M^2 (n) r = 0.03 ns



1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

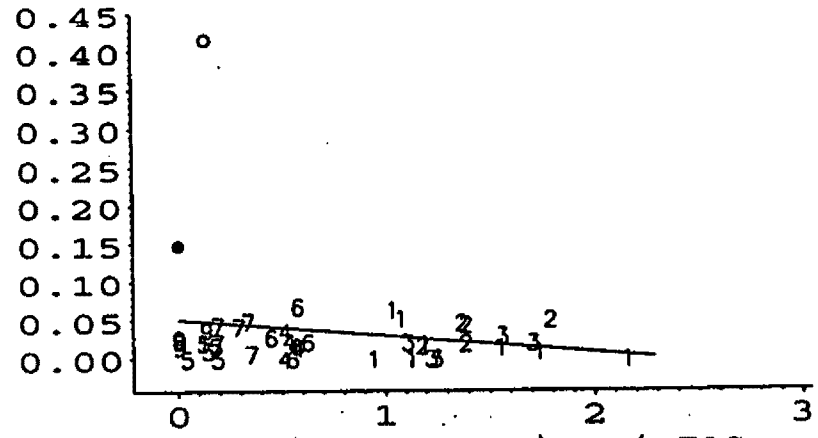
MOLLUSCA BIOMASS, g/0.015 M^2 (o) r = 0.48 ***



1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

CRUSTACEA BIOMASS, g/0.015 M^2 (p) r = 0.20 ns



1–9 = STATIONS 1–9

• = STATION 22 o = STATION 23

9.2. SEDIMENT TOXICITY TO AMPHIPODS

9.2.1. Introduction

The first objective of this portion of the study was to examine relations between sediment contamination by ΣDDT and dieldrin, acute sediment toxicity to the amphipod, Eohaustorius estuarius, and the abundance of amphipods at nine sites in the Lauritzen Channel/Santa Fe Channel/Richmond Harbor area. The second objective was to identify the lowest concentrations of DDT and dieldrin that were associated with effects on the survival and abundance of amphipods. The final objective was to evaluate the relative contribution of these compounds to toxicity and biological impacts in the study area. This evaluation was based on independent data on the toxicity and effects of DDT and dieldrin on amphipods (Swartz *et al.*, 1986, 1991; Nebeker *et al.*, 1989; Ferraro *et al.*, 1991; Hoke and Ankley, 1991).

9.2.2. Methods

Sediment and macrobenthos samples were collected on October 7-10, 1991 at a total of 9 stations: 4 in the Lauritzen Channel (Stations 1-4), 2 in the Santa Fe Channel (Stations 5-6), and 3 in the Richmond Harbor Channel (Stations 7-9). Five 0.1 m^2 van Veen grabs were taken at each station, except for Station 1 where 8 grabs were taken. Three 8.0 cm id cores, with a total area of 0.015 m^2, were taken from each grab for macrobenthic analysis. In contrast to the community analysis (Section 9.1.) in which only the 1.0 mm results were analyzed, the analysis of the amphipods in this section is based on the combination of amphipods retained on the 0.5 and 1.0 mm screens. Other procedures are described in Section 9.1.

Ten-day sediment toxicity tests were conducted with the amphipod Eohaustorius estuarius according to the ASTM (1990) guide. E. estuarius was selected as the test species because it is relatively sensitive to sediment contaminants and tolerant of a broad range of natural sediment features (*e.g.*, grain size, interstitial water salinity). On October 14, 1991 the sediment sample from each grab was mixed by hand with a spatula. Large pieces of shell or other debris were removed and approximately 175 mL of sediment were placed in a 1 L glass beaker to form a 2 cm deep layer. The sediment was covered by 775 mL of filtered, 28 °/‥ seawater, which was aerated through a 1 mL glass pipet. The beakers were placed in a 15° C water bath.

On October 9, 1991, approximately 2000 amphipods (3-5 mm total length) were collected from the shallow subtidal area along the north bank of Beaver Creek at Ona Beach State Park, Oregon. The amphipods were counted and placed in glass bowls containing a 2 cm deep layer of 0.5 mm-screened sediment from the Ona Beach site. The bowls were submerged in a flowing seawater table until the initiation of the experiment.

On October 15, 1991, the amphipods were recovered from sediment by sieving the contents of the bowls through a 1.0 mm screen. Amphipods were sequentially sorted into lots of 20 indi-

viduals, were recounted, and then each lot was placed into a randomly assigned 1 L experimental beaker. Most amphipods quickly swam or sank to the bottom and buried in the sediment. After 10 days the contents of the beakers were sieved through a 0.5 mm screen and the survivors counted. Apparently dead individuals were examined under a microscope for signs of life. Missing specimens were counted as dead (Swartz *et al.*, 1985a). Survivors were tested to see if they were able to rebury in clean sediment (ASTM, 1990). Since 99.2% of the survivors were able to rebury, we only report the mortality data in this report.

A toxicity test beaker was prepared for each sediment sample from each grab. Thus, there were 8 replicates for Station 1 and 5 replicates for Stations 2-9. In addition, negative and positive controls were conducted. Negative controls were 10 day mortality tests in 5 beakers with amphipod collection site (Ona Beach) sediment. The QA/QC requirement for the negative control is $\leq$ 10% mortality (ASTM, 1990). The negative control sediment was a fine-grained (178.0 μm median diameter, 98.8% sand), low-organic content (0.03% TOC) sand (DeWitt *et al.* 1992). Positive control was a 4 d mortality test in an unreplicated dilution series of cadmium (spiked as $CdCl_2$) in 28 ppt seawater (without sediment) at nominal concentrations of 30, 15, 7.5, 3.8, 1.9, 0.94, and 0.00 mg Cd/L. The QA/QC requirement for the positive control is that the cadmium 4 d LC_{50} be within 2 standard deviations of the mean 4 d LC_{50} in previous tests. The mean cadmium 4 d LC_{50} for the previous 16 positive control tests with E. estuarius in our laboratory is 11.8 mg/L (mean $\pm$ 2 SD = 6.3 - 17.2 mg/L). The negative and positive control beakers were processed exactly the same as the beakers containing sediment from Stations 1-9, except for the lack of sediment and shorter test duration of the positive control. Apparent, field-derived LC_{50} estimates for ΣDDT were calculated for concentrations based on sediment samples from both the grab cores at the time of field collection and the bioassay chambers at the initiation of the toxicity tests (*i.e.*, the T_0 bioassay storage sediment). There was no statistically significant difference between these LC_{50} estimates. Sediment chemical concentrations in the samples from the grab cores are, therefore, used in this report to allow comparisons of toxicity, biology, and bioaccumulation results with a single data set for sediment chemistry.

Statistical comparisons between chemical, toxicological, and biological analyses were made by correlation, ANOVA, and Student-Neuman-Keuls multiple range test (Sokal and Rohlf, 1981). LC_{50}s were calculated by probit analysis (Finney, 1971).

9.2.3. Results

Sediment Toxicity: The control results of the sediment toxicity test with E. estuarius met all QA/QC requirements. Mean mortality in the negative control sediment (amphipod collection site, Ona Beach, Oregon) was 3.0%, well within the ASTM (1990) requirement of negative control mortality $\leq$ 10% (Table 9-7). The cadmium 4 d LC_{50} in seawater was 16.9 mg/L, within 2 standard

deviations of the mean 10 day LC_{50} (11.8 mg/L) of the positive control for the previous 16 experiments with E. estuarius in our laboratory.

Mean mortality of E. estuarius exposed to sediment from the Lauritzen Channel/Santa Fe Channel/Richmond Harbor transect ranged from 23.0 to 66.3% (Table 9-7). Mean mortality in sediment from Station 1 was significantly different from the negative control (Ona Beach) sediment and sediment from all other stations on the transect. Mean mortality in sediment from Stations 1, 2, 3, and 6 was significantly different from the negative control sediment. Sediment from two of the eight grab samples from Station 1 caused 100% mortality of test specimens.

The bioassay data indicate a toxicity gradient along four stations from the head to the mouth of Lauritzen Channel (Table 9-7). Also, the mean toxicity at the stations in the Lauritzen Channel (42.0%) was greater than mean toxicity in the Santa Fe Channel (30.0%) or Richmond Harbor Channel (23.7%).

Amphipod Fauna: Eight species of amphipods were collected along the Lauritzen Channel/Santa Fe Channel/Richmond Harbor transect (Table 9-8). Three species were found in the Lauritzen Channel, two in the Santa Fe Channel, and seven in the Richmond Harbor Channel. Only two individuals were collected at Station 8, a site at which the macrobenthic community is greatly disturbed by prop scour (see Section 7.2. and Specht, 1991). Only one amphipod species, Grandidierella japonica was abundant in the Lauritzen Channel. Corophium heteroceratum was abundant in the Santa Fe and Richmond Harbor Channels. Ampelisca abdita was abundant at Richmond Harbor Station 9.

The abundance of G. japonica was inversely related to that of other amphipods (Figure 9-3). Where G. japonica was abundant or common (Lauritzen Stations 1-4), all other amphipods were rare or absent. Where other amphipods were abundant or common (Stations 5, 6, 7, 9), G. japonica was rare or absent.

Sediment Chemistry: The mean concentration of dieldrin and the sum of DDT metabolites (ΣDDT) and the total organic carbon along the Lauritzen Channel/Santa Fe Channel/Richmond Harbor transect are given in Tables 8-1 and 8-2. There was a very strong gradient of chemical contamination along the transect. The first three stations in the Lauritzen Channel (Stations 1-3) were highly contaminated by both dieldrin (25.8-35.2 μg/g OC) and ΣDDT (1,520-3,500 μg/g OC). The concentrations of dieldrin and ΣDDT at the other six stations were about an order-of-magnitude or more below the concentrations at Lauritzen Stations 1-3. However, a contamination gradient is evident even at these six stations, with minimal concentrations of dieldrin (0.07 μg/g OC) and ΣDDT (1.34 μg/g OC) at Station 9, farthest from the Lauritzen Channel. Concentrations of organic carbon-normalized dieldrin and ΣDDT were highly correlated among the nine stations ($r = 0.93$, $p < 0.001$).

The concentration of Aroclor 1254 was 118 μg/kg (6.7 μg/g OC) at Station 2, and 607 μg/kg (20 μg/g OC) at Station 6 (Table 8-8). The concentration of Aroclor 1260 was about half that of

Aroclor 1254 at Stations 2 and 6. The presence of neither Aroclor 1254 nor 1260 was confirmed in the sediment from Station 9. Aroclor 1242 was not detected at Stations 2, 6, or 9. PAH concentrations were also substantially higher at Station 6 than at Station 2, and lowest at Station 9 (Table 8-9).

The concentrations of AVS and most metals were highest at Stations 1 and 6 and lowest at Stations 8 and 9 (Table 8-10). The ratio of total SEM/AVS on a molar basis ranged from 0.05 at Station 1 to 0.48 at Station 9.

Relations between Chemistry, Toxicity and Amphipod Abundance: The high correlation between dieldrin and ΣDDT results in virtually identical patterns in the relations between both chemical parameters and the biological response indicators (toxicity and amphipod abundance). There were no substantial differences in mortality of E. estuarius at concentrations of dieldrin < 3 μg/g OC or at concentrations of ΣDDT < 200 μg/g OC (Figures 9-4, 9-5). At Lauritzen Stations 1-3, where the concentrations of dieldrin and ΣDDT ranged from 25-35 μg/g OC and 1,500-3,500 μg/g OC, respectively, there were significant increases in sediment toxicity.

The abundance of amphipods (excluding G. japonica) in the field was reduced at stations with lower chemical contamination than that at which toxicity began to increase. Amphipod abundance was variable, but often > 40 individuals/0.1 m^2 at dieldrin and ΣDDT concentrations of < 3 μg/g OC and < 100 μg/g OC, respectively (Figures 9-6, 9-7). Few or no amphipods (except G. japonica) were collected at Stations 1-4, where the concentrations of dieldrin and ΣDDT ranged from 2.46-35.2 μg/g OC and 189-3,500 μg/g OC, respectively.

The abundance of G. japonica was positively correlated to the toxicity ($r = 0.29$, $p < 0.05$; Figure 9-10) and ΣDDT and dieldrin contamination gradients ($r = 0.54$-0.58, $p < 0.001$; Figures 9-8, 9-9). G. japonica was most abundant when the concentration of the sum of DDT metabolites exceeded 1,500 μg/g OC, the concentration of dieldrin exceeded 25 μg/g OC, and the mortality of E. estuarius in laboratory toxicity tests exceeded 35%.

The abundance of amphipods other than G. japonica was inversely correlated to sediment toxicity ($r = -0.34$, $p < 0.05$, Figure 9-11). Mortality of E. estuarius was highest at Lauritzen Stations 1-3 where few or no amphipods except G. japonica were collected.

9.2.4. Discussion

Relative Contributions of DDT and Dieldrin to Impacts: The correlations between dieldrin, ΣDDT, toxicity, and amphipod abundance along the Lauritzen Channel/Santa Fe Channel/Richmond Harbor transect are characteristic of sediment contamination gradients within which many parameters of chemistry, biology and toxicity may be significantly related to one another (Swartz *et al.*, 1985b). The correlations demonstrate covariance, not causality. Experimental evidence is necessary to discriminate the relative contributions of dieldrin and ΣDDT to biological

impacts. Fortunately, experimental and field data are available to make this discrimination. The experimental data come from sediment toxicity tests in which DDT or dieldrin were spiked into uncontaminated sediment (Nebeker *et al.*, 1989, Hoke and Ankley, 1991). The field data are comparisons of contaminant-toxicity-biology relations between Lauritzen Channel/Santa Fe Channel /Richmond Harbor Channel transect (present study) and the DDT contamination gradient on the Palos Verdes Shelf, CA (Swartz *et al.*, 1985b, 1986, 1991; Ferraro *et al.*, 1991).

The 10 day LC_{50} of dieldrin in sediment to the amphipod, Hyalella azteca is 1,955 μg/g OC (mean for three sediments, range: 1,073 - 3,682 μg/g OC; Hoke and Ankley, 1991). The 10 day LC_{50} of 4,4'DDT in sediment to the amphipod, H. azteca is 371 μg/g OC (mean for three sediments, range: 272 - 473 μg/g OC; Nebeker *et al.*, 1989). The 10 d LC_{50} of PCB (Aroclor 1254) in sediment to R. abronius is 2,600 μg/g OC (Swartz *et al.*, 1988). These LC_{50} estimates can be used to convert the concentrations of dieldrin and ΣDDT to toxic units. A toxic unit is the concentration of a chemical that kills 50% of test specimens in a toxicity test (Sprague, 1970). The number of toxic units of PCB, dieldrin or DDT in a sediment sample is therefore the measured concentration divided by the LC_{50}. Sediment concentrations of PAH compounds were converted to toxic units by the ΣPAH model (Swartz *et al.*, in review).

When the dieldrin concentrations on the Lauritzen Channel/Santa Fe Channel/Richmond Harbor transect (Figure 9-4) are divided by the dieldrin LC_{50} for H. azteca, the maximum concentration of dieldrin toxic units is: (Station 1 conc.)/(LC_{50}) = (35.2 μg/g OC)/(1,955 μg/g OC) = 0.018 toxic units (Figure 9-12). At a maximum concentration of 0.018 toxic units, dieldrin was unlikely to contribute to the toxicity of Lauritzen Channel sediment to E. estuarius. At Station 2 there were 0.003 sediment toxic units of Aroclor 1254 and 0.284 sediment toxic units of PAHs. In contrast, the maximum concentration of ΣDDT toxic units in the Lauritzen Channel was: (3,500 μg/g OC)/(371 μg/g OC) = 9.43 toxic units (Figure 9-13). ΣDDT toxic units at Stations 2 and 3 were 7.30 and 4.10, respectively. These concentrations were more than sufficient to account for the observed mortality. At Lauritzen Station 2, the toxic unit ΣDDT concentration was 25 times greater than that of PAHs, 500 times greater than that of dieldrin, and 2,500 times greater than that of Aroclor 1254. The ΣDDT concentration in sediment decreased substantially at Stations 4 and 6 to 0.50 and 0.25 toxic units, respectively. Such levels of contamination might cause a slight increase in percent mortality, although a statistically significant increase relative to the negative control was only found at Station 6 (Table 9-7), where the PAH toxic unit concentration increased to 1.66. As noted in Section 6.6.7., Station 6 was chosen as a "worst case" for oil contamination. Sediment at Stations 5 and 7-9 contained less than 0.1 toxic units of ΣDDT, and exerted no significant toxicity relative to the control. Sediment at Station 9 had less than 0.01 toxic units of Aroclor

1254 and PAHs.

AVS is the sediment phase that determines the toxicity of metals that form insoluble sulfides (DiToro *et al.*, 1990). No metal toxicity would be expected when the sum of the molar concentrations of divalent metals is less than that of AVS. Thus, metals (Cu, Zn, Pb, Ni, Cd, Ag) were unlikely to contribute to the toxicity observed at the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations because the metals/AVS molar ratio was always $\leq$ 0.48.

The toxic unit analysis of the relative contributions of PAH, Aroclor 1254, dieldrin and ΣDDT is based on an interspecies extrapolation of LC_{50} values. H. azteca and E. estuarius have similar sensitivity to fluoranthene, another neutral organic compound. Comparative toxicological data show that the 10 d LC_{50} for fluoranthene in sediment differs only by a factor of 3 among these amphipod species (15.4, 10.6, and 5.1 mg/kg for H. azteca, E. estuarius and R. abronius, respectively; DeWitt *et al.*, 1989). Thus, the potential error associated with interspecies extrapolation of LC_{50}s is probably small relative to the major difference in the toxic unit concentrations of DDT, dieldrin, PAH, and Aroclor 1254 in the Lauritzen Channel.

The second independent evaluation of DDT as the cause of toxic effects is based on the apparent correspondence between DDT-toxicity relations in the Lauritzen Channel and the Palos Verdes Shelf, another site where DDT is a dominant contaminant. In surveys of surficial sediment near the Los Angeles County sewage outfall on the Palos Verdes Shelf in 1980, 1983, and 1986, mortality in toxicity tests with the amphipod, Rhepoxynius abronius, never exceeded 23%, ΣDDT never exceeded 300 μg/g OC, and there was no significant correlation between DDT and toxicity (Figures 9-5, 9-13; Swartz *et al.*, 1985b, 1986; Ferraro *et al.*, 1991). In 1985, sediment toxicity tests with R. abronius were conducted on sediments from sections of 50 cm deep cores collected on the Shelf near the outfall (Figures 9-5, 9-13; Swartz *et al.*, 1991). ΣDDT in the core sections was as great as 2,000 μg/g OC, mean mortality was as high as 90%, and mortality was significantly correlated with ΣDDT. The relation between ΣDDT and sediment toxicity to amphipods is very similar on the Palos Verdes Shelf and along the Lauritzen Channel/Santa Fe Channel/Richmond Harbor transect (Figure 9-5). The apparent 10 day LC_{50} for ΣDDT based on field data for the Lauritzen (2,500 μg/g OC) and Palos Verdes (1,044 μg/g OC) investigations agree within a factor of three, despite differences in test species.

There is also good agreement between ΣDDT and the distribution of amphipods in the Lauritzen Channel and on the Palos Verdes Shelf (Figure 9-7; Swartz *et al.*, 1985b, 1986; Ferraro *et al.*, 1991). In the Lauritzen Channel, few or no amphipods (except G. japonica) were collected at ΣDDT greater than about 100 μg/g OC. On the Palos Verdes Shelf few or no amphipods were collected at ΣDDT greater than about 200 μg/g OC.

Thus, two independent kinds of data implicate ΣDDT as a major factor causing sediment toxicity and amphipod population impacts

in the Lauritzen Channel. First, the metals/AVS molar ratio and the toxic units of dieldrin, Aroclor 1254 and ΣPAH are too low, while the toxic units concentration of ΣDDT is sufficient to exert acute toxicity. Second, the concentration-response patterns among sediment toxicity, amphipod abundance, and ΣDDT are very similar in two independent sites of DDT contamination. Together, these data provide good evidence that ΣDDT is the dominant ecotoxicological factor in the Lauritzen Channel.

Effects Concentrations for ΣDDT: The 10 day LC_{50} for ΣDDT in field-collected sediment was 1,044 μg/g OC for R. abronius in the Palos Verdes Shelf study and 2,500 μg/g OC for E. estuarius in the Lauritzen Channel. The threshold for 10 day toxicity occurred at about 300 μg/g OC in both studies (Figure 9-5). Chronic effects of ΣDDT on amphipod populations in the field were evident at concentrations lower than those that caused acute toxicity in the laboratory. Amphipod abundance sharply declined at ΣDDT concentrations between 100-300 μg/g OC on the Palos Verdes Shelf and in the Lauritzen Channel (Figure 9-7). The minimum ecotoxicological effects concentration appears to be about 100 μg ΣDDT/g OC.

Comparison of Toxicity to Other Coastal Sites: Mean mortality of E. estuarius in the Lauritzen Channel/Santa Fe Channel/-Richmond Harbor (33.2%) is comparable to mean mortality of E. estuarius in the NW portion of the Hylebos Waterway, WA (38.3%), another Superfund site (DeWitt *et al.*, 1989). Few other data are available on the toxicity of sediment from contaminated field sites to E. estuarius. However, the present bioassay data can be compared with sediment toxicity surveys conducted with R. abronius in other U.S. coastal areas (Table 9-9), since E. estuarius is typically only slightly less sensitive than R. abronius (DeWitt *et al.*, 1989). Lauritzen Station 1 with a mean mortality of 66.2% is a severe case of sediment toxicity. It ranks between the Houston Ship Channel, TX and the creosote-contaminated Elizabeth River, VA among the cases reported by Swartz *et al.* (1989). The mean mortality in sediment from all stations in the Lauritzen Channel/Santa Fe Channel/Richmond Harbor (33.2%) is typical of urbanized embayments that contain some sites of severe sediment contamination, *e.g.*, San Diego Bay, CA.

Pinza *et al.* (1992) conducted sediment toxicity tests with R. abronius on composites of sediment samples from each of three groups of five stations in Richmond Inner Harbor. They found no significant difference from their reference sediment at two sets of stations within the Richmond Inner Harbor Channel (mean percent mortality = 17% and 13%). There was a statistically significant increase in mortality of amphipods exposed to a composite sediment sample from five stations on the southeast bank of the Richmond Inner Harbor Channel (mean percent mortality = 42%). The mean percent mortality of R. abronius for all of the Pinza *et al.* (1992) toxicity tests of Richmond Harbor sediment (24.0%) is virtually identical to the mean percent mortality of E. estuarius in the present toxicity tests of Richmond Inner Harbor sediment (23.7%, Table 9-7).

White *et al.* (1993) conducted sediment toxicity tests with R. abronius on composites of sediment samples from the Lauritzen Channel, Santa Fe Channel, and Richmond Inner Harbor. They observed 100% mortality of R. abronius exposed to four composites from the Lauritzen Channel, 26% mortality in composites from the mouth of Lauritzen Channel and the Santa Fe Channel, and 11-15% mortality in the Richmond Inner Harbor. The spatial distribution of mortality of R. abronius reported by White *et al.* (1993) is very similar to that reported here for E. estuarius, except that R. abronius was more sensitive to Lauritzen Channel sediment than E. estuarius.

Sediment contamination and toxicity can be very patchy, even on small spatial scales (Swartz *et al.*, 1982, 1989). Patchiness is shown at Station 1 by the wide range in percent mortality from 35 - 100% for sediment samples from the eight grabs taken only meters apart (Table 9-7, 9-9). Mortality was 100% in sediment from two of the grabs, indicating the occurrence of small patches of sediment that cause extremely high toxicity.

In the absence of chemical contamination, E. estuarius is tolerant of a very broad range of sediment conditions. Mortality of E. estuarius in apparently uncontaminated sediment collected in Puget Sound and the central Oregon coast averaged 5.6% and rarely exceeded 20% (Figure 9-14; DeWitt *et al.*, 1989). In contrast, even at stations with low ΣDDT and dieldrin concentrations (*e.g.*, Stations 8 and 9) mean mortality of E. estuarius was ≥ 23% (Table 9-7, Figures 9-4,9-5, 9-12, 9-13). Some unknown factor(s), apparently, are causing a higher background level of mortality in the present study area than in the Puget Sound/Oregon reference stations. These unknown factor(s) may be unmeasured contaminants, natural sediment features, or interactions between several factors. They are probably not sediment grain size, salinity, dieldrin, or ΣDDT.

Tolerance of Grandidierella japonica to Sediment Contamination: G. japonica has been recommended as a sediment toxicity test species in a number of research and regulatory projects including the U.S. program for evaluating the acceptability of dredged material for ocean disposal (Reish and LeMay, 1988; Nipper *et al.*, 1989; ASTM, 1990; U.S. EPA/COE, 1991). Limited toxicological comparisons with other amphipods have shown that G. japonica has comparable or lower sensitivity to field sediments and reference toxicants (Anderson *et al.*, 1988; Hong and Reish, 1987). Field validation of the acceptability of an amphipod as a toxicity test species assumes that the distribution of the species along pollution gradients is positively related to the distribution of sensitive species and negatively related to contamination and toxicity (Swartz *et al.*, 1985b). However, the abundance of G. japonica along the Lauritzen Channel/Santa Fe Channel/Richmond Harbor transect was negatively related to that of other amphipods (Figure 9-3) and positively related to sediment toxicity to both E. estuarius and R. abronius and chemical contamination gradients (Figures 9-8, 9-9, 9-10; White *et al.*, 1993). G. japonica had a similar distribution in San Diego Bay,

where it was most common when other amphipods were rare or absent and 10 day sediment toxicity to the amphipod, R. abronius was highest (Table 9-10). The suitability of G. japonica as a sediment toxicity test species is questionable because of its natural presence and possible stimulation in contaminated, toxic sediment avoided by other amphipods.

Field and toxicological data show that most amphipod species have a relatively high sensitivity to sediment contaminants, especially for chemicals such as DDT that are designed to kill arthropods (Swartz *et al.* 1985b). This does not appear to be true for G. japonica, because it was abundant in habitats occupied by few or no other amphipod species, and it tolerated sediments that caused severe acute toxicity to E. estuarius and R. abronius. G. japonica should therefore be excluded in the present study from analyses of relations between toxicity, chemical contamination, and the abundance of sensitive species.

TABLE 9-7. Mortality of the infaunal amphipod, Eohaustorius estuarius, during 10-day exposures to sediment from the Lauritzen Channel/Santa Fe Channel/Richmond Inner Harbor transect.

Station	Number Dead (N=20) Grab Number 1	2	3	4	5	6	7	8	Mean Number Dead
1	20	14	9	11	12	13	7	20	13.2
2	3	11	8	9	12				8.6
3	5	10	7	8	6				7.2
4	4	7	3	4	5				4.6
5	5	4	5	9	5				5.6
6	5[1]	12	8	3	4				6.4
7	2	5	2	6	8				4.6
8	5	2	8	2	7				4.8
9	1	5	7	4	7				4.8
Negative Control	0	1	0	0	2				0.6

Student-Newman-Keuls multiple range test[2]:

Station	1	2	3	6	5	8	9	4	7	Negative Control
Percent Mortality	66.2	43.0	36.0	32.0	28.0	24.0	24.0	23.0	23.0	3.0

Positive Control: Cadmium (mg/L): 35 18 8.7 4.6 2.2 1.1 0.0

Mortality (N = 20): 19 6 7 3 2 1 1

4 d LC_{50} (± 95% C.L.) = 16.9 (13.1-21.7) mg/L cadmium

1 = Mortality adjusted for double seeding.

2 = Means connected by a solid line are not significantly different ($p \geq 0.05$).

TABLE 9-8. Species composition and abundance of amphipods along the Lauritzen Channel/Santa Fe Channel/Richmond Inner Harbor transect.

	Number of Individuals/0.075 m^2 Station								
Amphipod Species	1	2	3	4	5	6	7	8	9
Grandidierella japonica	28	64	14	6			2		1
Caprella mutica		2							
Corophium heteroceratum				1	7	34	70	2	18
Ampelisca abdita						1	3		77
Listriella goleta							1		
Dulichia rhabdoplastis									6
Caprella incisa									2
Aorides columbiae									1

Note: abundance figured as the total for all five grabs; Station 1 (eight grabs) adjusted to five grab basis.

TABLE 9-9. Comparison of the mortality of Eohaustorius estuarius in sediment from the Lauritzen Channel/Santa Fe Channel/Richmond Inner Harbor transect (*) with the mortality of Rhepoxynius abronius at other U.S. Coastal sites (Swartz, 1989).

Sediment Source	Percent Mortality Mean	Range	Comment
Yaquina Bay, OR	2	0 - 5	Amphipod collection site
NW Santa Monica Bay, CA	7	0 - 5	Reference site
Palos Verdes Shelf, CA	16	0 - 40	Near sewage outfall
San Diego Bay, CA	29	0 - 100	56 stations
Mean Lauritzen/Santa Fe/Richmond*	33.2	5 - 100	Mean, Present study
Nearshore Commencement Bay, WA	38	5 - 95	9 stations
South San Francisco Bay, CA	45	20 - 100	26 stations
Hylebos Waterway, WA	52	5 - 100	26 stations, Superfund site
Houston Ship Channel, TX	60	45 - 80	5 stations
Lauritzen Channel (Station 1)*	66.2	35 - 100	Worst case, Present study
Elizabeth River, VA	78	30 - 100	Creosote contamination
New York Bight, NY	88	80 - 95	Sludge Dump site
San Diego, CA, Station 28	97.5	90 - 100	Worst case, San Diego
Eagle Harbor, WA, Station 8	100		Creosote contamination

TABLE 9-10. Abundance of Grandidierella japonica in relation to that of all other amphipod species, and 10-day sediment toxicity to Rhepoxynius abronius at nine stations in San Diego Bay (R. C. Swartz, unpublished data).

Station	Percent Mortality	Amphipod Density ($N/0.1\ m^2$) Grandidierella japonica	Other Amphipods
28	97.5	5.2	6.5
27	58.0	2.6	0.0
29	24.0	5.2	9.1
11	12.5	10.4	19.5
15	12.0	3.9	2.6
40	11.0	0.0	72.8
33	10.0	0.0	102.6
13	8.0	1.3	75.4
44	2.5	0.0	843.2

FIGURE 9-3. The abundance of Grandidierella japonica in relation to the abundance of other amphipods.

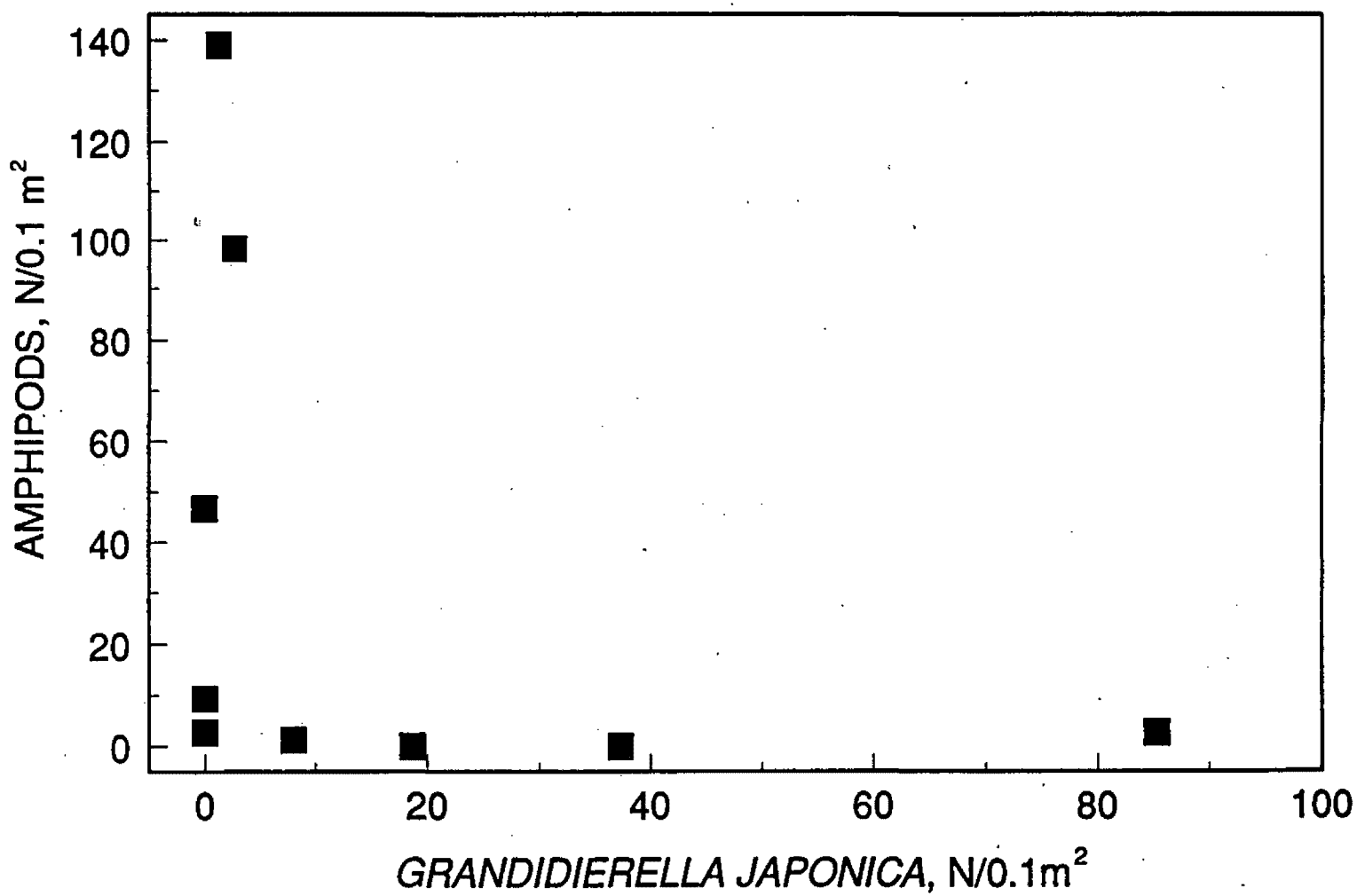

140
120
100
80
60
40
20
0
AMPHIPODS, N/0.1 m²
0
20
40
60
80
100
GRANDIDIERELLA JAPONICA, N/0.1m²

FIGURE 9-4. Mortality of Eohaustorius estuarius in relation to the concentration of dieldrin.

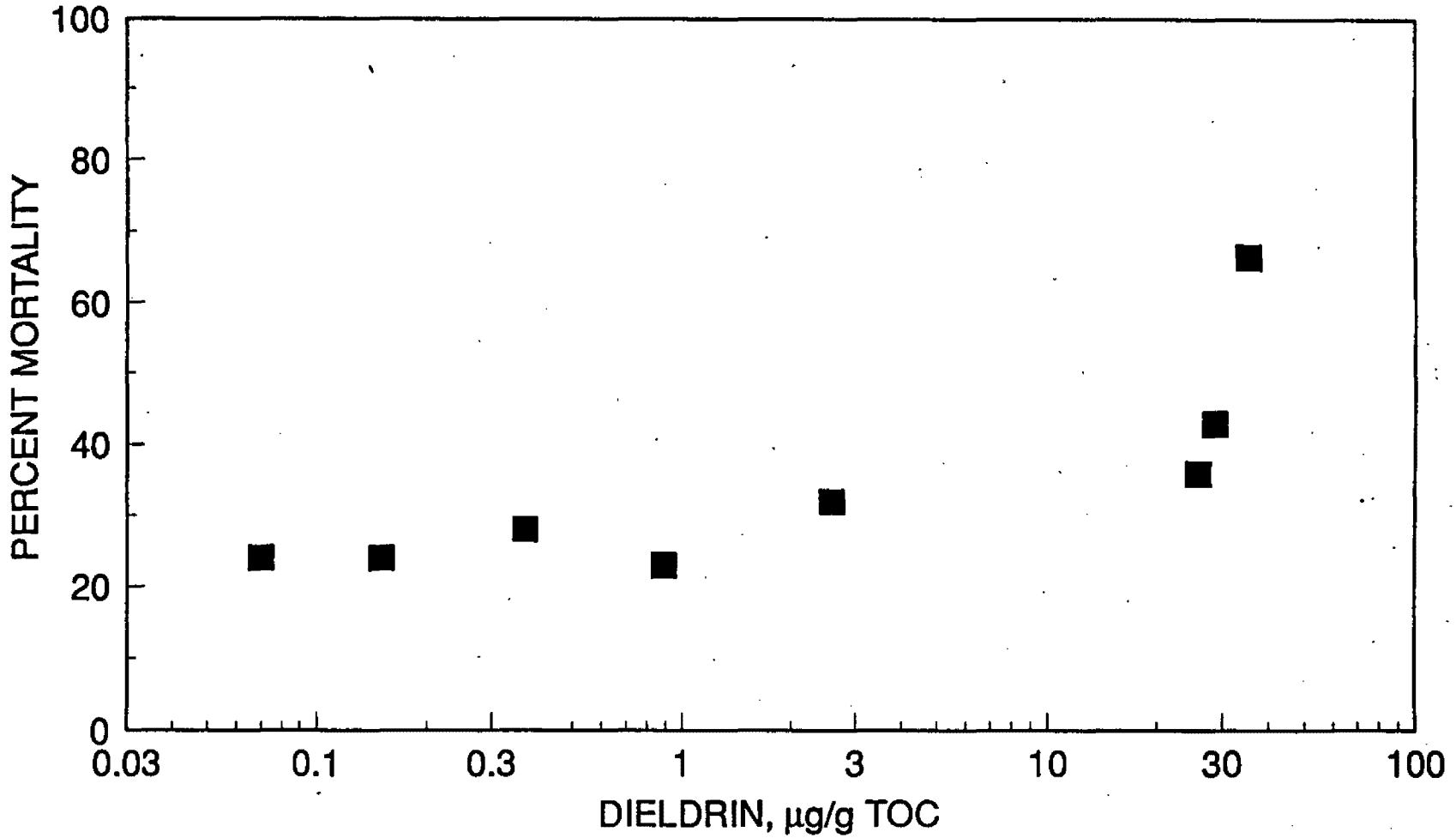

100
80
60
40
20
0
PERCENT MORTALITY
0.03
0.1
0.3
1
3
10
30
100
DIELDRIN, µg/g TOC

FIGURE 9-5. Mortality of Eohaustorius estuarius in Lauritzen Channel/Richmond Harbor sediment and Rhepoxynius abronius in Palos Verdes Shelf sediment in relation to the sum of the concentrations of DDT metabolites.

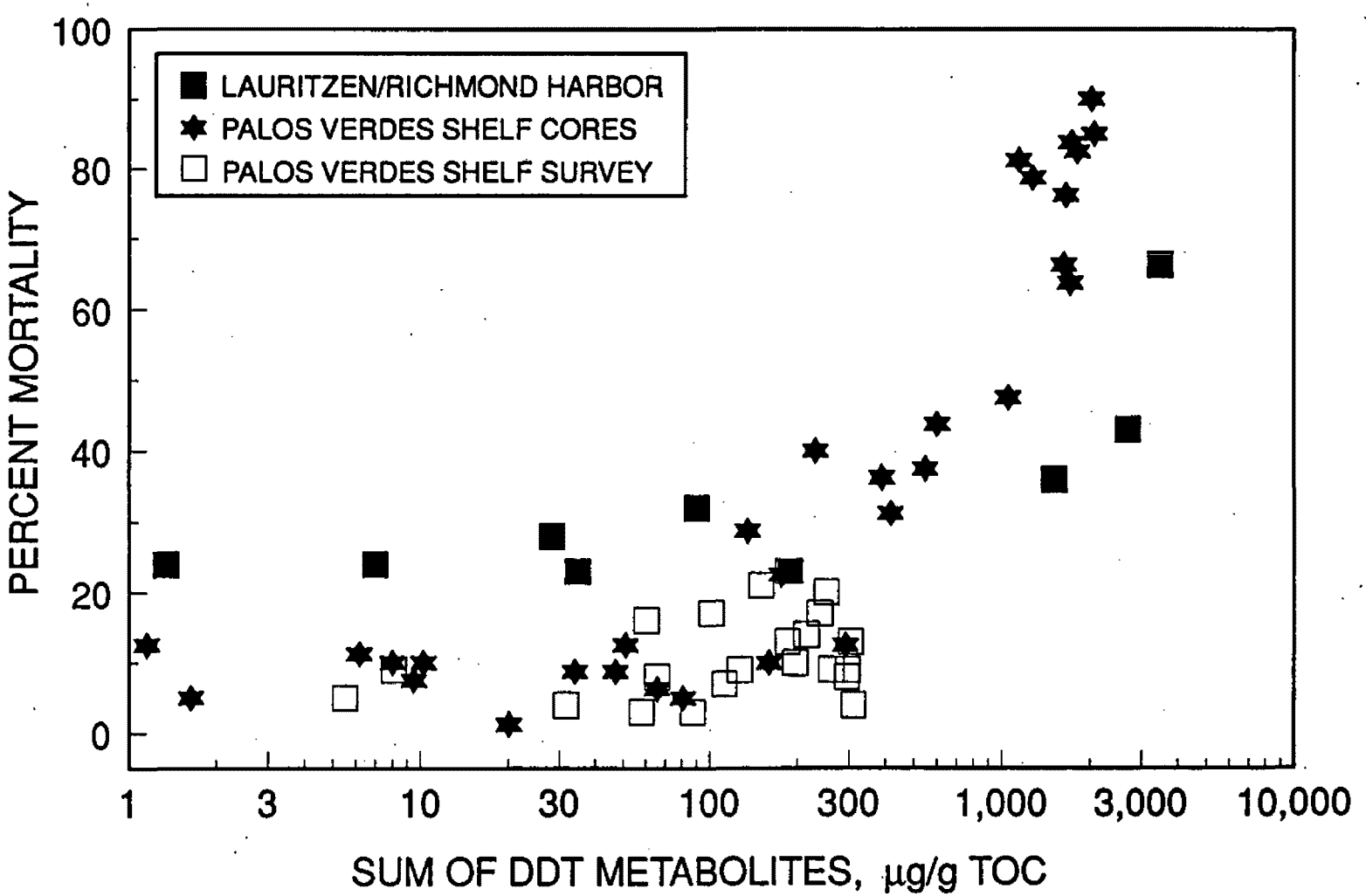

LAURITZEN/RICHMOND HARBOR
PALOS VERDES SHELF CORES
PALOS VERDES SHELF SURVEY
PERCENT MORTALITY
100
80
60
40
20
0
1
3
10
30
100
300
1,000
3,000
10,000
SUM OF DDT METABOLITES, µg/g TOC

FIGURE 9-6. The abundance of amphipods (excluding Grandidierella japonica) in relation to the concentration of dieldrin.

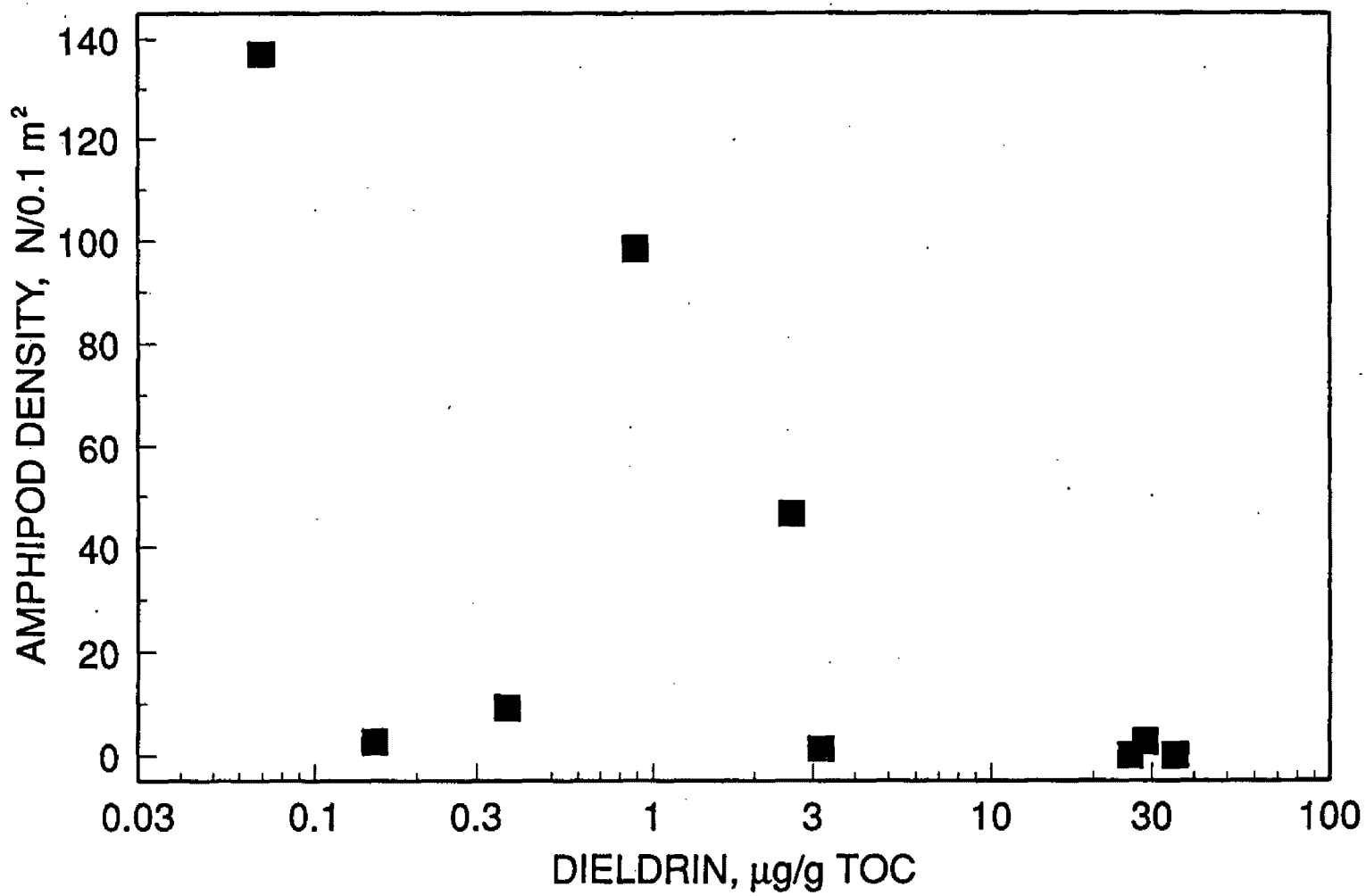

AMPHIPOD DENSITY, N/0.1 m²
140
120
100
80
60
40
20
0
0.03
0.1
0.3
1
3
10
30
100
DIELDRIN, µg/g TOC

FIGURE 9-7. The abundance of amphipods (excluding Grandidierella japonica) in relation to the sum of the concentrations of DDT metabolites.

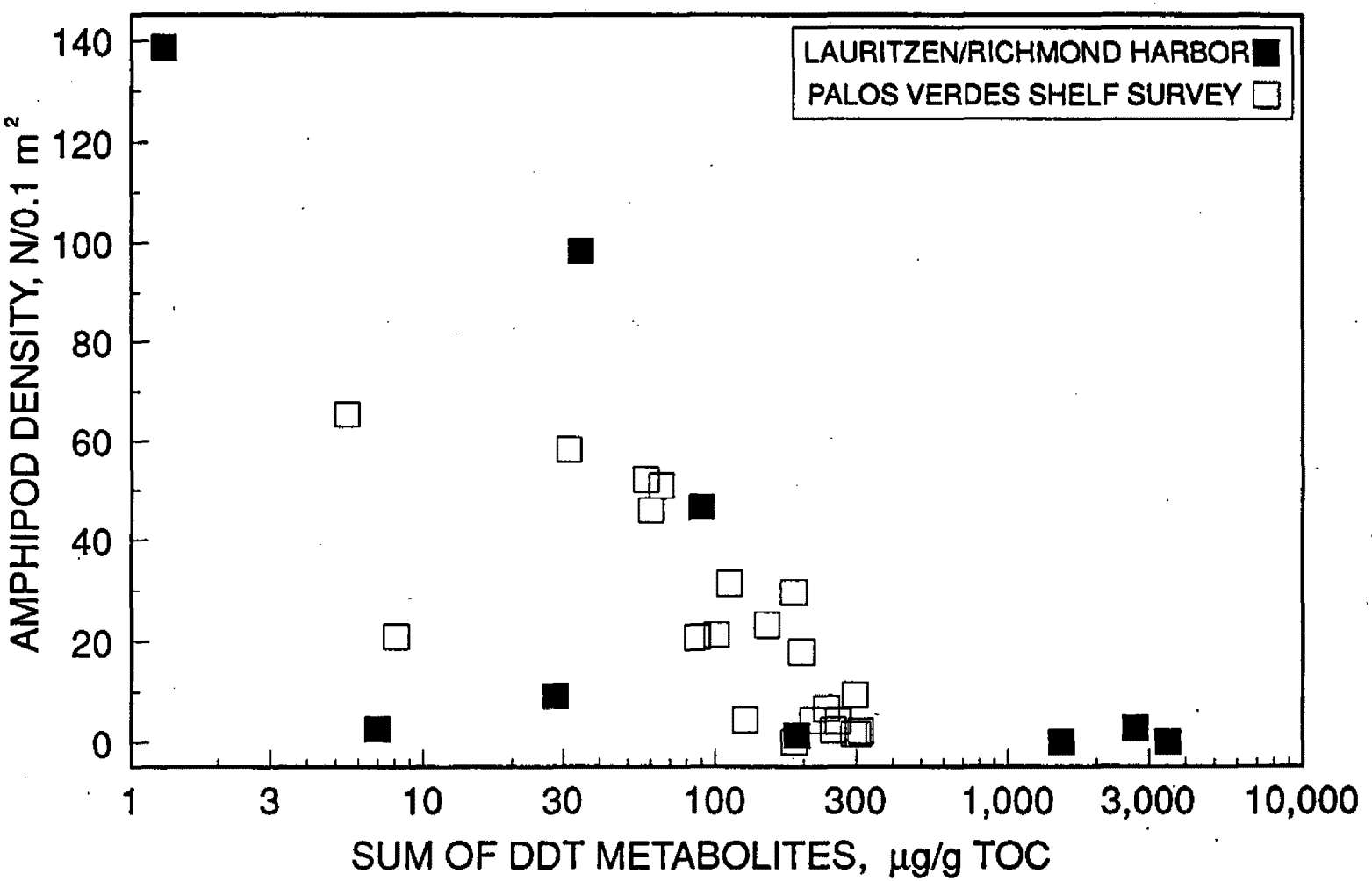

LAURITZEN/RICHMOND HARBOR
PALOS VERDES SHELF SURVEY
AMPHIPOD DENSITY, N/0.1 m²
140
120
100
80
60
40
20
0
1
3
10
30
100
300
1,000
3,000
10,000
SUM OF DDT METABOLITES, µg/g TOC

FIGURE 9-8. The abundance of Grandidierella japonica in relation to the concentration of dieldrin.

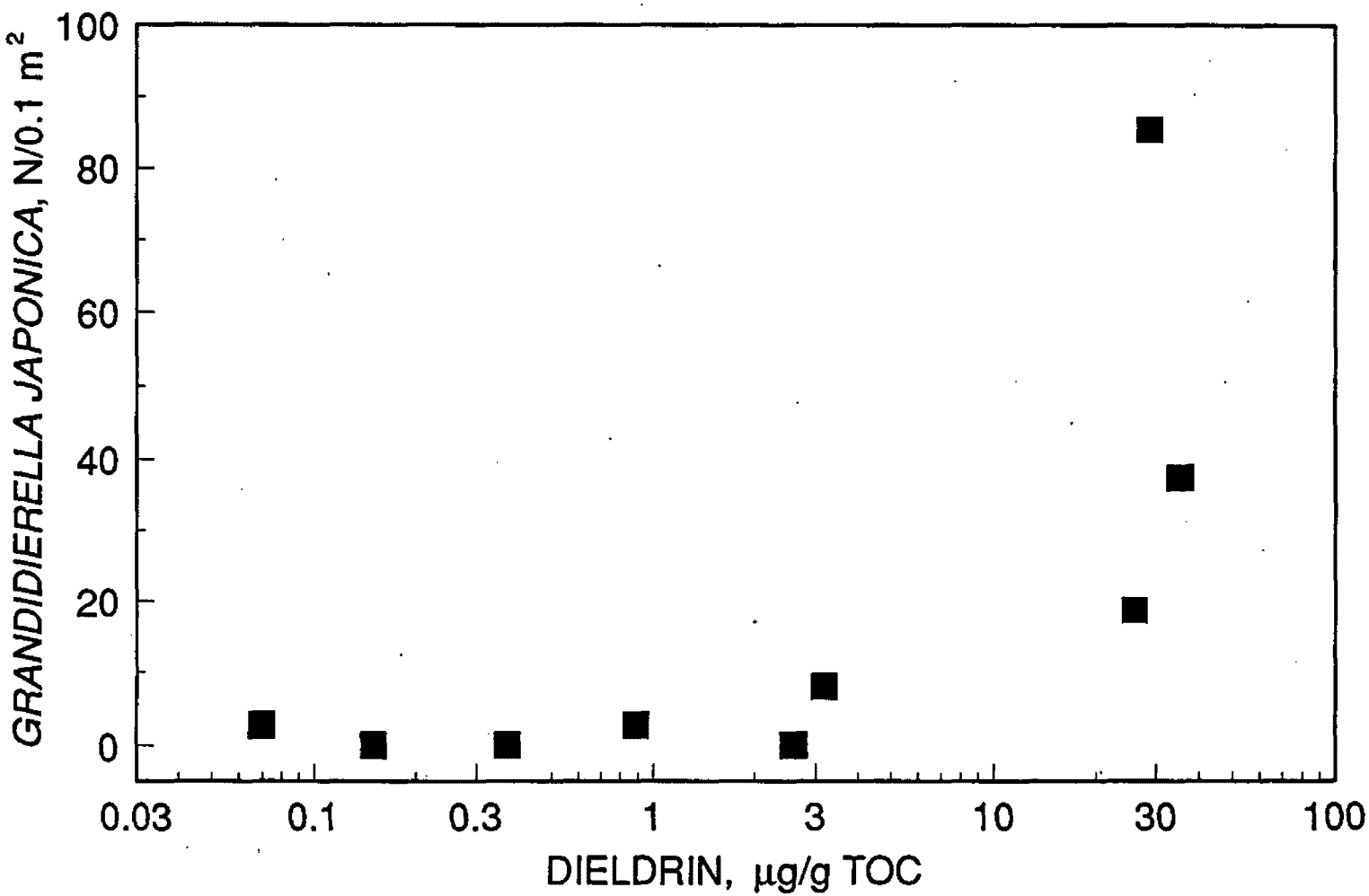

GRANDIDIERELLA JAPONICA, N/0.1 m²
100
80
60
40
20
0
0.03
0.1
0.3
1
3
10
30
100
DIELDRIN, µg/g TOC

FIGURE 9-9. The abundance of Grandidierella japonica in relation to the sum of the concentrations of DDT metabolites.

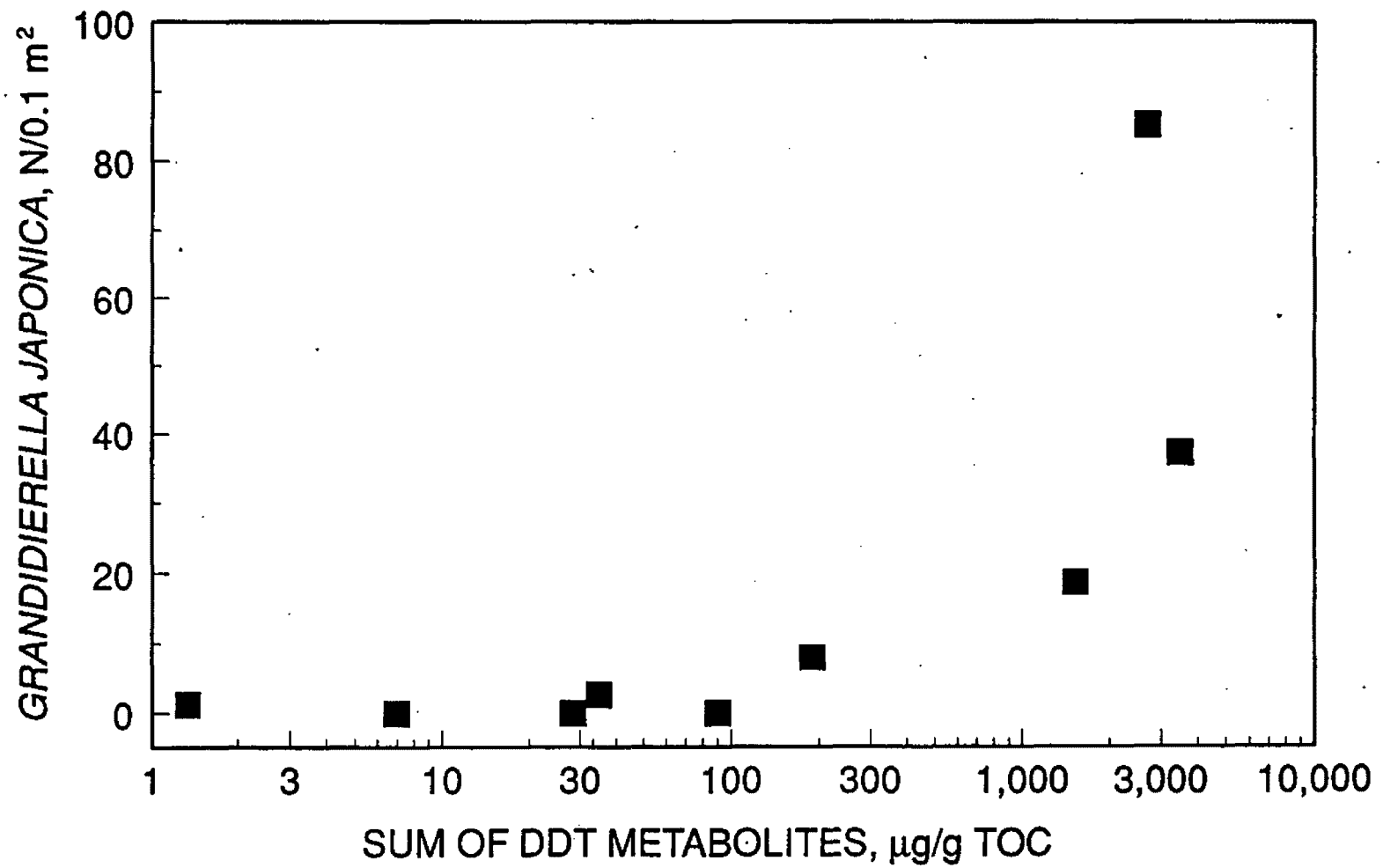

GRANDIDIERELLA JAPONICA, N/0.1 m²
100
80
60
40
20
0
1
3
10
30
100
300
1,000
3,000
10,000
SUM OF DDT METABOLITES, µg/g TOC

FIGURE 9-10. Mortality of Eohaustorius estuarius in relation to the abundance of Grandidierella japonica.

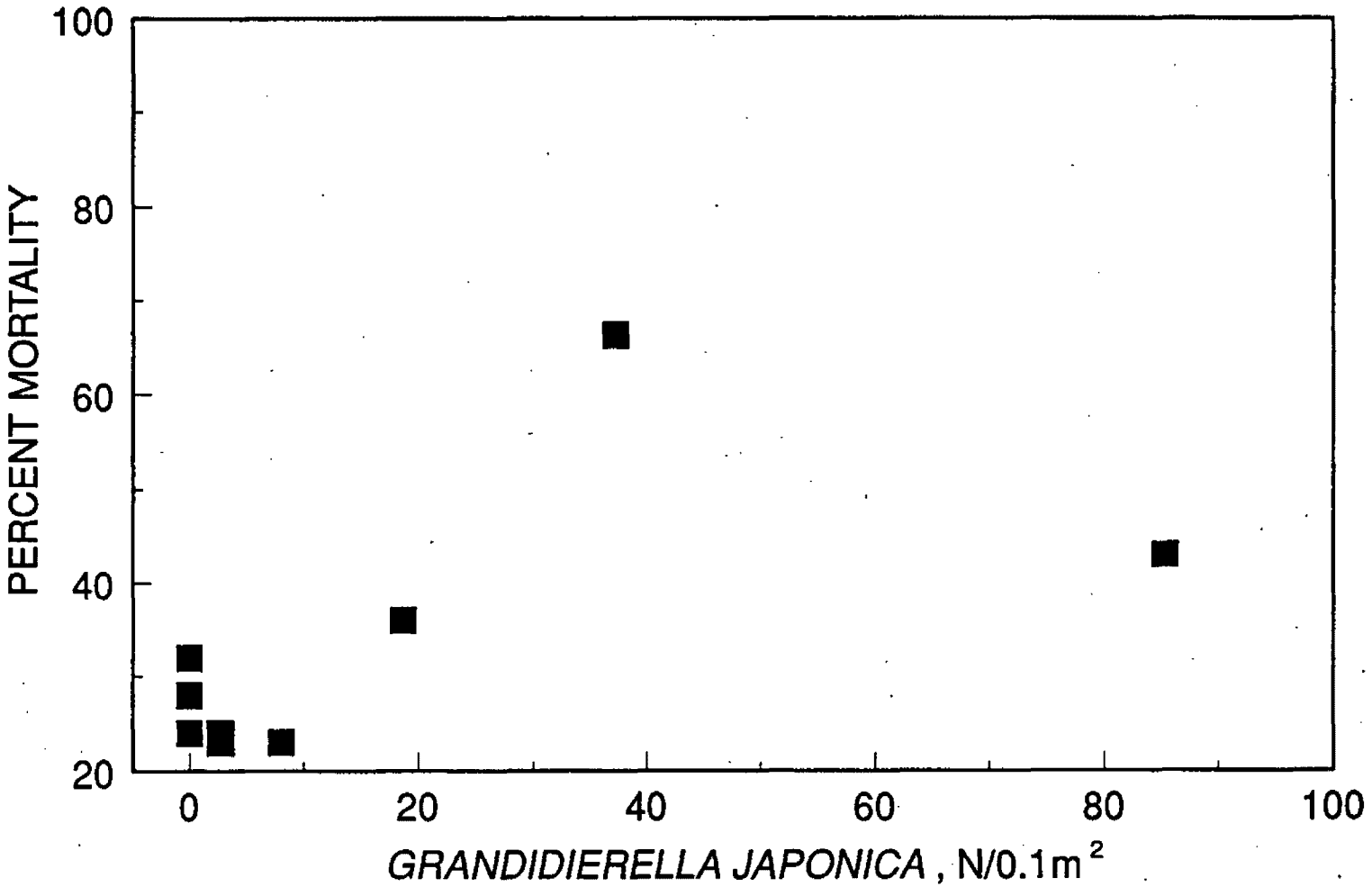

100
80
60
40
20
PERCENT MORTALITY
0
20
40
60
80
100
GRANDIDIERELLA JAPONICA , N/0.1m 2

FIGURE 9-11. Mortality of Eohaustorius estuarius in Lauritzen Channel/Richmond Harbor sediment and Rhepoxynius abronius in Palos Verdes Shelf sediment in relation to the abundance of amphipods (excluding Grandidierella japonica).

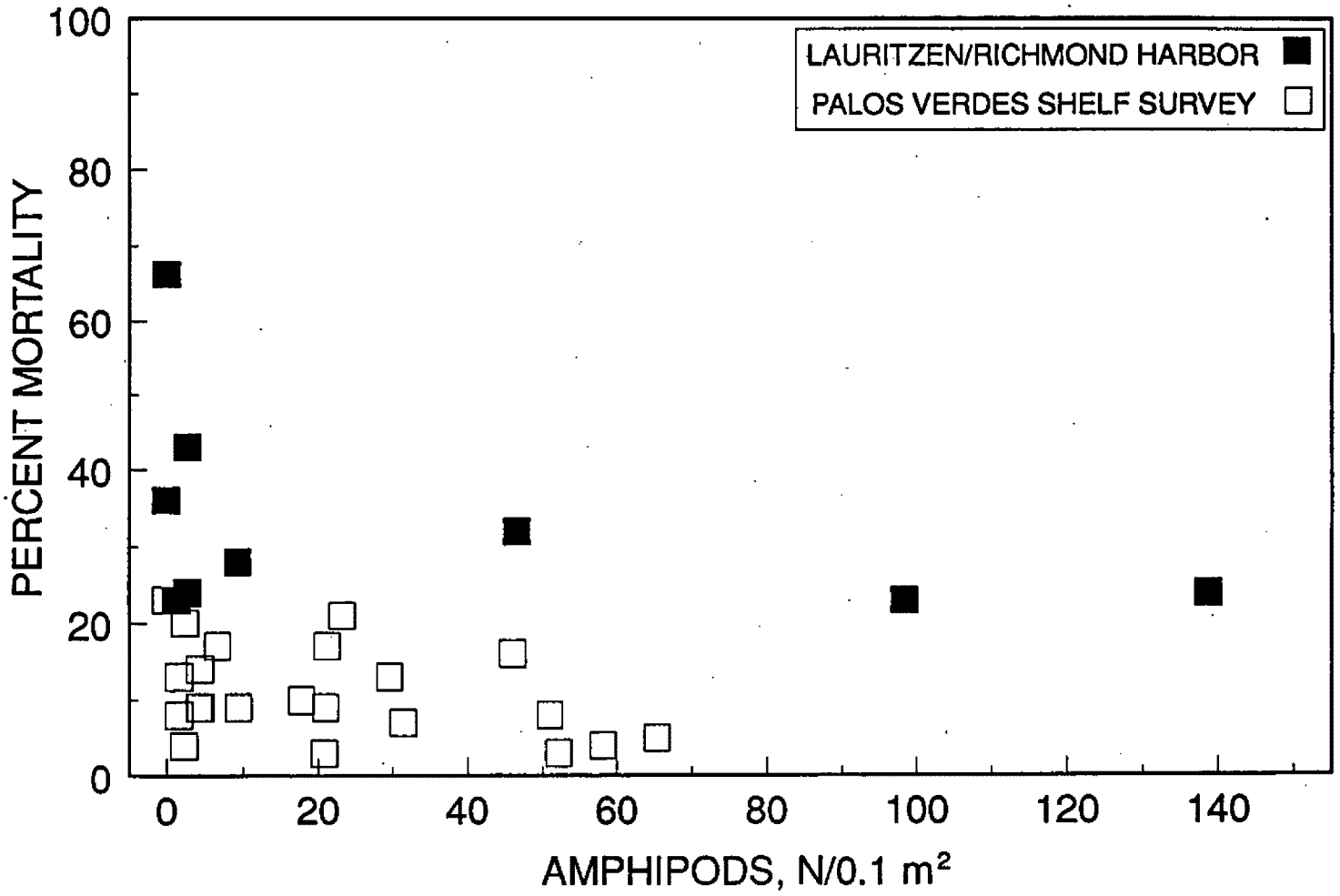

LAURITZEN/RICHMOND HARBOR
PALOS VERDES SHELF SURVEY
PERCENT MORTALITY
0
20
40
60
80
100
0
20
40
60
80
100
120
140
AMPHIPODS, N/0.1 m²

FIGURE 9-12. Mortality of Eohaustorius estuarius in relation to toxic units of dieldrin.

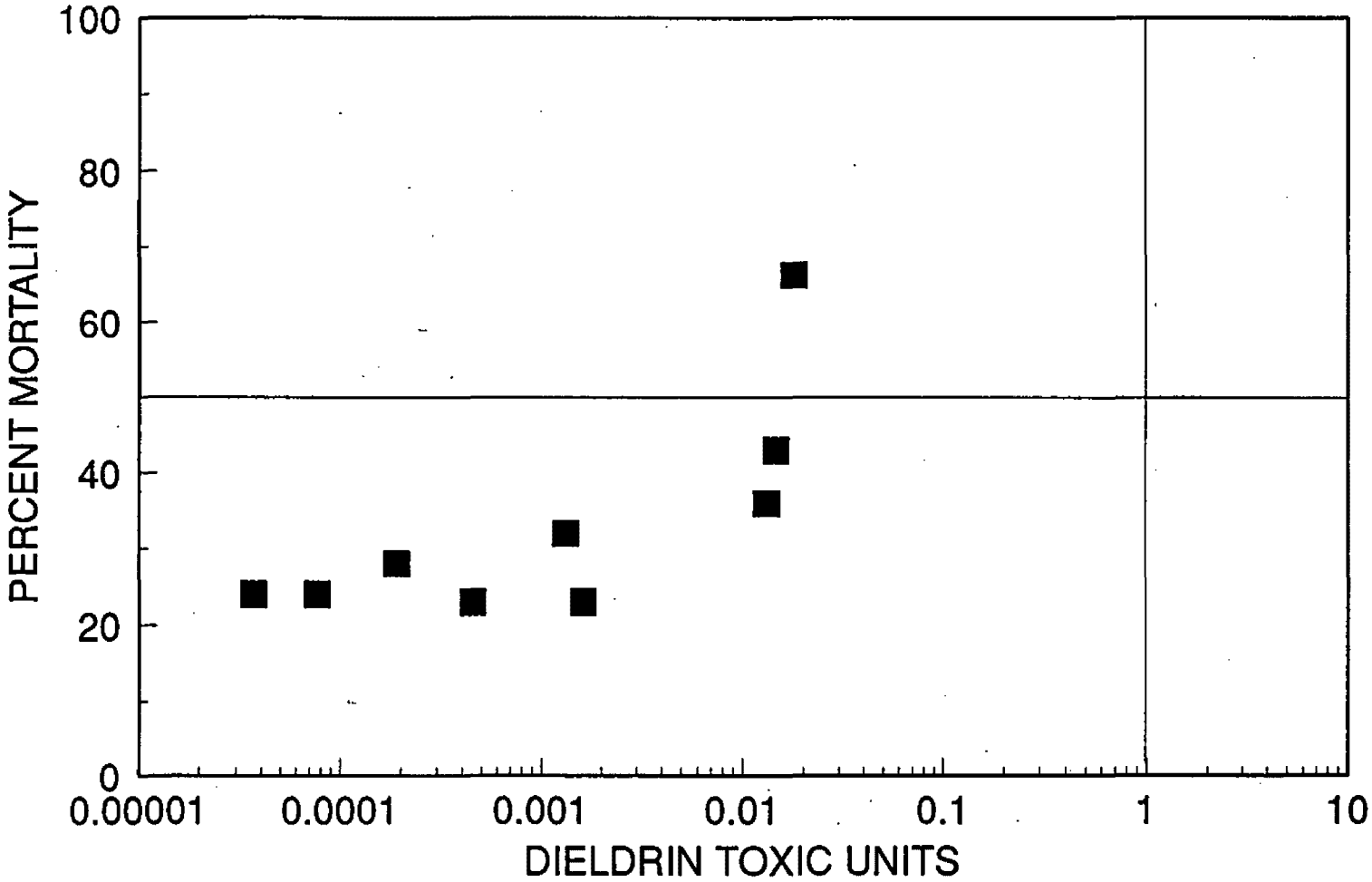

100
80
60
40
20
0
PERCENT MORTALITY
0.00001
0.0001
0.001
0.01
0.1
1
10
DIELDRIN TOXIC UNITS

FIGURE 9-13. Mortality of Eohaustorius estuarius in Lauritzen Channel/Richmond Harbor sediment and Rhepoxynius abronius in Palos Verdes Shelf sediment in relation to toxic units of the sum of DDT metabolites.

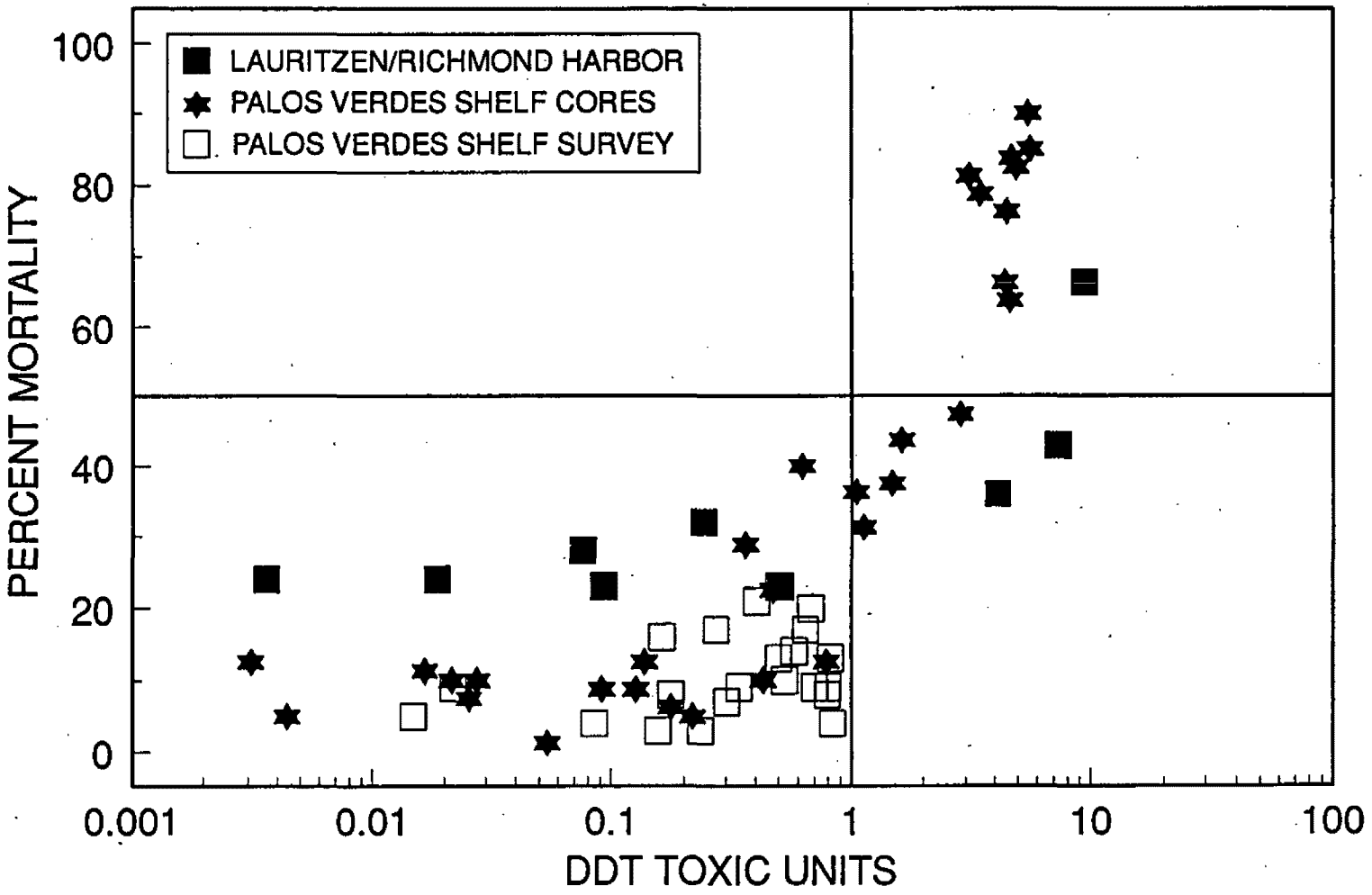
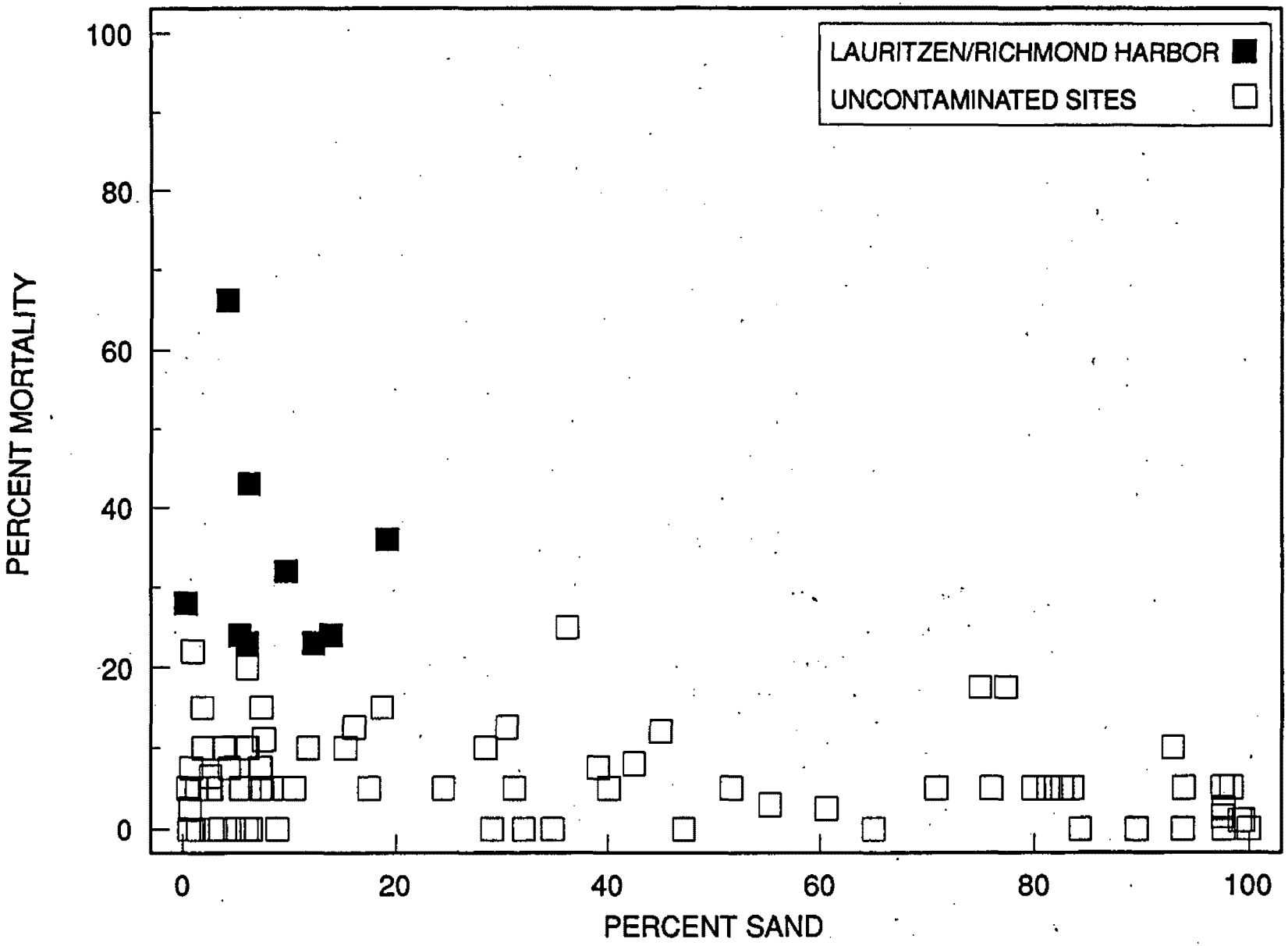

LAURITZEN/RICHMOND HARBOR
PALOS VERDES SHELF CORES
PALOS VERDES SHELF SURVEY
PERCENT MORTALITY
100
80
60
40
20
0
0.001
0.01
0.1
1
10
100
DDT TOXIC UNITS

FIGURE 9-14. Mortality of Eohaustorius estuarius in relation to sediment grain size.


LAURITZEN/RICHMOND HARBOR
UNCONTAMINATED SITES
PERCENT MORTALITY
PERCENT SAND
0
20
40
60
80
100

9.3. MULINIA GROWTH TEST

9.3.1. Introduction

Growth and scope-for-growth have been used to assess the sublethal effects pollutants have on sensitive organisms. A new sublethal test that uses growth in juveniles of the bivalve Mulinia lateralis is under development by EPA and SAIC at ERL-Narragansett (Burgess, 1991). This test has the advantages that it is short-term, easy to perform, and uses laboratory-cultured organisms (Burgess, 1991). Although in the developmental stage, the Mulinia growth test was used for a preliminary assessment of any sublethal effects due to sediment contamination.

9.3.2. Methods

The Mulinia growth test was conducted by SAIC at the ERL-Narragansett laboratory. Ten laboratory-cultured Mulinia (2-3 mm shell length, 0.8 ± 0.1 mg each) were placed into separate 150 ml glass chambers which contained about 50 ml of test or control sediment and 100 ml of seawater. These sediments were the unused portions of sediment collected for the 28-day bioaccumulation test from Stations 2-9 and Yaquina Bay control sediment (Station 0). Station 1 was excluded as not enough was available to perform the test. These sediments had been stored for approximately 30 days (4°C) before the start of the Mulinia test. Additionally, Mulinia were exposed to two additional controls: Long Island Sound sediment (a local reference sediment) and to a performance control beach sand that had been rinsed with deionized water.

Each sediment treatment consisted of five replicates. Exposure was for seven days at ambient temperature (about 20°C). During the exposure, small volumes (<1 ml/day) of concentrated algal culture were added to each chamber as a food source. This culture consisted of Isochrysis and Tetraselmis, each at an approximate concentration of 50,000 cells per ml.

After seven days, bivalves were separated from the sediment, the number of live organisms determined for each replicate, dried, weighed, and the change in weight during the exposure determined.

9.3.3. Results and Discussion

Table 9-11 presents the results of this test. No significant differences were noted in mortality between test and control sediments. Reduced growth relative to the sand control was noted in Mulinia exposed to sediment from Stations 5, 7 and 9. Station 9 also exhibited reduced growth in comparison to the Long Island Sound reference sediment. The lack of a reduction in growth at the highly contaminated Lauritzen stations suggests that Mulinia is not sensitive to the concentrations of these pesticides. This result is consistent with previous experience with this test that has shown Mulinia to be relatively insensitive to organic contaminants but quite sensitive to metals (Burgess, personal communication).

The apparent reductions in growth that occurred at Stations 5, 7, and especially 9 are difficult to explain. ΣDDT and dieldrin concentrations were lower at these stations than at the Lauritzen stations that did not show reduced growth (Table 8-1 and 8-2) and metal concentrations were relatively low at these sites (Tables 8-10). Although analysis for acid volatile sulfides (Table 8-10) suggests that metal in sediment from Station 9 may have been more bioavailable than at other sites, AVS was still present in excess relative to simultaneously-extracted metals, implying that there should not have been any toxic effects attributable to metals. Therefore, the growth effect noted at these stations is unexplained.

TABLE 9-11. Results of seven day Mulinia growth test.

STATION	MORTALITY (%)	WEIGHT CHANGE (mg)
SAND CONTROL	2±5	0.9±.01
0 YAQUINA BAY CONTROL	0	0.7±0.1
2	2±5	0.7±0.1
3	4±9	0.7±0.2
4	4±9	0.7±0.1
5	2±5	0.6±0.2[a]
6	2±5	0.7±0.1
7	0	0.6±0.3[a]
8	8±8	0.7±0.2
9 REFERENCE	2±5	0.4±0.2[ab]
LONG ISLAND REFERENCE	0	0.7±0.2

Values are Means ± SD.
a = Significantly different than Sand Control (t-test, $p<0.05$)
b = Significantly different than Long Island Sound Reference Station (t-test, $p<0.05$).

9.4. TROPHIC TRANSPORT OF SEDIMENT-ASSOCIATED ΣDDT TO PREY OF FISH-EATING BIRDS

9.4.1. Introduction

Contaminated sediments can serve as the ultimate pollutant source for higher trophic levels by the trophic transport of pollutants from contaminated benthic organisms to their predators and then through the pelagic food web. Because DDT and its metabolites biomagnify, the potential for contamination of the prey of aquatic birds and marine mammals is of particular concern. This section predicts the transport of sediment ΣDDT to higher trophic levels by coupling the equilibrium partitioning (EqP) bioaccumulation model for infaunal prey with the exponential biomagnification model (EBM) for their fish predators. These fish residues are then compared to the NAS standard to protect fish-eating birds. The purpose of this analysis is to identify trends. That is, the models are used to predict the general pattern of uptake at the various sites and to bracket likely exposures.

9.4.2. Methods

The EBM assumes a constant percentage increase in tissue residues between each trophic level, which is represented in the model by the "trophic step amplification factor" (TSAF)(Young, 1988). The EBM makes a different set of assumptions than the application of the EqP model to fish. In particular, it assumes that prey concentrations are the main factor controlling concentrations at higher trophic levels and it does not assume thermodynamic equilibrium between the organism and sediment, though it implicitly assumes steady-state conditions. One advantage of the EBM model is that it has been field validated for 4,4'-DDE using several marine food webs in Southern California (Young, 1988). Another advantage is that it predicts residues in higher trophic levels for which the EqP model is inappropriate because of the higher trophic level species' limited direct contact with sediment.

According to the exponential biomagnification model, the pollutant concentration at a particular trophic level can be predicted by:

$$TRy = TRx * TSAF^{(TLy-TLx)} \qquad \text{Eq. 9-5}$$

where:

TRy = tissue residue in predator, trophic level y (μg/kg, wet)
TRx = tissue residue in prey, trophic level x (μg/kg, wet)
TSAF = trophic step amplification factor (unitless)
TLy = trophic level of predator (unitless)
TLx = trophic level of prey (unitless)

This formula is equivalent to:

$$\ln TRy = \ln (TRx) + (TLy - TLx) * \ln TSAF. \quad \text{Eq. 9-6}$$

The trophic level of the predator has to be greater than its prey (*i.e.*, TLy > TLx) though trophic levels can have fractional values (*e.g.*, 3.5). Tissue residues are on a whole body basis and can be in wet, dry, or lipid-normalized weights. Wet weights are used in the present analysis. The trophic step amplification factor for 4,4'-DDE in 11 Pacific ecosystems averaged 3.3, with a median of 1.8 and a range of 0.48 to 10 (Young, 1988). For this trend analysis, the median value for 4,4'-DDE (1.8) is used as the estimate for ΣDDT. No TSAF is available for dieldrin, though it is likely to be smaller because of dieldrin's lower K_{ow}.

Tissue residues in infauna, the base of the benthic food web, are calculated using the EqP bioaccumulation model (see Section 8.3.). Substituting the EqP model for TRx in Equation 9-6, the formula for a fish feeding on the benthos becomes:

$$TRy = (AF * L * Cs/TOC) * TSAF^{(TLy-TLx)} \quad \text{Eq. 9-7}$$

where:

AF = accumulation factor (g OC/gL)

L = lipids (decimal fraction, wet weight)

Cs = pollutant conc. in sediment (μg/kg, dry)

TOC = total organic carbon of sediment (decimal fraction, dry weight)

Different sediment conditions can be modeled by substituting different values for the bulk sediment concentration and/or TOC. Modeling different benthic prey items can be accomplished by substituting different lipid and/or AF values. The average site sediment concentrations of ΣDDT and TOC for trawls (Tables 8-28A and 8-30) are used in this analysis.

The benthos is assigned a trophic level of 2.0 (Table 8-23), which should closely approximate the trophic level of either filter or deposit feeders. The wet lipid concentration of the benthic prey is set at 1.0%, which is equivalent to a dry lipid weight of about 5-7%. Use of a wet lipid concentration generates a wet tissue residue in the benthos and higher trophic levels.

To bracket residues in different prey types, three sets of AFs are used for each of the sites. The first is the overall average AF (0.289) based on all the species caught in the grabs and trawls at all the sites (Table 8-20). Since it is based on a larger number of replicates, the overall average for all sites should be a better predictor than the site-specific AFs, which in

one case is based on only two samples. The second is the average of all the field-caught filter feeders (0.10; Table 8-20). The third AF is the mean of the laboratory values for Macoma nasuta corrected to steady-state residues. The Macoma AF is used as an estimate for deposit-feeding infauna. To give each station equal weight, the Macoma AF is calculated as the mean of the station means, which results in a slightly different value than the mean of the individual AFs given in Table 8-18 (0.924 vs. 0.90).

The predicted results are compared to observed residues in gobiids and shiner surfperches. Both are relatively non-mobile so their tissue residues should reflect local conditions, and whole body tissue concentrations are available at all of the sites. Both species are assigned a trophic level of 3.25 (Table 8-23). Gobiids primarily feed on benthic prey, and so are an excellent species to model the trophic transport of sediment-associated pollutants. However, as a bottom species, they are less likely to be prey for fish-eating birds. Shiner surfperch, by feeding on benthic invertebrates as well as water-column organisms, have a connection with sediment-associated pollutants though it is not as strong as with the gobiids. However, as a water-column organism, they are more likely to be preyed upon by fish-eating birds.

9.4.3. Results/Discussion

The equilibrium partitioning bioaccumulation and exponential biomagnification models, like all models, are subject to a number of uncertainties. These are examined in relation to the accuracy required for a trend analysis. The EBM is relatively insensitive to the likely range of errors in trophic level assignments. For example, underestimating the trophic level (TL) by 0.5 trophic level would result in underestimating residues by about 34%. An error of a full trophic level, which is unlikely, would result in an error equivalent to the TSAF, or 1.8-fold in this analysis. Because it is multiplicative, errors in the value of the TSAF can have a greater effect, especially on higher trophic levels. When starting at trophic level 2 as the base of the food web, use of the mean TSAF for 4,4'-DDE instead of the median (3.3 vs. 1.8) (Young, 1988) results in about a 2-fold higher residue at trophic level 3 and more than a 3-fold higher residue at trophic level 4.

Potentially greater sources of error relate to estimating the residues in infaunal prey at the base of the benthic food web. One assumption is that the fish feed only in the modeled sites (*e.g.*, Lauritzen Channel). Fish that spend time feeding in less contaminated sites will have a lower tissue residue than predicted. Conversely, fish that feed in more contaminated systems will have a higher than predicted residue. The extent of this error is related to the mobility of the species and the spatial pattern of contamination. The error should be greater with higher trophic level predators (TL > 3.5), which tend to be larger and more mobile, than with smaller benthic predators such as Cymatogaster and gobiids.

Another source of uncertainty relating to the prey is the

composition of the infaunal prey species. Residues in deposit-feeding infauna can be 2-fold to 20-fold higher than in filter-feeding species (Table 8-20), so that prey composition can have a major effect on a predator's exposure. To bracket this source of ecological variation, infaunal prey residues were estimated using AFs for three different prey types. Prey residues predicted using the overall field AF were used to model the diet of generalist predators feeding on filter-feeding bivalves and deposit-feeding Macoma and polychaetes. Use of this overall average field AF assumes that the invertebrates caught in the grabs/-trawls approximate the biomass of the consumed prey, or at least that the AFs for the various taxa are in similar ratios to those consumed. The AF for filter feeders was used as the predictor for predators specializing on filter-feeding bivalves while the AF for Macoma was used as the predictor for fish ingesting sediment-ingesting infauna, such as deposit-feeding polychaetes and Macoma. The average lipid content used in calculating benthic tissue residues also could introduce an error, though it is unlikely that differences from the 1.0% (wet) would introduce more than a 50-100% error.

Predicted tissue residues for trophic levels 2 to 4 and observed residues in gobiid and shiner surfperch at each of the three sites are presented in Figures 9-15, 9-16, and 9-17. Comparison of observed to predicted fish tissue residues offers a check on the validity of this approach. Both the observed gobiid and shiner surfperch tissue residues were within 3-fold of the residues predicted with the overall field AF at all three sites. The overall AF overestimated goby and shiner perch residues in the Lauritzen Channel but underestimated them in the Santa Fe and Richmond Channels. The higher than predicted residues in the Santa Fe and Richmond Channels may be due to movement of heavily contaminated benthic prey from Lauritzen into less contaminated sites or feeding forays by the gobys and shiner surfperch into Lauritzen Channel. In any case, the similarity between observed and predicted residues is encouraging, especially when it is considered that ΣDDT sediment concentrations varied by more 600-fold among the three sites compared to the less than 3-fold difference in observed/predicted residues.

The models also correctly predicted expected patterns. For both species at all three sites, the observed residues fell between those predicted for fish feeding exclusively on filter-feeding prey and those feeding on deposit-feeding prey. This result was expected as few fish feed exclusively on filter feeders or deposit feeders, and the result supports the contention that the filter-feeding and deposit-feeding AFs bracket the residues in benthic prey.

Although it does not validate the models *per se*, these results support the use of these coupled models as a predictive tool over a wide range of sediment concentrations for fish feeding at trophic level 3.0 to 3.5 feeding on benthic prey. No large fish predators were analyzed, so it is not possible to check the models for higher trophic levels (TL > 3.5). Because of the

uncertainties involving mobility, the predictions for trophic levels above 3.5 or for highly mobile species should be considered as potential rather expected residues.

One method of assessing the significance of the predicted residues is to compare them to the National Academy of Science's standard of 50 μg/kg ΣDDT (wet weight) in fish to protect fish-eating birds. Because small bottom-feeding fish, such as gobiids and shiner surfperches, are potential prey for certain fish eating birds, this is an appropriate ecological criterion. The models predicts that fish residues will exceed this criterion by over 100-fold to almost 1000-fold in Lauritzen Channel, depending upon which AF is used (Figure 9-15). In the Santa Fe Channel, the NAS criterion will be exceeded by 2- to 20-fold (Figure 9-16). Given the extent of these violations, it is apparent that fish-eating birds in either of these channels would be at substantial risk. In Richmond Harbor Channel, the models predict fish residues below the criterion when the overall average field and filter-feeding AFs are used but above the criterion when the deposit-feeder AF is used (Figure 9-17). Thus, whether a violation occurs will depend upon the specific diet of a given fish species.

The coupled models also can be used to predict the sediment concentrations required to achieve a 50 μg/kg residue in trophic level 3.25 benthic feeding fish (Figure 9-18). The predicted sediment concentrations are 2.6 μg/g OC when the deposit-feeder AF is used, 8.3 μg/g OC when the overall field AF is used, and 23.9 μg/g OC with the filter-feeder AF. Assuming a TOC of 0.019 g OC/g, these values are equivalent to bulk sediment concentrations of 49 μg/kg, 158 μg/kg, and 454 μg/kg, respectively. Lower sediment TOCs would require proportionately lower bulk sediment concentrations. As mentioned, it is unlikely that fish prey exclusively on deposit feeders or filter feeders, so the prediction based on the overall AF is probably the best general estimate.

Any other criterion can be calculated as a direct proportion of the NAS standard. For example, to meet the human health criterion (32 μg/kg, see Section 5.1.), the required sediment concentration would be 64% (*i.e.*, 32/50) of the residues based on the NAS standard. This results in bulk sediment concentrations of 31 μg/kg, 101 μg/kg, and 291 μg/kg, for the three AFs, respectively, at a TOC of 1.9%. These predictions are for trophic level 3.25 fish, which tend to be small. Predictions for larger, higher trophic level sport fish would generate lower sediment concentrations.

The above analysis assumes that birds consume fish only from the respective channels. Because the NAS standard is based on chronic rather than acute effects, it is possible to predict bird exposure to DDT as a function of the time spent feeding in each channel by calculating a yearly average DDT residue in their fish prey. A year is an ecologically relevant time frame for predicting chronic effects of DDT because of the very slow depuration rate in birds. Additionally, a year relates to the breed-

ing and/or migratory cycle of most birds.

The annual average fish residue is calculated by multiplying the number of days a bird feeds in a particular channel by the average fish residue in the channel divided by 365 days. An average fish residue above 50 μg/kg would indicate that the average annual DDT residue in fish exceeded the NAS standard, whereas an average below 50 ug/kg would indicate that the standard was not exceeded. This approach assumes that any fish the birds consume outside of the modeled channel contain no DDT. To the extent that this assumption is violated, the approach underestimates the annual exposure to DDT, although not the contribution from the particular channel. This approach also assumes a constant feeding rate and that the timing of the consumption of DDT does not influence its physiological effect. These calculations are made using the overall field AF, which should be the best predictor for fish predators consuming a variety of benthic taxa. Use of the filter-feeder AF or deposit-feeding AF would increase or decrease, respectively, the allowable duration in proportion to the differences in the AFs.

Figure 9-19 shows the annual average prey residues as a function of the number of days feeding in each of the three channels. The predicted ΣDDT fish residues in Lauritzen Channel are so high that the annual average prey concentration exceeds the 50 μg/kg NAS standard with slightly more than one day (about 28 hours) of feeding by birds. With such a short exposure period, even incidental feeding within Lauritzen Channel would exceed the NAS standard. Pelicans and other fish-eating birds were observed feeding in Lauritzen (see Section 4.2.2.); so it is highly likely that individual birds, if not populations, spend at least several days over a year feeding in the Lauritzen Channel. Additionally, migrating birds flying through the area could feed for several days within Lauritzen, thereby exceeding the NAS standard on a yearly basis.

Fish-eating birds would have to feed for about 2 months, or about 17% of a year, in the Santa Fe Channel to exceed the NAS standard on an annual basis. At this duration, more than incidental feeding would be required to exceed the NAS standard and migratory birds would be at a lower risk. However, local individuals and populations could be at risk. The Santa Fe comprises about 25.8% of the combined Richmond, Santa Fe, and Lauritzen Channels. Therefore, a bird that fed proportionally among the three channels would ingest about 26% of its prey from the Santa Fe Channel, resulting in an annual average fish residue exceeding the NAS standard based on fish consumed from the Santa Fe Channel.

In the Richmond Channel, the predicted fish residue, 40 μg/kg, is below the NAS standard so according to the models, birds could consume their entire diet in Richmond Channel without exceeding this standard. However, it is important to recognize that the 3-fold uncertainty observed when comparing the goby and surfperch residues to predicted values would include the NAS standard. Additionally, fish selecting for deposit-feeders would be likely

to exceed the NAS standard. Therefore, it is possible that individual birds or bird populations that feed extensively in Richmond Channel could exceed the NAS standard on an annual basis. Birds that only feed intermittently in Richmond Harbor Channel should not be at risk. In any case, the risk is much less than in the other two channels.

FIGURE 9-15. Predicted tissue residues of ΣDDT by trophic level in Lauritzen Channel.

Overall field accumulation factor (0.289) used to model prey for a generalist fish predator. Filter-feeding accumulation factor (0.010) used to model prey for a predator feeding on filter-feeding bivalves. Deposit-feeding accumulation factor (0.924) used to model residues in prey for a fish preying on sediment-ingesting polychaetes and Macoma.

PREDICTED SUM DDT RESIDUES
LAURITZEN CHANNEL

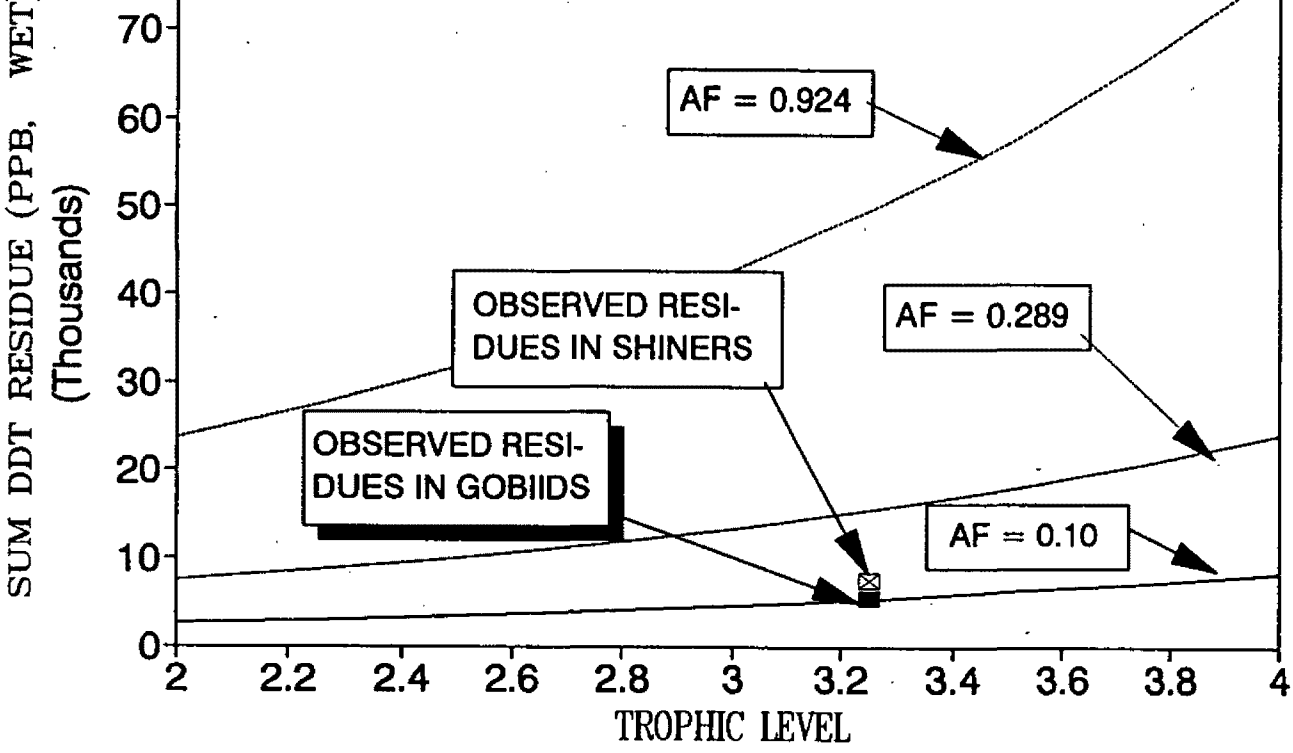


FIGURE 9-16. Predicted tissue residues of ΣDDT by trophic level in Santa Fe Channel.

Overall field accumulation factor (0.289) used to model prey for a generalist fish predator. Filter-feeding accumulation factor (0.010) used to model prey for a predator feeding on filter-feeding bivalves. Deposit-feeding accumulation factor (0.924) used to model residues in prey for a fish preying on sediment-ingesting polychaetes and Macoma.

PREDICTED SUM DDT RESIDUES
SANTA FE CHANNEL

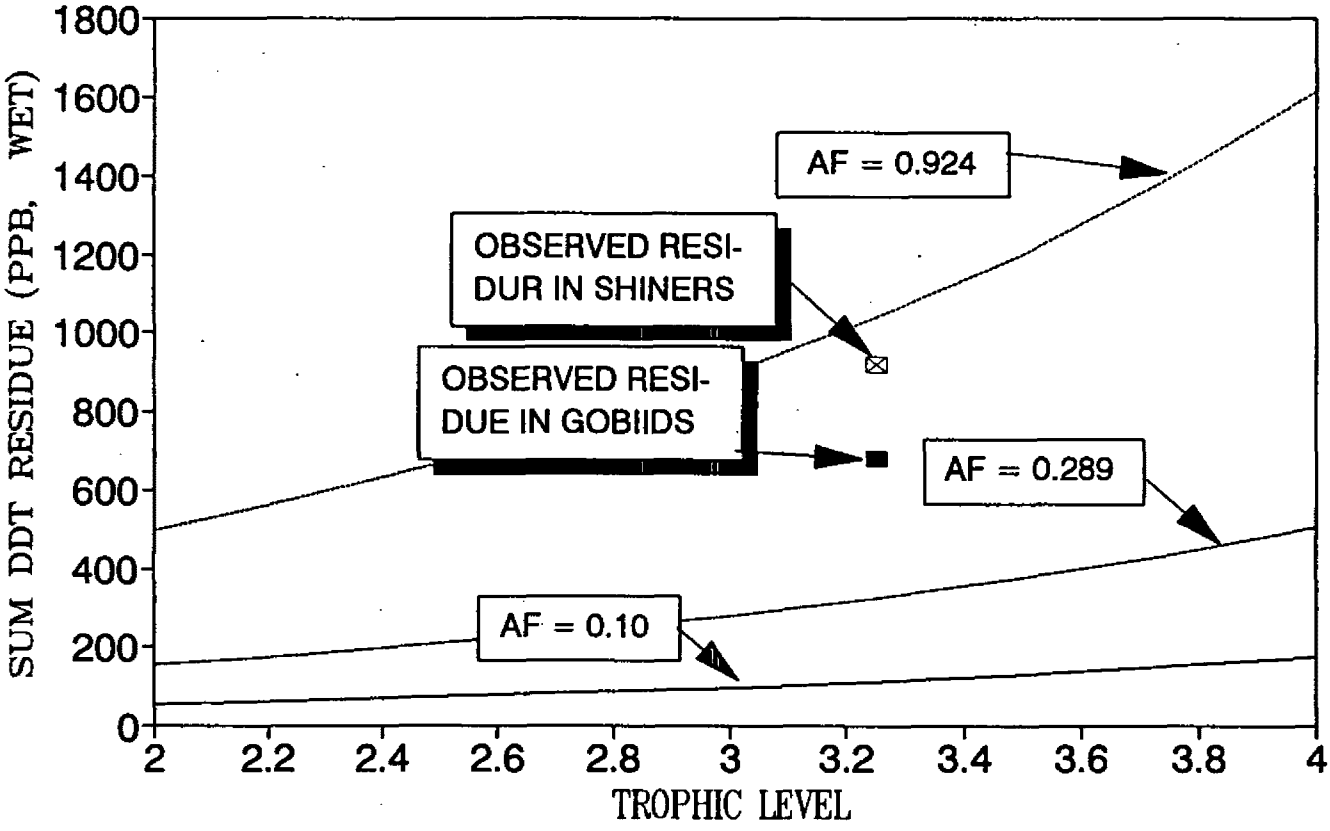


FIGURE 9-17. Predicted tissue residues of ΣDDT by trophic level in Richmond Inner Harbor Channel.

Overall field accumulation factor (0.289) used to model prey for a generalist fish predator. Filter-feeding accumulation factor (0.010) used to model prey for a predator feeding on filter-feeding bivalves. Deposit-feeding accumulation factor (0.924) used to model residues in prey for a fish preying on sediment-ingesting polychaetes and Macoma.

PREDICTED SUM DDT RESIDUES
RICHMOND CHANNEL

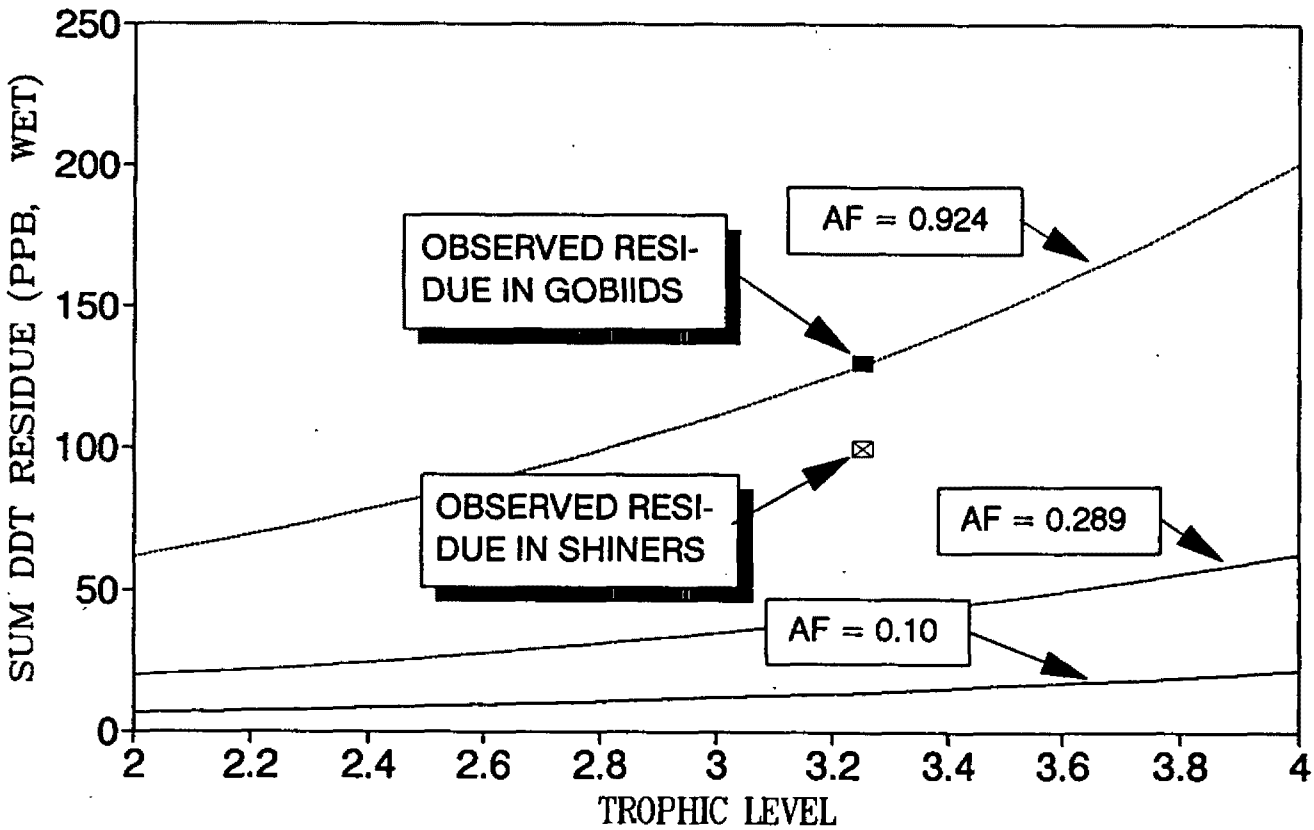


FIGURE 9-18. Predicted tissue residues of ΣDDT versus organic-normalized sediment concentrations in trophic level 3.25 fish, feeding on different types of benthic prey.

Overall field accumulation factor (0.289) used to model prey for a generalist fish predator. Filter-feeding accumulation factor (0.010) used to model prey for a predator feeding on filter-feeding bivalves. Deposit-feeding accumulation factor (0.924) used to model residues in prey for a fish preying on sediment-ingesting polychaetes and Macoma.

PREDICTED SUM DDT IN TROPHIC LEVEL 3.25 FISH

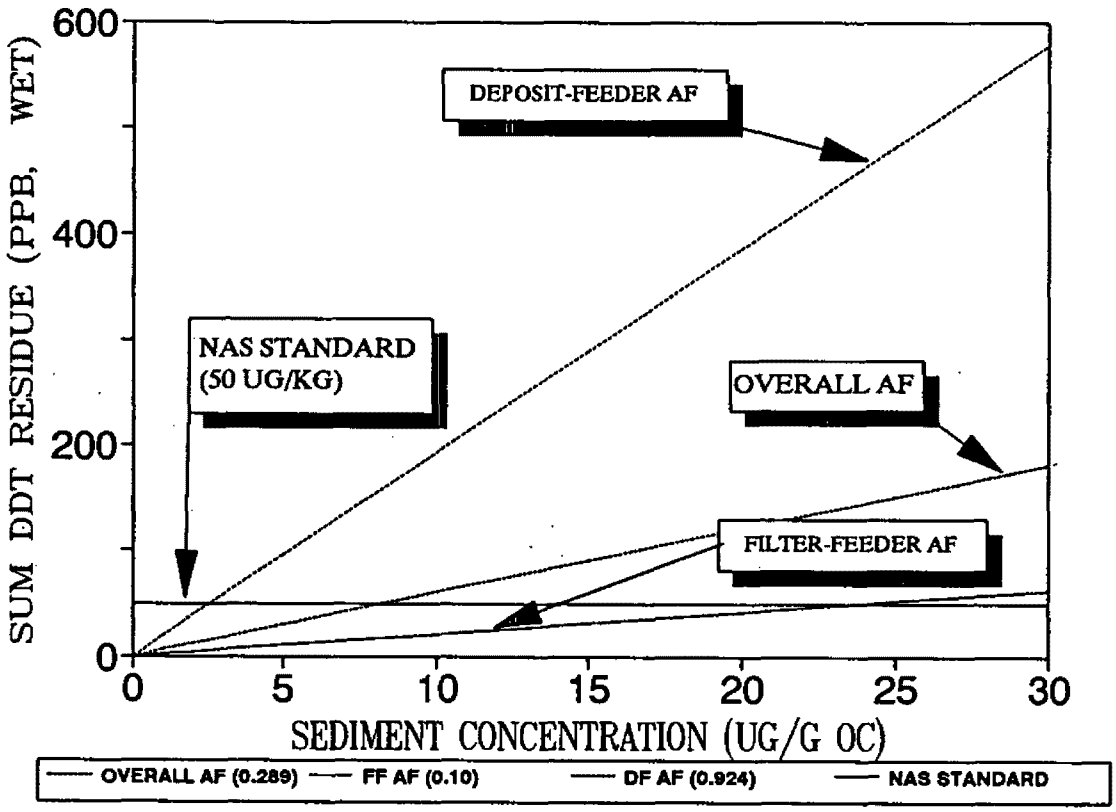
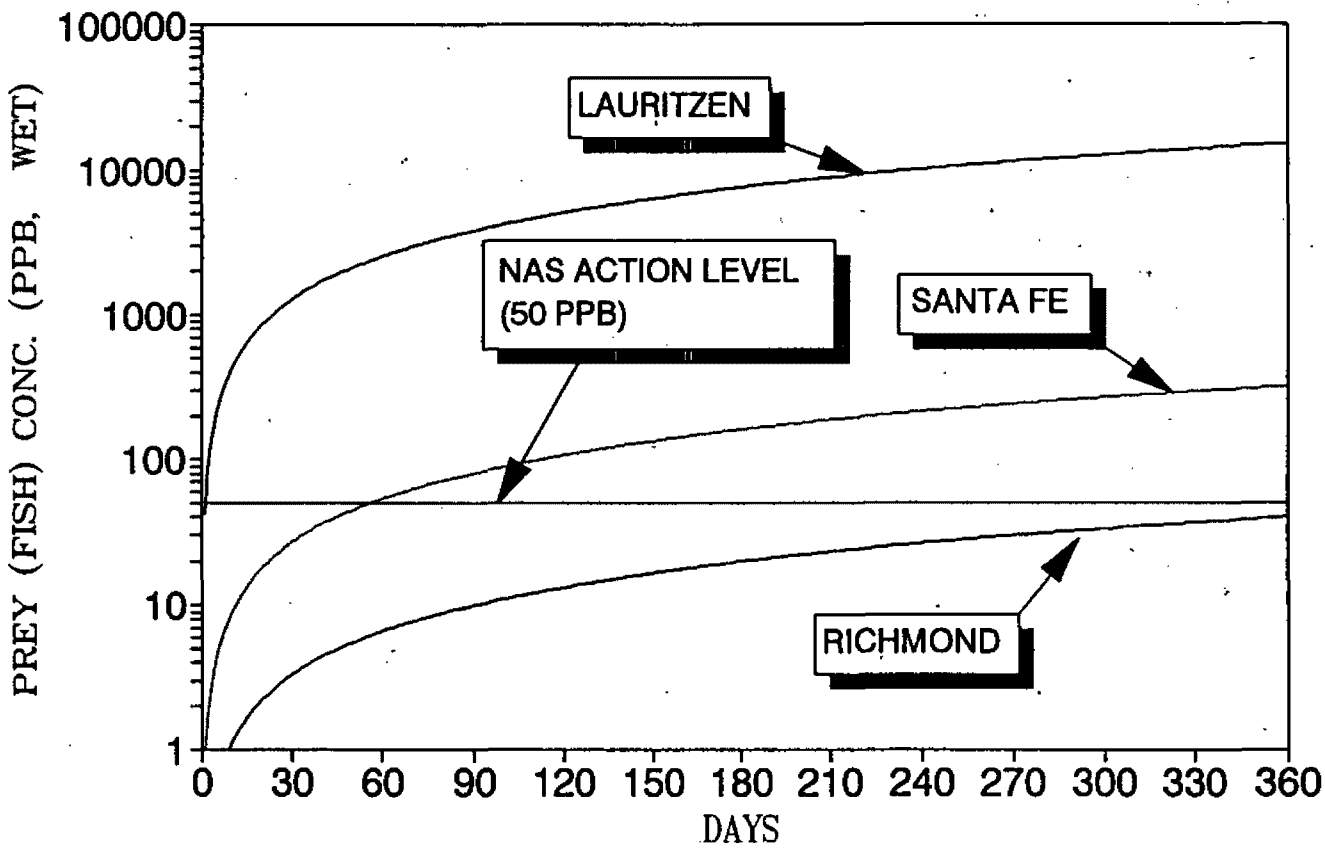


FIGURE 9-19. Predicted average annual ΣDDT residue in prey (fish) as a function of the duration birds feed in each of the channels. Prey residues above 50 μg/kg indicate that the NAS standard for DDT is exceeded on an annual basis. Fish residues predicted using overall field AF (0.289) for benthic prey.

PREDICTED AVERAGE PREY (FISH) CONC. AS FUNCTION OF DAYS BIRDS FEED IN CHANNEL



10.0 RISK CHARACTERIZATION AND POTENTIAL REMEDIATION LEVELS

10.1. INTRODUCTION

This section summarizes the exposure and effects data from the previous sections (Table 10-1 to 10-6). These data are used to estimate the overall existing ecological risk, the importance of Lauritzen Channel as a contaminant source for other sites, and the relative toxicological importance of ΣDDT. Then, sediment concentrations of ΣDDT and dieldrin required to achieve different remediation criteria are predicted (Table 10-7 to 10-12) using the models developed in previous sections.

Several of the ecological criteria summarized in the tables overlap, as for example, the water concentration and the corresponding residue for a residue-based Water Quality Criterion. When possible, it is worthwhile examining criteria from different perspectives as independent models could generate different values with different levels of uncertainty. Additionally, there is a higher level of certainty if two or more approaches converge to a similar prediction. End-points of less direct ecological significance were included in the tables. In particular, chronic WQC for ΣDDT and dieldrin, which are based on protecting fish-eating birds, are applied to the interstitial water concentrations. Such comparisons are useful in comparing the extent of contamination across sites, generating insight into the potential for bioaccumulation in demersal fishes or benthic invertebrates that may be prey for birds (*e.g.*, certain shrimp), and as an indication of the upper limit for overlying water concentrations during resuspension events. Nonetheless, such comparisons should not be given the same weight in determining remediation levels as the comparison of the overlying water concentrations to the chronic WQC. The summary and predictive tables include values based on human health end-points. These values are presented for comparative purposes only.

It is important to note that this document does not establish or advocate any remediation level, and the sediment clean-up concentrations presented here are for consideration and evaluation only. This study is part of a larger evaluation of Richmond Harbor and the final decision on remediation will take into account these additional data (*e.g.*, White *et al.*, 1993) and any other pertinent information.

10.2. DDT AS THE MAJOR TOXICOLOGICAL STRESS

Richmond Harbor, like most urbanized near-coastal ecosystems, is subject to a variety of anthropogenic stresses. Other than ΣDDT and dieldrin, the most important types of anthropogenic stresses in Richmond Harbor appear to be physical disruption from ships and other sediment-associated neutral organic pollutants. Depending upon the extent and spatial pattern of these stresses, they could obscure the ecological effects of ΣDDT and dieldrin, thereby making these pollutants seem less toxic (*e.g.*, a toxicant

seem less toxic (e.g., a toxicant with the opposite pattern of ΣDDT could obscure the relationship between sediment concentrations and amphipod mortality). Conversely, if these stresses were spatially correlated with ΣDDT and dieldrin, they could result in spurious correlations.

It is critical to recognize, however, that the effects of these additional stresses potentially affect only some of the measurements of ecological exposure and risk (Table 5-13). In particular, the predictions based on amphipod toxicity and benthic community structure could be affected by additional stresses. End-points based on Water Quality Criteria or tissue residues rely solely on the concentration of the compound, and the presence or absence of other pollutants or stresses is irrelevant.

There are a number of lines of evidence establishing ΣDDT as the major cause of toxicity and benthic community alterations. First is the very high ΣDDT sediment concentrations (Tables 8-11A). Although the AET and ER-M were not retained as end-points, the comparison of the ΣDDT sediment concentrations to these indices indicates that ΣDDT far exceeds potentially adverse levels in many areas of Richmond Harbor (see Section 3.2.). Additionally, the comparison of the concentrations of other neutral organic pollutants to these indices indicated that they were not the major stressor in the more heavily contaminated portions of Richmond Harbor (Section 3.2.).

The second line of evidence is that ΣDDT sediment concentrations explained a significant amount of the variation in 10 of 16 measures of benthic community structure (Table 9-2). In addition, ΣDDT sediment concentration explained a highly significant amount of the variation in the two sensitive macrobenthic indicators, Infaunal Index and number of amphipods excluding G. japonica, after accounting for the possible effects of sediment %silt-clay and TOC. Dieldrin sediment concentration did not explain a significant amount of the variation in these two macrobenthic indicators after accounting for the possible effects of sediment %silt-clay, TOC and ΣDDT sediment concentration (Tables 9-4 and 9-5).

The third line of evidence is a correlative-comparative approach, in which sediment toxicity and amphipod abundance were found to be similarly related to ΣDDT sediment concentrations at Richmond Harbor and in Southern California (Section 9.2.). It is unlikely that similar spurious relationships between amphipod distributions and sediment ΣDDT concentrations would occur at two independent sites.

The fourth, and perhaps the strongest, line of evidence is the application of toxic units to the amphipod toxicity test data (Section 9.2.). The maximum concentrations of dieldrin and ΣDDT in Lauritzen Channel in toxic units were 0.018 and 9.43, respectively, indicating that ΣDDT was the dominant ecotoxicological factor. The toxic units of PAHs and PCBs were not sufficient to explain the toxicity in Lauritzen. Toxic units of ΣDDT were lower in Santa Fe, but may have been sufficient to contribute to

acute toxicity and/or chronic toxicity. PAHs were relatively and absolutely more important in the Santa Fe Channel, especially in a localized oiled area, where PAHs may have been the major stressor. ΣDDT toxic units were low in Richmond Inner Harbor Channel, indicating that this pollutant was not a dominant stressor. The application of the ΣDDT chronic effects levels to the interstitial water concentrations (Table 10-1) supports the contention that ΣDDT concentrations are sufficient to explain the observed acute toxicity in Lauritzen and may contribute to chronic toxicity in the Santa Fe.

Taken in total, the comparative, correlative, and experimental evidence demonstrates that ΣDDT is the major toxicological stress in Lauritzen Channel. ΣDDT probably contributes to any chronic toxicity in Santa Fe Channel, though total PAHs may be the dominant toxicant in localized "hot spots" of oil contamination. In Richmond Inner Harbor Channel, the toxicological importance of ΣDDT, and dieldrin, is greatly diminished and other pollutants or physical disturbances are probably the major stresses.

10.3. LAURITZEN CHANNEL AS A CONTAMINATION SOURCE

The summary in Table 8-30 illustrates the steep spatial gradients in ΣDDT concentrations in five distinct ecosystem components (sediment, overlying water, and mussel, goby, and surfperch residues) over the relatively short distances separating the three channels. With the exception of the 50-fold gradient in average sediment concentration, concentrations drop by factors of about 5X to 10X from Lauritzen Channel to Santa Fe Channel. The decrease obtained over the interval between the Santa Fe and Richmond Inner Harbor Channels range from about 6 to 12 fold. A corresponding comparison for dieldrin is presented in Table 8-31. Again, the steepest gradient between Lauritzen and Santa Fe is observed in the average sediment concentrations (approximate 20-fold decrease), with the other values decreasing by factors of 5X to 12X. Between the Santa Fe and Richmond Inner Harbor Channel sites, concentrations decrease by factors of 4X to 20X.

These sharp gradients in both sediment and water concentrations away from Lauritzen Channel in both ΣDDT and dieldrin strongly indicate that Lauritzen Channel is the ultimate source of contamination to the surrounding sites. Because of the high concentrations in Lauritzen Channel, the mass transport of minimal amounts of water or sediment could result in ecologically significant transport of ΣDDT or dieldrin. For example, mixing 1 liter of unfiltered Lauritzen overlying water with 49 liters of Richmond Inner Harbor water (assuming 0 ΣDDT concentration) would result in a water concentration at the ΣDDT Water Quality Criterion of 1 ng/L. With the 5-6 foot tidal range, there is a mechanism for the daily movement of water and suspended particles out of Lauritzen Channel into Santa Fe. Because ΣDDT water and sediment concentrations are about an order-of-magnitude higher in

Santa Fe Channel than in Richmond Inner Harbor Channel, the Santa Fe Channel is probably a secondary source of contamination to the Richmond Inner Harbor Channel. Exchange of water between Richmond Inner Harbor Channel and San Francisco Bay would ultimately transport Lauritzen contaminants not sequestered in Santa Fe and Richmond Channels into the San Francisco Bay ecosystem.

Besides the strong spatial gradients, three additional lines of evidence support the contention that Lauritzen Channel is a pollutant source. First, the higher water-to-sediment ratios for both ΣDDT and dieldrin in the Santa Fe and Richmond Inner Harbor Channels are suggestive of the movement of more contaminated water into less contaminated sites (see Section 8.2.). Second, the higher AFs found in the field infauna in Richmond Inner Harbor Channel suggests ingestion of suspended particles with a higher concentration than found in the Richmond Inner Harbor bulk sediment (see Section 8.3.). The last set of data are the higher mussel residues/sediment concentration ratios for ΣDDT and dieldrin in Richmond Inner Harbor Channel, which again suggest transport of more contaminated particles and/or water from Lauritzen Channel (see Section 8.3.)

With the high K_{oc}s of dieldrin, DDT and its metabolites, transport associated with particulate matter probably is more important than movement of dissolved compounds. Sorbed contaminants can either be transported on suspended particulates or by bed load transport of surficial sediment particles.

The practical consequence of this outfluxing of pollutants is that remediation of Lauritzen Channel should reduce exposure in Santa Fe and Richmond Inner Harbor Channels, improving ecological conditions even if no remediation actions were taken in these two channels. Remediating Lauritzen Channel should have the largest, or at least the most rapid, impact on the water concentrations and water-borne exposures in the Santa Fe and Richmond Inner Harbor Channels. To account for this possibility, both the site-specific WSR and the Lauritzen Channel WSR are used to predict WQC related remediation levels in the Santa Fe and Richmond Inner Harbor Channels. This assumes that the Lauritzen Channel WSR reflects the "normal" pollutant flux from sediment in the absence of transport of higher concentration material. Reducing the pollutant input should also reduce ΣDDT and dieldrin sediment concentrations in the Santa Fe and Richmond Inner Harbor Channels, though any measurable reduction could take years.

10.4 EXTENT OF EXISTING EXPOSURE

10.4.1. Sediment and Interstitial Water Concentrations

The sediment data from this study unequivocally demonstrate that portions of Richmond Harbor are extensively contaminated with ΣDDT (Tables 8-1, 8-2, 8-11A, B, 8-30, 8-31). Sediment concentrations of ΣDDT in Lauritzen Channel are the highest reported in San Francisco Bay (Table 8-11A) and among the highest in the country. The Santa Fe Channel also has high concentrations when compared to other parts of San Francisco Bay. The average sedi-

ment concentration of the Richmond Inner Harbor Channel does not indicate extensive ΣDDT sediment contamination, with sediment concentrations about equal to the regional averages, though several stations at the end of the channel have moderate concentrations (Tables 8-1).

Presently, there are no regulatory criteria with which to compare these bulk ΣDDT sediment concentrations. However, the ΣDDT interstitial water concentrations can be compared to the toxicity-based acute WQC and chronic toxicity effects levels predicted from acute/chronic ratios (see Section 5.1.2.). The mean ΣDDT interstitial water concentration in Lauritzen Channel exceeds the acute WQC by 13-fold (Table 10-1). This strongly suggests that bulk sediment concentrations are sufficiently high to result in acute mortality of certain benthic species. The mean Santa Fe ΣDDT interstitial water concentration is 2-fold higher than the acute value, but this violation is due to high concentration at a single station (Station 6). Thus, ΣDDT-related impacts on benthic species may be expected at least at certain locations in the Santa Fe Channel. The ΣDDT interstitial water concentrations in Richmond Inner Harbor Channel bracket the chronic effect predicted from acute/chronic ratios, suggesting that, at worst case, ΣDDT could cause chronic impacts on the most sensitive benthic species.

Dieldrin sediment concentrations showed the same spatial pattern as ΣDDT with very high concentrations in Lauritzen Channel, regionally high concentrations in the Santa Fe Channel, and about regional values in the Richmond Inner Harbor Channel (Table 8-11B). These sediment concentrations can be compared to the interim sediment quality criterion for dieldrin (17 μg/g OC). The mean concentration in Lauritzen Channel exceeds the SQC value (Table 10-6), with the highest station twice the criterion concentration. Neither site averages nor the stations with the highest concentrations exceeded the criterion in Santa Fe or Richmond Inner Harbor Channel.

10.4.2. Overlying Water Concentrations

Concentrations in the overlying water showed the same pattern as the sediment pollutants, with average concentrations of ΣDDT in whole water samples decreasing approximately 50-fold between Lauritzen and Richmond Inner Harbor Channels (Table 8-30). The corresponding decline for dieldrin was at least 20-fold (Table 8-31). The average concentrations of ΣDDT for whole water samples from the Lauritzen Channel (50 ng/L) and the Santa Fe Channel (8.6 ng/L) both substantially exceed EPA's ambient Water Quality Criteria chronic value of 1.0 ng/L (Table 10-1), indicating that adverse levels of bioconcentration would occur at these sites. Although the acute WQC, which is based on toxicity, was not violated, the Lauritzen ΣDDT water concentration was 40% of the value, suggesting that the water concentration may be high enough to result in chronic toxicity in sensitive water-column species.

Dieldrin concentrations in the overlying water were less

than 1/10th the acute WQC at all sites (Table 10-4), suggesting that the concentrations are below those that would cause acute or chronic water column toxicity. However, the average concentration in the Lauritzen Channel whole water samples substantially exceeds the residue-based WQC chronic value of 1.9 ng/L. The overall mean Santa Fe value just exceeds the criterion. The Richmond water is below the detection limit of 1.0 ng/L. These results indicate that the overlying water dieldrin concentrations are high enough to result in detrimental levels of bioconcentration in the Lauritzen Channel and possibly in the Santa Fe Channel.

10.4.3. Exposure as Determined by Tissue Residues

One ecological impact of tissue residues in fish and benthic organisms is as a biologically-mediated exposure route to higher trophic levels, including fish-eating birds and wildlife. The chronic Water Quality Criteria and NAS residues for ΣDDT and dieldrin are designed to protect these higher trophic levels. Besides impacts on higher trophic levels, residues can impact the contaminated organism directly. Chronic DDT residue levels of 3,000 - 6,250 μg/kg (wet), based on reduced survival in fish fry, were reported in EPA's AWQC document (U.S. EPA, 1980a) (Table 5-4). These residues are used herein to predict the potential for reproductive impacts in both fish and benthic invertebrates.

Mussels: Mussels are frequently used as a bioindicator organism for water exposures. The average concentrations of ΣDDT and dieldrin in the whole soft tissues of Mytilus galloprovincialis from Lauritzen Channel are three and at least two orders-of-magnitude greater, respectively, than those measured in control specimens of M. californianus from the central Oregon coast. Residues of ΣDDT and dieldrin in Lauritzen mussels were also 72X and 24X, respectively, higher than in Richmond Inner Harbor Channel. The EPA chronic WQC-based residue and NAS standards are based on concentrations in fish, but are used here for comparison since several bird species feed on mussels. ΣDDT residues exceed the chronic WQC based residue and the NAS standard for DDT in both Lauritzen and Santa Fe Channels (Table 10-2). With dieldrin, the chronic WQC residue is not violated in any of the sites, versus the NAS standard which is violated in Lauritzen and Santa Fe Channels (Table 10-5).

Fish: To gain a better statistical representation, the "fish" residues in Tables 10-2 and 10-5 are based on the average of the whole body residues in gobiids and shiner surfperch. Both species appear to be residents and are assigned to trophic level 3.25. Surfperch are water-column organisms but prey on bottom fauna, potentially accumulating pollutants both from the water column and from the benthos. Gobys are benthic and presumably accumulate most or all of their contaminant load from benthic prey and/or direct contact with the sediment. The combined average "fish" residue should generally represent residues in resident fish of a similar trophic level. In any case, the residues of the two species are within about 50% of each other

The average whole body concentrations in the composite fish exceed the FDA interstate commerce seafood limit for ΣDDT in Lauritzen Channel and is 98% of the FDA value for dieldrin (Tables 10-2 and 10-5). The combined fish residue also substantially exceeds the chronic ΣDDT WQC residues to protect fish-eating birds in Lauritzen and Santa Fe Channels. The more stringent NAS standard for ΣDDT is violated in Lauritzen, Santa Fe, and Richmond Inner Harbor Channels. These data indicate that existing ΣDDT fish residues are sufficiently high to pose a risk to higher trophic levels, including pelicans, throughout much of Richmond Harbor.

Anchovy, an important prey item for the endangered brown pelican, was captured in all three channels. Anchovy differ from surfperch and gobiids in that they have little direct association with the benthos and are highly mobile and, presumably, any individual spends limited time within Richmond Harbor. Anchovy residues exceed the WQC residue for ΣDDT in both Santa Fe and Richmond Inner Harbor Channels. The NAS residue for ΣDDT was exceeded in all three channels (Table 8-26A), with a residue more than 10-fold greater than the standard in the Santa Fe Channel. The NAS dieldrin residue also was exceeded by several fold in both the Lauritzen and Santa Fe Channels. These results indicate that anchovy, and presumably other mobile fish, swimming into Richmond Harbor can accumulate ecologically harmful levels of ΣDDT and dieldrin during a transient exposure. Because of the slow depuration of ΣDDT and dieldrin (*e.g.*, DDT half life of 63-428 days in menhaden, Warlen *et al.*, 1977), it is likely that these contaminant loads will be retained for a sufficient duration for the anchovy to reenter the San Francisco Bay food web. Thus, the contamination in Richmond Harbor can result in the exposure of fish, wildlife, and piscivorous birds in San Francisco Bay even if these species or individuals do not feed in Richmond Harbor.

Benthic Invertebrates: For sediment-associated pollutants, the residues in the laboratory-exposed Macoma nasuta corrected to steady-state offer the most accurate measure of sediment bioavailability in benthic organisms. ΣDDT and dieldrin residues ranged over 2 to 3 orders-of-magnitude from Lauritzen to Richmond Inner Harbor Channel (Table 8-16, 10-2, 10-4). The ΣDDT and dieldrin tissue residues in clams exposed to Lauritzen Channel sediment are among the highest ever reported. Residues in filter-feeding benthos are several fold lower than those in M. nasuta but still contain high residues and show the same spatial pattern (Table 8-19).

The Macoma residues can be compared to the WQC-based residues and NAS standards for fish (Table 10-2 and 10-5). As discussed previously, benthic bivalves are potential prey for diving birds, so that the application of standards related to protecting birds is ecologically relevant. The average ΣDDT residue in M. nasuta exceeds both the chronic WQC and the NAS residues in all stations out to Station 7 in the Richmond Inner Harbor Channel. For dieldrin, the WQC residue criterion is exceeded only in Lauritzen

Channel while the dieldrin NAS standard is closely approached or violated in both Lauritzen and Santa Fe Channels.

10.5. EXTENT OF EXISTING ECOLOGICAL EFFECTS

Relating changes in benthic community structure to pollutant stress is a frequently used method of determining the extent and nature of the biological effects of sediment contamination (*e.g.*, Long and Chapman, 1985; Swartz *et al.*, 1986; Chapman *et al.*, 1987; Ferraro *et al.*, in press). While no single or set of measures of community indices captures the complexity of a benthic community, the Infaunal Index and the abundance of amphipods have been used successfully to describe or predict changes in a variety of benthic communities (Mearns *et al.*, 1982; Swartz *et al.*, 1982). These two measures also had the strongest relationship with sediment ΣDDT concentrations (Table 9-2).

Bascom *et al.*'s (1978) criteria for "control" (II > 83) and "changed" (II <=60) benthic conditions are used as criteria for the Infaunal Index. The former value would indicate the sediment pollutants were having no or a negligible effect on the community, while the latter would indicate that sediment pollutants were having a sufficient impact to change the relative abundance of various key species groups. Although none of the site means were below the 60 criterion, there were several individual values below 60 in Lauritzen Channel, indicating a stressed condition. With one exception,,the control value of >83 was obtained only in the Richmond Inner Harbor Channel, Station 9.

Sediment bioassays are the second general method of assessing the biological impacts of sediment contamination. These controlled laboratory tests complement the field survey. In particular, they are not subject to effects from non-chemical stresses (*e.g.*, temperature, salinity, physical disturbance) or biological interaction (*e.g.*, predation, competition) that can influence the results of a benthic survey. Eohaustorius estuarius displayed toxicity in all of the stations, with the mean toxicity in Lauritzen Channel significantly greater than the control sediment. The amphipod mortality at Lauritzen Channel Station 1 ranks among the worst cases of sediment toxicity reviewed by Swartz *et al.* (1989). The cause of the approximately 20% reduction in survival of E. estuarius at the low ΣDDT contaminated stations in Richmond Inner Harbor Channel relative to the amphipod's native Yaquina Bay, Oregon sediment is unexplained and probably not related to ΣDDT or dieldrin.

Relations between ΣDDT, sediment toxicity, and amphipod abundance in this investigation correspond with those on the marine Palos Verdes Shelf. The threshold for 10-day toxicity occurred at about 300 μg/g OC. The abundance of amphipods in the field (excluding Grandidierella japonica) was reduced at ΣDDT concentrations greater than about 100 μg/g OC. This 100 μg/g OC is taken as the minimum ecological effects concentration for amphipods. The average ΣDDT concentration in Lauritzen Channel exceeds this effects level by almost 20-fold. The minimum ecological effects concentration is not exceeded in the Santa Fe

or Richmond Inner Harbor Channels.

10.6. POTENTIAL REMEDIATION LEVELS

The relationships established from the laboratory and field studies allow predictions of sediment concentrations of ΣDDT and dieldrin required to achieve various water column, tissue residue, or benthic community end-points. Predictions of these potential remediation levels are based on the assumption that the local marine sediments are the ultimate source for the ΣDDT and dieldrin contamination at each of the sites. To the extent that runoff, erosion, or transport from other sites contribute to the exposure, the predictions for the required sediment clean-up levels may be overly conservative (*i.e.*, lower than required). In particular, transport of contaminants from the Lauritzen Channel is likely to make predictions for other channels conservative.

There is uncertainty associated with all these predictions, and the reader is referred to the appropriate section for discussion of the underlying assumptions and the empirical measurements of variability (*e.g.*, standard error, etc.). In nearly all cases, the uncertainties associated with a measurement or model are relatively small compared to the more than three order-of-magnitude range in sediment ΣDDT concentration. A few of the predictions are either based on the tails of the regression or using/predicting dieldrin sediment concentrations that are close to or below the sediment detection limit. There is more uncertainty in these predictions.

A standard TOC of 1.9% was used to convert any predictions based on organic carbon normalized sediment concentrations to bulk sediment concentrations.

10.6.1. Remediation Levels Based on Water Quality Criteria

It is possible to predict the sediment concentration required to achieve an overlying water criterion if there is a consistent relationship between sediment and water concentrations. The range in the maximum and minimum water-to-sediment ratios (WSR) for both ΣDDT and dieldrin is only about 2-fold to 5-fold over both sampling periods (Tables 8-13 and 8-14). This relatively low variability indicates that whole water concentrations can be predicted with a reasonable degree of accuracy from sediment concentrations, at least during non-storm conditions. The sediment concentration required to achieve a water concentration can be predicted from:

$$Cs = Cw/WSR \qquad \text{Eq. 10-1}$$

where

Cs = sediment concentration (μg/kg dry)

Cw = target water concentration (μg/l)

WSR = water-sediment ratio (kg/l)

Table 10-7 gives the predicted ΣDDT sediment concentrations required to achieve various water quality criteria, while Table 10-8 gives the equivalent sediment concentrations based on dieldrin WQC. Although specific stations are identified in Tables 10-7 and 10-8, achieving water concentrations is best based on channel averages or major portions of channels rather than a localized "hot spot" (*i.e.*, Station 6).

None of the Lauritzen Channel would have to be remediated based on the sediment concentrations to achieve the acute WQC for ΣDDT or dieldrin. However, all of the Lauritzen stations would have to be remediated to achieve the ΣDDT sediment concentration based on the chronic ΣDDT WQC, while Stations 1-3 would have to be remediated based on the chronic dieldrin WQC.

The extent of remediation required in the Santa Fe and Richmond Inner Harbor Channels depends upon which WSR is used. To achieve the chronic ΣDDT WQC with the channel-specific WSRs, all of the Santa Fe would have be remediated as well as the Richmond Inner Harbor Channel to between Stations 7 and 8. For dieldrin, the Santa Fe and Richmond Inner Harbor Channel averages are below the sediment concentrations.

As discussed in Section 8.2., the higher WSRs in the Santa Fe and Richmond Inner Harbor Channels may be due, at least in part, to the flux of contaminated water from Lauritzen Channel. If the Lauritzen Channel is remediated, any elevation due to this flux would be greatly reduced or completely eliminated. Under this scenario, it would be more appropriate to use the Lauritzen Channel WSRs and the corresponding sediment concentrations. Using the Lauritzen Channel WSR, the average Santa Fe Channel ΣDDT sediment concentration is below the 1010 μg/kg ΣDDT sediment remediation level. This approach suggests that the Santa Fe would not have to be cleaned-up to achieve the chronic ΣDDT and dieldrin WQC if sediment concentrations in the Lauritzen Channel were reduced.

10.6.2. Remediation Levels Based on Tissue Residue Criteria

Pelagic Tissue Residues: With a method of predicting water concentrations, it is possible to determine the sediment concentrations required to achieve tissue residue criteria in organisms whose major route of exposure is through the overlying water. The equation for this prediction is based on the following equation:

$$Cs = TR/(WSR * BCF_{uw}) \qquad \text{Eq. 10-2}$$

where

TR = Tissue residue (μg/kg wet)

where

TR = Tissue residue (μg/kg wet)

BCF_{uw} = Bioconcentration factor based on unfiltered water, for whole body (l/kg)

This approach is applied to mussels and shiner surfperch for ΣDDT and dieldrin (Tables 10-9 and 10-10). Because some of the surfperch uptake may be from ingestion of benthic prey (Table 8-23), the use of a BCF_{uw} model may overestimate the actual uptake from water alone. Therefore, the remedial sediment concentrations predicted from surfperch may represent lower limits for sediment remediation for organisms with a water uptake exposure.

Using the Lauritzen Channel WSR for all channels, Stations 1-3 would have to be cleaned-up based on the chronic WQC ΣDDT tissue residues in mussels (Table 10-9). Up to Station 4 in Lauritzen Channel would also have to be cleaned-up based on the chronic ΣDDT WQC in shiner surfperch and the NAS ΣDDT residue in mussels. The Santa Fe Channel would have to be cleaned-up to achieve the NAS ΣDDT residue in surfperch. Remediating for ΣDDT would reduce dieldrin below its required sediment concentration for the equivalent ecological criterion (Table 10-10) except at Station 7 which exceeds the NAS remediation sediment value based on shiner surfperch.

Using the channel-specific WSRs results in about 10-fold and 2-fold lower sediment concentrations for ΣDDT and dieldrin, respectively. All stations in the Lauritzen and Santa Fe Channels would require remediation to achieve chronic WQC ΣDDT residues in mussels, while the end of Richmond Inner Harbor, Stations 7 and 17, would also require remediation based on the surfperch. The ΣDDT sediment concentration to achieve the more stringent NAS residue is achieved at around Station 8 and seaward based on mussels. When surfperch are used, the ΣDDT concentrations are below the NAS remediation concentration only at the reference station (Station 9) and stations in San Francisco Bay (Stations 19-23). A number of peripheral and basin sites in San Francisco Bay exceed the ΣDDT sediment concentration required to achieve the NAS ΣDDT residue in surfperch (Table 8-11A), suggesting the violations in cleaner portions of Richmond Inner Harbor Channel may represent regional contamination.

Anchovy are important prey items for pelican and other fish-eating birds, and their present residues of ΣDDT or dieldrin violate the chronic WQC or NAS standard in each of the three channels. Unfortunately, the mobility of anchovy violates the assumption that the fish will be exposed to a constant water concentration for a sufficient duration to approach steady-state tissue residues; therefore it is not appropriate to use Equation 10-2. Rather, a kinetic approach is needed to accurately predict residues in mobile species with a transient exposure (Lee, 1992), but the rate constants are not available. However, any remediation that protects a resident species, such as the surfperch,

should generally protect a mobile species with a shorter exposure duration.

Benthic Tissue Residues: Residues in benthic fish and invertebrates are ecologically important both as a source of contamination for birds and marine mammals, and in the detrimental effects they have on the contaminated organisms. Accurate predictions of residues in M. nasuta can be made either using the equilibrium partitioning (EqP) bioaccumulation model or linear regressions. The EqP model is used in this analysis since it allows for direct incorporation of TOC, which could change during a remediation involving sediment removal. ΣDDT residues in a trophic level 3.25 benthic fish are predicted using the exponential biomagnification model (EBM) trophic model and the EqP model. Dieldrin residues are predicted by the EqP model only (Table 10-11). Because the AFs for gobiids are used in the EqP models, the fish is referred to as "gobiid" in Table 10-11, though the results should generally apply to any benthic-feeding trophic level 3.25 fish (see Section 9.4.).

The sediment concentrations to achieve the chronic WQC ΣDDT residues in fish and invertebrates range from about 200 to 500 μg/kg (Table 10-11). These ΣDDT sediment concentrations are exceeded at all stations up to about Station 7 in Richmond Inner Harbor Channel. The NAS standard for ΣDDT results in violations up to Station 8 using the sediment concentrations based on the EqP predictions with gobiids and M. nasuta.

With dieldrin, the sediment concentrations based on the chronic WQC residues in gobiids are exceeded only in the most contaminated section of Lauritzen Channel (Stations 1-3). The dieldrin sediment concentrations based the NAS standard using M. nasuta is presently violated through Station 7, while the lower concentrations based on the gobiids is also violated at Station 17. The dieldrin concentration derived from gobiids is also violated at Station 23 even though the stations between Stations 7 and 17 had lower or non-detectable concentrations. The violation at Station 23 could reflect analytical variation at these low sediment concentrations or spatial heterogeneity.

Using the ΣDDT effects-based tissue residue for fish fry, remediation would be required at Stations 1-3 based on both benthic invertebrates and benthic fish.

10.6.3. Remediation Based on Benthic Community Structure and Sediment Toxicity:

Sediment concentrations and benthic community parameters tended to group into "high", "intermediate", and "low" pollutant groups (Section 9.1.). One approach to determining remediation concentrations is to compare these group Infaunal Indices values to the "control" and "changed" values. None of the mean group Infaunal Indexes are below the criterion for "changed", though values less than 60 were observed at the high pollutant stations (Stations 1-3) (Table 10-12). The occurrence of these localized "changed" portions of the benthic community in Lauritzen Channel suggest that the community as a whole is approaching its assim-

tion should result in "normal" (>60) Infaunal Index conditions at all sites within the channel. If the goal were to obtain "control" Infaunal Index conditions (II = 83) at all sites, remediation would require cleaning up the Lauritzen, Santa Fe, and parts of the Richmond Inner Harbor Channels (Stations 1-7) to approximately the ΣDDT concentrations in the low pollutant group.

A regression approach can be used with the same data and criteria to predict a remedial ΣDDT sediment concentration. The advantage of using a regression is that the sediment concentration for a specific Infaunal Index value can be predicted, but it must be recognized that the regression explained only about 50% of the variation. Using the goal of obtaining "control" conditions at all sites, the predicted clean-up concentration is 9.7 μg/g OC (184 μg/kg @ 1.9% TOC). This would require clean-up of all the Lauritzen and Santa Fe Channels, as well as Station 7 of Richmond Inner Harbor Channel, a result similar to that obtained by comparing group means.

It is also possible to use the regressions to predict what ΣDDT sediment concentration would cause a specified value or percentage decline compared to a zero pollutant exposure (*i.e.*, change from the value of the Y-intercept in the regressions in Tables 9-2 through 9-5). The advantage of this approach is that it normalizes any change to local conditions rather than using standards (*i.e.*, 83 and 60) developed for Southern California Bight communities. A 20% decline in the Infaunal Index is tentatively suggested here as a indicator of an ecologically significant change in benthic community structure and function. The 20% decline is consistent with the use of 20% amphipod mortality as a decision point in the evaluation of dredge materials (EPA/ACE, 1991). The ΣDDT sediment concentration resulting in an Infauna Index equal to 80% of the 0 ΣDDT value (II_{80}=71.59) is 533 μg/g OC (10,126 μg/kg at 1.9% TOC). Stations 1-3 in Lauritzen Channel violate this concentration and would require clean-up.

The same relative approach to local zero pollutant conditions can be used with amphipod density. The predictions can be made with all amphipod species or with Grandidierella japonica, a ΣDDT tolerant species, excluded. Both regressions are highly significant (P<0.01) (Table 9-2). A 20% reduction in amphipods w/o G. japonica is predicted to occur at a ΣDDT concentration of 3.96 μg/g OC (75 μg/kg at 1.9% TOC), resulting in present violations up to and including Station 8 in Richmond Inner Harbor Channel. Including G. japonica results in a slightly higher sediment remediation concentration (4.75 μg/g OC or 90 μg/kg), as expected by the addition of a pollutant tolerant species, but the difference is minimal.

The laboratory toxicity test offers an independent method of predicting remedial concentrations protective of benthic organisms (Table 10-12). The 10-day LC_{50} for ΣDDT with Eohaustorius estuarius was 2,500 μg/g OC (47,500 μg/kg, dry). Because this sediment concentration results in 50% acute mortality, it is not ecologically protective. A more protective value from the sedi-

isms (Table 10-12). The 10-day LC50 for ΣDDT with Eohaustorius estuarius was 2,500 μg/g OC (47,500 μg/kg, dry). Because this sediment concentration results in 50% acute mortality, it is not ecologically protective. A more protective value from the sediment bioassay is the threshold for toxicity in the 10-day test, 300 μg/g OC (5700 μg/kg @ 1.9% TOC). This concentration would require the clean-up of most of Lauritzen Channel (Stations 1-3).

Because 300 μg/g OC is a threshold for toxicity, species more sensitive than E. estuarius may experience toxicity at this sediment concentration. And, in fact, since these tests were completed, Rhepoxynius abronius showed 0 survival in Lauritzen Channel sediment (White *et al.*, 1993), suggesting it is more sensitive than E. estuarius to ΣDDT. As a method to account for more sensitive species as well as chronic exposures, the amphipod toxicity data were combined with the field distribution data for seven amphipod species (excluding G. japonica) to generate a minimum ecological effects concentration for ΣDDT of 100 μg/g OC (1,900 μg/kg ΣDDT at 1.9% TOC). (The field data used in deriving the minimum ecological effects concentration included small amphipods (>0.5mm <1.0 mm) which were not used in the amphipod regressions and, therefore, is a different data set.) This ecological criterion would require remediation of all of Lauritzen Channel.

The minimum ecological effects concentration is several fold higher than the predictions based on the amphipod regressions. Part of the reason for the difference is that the minimum ecological effects concentration approach assumes there is a toxicological threshold while the linear regression has no threshold. Additionally, the variability in the field density of amphipods, especially at low ΣDDT sediment concentrations was high (Figures 9-1, i, j). Consequently, the accuracy of the regressions at low ΣDDT sediment concentrations is low. Nonetheless, these two approaches should bracket the ΣDDT sediment concentration needed to protect amphipods, with the minimum ecological effect concentration the more accurate.

Although ΣDDT is considered the major cause of sediment toxicity, the question remains whether remediating the DDT-contaminated sediment would reduce dieldrin to a sufficient extent. An interim sediment quality criterion has been developed for dieldrin that can be used to determine what areas would require remediation based on toxic effects on benthos by this compound alone. The interim SQC of 17 μg/g OC indicates that Stations 1-3 in Lauritzen should be remediated based on dieldrin concentrations. All the ecologically protective benthic predictors considered here require clean-up of these stations, as do most of the other water and tissue criteria considered previously. Therefore, at least in terms of effects on the benthos, cleaning-up the ΣDDT contaminated sediment would assure sufficient remediation of dieldrin contamination.

10.7. SUMMATION OF VIOLATIONS

Gaining a perspective of the overall ecological risk in an assessment with multiple end-points is difficult for no other reason than the magnitude of numbers to comprehend. One qualitative method of summarizing the data is to sum the number of end-points that were violated in each channel. That is, simply count all the end-points listed in Tables 10-1 to 10-3 that exceeded each of the criteria. Table 10-13 summarizes the violations by sets, while Figure 10-1 presents these data graphically. One limitation of simply summing the number of violations is that it does not differentiate between a site that violates a criterion by orders-of-magnitude versus one that barely exceeds it. To account for differences in magnitude, the "exceedance factors" (the values given in Tables 10-1 to 10-3) are summed for each of the channels (Figure 10-2). This analysis is limited to DDT since it is the dominant pollutant. All human health criteria are excluded from both calculations.

It must be recognized that these summations are qualitative aids to visualizing the data and not quantitative assessments. The summations would change depending upon the number of criteria included and how the criteria are expressed. For example, including both the water concentration for a residue-based WQC and the corresponding residue tends to give this criterion more weight. Additionally, all criteria are treated equally, when, in fact, all criteria are not of the same ecological or regulatory significance. In particular, prediction of sediment concentrations to achieve the WQC in overlying water should be given special consideration because of the WQC's regulatory standing. Nonetheless, Figures 10-1 and 10-2 are useful in gaining a qualitative understanding of spatial distribution of the total ecological impact.

The sites divide into three general groups based on the distribution of the number of violations (Figure 10-1). The distribution of exceedance factors by channel (Figure 10-2) shows a much sharper gradient than the distribution based on the presence/absence of violations. This sharper gradient is not surprising given the large concentration difference among the channels. Although the gradient is sharper, the pattern is similar to that with the presence/absence data.

The difference between the two types of representations points out that while there were a number of criteria violations in the Santa Fe Channel and at the end of the Richmond Inner Harbor Channel, the extent of these violations tended to be small relative to the Lauritzen Channel. Clearly, cleaning-up Lauritzen Channel would result in the greatest reduction in the extent of violations. As discussed previously, cleaning-up the Lauritzen Channel also would reduce the flux of ΣDDT and dieldrin into the Santa Fe and Richmond Inner Harbor Channels, which may be sufficient to correct some of the violations observed at these stations.

TABLE 10-1. Summary of existing violations of water quality end-points for ΣDDT (factor by which average channel concentrations violate criterion). Violations (factor ≥ 1.0) underlined.

CRITERION	CRITERION VALUE	LAURITZEN OVERLYING WATER	LAURITZEN INTERSTITIAL WATER[1]	SANTA FE OVERLYING WATER	SANTA FE INTERSTITIAL WATER[2]	RICHMOND OVERLYING WATER	RICHMOND INTERSTITIAL WATER[3]
Concentration	-	0.050 µg/l Table 8-30	1.62 µg/l Table 8-1	0.008 µg/l Table 8-30	0.260 µg/l Table 8-1	0.001 µg/l Table 8-30	0.019 µg/l Table 8-1
EPA Saltwater Acute WQC	0.13 µg/l	0.4x	13x	0.06x	2x	0.008x	0.1x
EPA Saltwater Chronic WQC[4]	0.001 µg/l	50x	1620x	8x	260x	1x	19x
Chronic Toxicity to Sensitive Invertebrates Predicted from Acute/Chronic Ratios	0.013 - 0.065 µg/l	.8x - 4x	25x - 125x	0.1x - 0.6x	4x - 20x	0.01x-0.08x	0.3x - 1.5x
EPA Human Health WQC[4]	0.00059 µg/l	85x	2746x	14x	441x	2x	32x
CA WQ Objective: Saltwater Aquatic Life, Daily Average[4]	0.001 µg/l	50x	1620x	8x	260x	1x	19x
CA WQ Objective: Human Health, 30-Day Average[4]	0.0006 µg/l	83x	2700x	13x	433x	2x	32x

1 = Mean of total interstitial water Stations 1-4 (Table 8-1)
2 = Mean of total interstitial water Stations 5 and 6 (Table 8-1)
3 = Mean of total interstitial water Stations 7, 8, and 9 (Table 8-1)
4 = Residue-based criterion for protection of fish-eating birds or humans. Of less ecological significance for IW.

TABLE 10-2. Summary of existing violations of tissue residue quality end-points for ΣDDT (factor by which average residue exceeds criterion). Violations (factor ≥ 1.0) underlined.

CRITERION	CRITERION VALUE	LAURITZEN MACOMA[1]	LAURITZEN MUSSELS	LAURITZEN FISH[2]	SANTA FE MACOMA[1]	SANTA FE MUSSELS	SANTA FE FISH[2]	RICHMOND MACOMA[1]	RICHMOND MUSSELS	RICHMOND FISH[2]
Whole body residue (µg/kg wet)		6970 Table 8-16	2900 Table 8-24A	6450 Table 8-26A	279 Table 8-16	350 Table 8-24A	800 Table 8-26A	78 Table 8-16	40 Table 8-24A	115 Table 8-26A
Basis of EPA Chronic WQC: Residues in fish to protect brown pelican	150 µg/kg (wet)	46x	19x	43x	2x	2x	5x	0.5x	0.3x	0.8x
Reduced survival in fish fry	3,000 -6,250 µg/kg (wet)	1x - 2x	0.5x - 0.97x	1.0x - 2.2x	0.04x - 0.09x	0.06 - 0.1x	0.1x - 0.3x	0.01 - 0.03x	0.006 - 0.01	0.02x - 0.04x
Basis of EPA Human Health WQC: 10^{-6} excess cancer risk	Fish = 32 µg/kg (wet); Shellfish = 260µg/kg (wet)	27x	11x	202x	1.1x	1.3x	25x	0.3x	0.2x	4x
NAS Residue in fish, to protect fish-eating birds and mammals	50 µg/kg (wet)	139x	58x	129x	6x	7x	16x	2x	0.8x	2x
FDA - Fish	5,000 µg/kg (wet)	1.4x	0.6x	1.3x	0.06x	0.07x	0.2x	0.02x	0.008x	0.02x

1 = Data are 28-day laboratory residues corrected to steady-state residues and converted to wet weights using a wet/dry ratio of 0.147.
2 = Average of whole body residues for gobiids and shiner surfperch.

TABLE 10-3. Summary of existing violations of sediment quality and benthic community structure end-points for ΣDDT (factor by which average channel values exceeds criterion). Violations (factor ≥ 1.0) underlined.

CRITERION	CRITERION VALUE	LAURITZEN CHANNEL	SANTA FE CHANNEL	RICHMOND CHANNEL
Mean ΣDDT Sediment Conc. (µg/kg dry wt / µg/g OC)	-	38,579/1981[1] Table 8-1	878/48[2] Table 8-1	109/9.7[3] Table 8-1
EPA SQC (µg/g OC)	0.828[4] µg/gOC	2393x	58x	12x
10-day LC_{50} w/Eohaustorius estuarius	47500µg/kg /2500µg/gOC	0.8x	0.02x	0.004x
Threshold of toxicity with E. estuarius	5700µg/kg /300µg/gOC	6.6x	0.2x	0.03x
Minimum ecotoxicological effects conc. on amphipods	1900µg/kg /100µg/gOC	19.8x	0.5x	0.1x
Mean Infaunal Index >83?	II >83 Control[5] II >60 Normal II <60, >30 Changed II <30 Degraded	NO (69.8) Fig. 9-1, d	NO (73.8) Fig. 9-1, d	YES (85.5[6]) Fig. 9-1, d
Any Infaunal Index values <60?	"	YES	NO	NO
Acute Amphipod Toxicity	20% toxicity compared to reference[7]	42% Table 9-7	34% Table 9-7	24% Table 9-7
Density of amphipods/0.075 m^2 (All spp / w/o Grandidierella)	-	28.8/0.75 Table 9-8	21/21 Table 9-8	91/89[6] Table 9-8
Percent reduction in amphipods compared to reference Station 9. (All spp / w/o Grandidierella)	20% toxicity compared to reference[7]	72%/99% Table 9-8	80%/80% Table 9-8	28%/30%[6] Table 9-8

1 = Stations 1-4
2 = Stations 5, 6, 13-16
3 = Stations 7-9, 009, 17, 18
4 = Residue-based for brown pelican and not considered appropriate for sediments. Included for comparative purposes only.
5 = Criteria from Bascom *et al.* (1978)
6 = Station 8 excluded because of prop wash.
7 = Based on 80% amphipod mortality in laboratory tests used in evaluating dredge materials (U.S. EPA/ACE, 1991).

TABLE 10-4. Summary of existing violations of water quality end-points for dieldrin (factor by which average concentration of dieldrin exceeds criterion). Violations (factor ≥ 1.0) underlined.

CRITERION	CRITERION VALUE	LAURITZEN OVERLYING WATER	LAURITZEN INTERSTITIAL WATER[1]	SANTA FE OVERLYING WATER	SANTA FE INTERSTITIAL WATER[2]	RICHMOND OVERLYING WATER	RICHMOND INTERSTITIAL WATER[3]
Concentration	-	0.018 µg/l (Table 8-31)	0.129 µg/l (Table 8-2)	0.002 µg/l (Table 8-31)	0.001 µg/l (Table 8-2)	<0.001 µg/l[4] (Table 8-31)	0.0002 µg/l (Table 8-2)
EPA Saltwater Acute WQC	0.71 µg/l	0.03x	0.2x	0.003x	0.001x	<0.001x	0.0003x
EPA Saltwater Chronic WQC[5] Based on 300 µg/kg in fish oil	0.0019 µg/l	9.5x	68x	1.1x	0.5x	<0.5x	0.1x
FDA Action Level[5] Based on 300 µg/kg in fish	0.12 µg/l	1.5x	11x	0.2x	0.01x	<0.08x	0.02x
Chronic toxicity to sensitive invertebrates	0.11 µg/l[6]	0.2x	1.2x	0.02x	0.01x	<0.009x	0.002x
EPA Human Health WQC (10^{-6})[5]	0.00014 µg/l	129x	921x	14x	7x	<7x	1.4x
CA WQ Objective: Saltwater Aquatic Life Daily Average[6]	0.0019 µg/l	9.5x	68x	1.1x	0.5x	<0.5x	0.1x
CA WQ Objective: Human Health 30-Day Average[6]	0.00014 µg/l	129x	921x	14x	7x	<7x	1.4x

1 = Mean of total interstitial water Stations 1-4
2 = Mean of total interstitial water Stations 5 and 6
3 = Mean of total interstitial water Stations 7, 8, and 9
4 = Concentration below detection limit. The detection limit is used as the maximum possible factor by which the criterion could be exceeded.
5 = Residue-based criterion for protection of fish-eating birds or humans. Of less ecological significance for IW.
6 = From Water Quality Criteria document

TABLE 10-5. Summary of existing violations of tissue residue quality end-points for dieldrin (factor by which average residue exceeds criterion). Violations (factor ≥ 1.0) underlined.

CRITERION	CRITERION VALUE	LAURITZEN MACOMA[1]	LAURITZEN MUSSELS	LAURITZEN FISH[2]	SANTA FE MACOMA[1]	SANTA FE MUSSELS	SANTA FE FISH[2]	RICHMOND MACOMA[1]	RICHMOND MUSSELS	RICHMOND FISH[2]
Whole body residue (µg/kg, wet)	-	168 Table 8-16	97 Table 8-24A	295 Table 8-26A	7.4 Table 8-16	19 Table 8-24A	34 Table 8-26A	1.4 Table 8-16	4 Table 8-24A	4 Table 8-26A
Basis of EPA Chronic WQC: Residues of 300 ppb in fish oil	300 µg/kg (wet)	0.6x	0.3x	0.98x	0.02x	0.06x	0.1x	0.005x	0.01x	0.01x
Basis of EPA Human Health WQC: 10^{-6} excess cancer risk	Fish =0.7 µg/kg (wet); Shellfish = 5.7 µg/kg (wet)	29x	17x	421x	1.3x	3.3x	49x	0.2x	0.7x	6x
NAS Residue in fish to protect fish eating birds	5 µg/kg (wet)	34x	19x	59x	1.5x	4x	7x	0.3x	0.8x	0.8x
FDA action limit - fish, shellfish, and fish oil	300 µg/kg (wet)	0.6x	0.3x	0.98x	0.02x	0.06x	0.1x	0.005x	0.01x	0.01x

1 = Data are 28-day laboratory residues corrected to steady-state residues and converted to wet weights using a dry/wet ratio of 0.147.
2 = Average of the mean whole body residues for gobiids and shiner surfperch.

TABLE 10-6. Summary of existing violations of sediment quality and benthic community structure end-points for dieldrin. Violations (factor ≥ 1.0) underlined.

Note: ΣDDT rather than dieldrin is considered the main pollutant stress (see text). Therefore, the predicted effects, other than the comparison to the dieldrin SQC, are better explained by the ΣDDT sediment concentration (Table 10-3).

CRITERION	CRITERION VALUE	LAURITZEN CHANNEL[1]	SANTA FE CHANNEL[2]	RICHMOND CHANNEL[3]
Mean Dieldrin Sediment Conc. (µg/kg dry wet / µg/g OC	-	438.35/23.04 Table 8-2	23.03/1.11 Table 8-2	2.47/0.22 Table 8-2
EPA Chronic SQC (µg/g OC) (Factor by which SQC is exceeded)	17 µg/g OC[4]	1.4x	0.07x	0.01x

1 = Stations 1-4.
2 = Stations 5, 6, 13-16.
3 = Stations 7-9, 009, 17, 18.
4 = Range from 7.7 to 36 µg/g OC.

TABLE 10-7. Predicted ΣDDT sediment concentrations (μg/kg dry) required to achieve water quality criteria. Specific stations violating criterion indicated by "(Sta #)".

CHANNEL	WATER-SEDIMENT RATIO[1] (KG/L)	SEDIMENT CONC. TO ACHIEVE ACUTE WQC (0.13 μG/L)	SEDIMENT CONC. TO ACHIEVE CHRONIC WQC (0.001 μG/L)	SEDIMENT CONC. TO ACHIEVE HUMAN HEALTH WQC (0.00059 μG/L)
LAURITZEN	0.99×10^{-6}	131,313 (none)	1010 (Sta# 1-4)	596 (Sta# 1-4)
SANTA FE	8.5×10^{-6}	15,294 (none)	118 (Sta# 5-6, 13-16)	69 (Sta# 5-6, 13-16)
SANTA FE[2]	0.99×10^{-6}	131,313 (none)	1010 (Sta # 6)	596 (Sta # 6, 14, 16)
RICHMOND	11.9×10^{-6}	10,924 (none)	84 (Sta# 7, 17)	50 (Sta# 7-8, 17)
RICHMOND[2]	0.99×10^{-6}	131,313 (none)	1010 (none)	596 (none)

1 = Maximum and minimum water sediment ratios given in Table 8-13 can be used to bracket the uncertainty in the predicted sediment concentrations due to variation in the water-sediment ratios.

2 = The Lauritzen WSR and the resulting sediment concentrations may be more appropriate than the channel-specific values if the Lauritzen Channel sediments are remediated.

TABLE 10-8. Predicted dieldrin sediment concentrations (µg/kg dry) required to achieve water quality criteria. Specific stations violating criterion indicated by "(Sta #)".

CHANNEL	WATER - SEDIMENT RATIO[1] (KG/L)	SEDIMENT CONC. TO ACHIEVE ACUTE WQC (0.71 µG/L)	SEDIMENT CONC. TO ACHIEVE CHRONIC WQC (0.0019 µG/L)	SEDIMENT CONC. TO ACHIEVE HUMAN HEALTH WQC (0.00014 µG/L)
LAURITZEN	31.6×10^{-6}	22,468 (none)	60 (Sta# 1-3)	4.4 (Sta# 1-4)
SANTA FE	62.1×10^{-6}	11,433 (none)	31 (Sta# 6)	2.3 (Sta# 5-6, 13-16)
SANTA FE[2]	31.6×10^{-6}	22,468 (none)	60 (none)	4.4 (Sta# 5-6, 13-16)
RICHMOND	<DL[3]	11,433[4] (none)	31[4] (none)	2.3[4] (Sta# 7, 17, 19, 23)
RICHMOND[2]	31.6×10^{-6}	22,468 (none)	60 (none)	4.4 (Sta# 7)

1 = Maximum and minimum water sediment ratios given in Table 8-14 can be used to bracket the uncertainty in the predicted sediment concentrations due to variation in the water-sediment ratios.

2 = The Lauritzen WSR and the resulting sediment concentrations may be more appropriate than the channel-specific values if the Lauritzen Channel sediments are remediated.

3 = Water concentration at or below detection limit so not possible to calculate WSR.

4 = Values calculated using the Santa Fe WSR.

TABLE 10-9. Predicted ΣDDT sediment concentrations (μg/kg dry) required to achieve tissue residue end-points from water exposure. Specific stations violating criterion indicated by "(Sta #)".

CHANNEL	WATER - SEDIMENT RATIO (KG/L)[1]	CHRONIC WQC RESIDUE (150 μG/KG) IN MUSSELS[2]	CHRONIC WQC TISSUE (150 μG/KG) IN SURFPERCH[3]	NAS RESIDUE (50 μG/KG) IN MUSSELS[2]	NAS RESIDUE (50 μG/KG) IN SURFPERCH[3]	HUMAN HEALTH WQC RESIDUE (260 μG/KG) IN MUSSELS[2]	HUMAN HEALTH WQC RESIDUE (32 μG/KG) IN SURFPERCH[3]
LAURITZEN	0.99×10^{-6} Table 8-13	3272 (Sta# 1-3)	1273 (Sta# 1-4)	1091 (Sta# 1-4)	424 (Sta# 1-4)	5672 (Sta# 1-3)	271 (Sta# 1-4)
SANTA FE	8.5×10^{-6} Table 8-13	381 (Sta# 5-6, 13-16)	148 (Sta# 5-6, 13-16)	127 (Sta# 5-6, 13-16)	49 (Sta# 5-6, 13-16)	660 (Sta# 6, 14, 16)	32 (Sta# 5-6, 13-16)
SANTA FE[4]	0.99×10^{-6}	3272 (none)	1273 (Sta# 6)	1091 (Sta# 6)	424 (Sta# 6, 13-16)	5672 (none)	271 (Sta# 5-6, 13-16)
RICHMOND	11.9×10^{-6} Table 8-13	272 (Sta# 7)	106 (Sta# 7, 17)	91 (Sta# 7, 17)	35 (Sta# 7-8, 17-18)	472 (none)	23 (Sta# 7-8, 009, 17-18)
RICHMOND[4]	0.99×10^{-6}	3272 (none)	1273 (none)	1091 (none)	424 (none)	5672 (none)	271 (Sta# 7)

1 = Maximum and minimum water sediment ratios given in Table 8-13 can be used to bracket the uncertainty in the predicted sediment concentrations due to variation in the water-sediment ratios.

2 = Overall average whole body bioconcentration factor based on unfiltered water (BCF_{uw}) for mussels = 46,300 (Table 8-25).

3 = Overall average whole body bioconcentration factor based on unfiltered water (BCF_{uw}) for shiner surfperch = 119,000 (Table 8-27).

4 = The Lauritzen WSR and sediment concentrations may be more appropriate than the channel-specific values if the Lauritzen Channel sediments are remediated.

TABLE 10-10. Predicted dieldrin sediment concentrations (µg/kg dry) required to achieve tissue residue criteria values from water exposure. Specific stations violating criterion indicated by "(Sta #)".

CHANNEL	WATER - SEDIMENT RATIO (KG/L)[1]	CHRONIC WQC RESIDUE (300 µG/KG) IN MUSSELS[2]	CHRONIC WQC RESIDUE (300 µG/KG) IN SURFPERCH[3]	NAS RESIDUE (5 µG/KG) IN MUSSELS[2]	NAS RESIDUE (5 µG/KG) IN SURFPERCH[3]	HUMAN HEALTH WQC RESIDUE (5.7 µG/KG) IN MUSSEL[2]	HUMAN HEALTH WQC RESIDUE (0.7 µG/KG) IN SURFPERCH[3]
LAURITZEN	31.6×10^{-6} Table 8-14	1159 (none)	432 (Sta# 1-3)	19 (Sta# 1-4)	7 (Sta# 1-4)	22 (Sta# 1-4)	1.0 (Sta# 1-4)
SANTA FE	62.1×10^{-6} Table 8-14	589 (none)	220 (none)	10 (Sta# 6, 13-14)	4 (Sta# 5-6, 13-16)	11 (Sta# 6, 13, 14, 16)	0.5 (Sta# 5-6, 13-16)
SANTA FE[4]	31.6×10^{-6} Table 8-14	1159 (none)	432 (none)	19 (Sta# 6)	7 (Sta# 6, 13-16)	22 (Sta# 6)	1.0 (Sta# 5-6, 13-16)
RICHMOND	<DL[5] Table 8-14	589 (none)	220 (none)	10 (none)	4 (Sta# 7, 23)	11 (none)	0.5 (Sta# 7-9, 17, 19, 20, 23)
RICHMOND[4]	31.6×10^{-6} Table 8-14	1159 (none)	432 (none)	19 (none)	7 (Sta# 7)	22 (none)	1.0 (Sta# 7-8, 17, 19, 20, 23)

1 = Maximum and minimum water sediment ratios given in Table 8-14 can be used to bracket the uncertainty in the predicted sediment concentrations due to variation in the water-sediment ratios.

2 = Overall average whole body bioconcentration factor based on unfiltered water (BCF_{uw}) for mussels = 8200 (Table 8-25).

3 = Overall average whole body bioconcentration factor based on unfiltered water (BCF_{uw}) for shiner surfperch = 22,000 (Table 8-27).

4 = The Lauritzen WSR and the resulting sediment concentrations may be more appropriate than the channel-specific values if the Lauritzen Channel sediments are remediated.

5 = Water concentration below or at detection limit, so not possible to calculate WSR concentrations. Value calculated using the Santa Fe WSR.

TABLE 10-11. Predicted ΣDDT and dieldrin sediment concentrations (μg/kg dry) required to achieve tissue residue end-points in organisms with primarily a sediment uptake route. Specific stations violating criterion indicated by "(Sta #)".

CRITERION (Σ DDT / DIELDRIN)	REDUCED SURVIVAL IN FISH FRY (3000-6250 μG DDT/KG) IN MACOMA[1,2]	REDUCED SURVIVAL IN FISH FRY (3000-6250 μG DDT/KG) IN "GOBIIDS"[1,3]	CHRONIC WQC RESIDUE (150/300 μG/KG) IN MACOMA[2]	CHRONIC WQC RESIDUE (150/300 μG/KG) IN "GOBIIDS"[3]	NAS RESIDUE (50/5 μG/KG) IN MACOMA[2]	NAS RESIDUE (50/5 μG/KG) IN "GOBIIDS"[3]	HUMAN HEALTH WQC RESIDUE (260/5.7 μG/KG) IN MACOMA[2]	HUMAN HEALTH WQC RESIDUE (32/0.7 μG/KG) IN "GOBIIDS"[3]
ΣDDD	6333 - 13194 (Sta# 1-3)	TROPHIC = 9460 - 19708 (Sta# 1-3) EqP = 4524 - 9425 (Sta# 1-3)	317 (Sta# 1-7, 13-16)	TROPHIC = 473 (Sta# 1-4, 6, 13-16) EqP = 226 (Sta# 1-7, 13-16)	106 (Sta# 1-7, 13-17)	TROPHIC = 158 (Sta# 1-7, 13-16) EqP = 75 (Sta# 1-8, 13-17)	549 (Sta# 1-4, 6, 13, 14, 16)	TROPHIC = 101 (Sta# 1-7, 13-17) EqP = 48 (Sta# 1-8, 13-17)
DIELDRIN	-	-	377 (Sta# 1-3)	EqP = 150 (Sta# 1-3)	6.3 (Sta# 1-4, 6-7, 13-16)	EqP = 2.5 (Sta# 1-7, 13-17, 23)	7.2 (Sta# 1-4, 6-7, 13-16)	EqP = 0.4 (Sta# 1-9, 13-17, 19-20, 23)

1 = Residue values reported in the AWQC document (U.S. EPA, 1980a) (see Table 5-4). No equivalent residue is available for dieldrin.

2 = Macoma residues calculated using the EqP bioaccumulation model with mean laboratory AFs corrected to steady-state values. AFs = 0.9 and 1.51 for ΣDDT and dieldrin, respectively (Table 8-18). Macoma lipid = 0.01 kg/kg wet weight and TOC = 0.019 kg/kg in all calculations.

3 = Residues in trophic level 3.25 fish, "gobiids", calculated using the exponential trophic model ("trophic") and the EqP model for ΣDDT and the EqP model only for dieldrin. EqP predictions based on gobiid AFs = 0.63 and 1.9 for ΣDDT and dieldrin, respectively (Table 8-32), and lipid = 0.02 kg/kg wet weight. Overall field AF (0.289, Table 8-20) used to calculate residues in benthic prey in the trophic model. TOC = 0.019 kg/kg in all calculations.

TABLE 10-12. Predicted ΣDDT or dieldrin sediment concentrations required to achieve benthic community structure and sediment toxicity end-points. Specific stations violating criterion indicated by "(Sta #)".

CRITERION	METHOD OF PREDICTING	ΣDDT OR DIELDRIN CONCENTRATION
10-DAY LC_{50} WITH *EOHAUSTORIUS ESTUARIUS*	10-DAY SEDIMENT TOXICITY TEST TABLE 9-7	2500 µG/G OC ΣDDT (47500 µG/KG @ 1.9% TOC) (Sta# 1, 2)
THRESHOLD OF TOXICITY WITH *E. ESTUARIUS*	10-DAY SEDIMENT TOXICITY TEST FIGURE 9-9	300 µG/G OC ΣDDT (5700 µG/KG @ 1.9% TOC) (Sta# 1-3)
MINIMUM ECOTOXICOLOGICAL EFFECTS CONCENTRATION ON AMPHIPODS	SEDIMENT TOXICITY AND COMPARATIVE TOXICOLOGY FIGURE 9-7	100 µG/G OC ΣDDT (1900 µG/KG @ 1.9% TOC) (Sta# 1-4, 6)
AMPHIPOD ABUNDANCE (ALL SPECIES) 80% OF 0 ΣDDT SEDIMENT CONCENTRATION[1] (# AMPHIPODS = 0.8 * 16.09 = 12.87)	REGRESSION OF ΣDDT SEDIMENT CONC. VS. TOTAL AMPHIPOD ABUNDANCE TABLE 9-2	4.75 µG/G OC ΣDDT (90 µG/KG @ 1.9% TOC) (Sta# 1-7, 13-17)
AMPHIPOD ABUNDANCE EXCLUDING *GRANDIDIERELLA JAPONICA* 80% OF 0 ΣDDT SEDIMENT CONCENTRATION[1] (# AMPHIPODS = 0.8 * 16.78 = 13.42)	REGRESSION OF ΣDDT SEDIMENT CONC. VS. AMPHIPOD ABUNDANCE EXCLUDING *G. JAPONICA* TABLE 9-2	3.96 µG/G OC ΣDDT (75 µG/KG @ 1.9% TOC (Sta# 1-8, 13-17)
CHANNEL MEAN INFAUNAL INDEX =60	REGRESSION OF INFAUNAL INDEX ON ΣDDT SEDIMENT CONC. TABLE 9-2	30,900 µG/G OC ΣDDT (587,000 µG/KG @ 1.9% TOC) (none)
NO STATIONS WITH INFAUNAL INDEX BELOW 60	RELATING PRESENT INFAUNAL INDEX VALUES TO ΣDDT SEDIMENT CONC. APPENDIX 9-1	2578 µG/G OC < ΣDDT SEDIMENT CONC. > 90 µG/G OC (Sta# 1-2 < SEDIMENT >Sta# 5, 7-9)
CHANNEL MEAN INFAUNAL INDEX =83	REGRESSION OF INFAUNAL INDEX ON ΣDDT SEDIMENT CONC. TABLE 9-2	9.7 µG/G OC ΣDDT (184 µG/KG @ 1.9% TOC) (Sta# 1-7, 13-16)
INFAUNAL INDEX 80% OF 0 POLLUTANT CONDITION[2] (II = 0.8 * 89.49 = 71.6)	REGRESSION OF INFAUNAL INDEX ON ΣDDT SEDIMENT CONC. TABLE 9-2	533 µG/G OC ΣDDT (10,126 µG/KG @ 1.9% TOC) (Sta# 1-3)
INTERIM SEDIMENT QUALITY CRITERION FOR DIELDRIN	PREDICTING BENTHIC TOXICITY FROM WQC ASSUMING EQUILIBRIUM PARTITIONING TABLE 5-12	17 µG/G OC DIELDRIN (323 µG/KG @ 1.9% TOC) (Sta# 1-3)

1 = Based on 80% amphipod mortality in laboratory tests used in evaluating dredge materials (U.S. EPA/ACE, 1991).

2 = The criterion of 80% of the no pollutant Infaunal Index is suggested here as a comparative approach to quantifying deviations from the local non-impacted benthic community. This approach is independent of the criteria given in Bascom *et al.* (1978).

TABLE 10-13. Summary of the presence ("X) of violations for ΣDDT end-points in Tables 10-1 to 10-3.

CRITERION	LAURITZEN	SANTA FE	RICHMOND
WATER:			
Acute WQC - Overlying Water	-	-	-
Chronic WQC - Overlying Water	X	X	~X
Acute WQC - Total IW	X	X	-
Chronic - Total IW	X	X	X
Chronic Toxicity to Inverts. - Overlying water	~X	-	-
Chronic Toxicity to Inverts. - Total IW	X	X	~X
CA WQ Objective - Overlying	X	X	~X
CA WA Objective - Total IW	X	X	X
TISSUE RESIDUES:			
Basis Chronic WQC - Macoma	X	X	-
Basis Chronic WQC - Mussels	X	X	-
Basis Chronic WQC - Composite Fish	X	X	-
Reduced Survival in Fish Fry - Macoma	X	-	-
Reduced Survival in Fish Fry - Mussels	-	-	-
Reduced Survival in Fish Fry - Composite Fish	X	-	-
NAS Residue - Macoma	X	X	X
NAS Residue - Mussels	X	X	-
NAS Residue - Composite Fish	X	X	X
SEDIMENT/BENTHIC COMMUNITY:			
Sed. Conc. > LC_{50} W/Eohaustorius	-	-	-
Sed. Conc. > Threshold toxicity w/Eohaustorius	X	-	-
Sed. Conc. > Min. Ecological Effects Conc.	X	-	-
Mean Infaunal Index > 83	X	X	-
Any Infaunal Index < 60	X	-	-
Greater than 20% Eohaustorius mortality	X	X	X
Greater than 20% reduction amphipods	X	X	X
SUMMATION:	21	15	9

~X = Value equal to criterion or range brackets criterion.

FIGURE 10-1. Summation of the number of violations of ΣDDT endpoints in tables 10-1 to 10-3 by channel. Human health criteria not included.

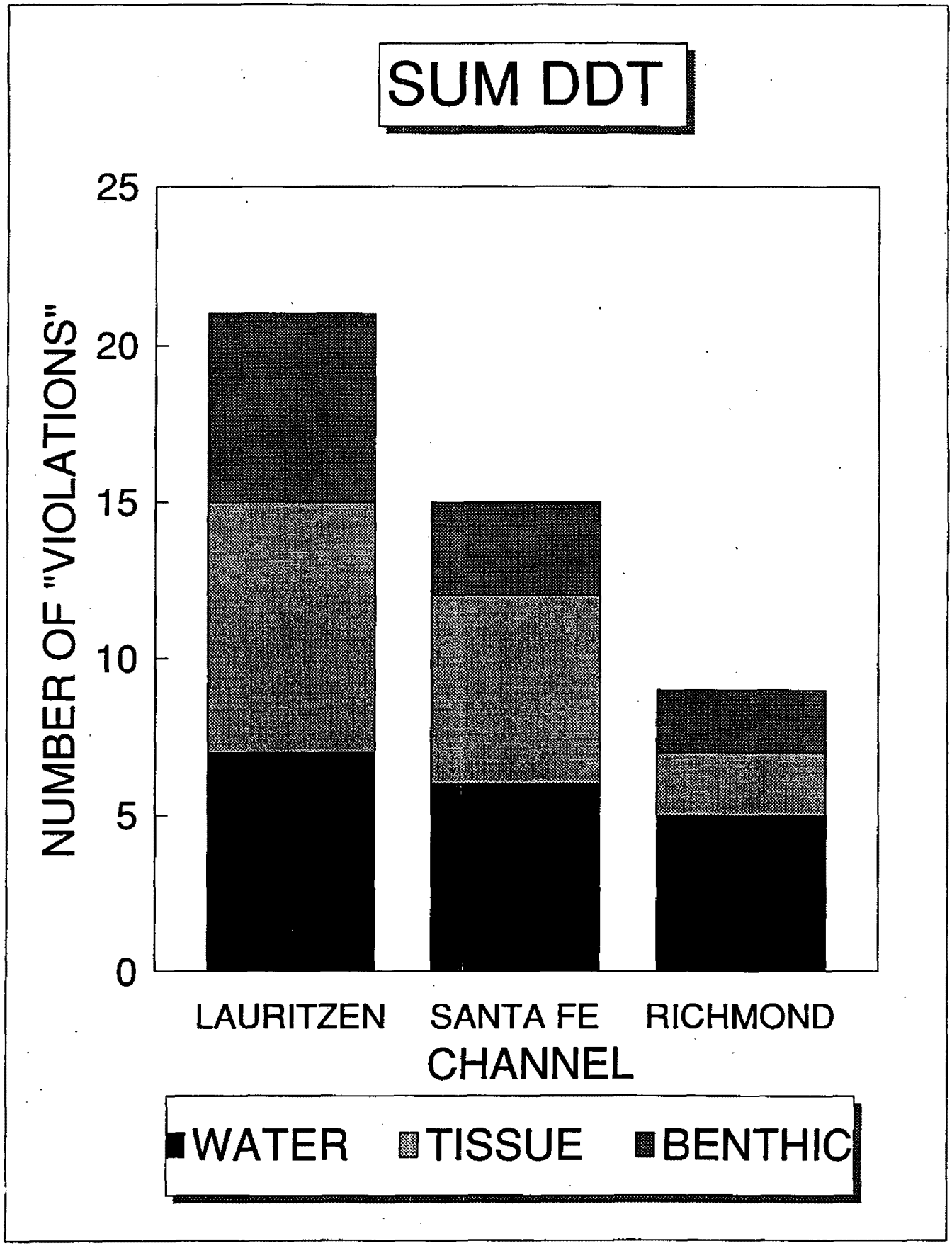
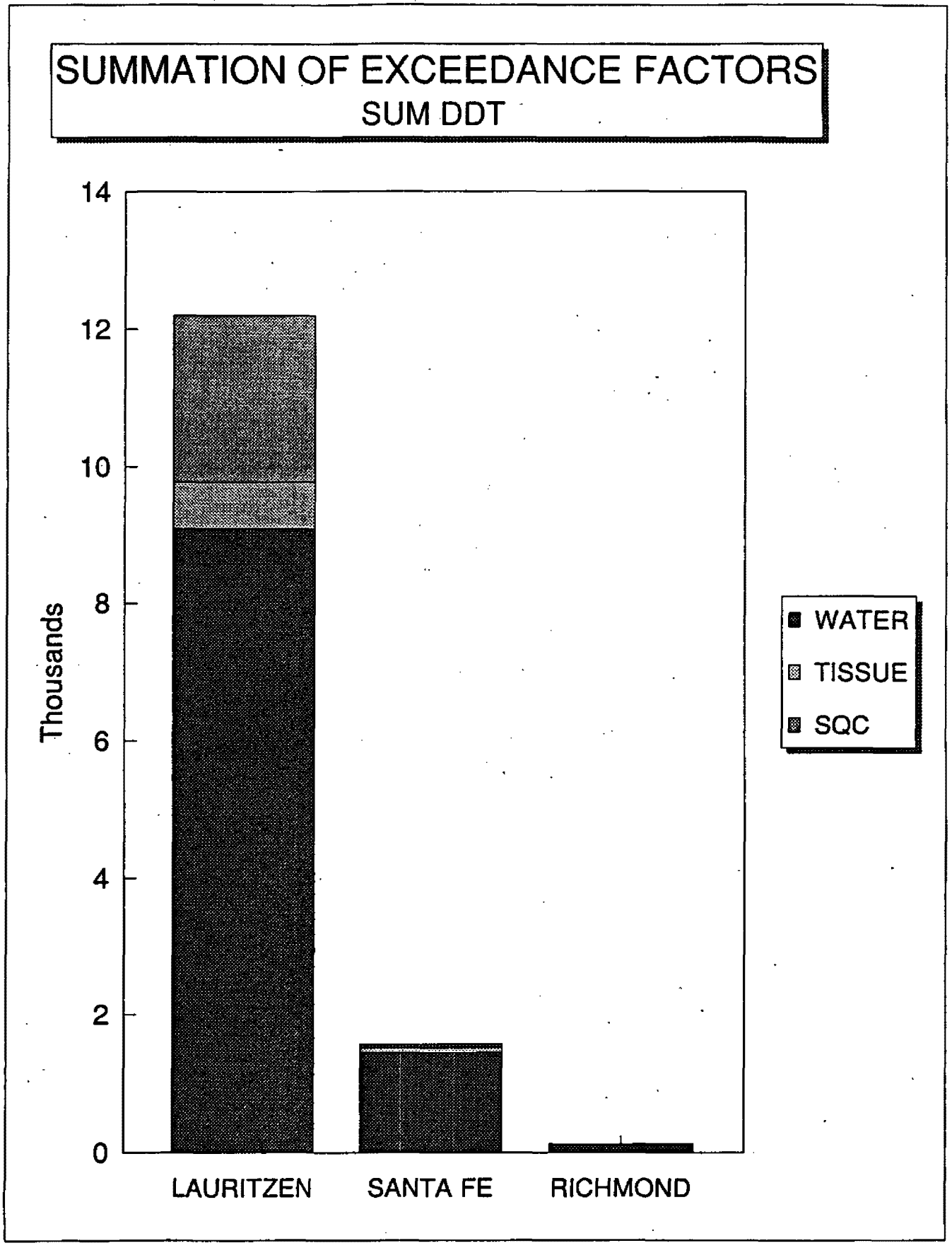

SUM DDT
NUMBER OF "VIOLATIONS"
25
20
15
10
5
0
LAURITZEN
SANTA FE
RICHMOND
CHANNEL
WATER
TISSUE
BENTHIC

FIGURE 10-2. Summation of the "exceedance factors" for ΣDDT endpoints in tables 10-1 to 10-6 by channel. Human health criteria not included.


SUMMATION OF EXCEEDANCE FACTORS
SUM DDT
14
12
10
8
6
4
2
0
Thousands
LAURITZEN
SANTA FE
RICHMOND
WATER
TISSUE
SQC

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APPENDIX 4-1. Andrew Lincoff memo of 12/8/93, photographs of birds at the United Heckathorn superfund site.

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION IX
75 Hawthorne Street
San Francisco, Ca. 94105-3901

December 8, 1993

MEMORANDUM

Subject: Photographs of birds at the United Heckathorn Superfund Site.

From: A. Lincoff
Remedial Project Manager

To: Site file

On the afternoon of November 30, 1993 a brief field visit was made to the Richmond Inner Harbor to obtain photographic documentation of presence of birds in the channels investigated in the ecological assessment and marine remedial investigation. The photographs were taken by myself from the Region 9 Research Vessel Johnson which was piloted by Peter Husby and crewed by Ken Hendrix of the Environmental Support Branch. The photographs were taken with a Ricoh single lens reflex camera with a 135 mm zoom lens. All photographs were taken at 135 mm. The film used was Kodak Gold 100. The locations are abbreviated as: Richmond Inner Harbor Channel, RIHC; Parr Canal, Parr; Lauritzen Channel, LC; and Santa Fe Channel, SFC.

Photo No.	Species/Type	Location	Time	Comments
1	loon	RIHC	1:04	ate a fish
2	hawk	RIHC	1:05	
3	gull	RIHC	1:06	one of two
-- no photo, tern? seen feeding in RIHC --				
4	grebe	RIHC	1:08	diving
5	grebe	RIHC	1:11	
6	grebe and loon	RIHC	1:11	
-- no photo, another grebe seen RIHC --				
7	grebe	RIHC	1:14	
8	coots	RIHC	1:16	two of ~ten
9	great blue heron	Parr	1:18	
10	tern?	Parr	1:19	
11	sandpiper?	Parr	1:20	at shoreline
-- no photo, several coots --				
12	grebe	Parr	1:24	diving
-- no photo, cormorant and coots seen in SFC --				
13	Cal. brown pelican	SFC	1:30	
14	cormorant	LC	1:32	took off from water

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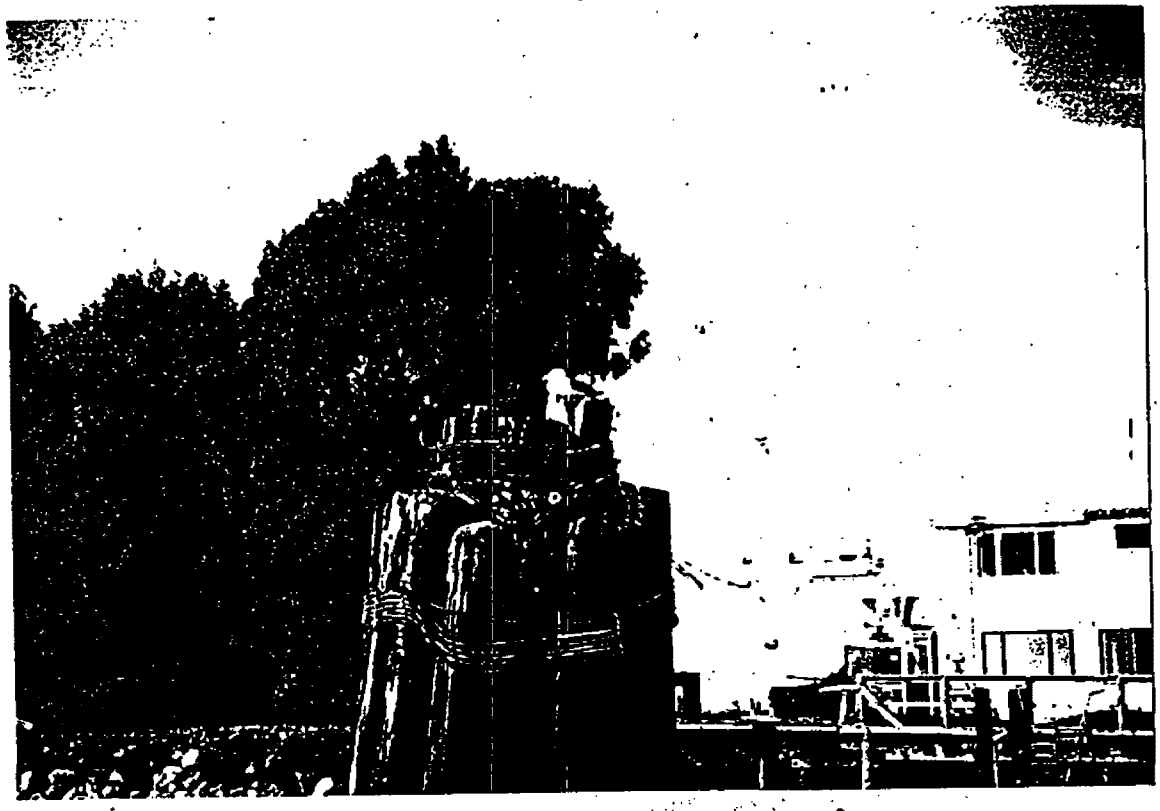
	-- no photo, grebe and long-billed brown shorebird seen in LC --			
15	grebe	LC	1:43	
16	loon	SFC	1:47	
17	loon	SFC	1:47	same bird as 17
18	wandering tattlers	SFC	1:49	sandpipers seen
19	wandering tattlers	SFC	1:49	same birds as 19
	-- no photo, coots seen --			
20	wandering tattler	SFC	1:53	
21	C.B. pelican and gulls	SFC	1:56	taking off
22	Cal. brown pelican	SFC	1:56	same bird as 22
	-- no photo, grebes seen in SFC --			
23	double-crested cormorant	SFC	2:00	
24	cormorants	SFC	2:04	
25	Cal. brown pelican	SFC	2:09	took off from water
26	loon	LC	2:14	grebe also seen
	-- no photo, grebe seen in SFC, paused to adjust engines --			
27	cormorant	SFC	2:27	
28	cormorant	SFC	2:27	same bird as 28
29	coots and grebes	Parr	2:33	
30	coots	Parr	2:33	
31	Cal. brown pelican	RIHC	2:42	

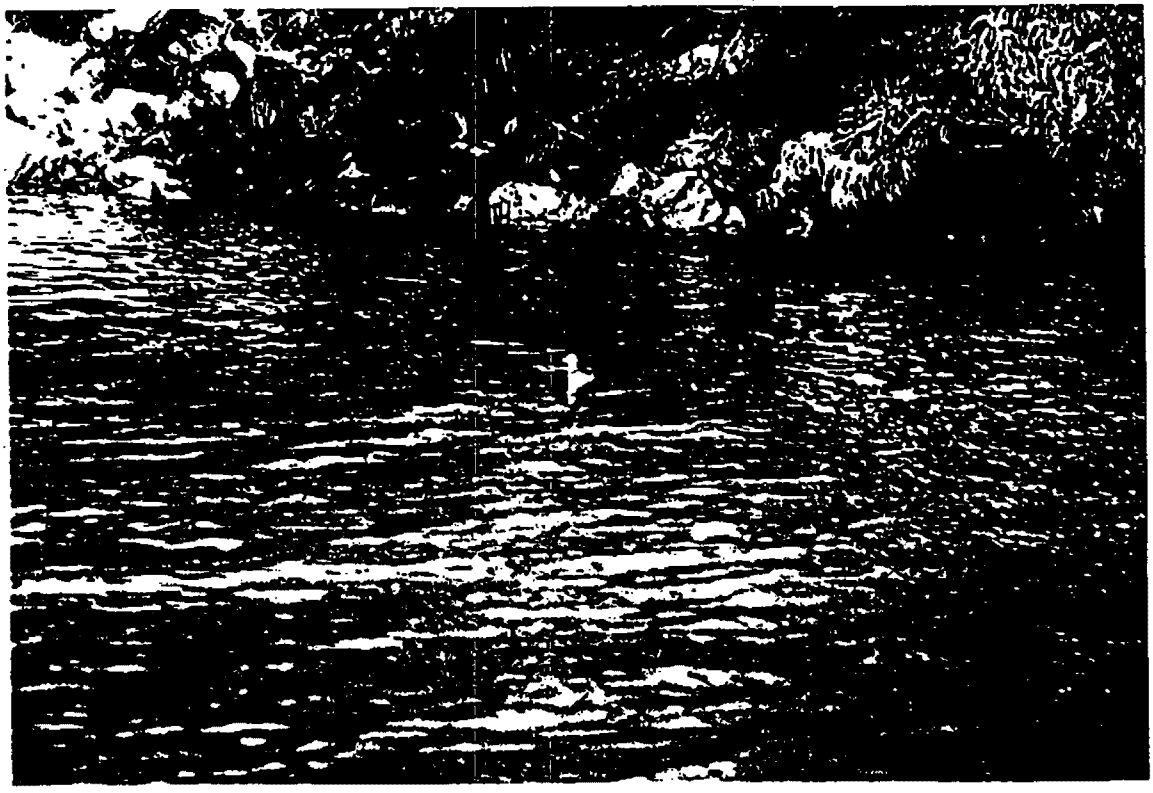
General observations: Coots and gulls were the most numerous birds in the Inner Harbor. Grebes were the most numerous bird which feeds primarily on fish (using the descriptions in Table 4-1 of the 1993 draft Ecological Assessment).

1. Loon, Richmond Inner Harbor Channel. 1:04 p.m., 11/30/93.

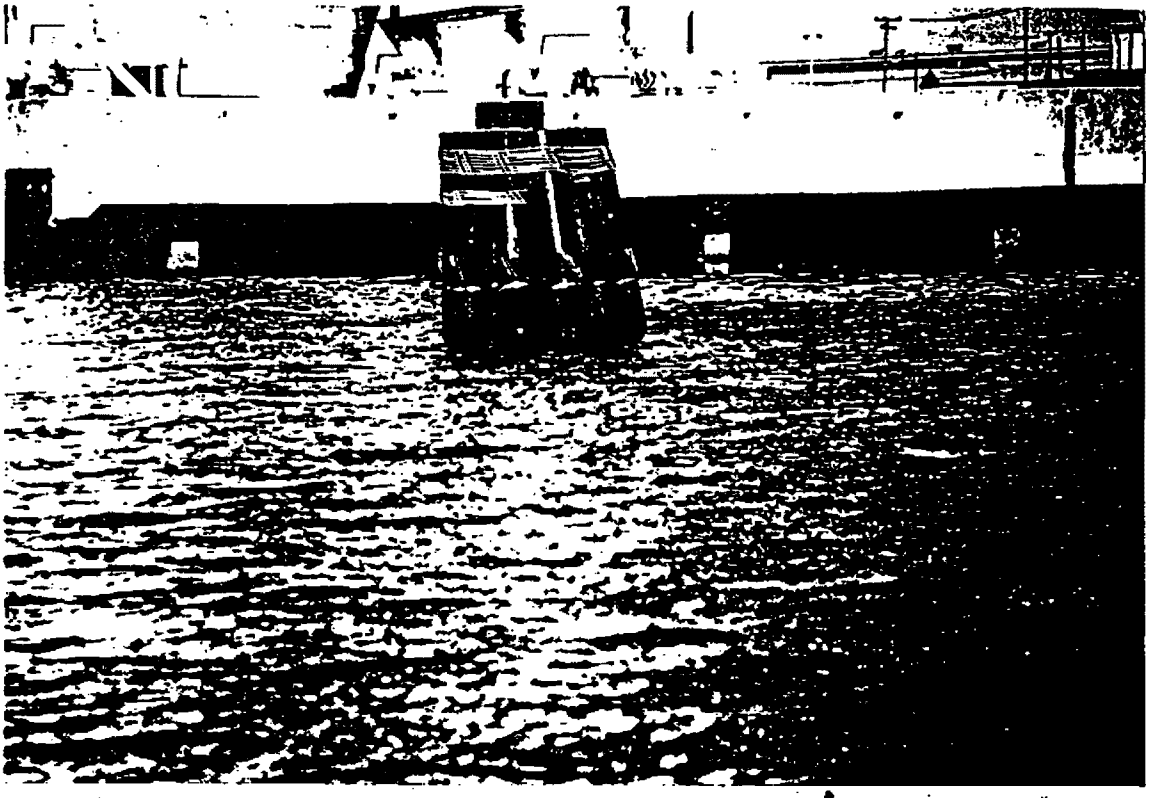
2. Hawk, Richmond Inner Harbor Channel. 1:05 p.m., 11/30/93.



3. Gull, Richmond Inner Harbor Channel. 1:06 p.m., 11/30/93.

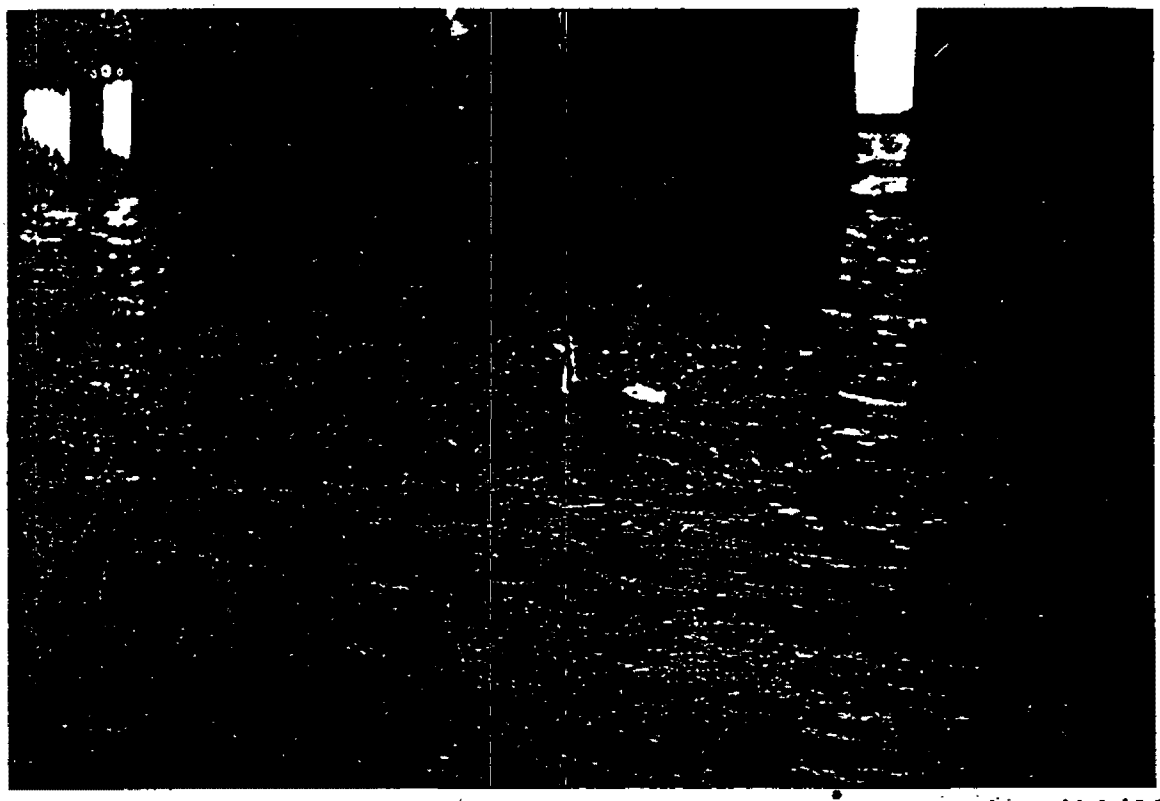


4. Grebe, Richmond Inner Harbor Channel. 1:08 p.m., 11/30/93.

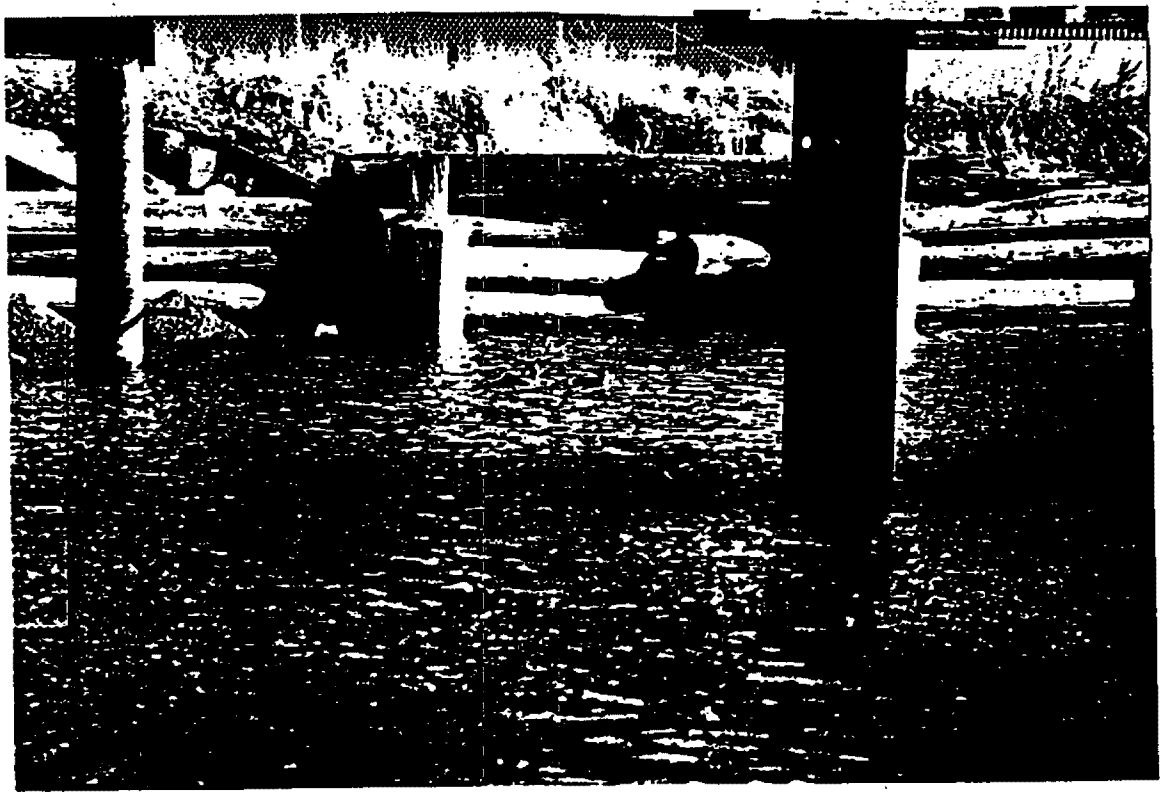


5. Grebe, Richmond Inner Harbor Channel. 1:11 p.m., 11/30/93.

6. Grebe and loon, Richmond Inner Harbor Channel. 1:11 p.m., 11/30/93.

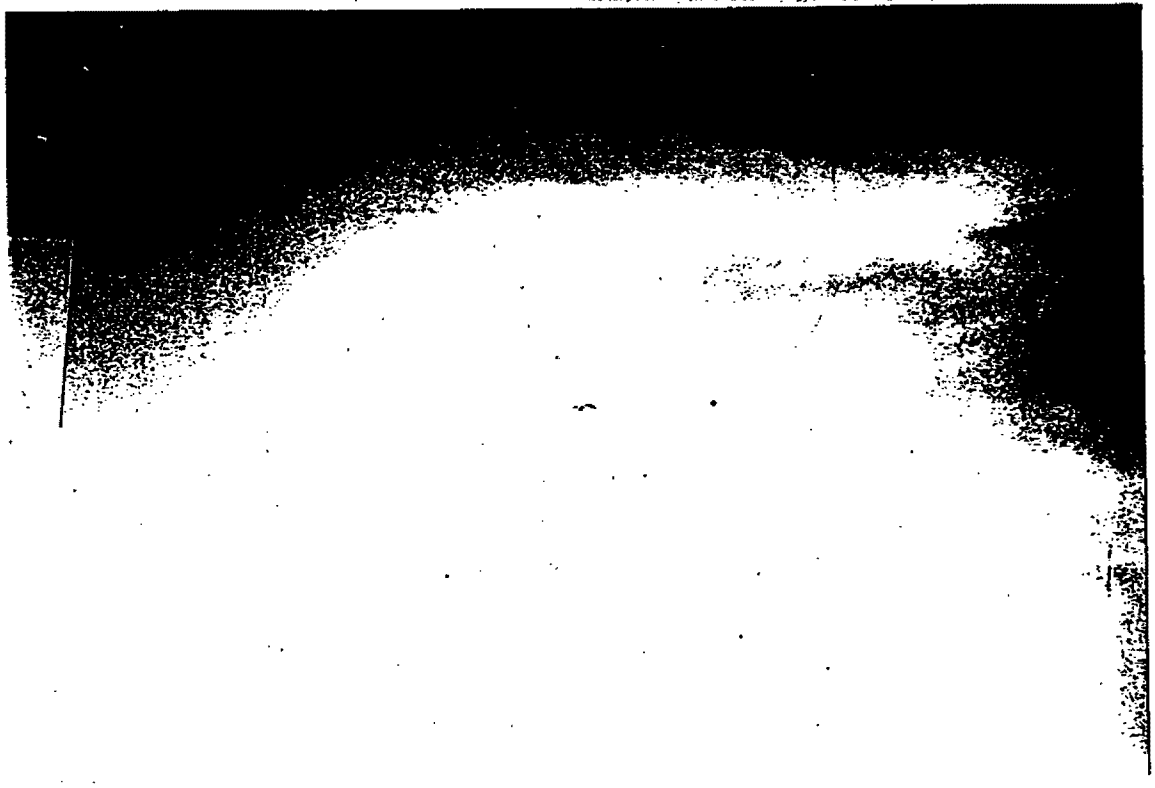


7. Grebe, Richmond Inner Harbor Channel. 1:14 p.m., 11/30/93.



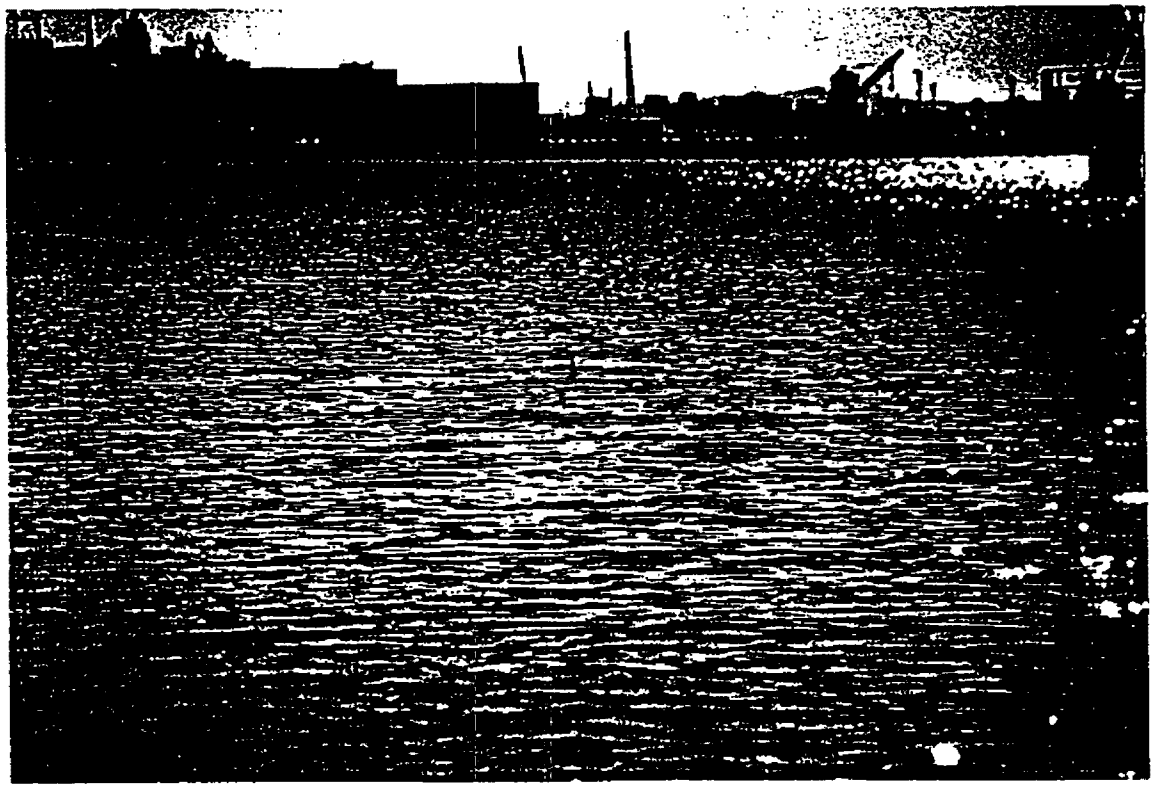
8. Coots, Richmond Inner Harbor Channel. 1:16 p.m., 11/30/93.

9. Great blue heron, Parr Canal. 1:18 p.m., 11/30/93.

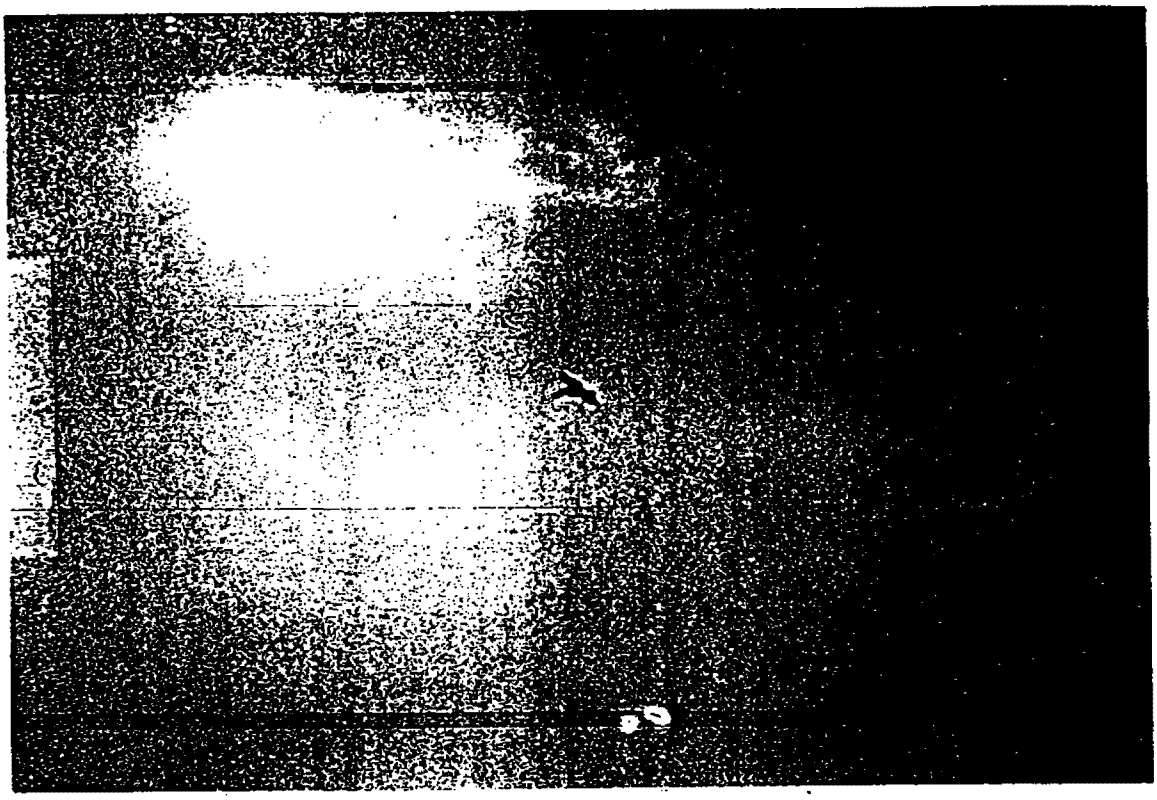


10. Tern?, Parr Canal. 1:19 p.m., 11/30/93.

11. Sandpiper?, Parr Canal. 1:20 p.m., 11/30/93.

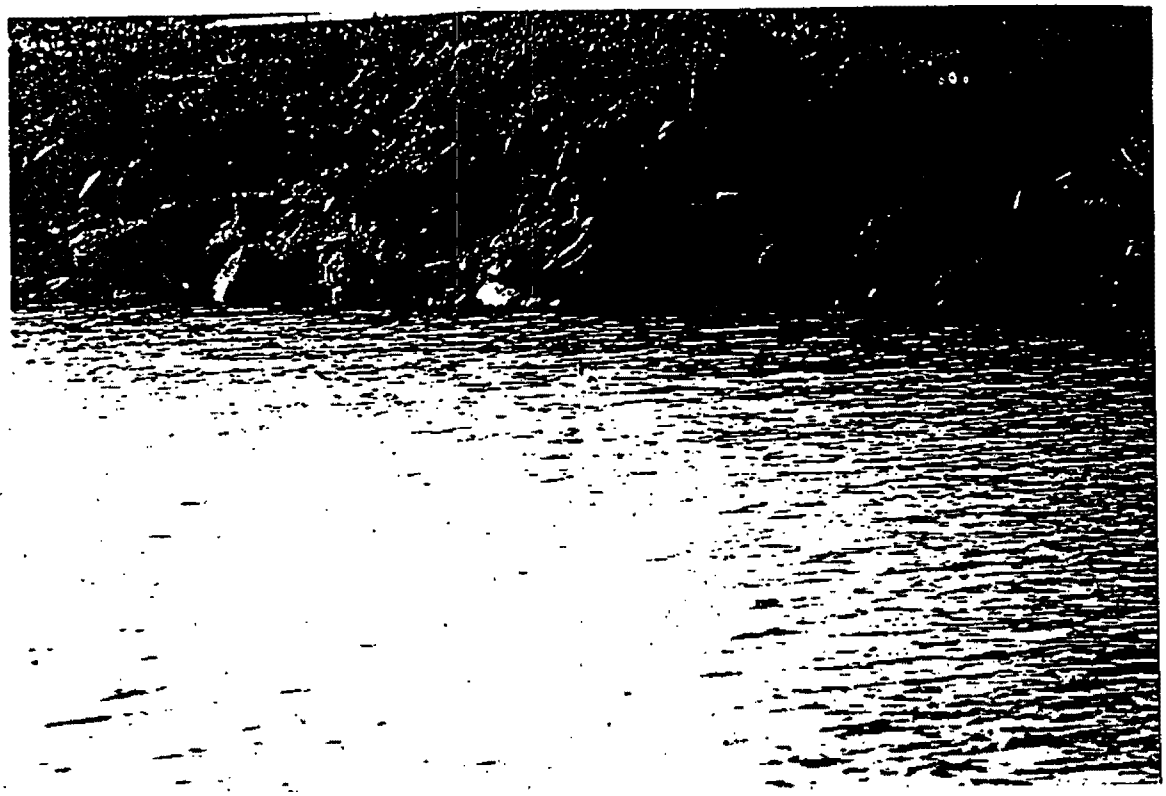


12. Grebe, Parr Canal. 1:24 p.m., 11/30/93.

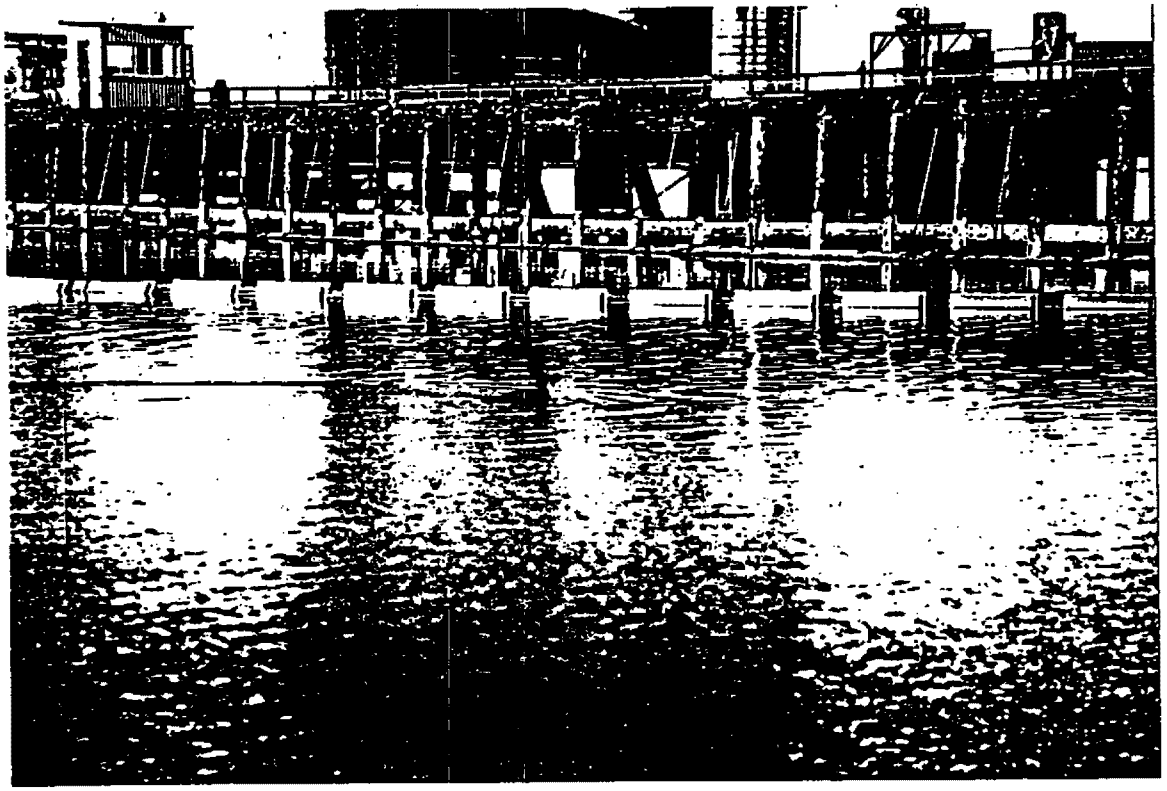


13. California brown pelican, Santa Fe Channel. 1:30 p.m., 11/30/93.

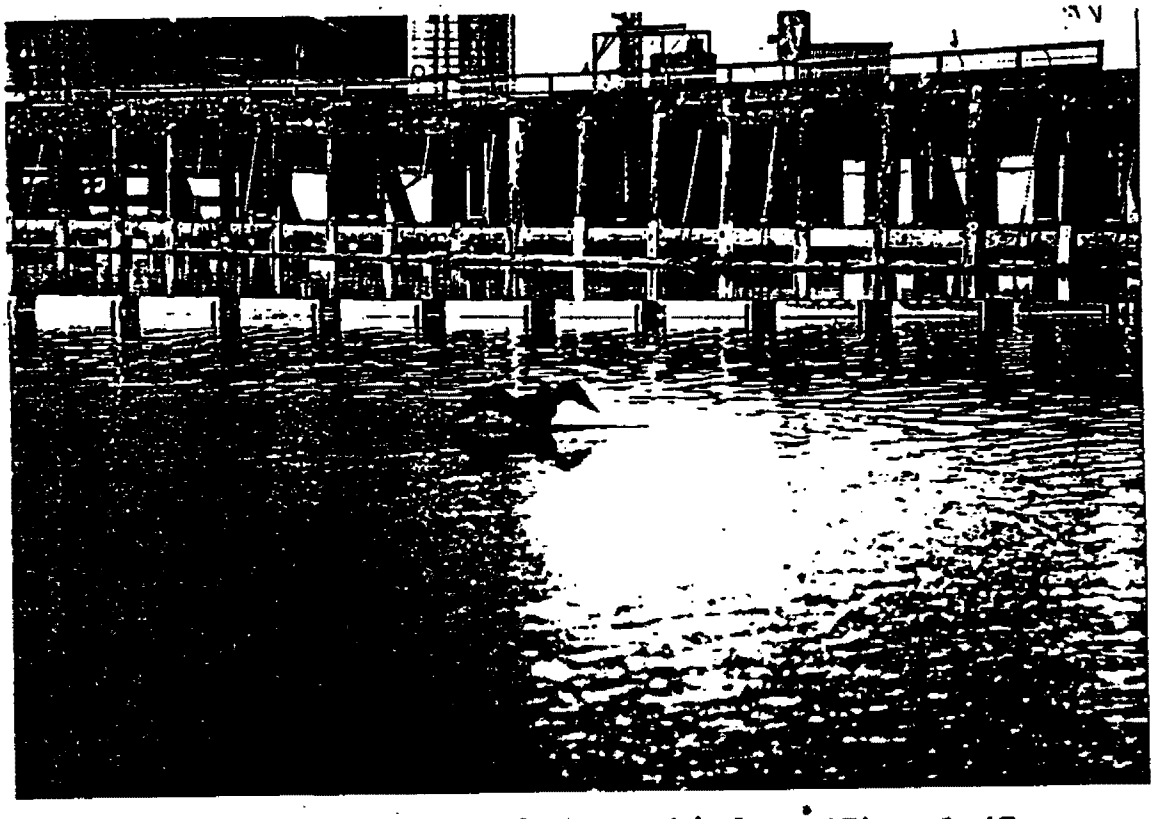
14. Cormorant, Lauritzen Channel. 11/30/93.



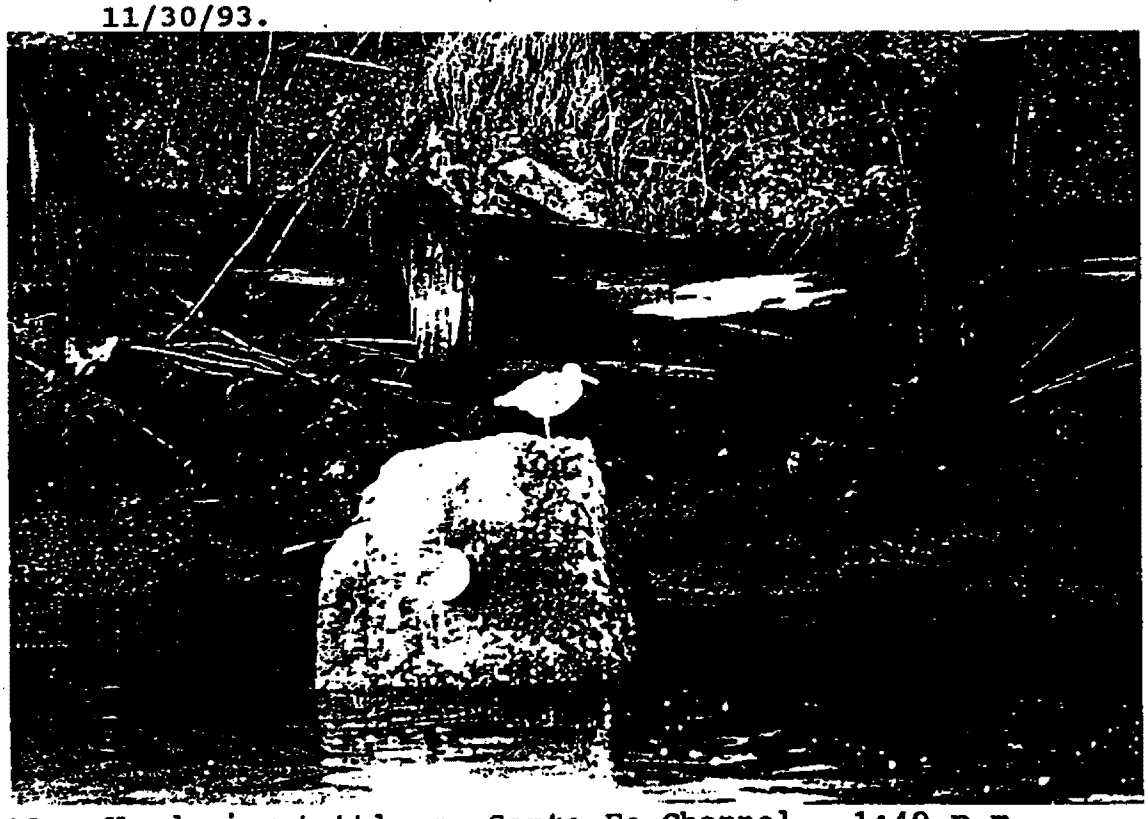
15. Grebe, Lauritzen Channel. 1:43 p.m., 11/30/93.



16. Loon, Santa Fe Channel. 1:47 p.m., 11/30/93.

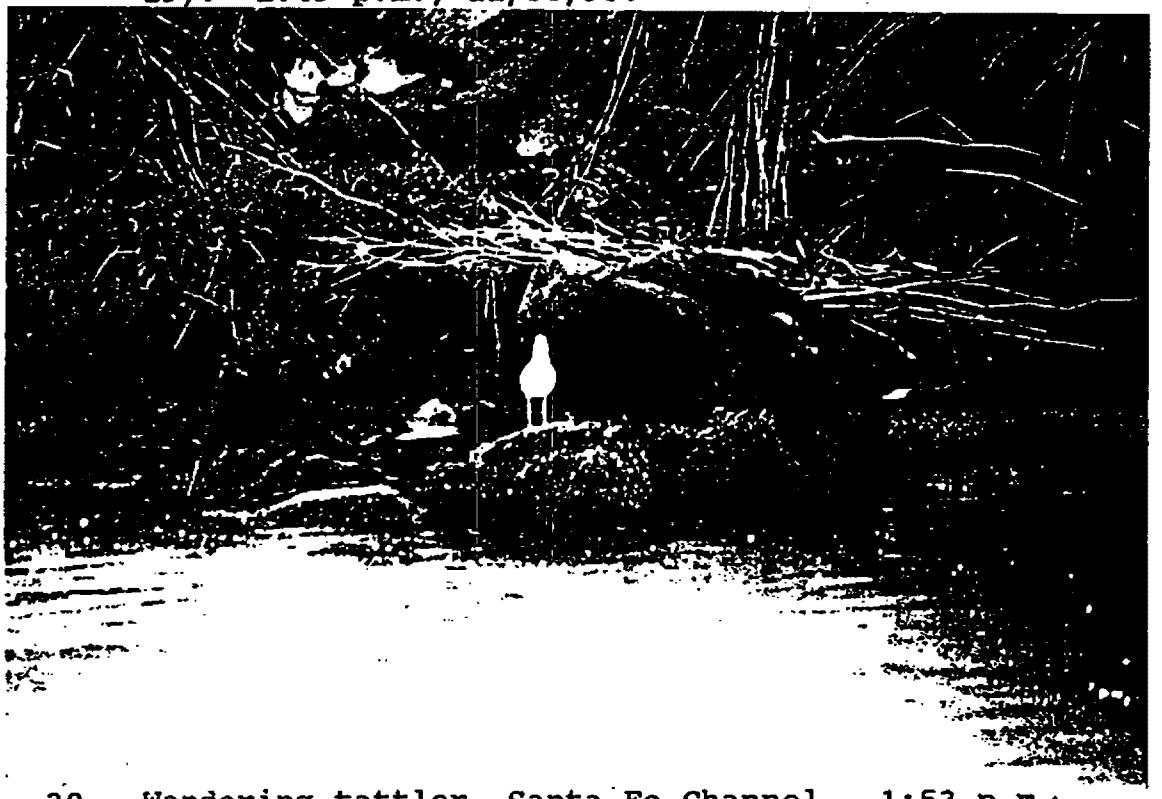


17. Loon, Santa Fe Channel (same bird as 17). 1:47 p.m., 11/30/93.

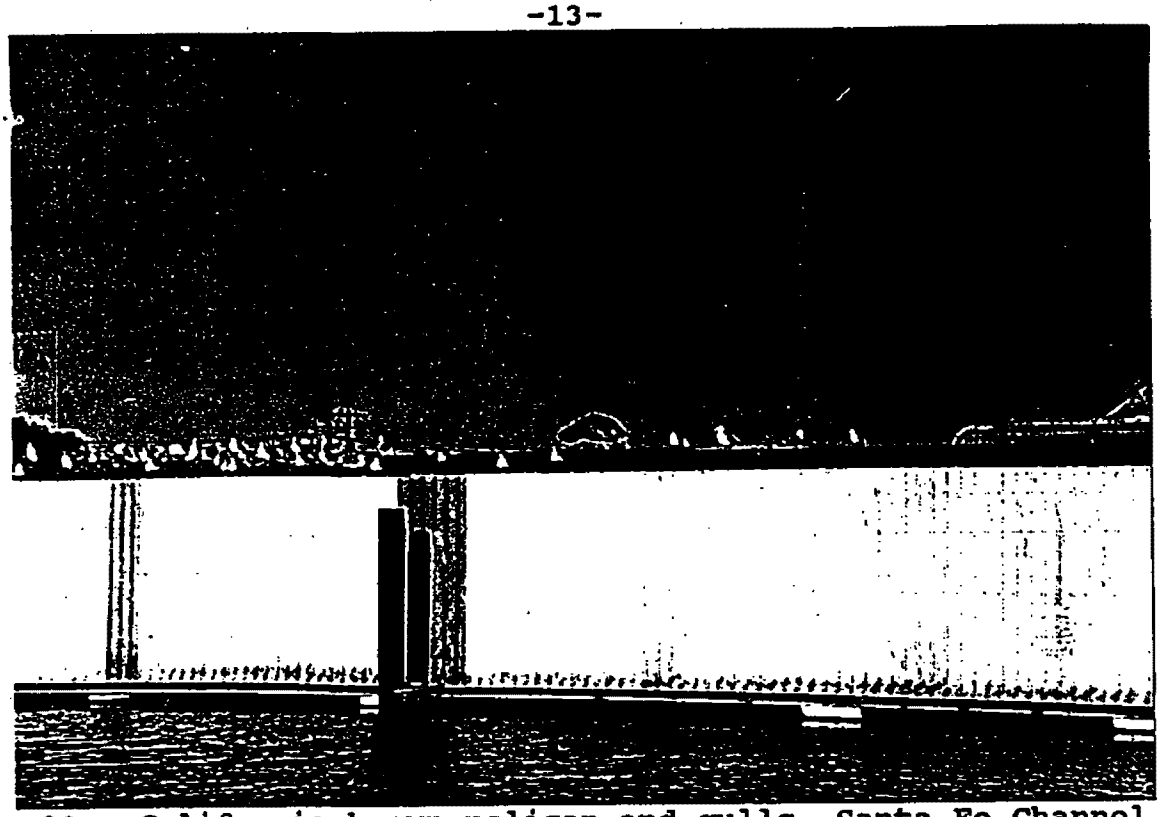


18. Wandering tattlers, Santa Fe Channel. 1:49 p.m., 11/30/93.

19. Wandering tattlers, Santa Fe Channel (same birds as 19). 1:49 p.m., 11/30/93.

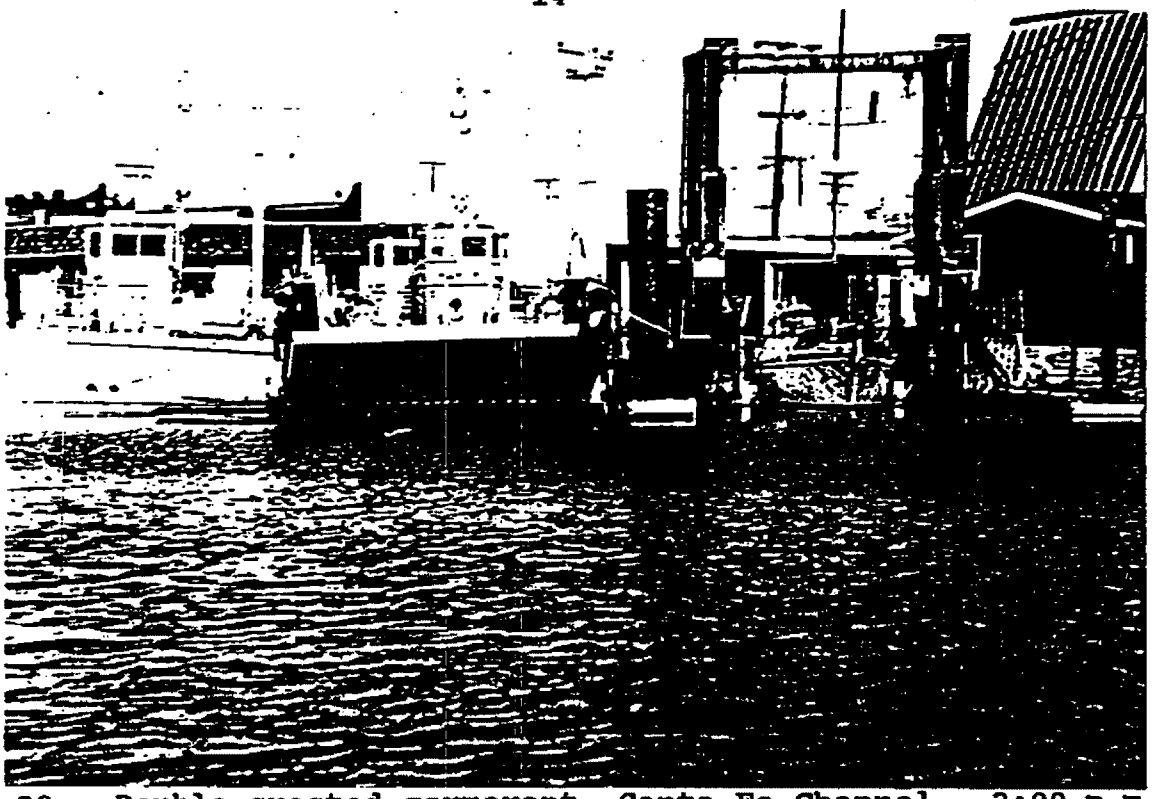


20. Wandering tattler, Santa Fe Channel. 1:53 p.m., 11/30/93.



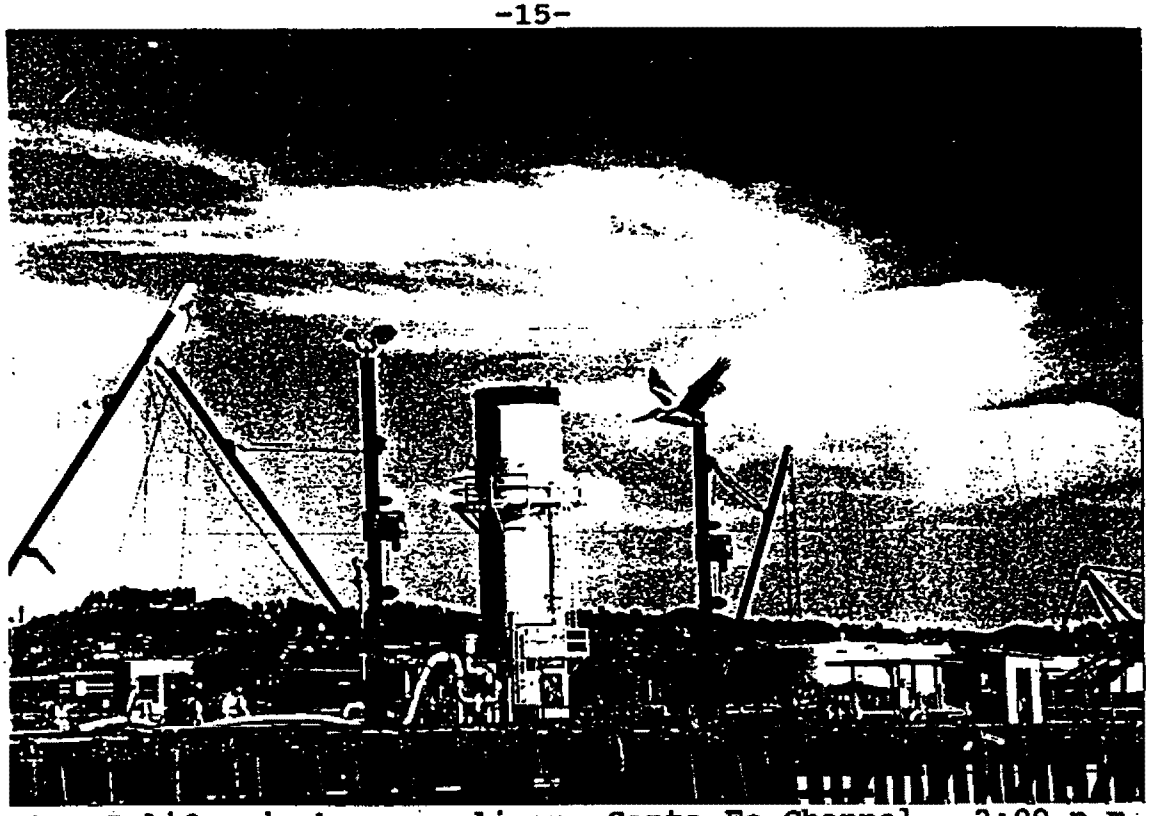
21. California brown pelican and gulls, Santa Fe Channel. 1:56 p.m., 11/30/93.

22. California brown pelican, Santa Fe Channel (same bird as 22). 1:56 p.m., 11/30/93.



23. Double-crested cormorant, Santa Fe Channel. 2:00 p.m., 11/30/93.

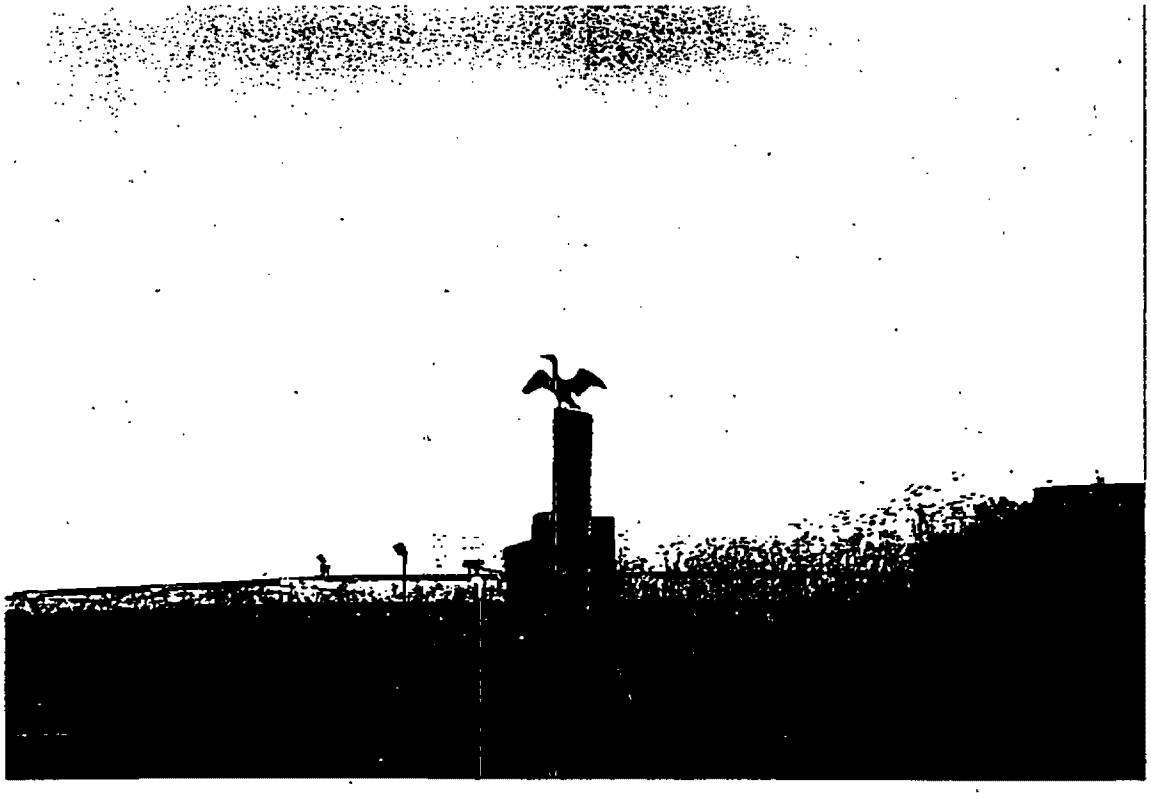
24. Cormorants, Santa Fe Channel. 2:04 p.m., 11/30/93.



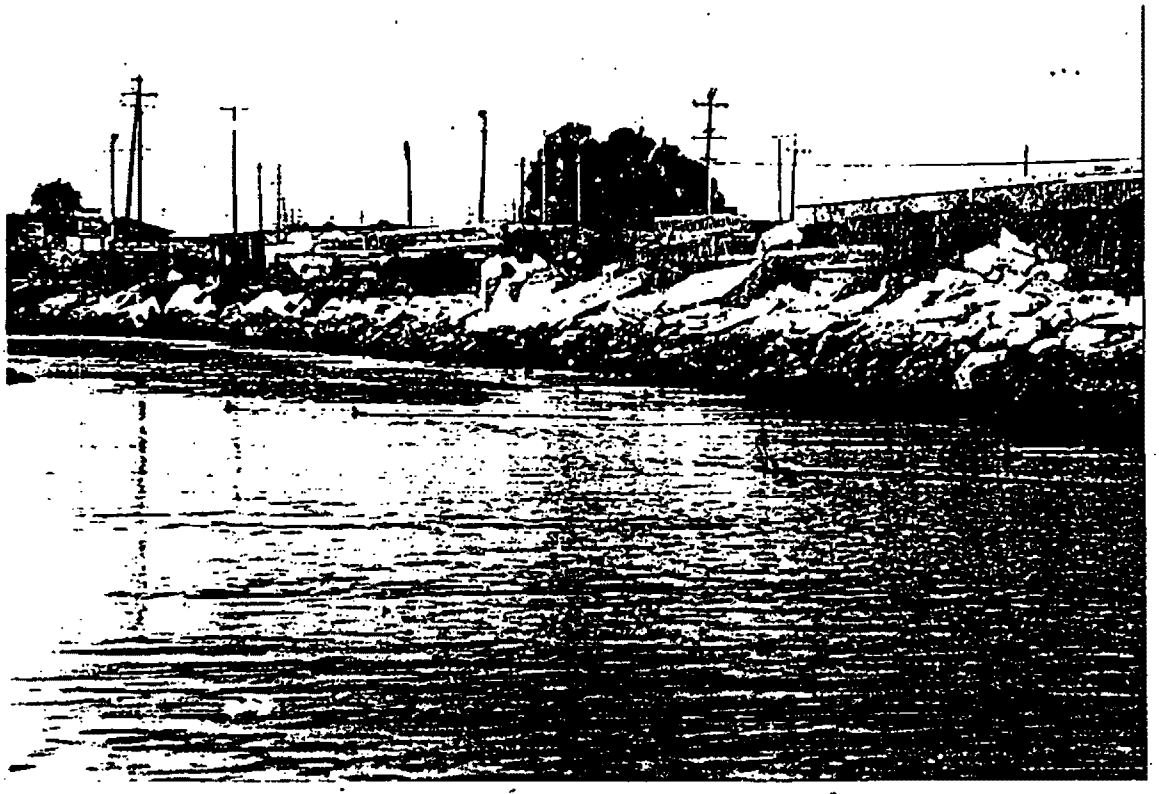
25. California brown pelican, Santa Fe Channel. 2:09 p.m., 11/30/93.

26. Loon, Lauritzen Channel. 2:14 p.m., 11/30/93.

27. Cormorant, Santa Fe Channel. 2:27 p.m., 11/30/93.

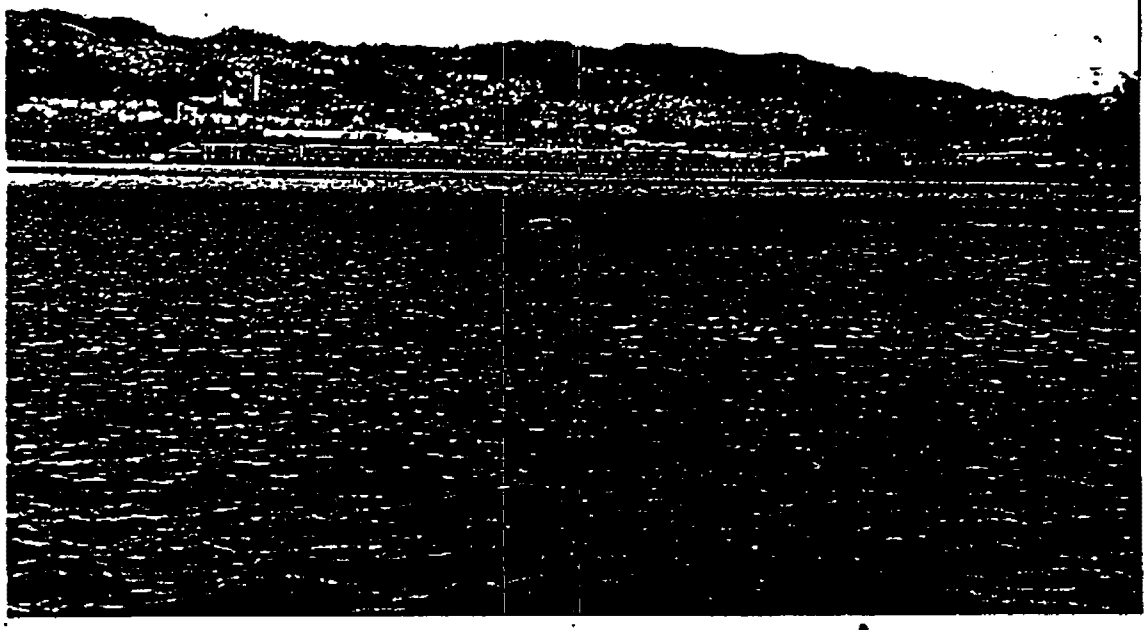


28. Cormorant, Santa Fe Channel. 2:27 p.m., 11/30/93.



29. Coots and grebes, Parr Canal. 2:33 p.m., 11/30/93.

30. Coots, Parr Canal. 2:33 p.m., 11/30/93.



31. California brown pelican, Richmond Inner Harbor Channel. 2:42 p.m., 11/30/93.

APPENDIX 4-1 (Cont'd). Andrew Lincoff memo of 12/16/93, Baykeeper tour/ photographs of birds at the United Heckathorn superfund site.

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION IX
75 Hawthorne Street
San Francisco, Ca. 94105-3901

December 16, 1993

MEMORANDUM

Subject: Baykeeper Tour/ Photographs of birds at the United Heckathorn Superfund Site.

From: A. Lincoff
Remedial Project Manager

To: Site file

On the morning of December 13, 1993 Denise Klimas, the Region's NOAA Coastal Resources Coordinator, arranged a field visit of the Inner Richmond Harbor and United Heckathorn Site aboard the Baykeeper to familiarize staff of other resource agencies with the site. In addition to myself, Ms. Klimas and Mike Herz, the pilot of the Baykeeper, were Michael Martin of the CA Dept. of Fish and Game, John Lindsay of NOAA, Jim Haas of the U.S. Fish and Wildlife Service, and Jim Bybee of the National Marine Fisheries Service. During the visit I discussed the site contamination and Remedial Investigation and Feasibility Study and took the following photographs. The photos were taken with a Ricoh single lens reflex camera with a 135 mm zoom lens. The film used was Kodak Gold 100. The locations are abbreviated as: Richmond Inner Harbor Channel, RIHC; Parr Canal, Parr; Lauritzen Channel, LC; and Santa Fe Channel, SFC. The first 28 photographs were taken between 9:55 and 10:10 a.m. The last 6 were taken between 10:10 and 10:30 a.m. Where not specifically identified, gulls are most likely western gulls, cormorants are most likely double-crested cormorants, and grebes are most likely western grebes.

Photo No.	Species/Type	Location	Time	Comments
1	Cal. brown pelicans	SFC	9:55	
2	Cal. brown pelican	SFC		
3	Cal. brown pelican	LC		
4	Cal. brown pelican	LC		same bird as 3
5	cormorant and gull	LC		double-crested and western
6	double-crested cormorant	LC		
7	coots	LC		
8	western gulls	LC		
9	Cal. brown pelican	LC		
10	Cal. brown pelican	LC		same bird as 9

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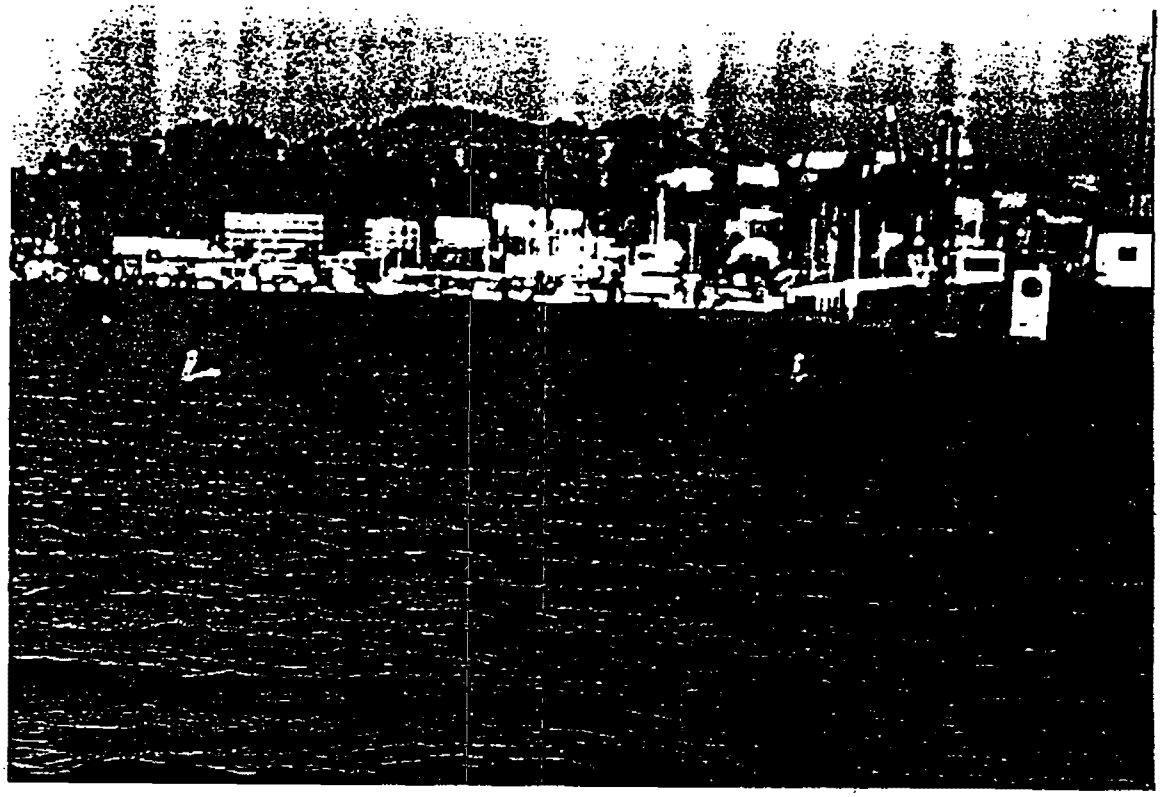
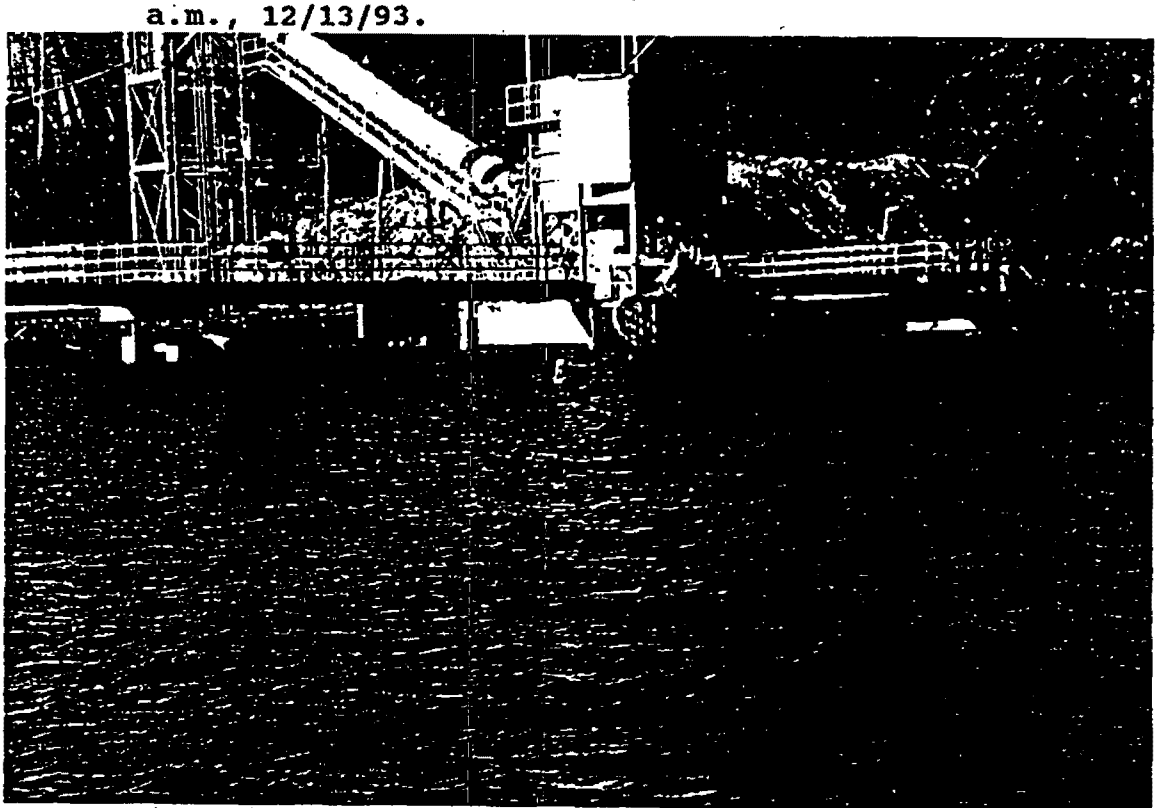
11	Cal. brown pelican	LC		same bird as 9
12	gull	LC		
13	Cal. brown pelican	LC		
14	Cal. brown pelican	LC		
15	Cal. brown pelican	LC		same bird as 14
16	coots	LC		
17	cormorants	LC		pelican, SFC
18	Cal. brown pelicans	SFC		
19	pelican, gull and cormorant	SFC	SFC	pelican feeding
20	Cal. brown pelicans	SFC		
21	Cal. brown pelicans	SFC		feeding
22	Cal. brown pelicans	SFC	feeding	(most same as 21)
23	double-crested cormorant	SFC		
24	western grebe	SFC		
25	Cal. brown pelican	SFC		
26	Cal. brown pelican	SFC		same bird as 26
27	cormorant	SFC		
28	grebes	SFC	10:10	
29	kingfisher	SFC		on pole
30	Cal. brown pelicans	LC		
31	cormorant	SFC		
32	human fishermen	Parr/RIHC		
33	sandpiper	Parr		
34	grebes	SFC/RIHC	10:30	group of ~14

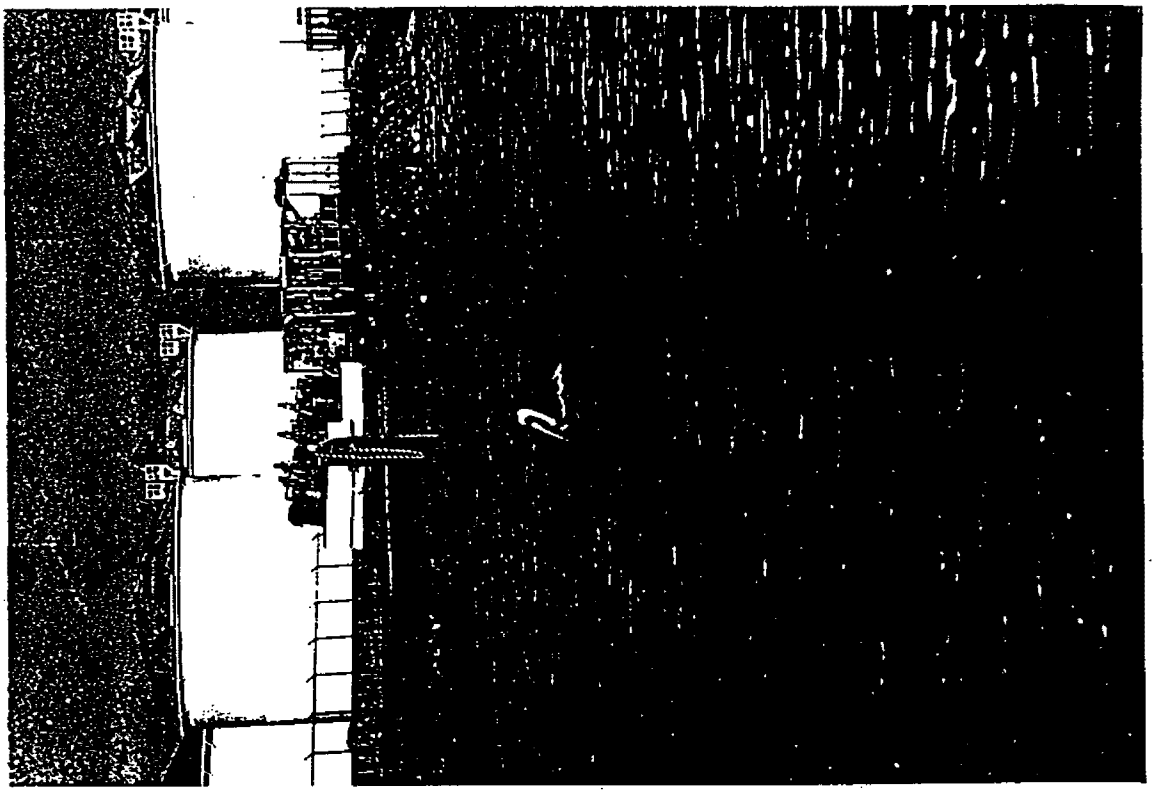
General observations: Numerous pelicans, grebes, cormorants, gulls and coots were seen in the Richmond Inner Harbor, particularly the Lauritzen and Santa Fe Channels. Loons were also seen. The group of pelicans seen feeding in the Santa Fe Channel (photos 19-22) was at the confluence with the Lauritzen and numbered 6 or 7.



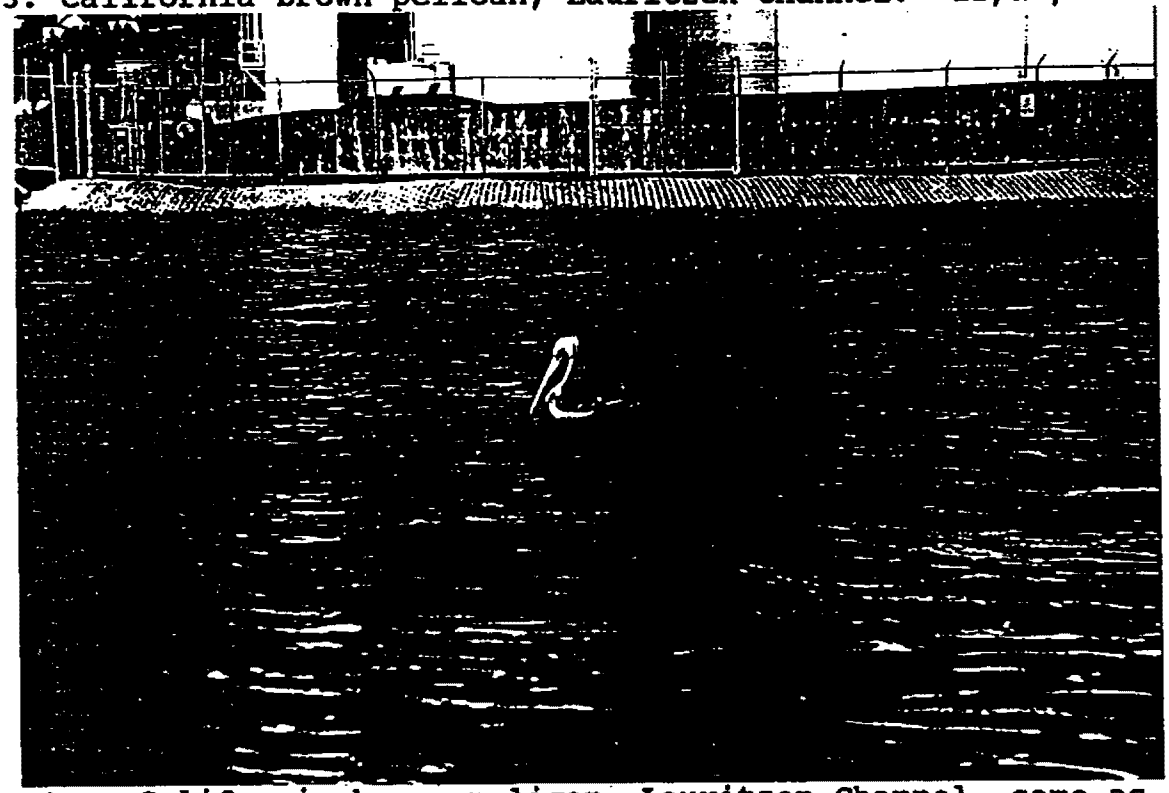
1. California brown pelicans, Santa Fe Channel. 9:55 a.m., 12/13/93.



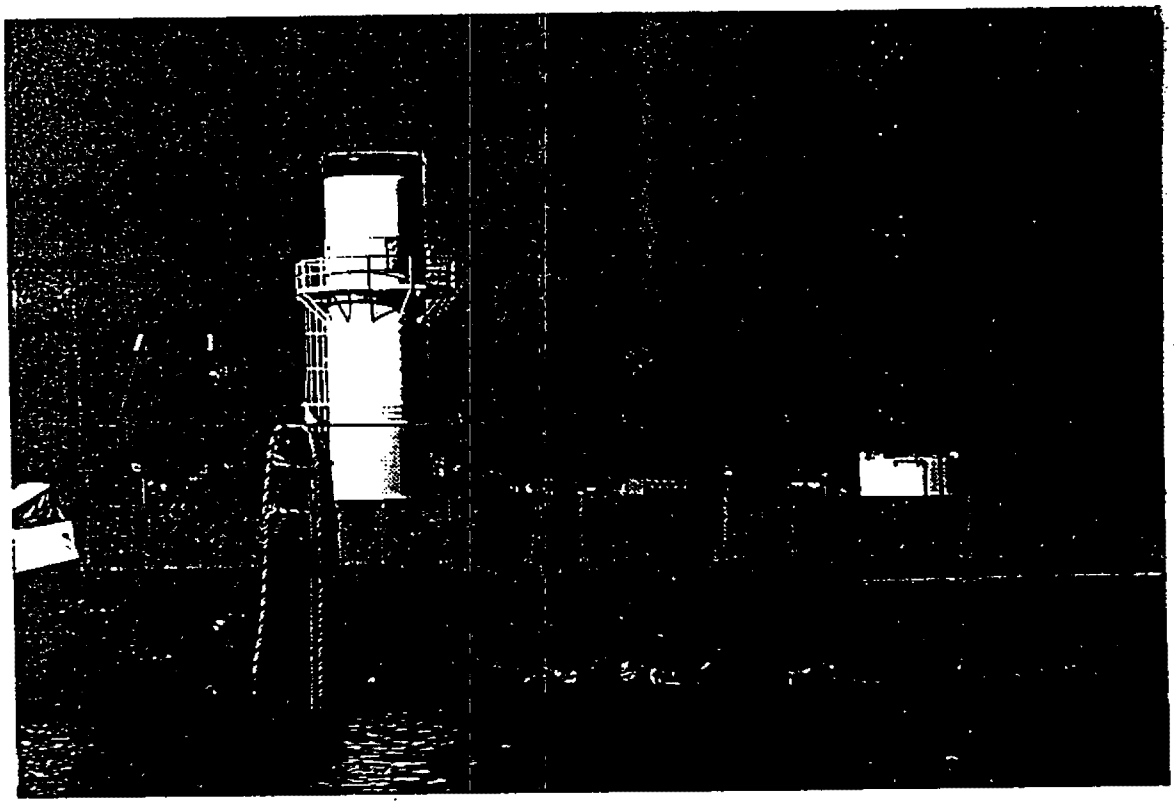
2. California brown pelican, Santa Fe Channel. 12/13/93.



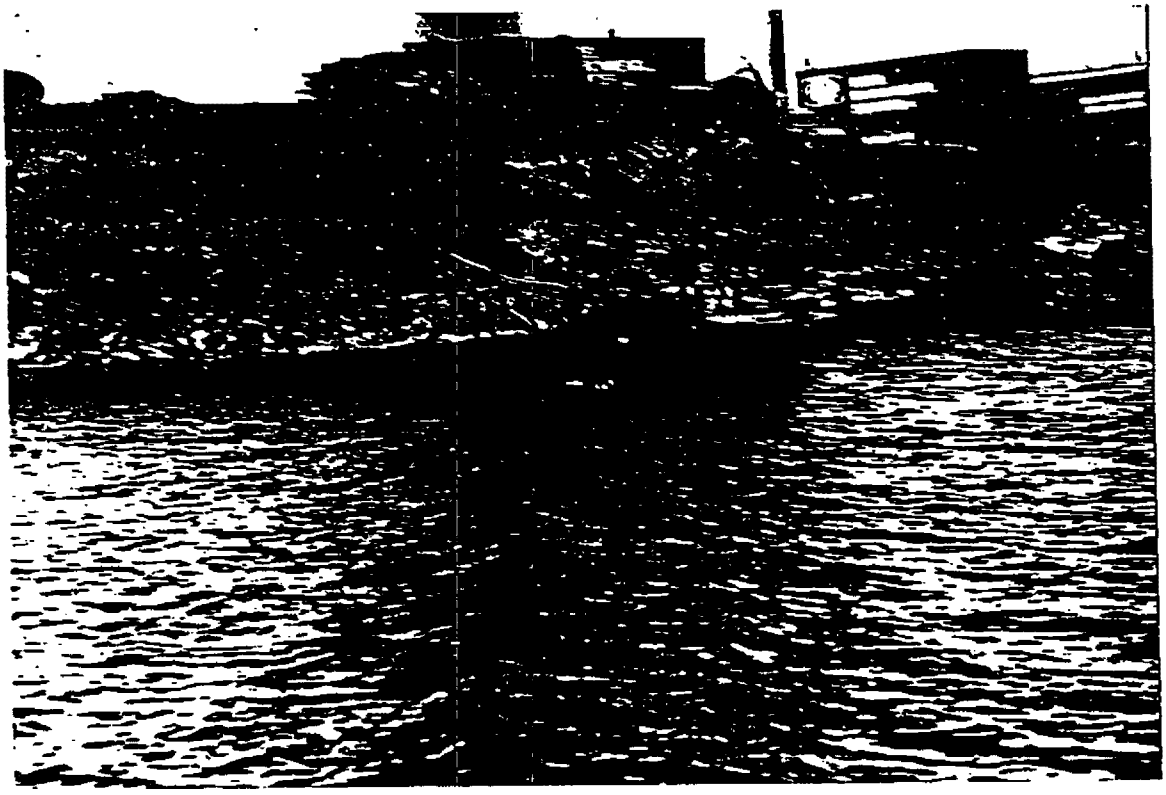
3. California brown pelican, Lauritzen Channel. 12/13/93.



4. California brown pelican, Lauritzen Channel, same as bird 3. 12/13/93.

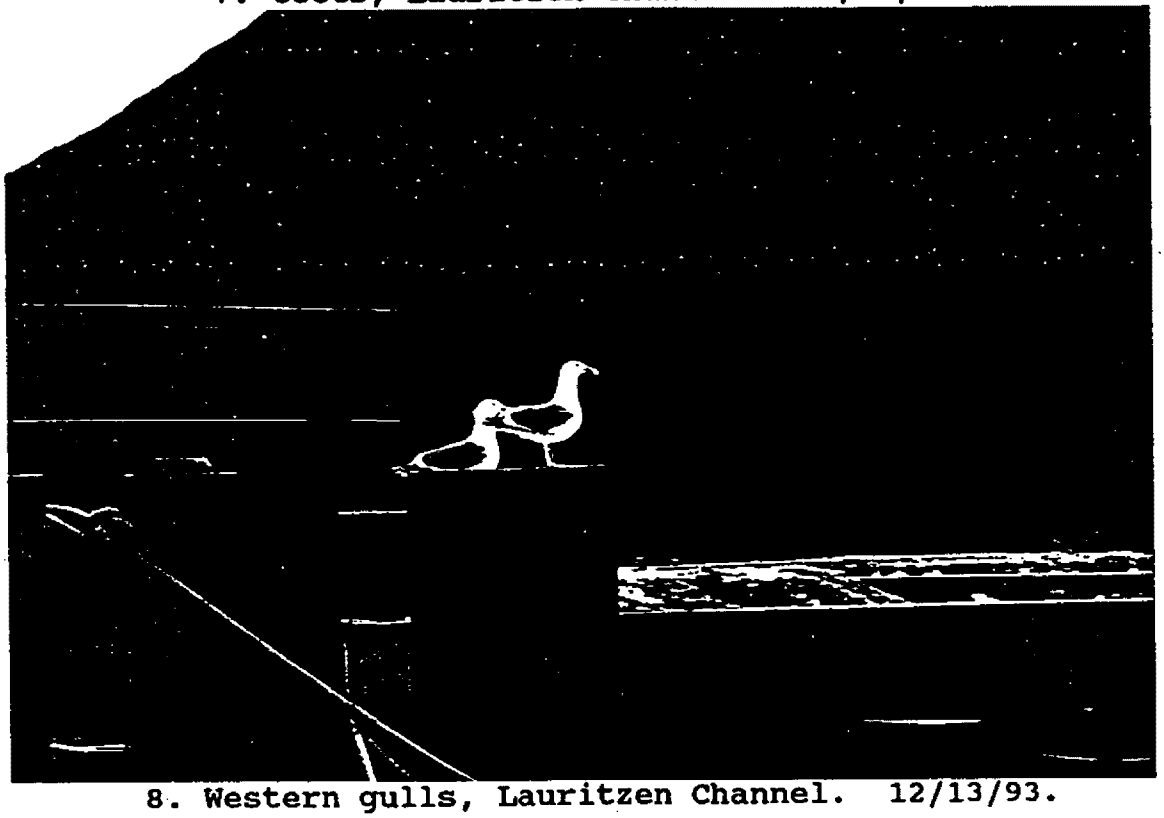


5. Double-crested cormorant and western gull, Lauritzen Channel. 12/13/93.

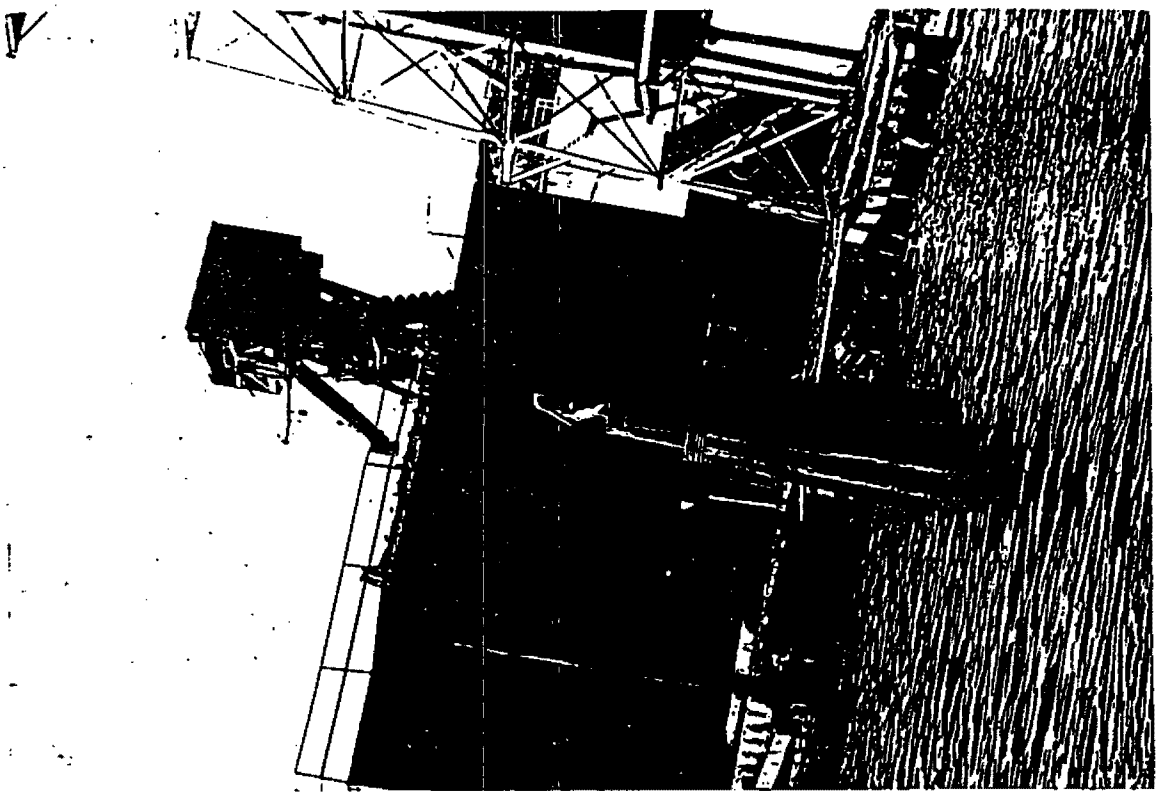


6. Double-crested cormorant, Lauritzen Channel. 12/13/93.

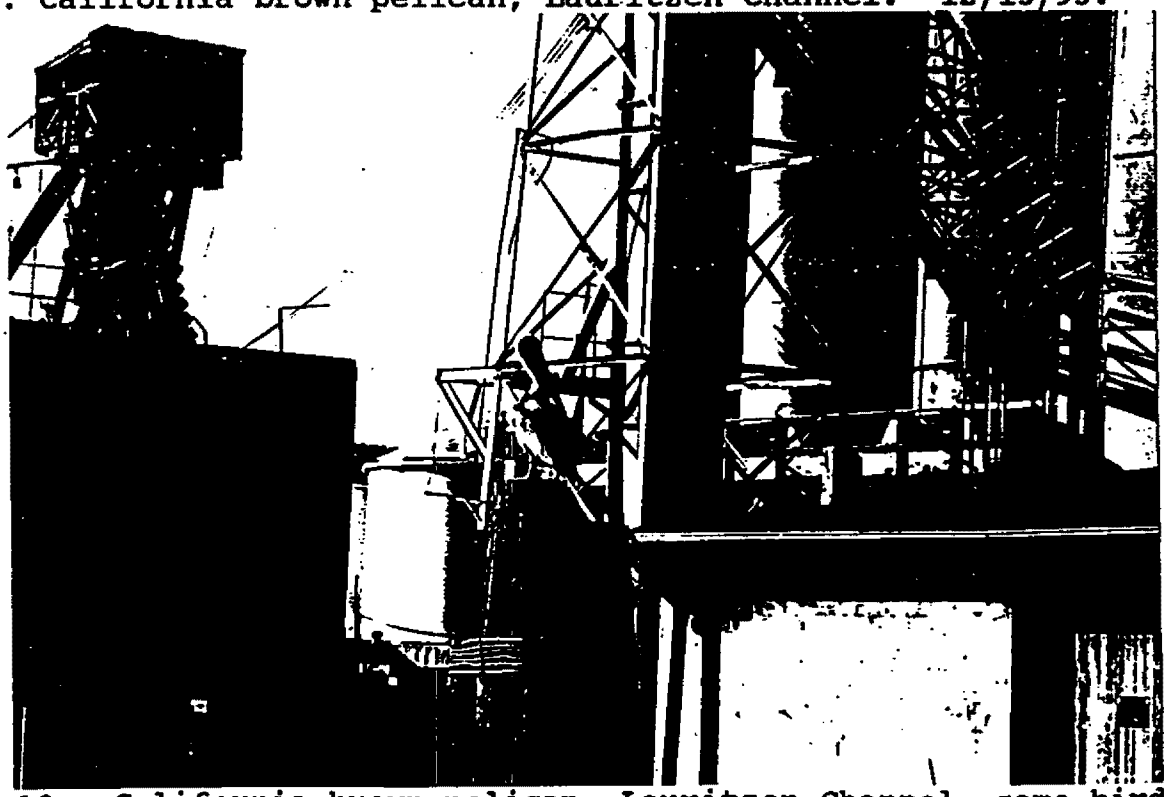
7. Coots, Lauritzen Channel. 12/13/93.



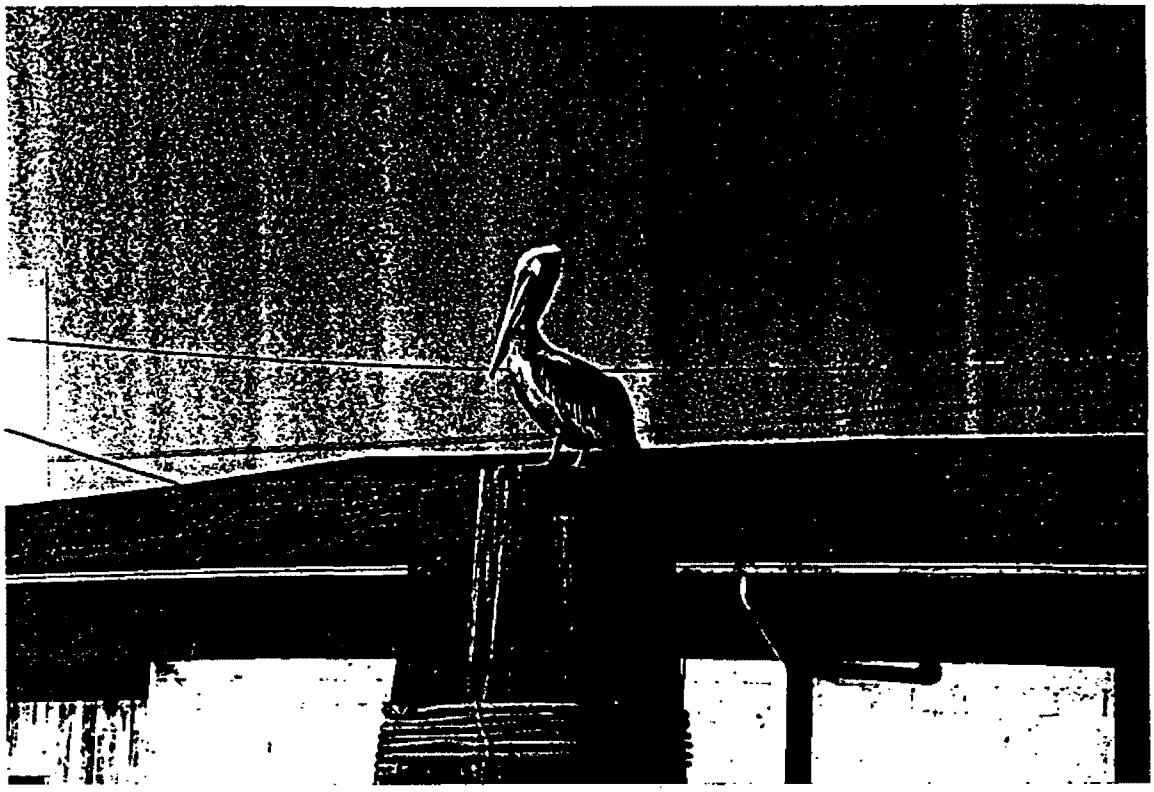
8. Western gulls, Lauritzen Channel. 12/13/93.



9. California brown pelican, Lauritzen Channel. 12/13/93.

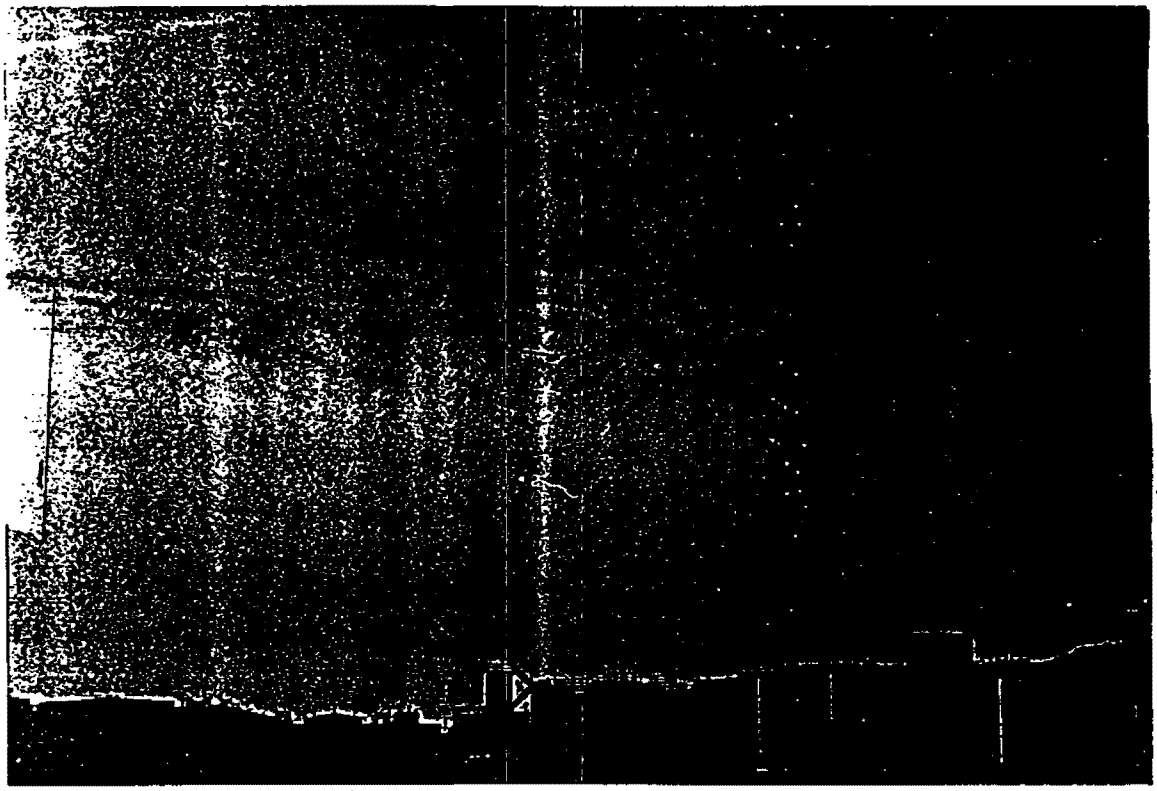


10. California brown pelican, Lauritzen Channel, same bird as 9. 12/13/93.

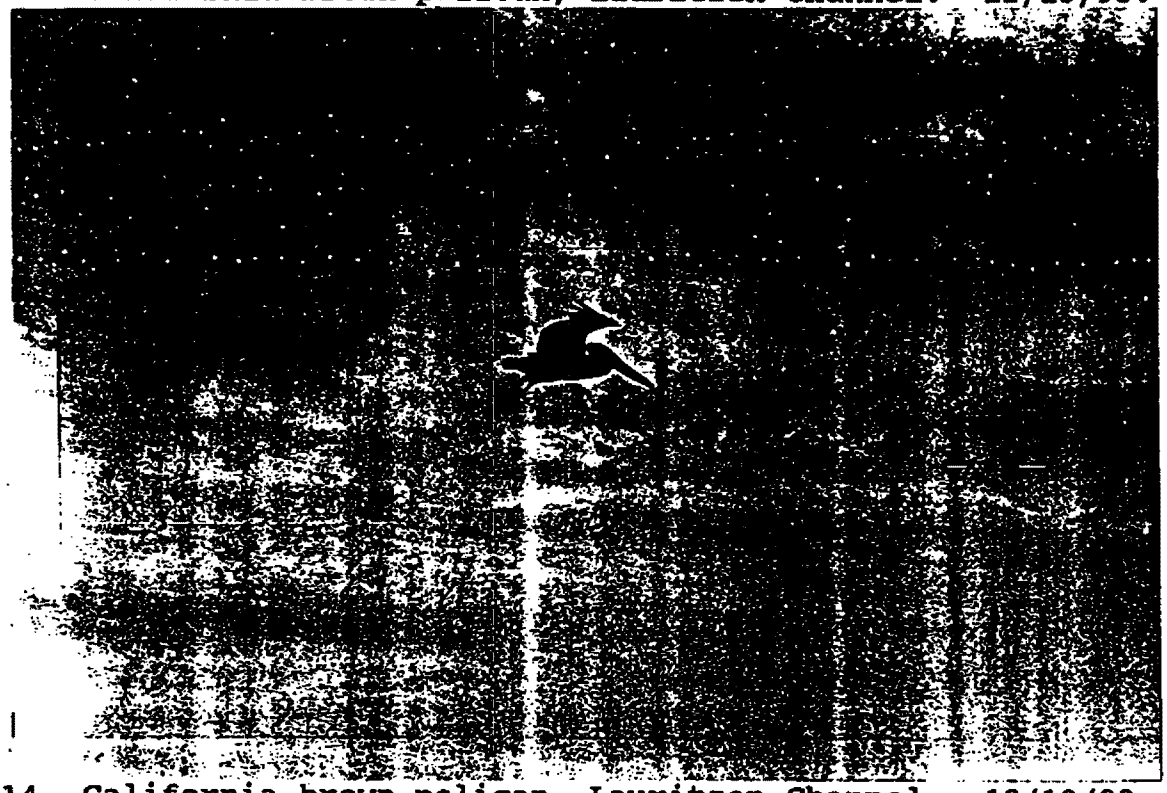


11. California brown pelican, Lauritzen Channel, same bird as 9. 12/13/93.

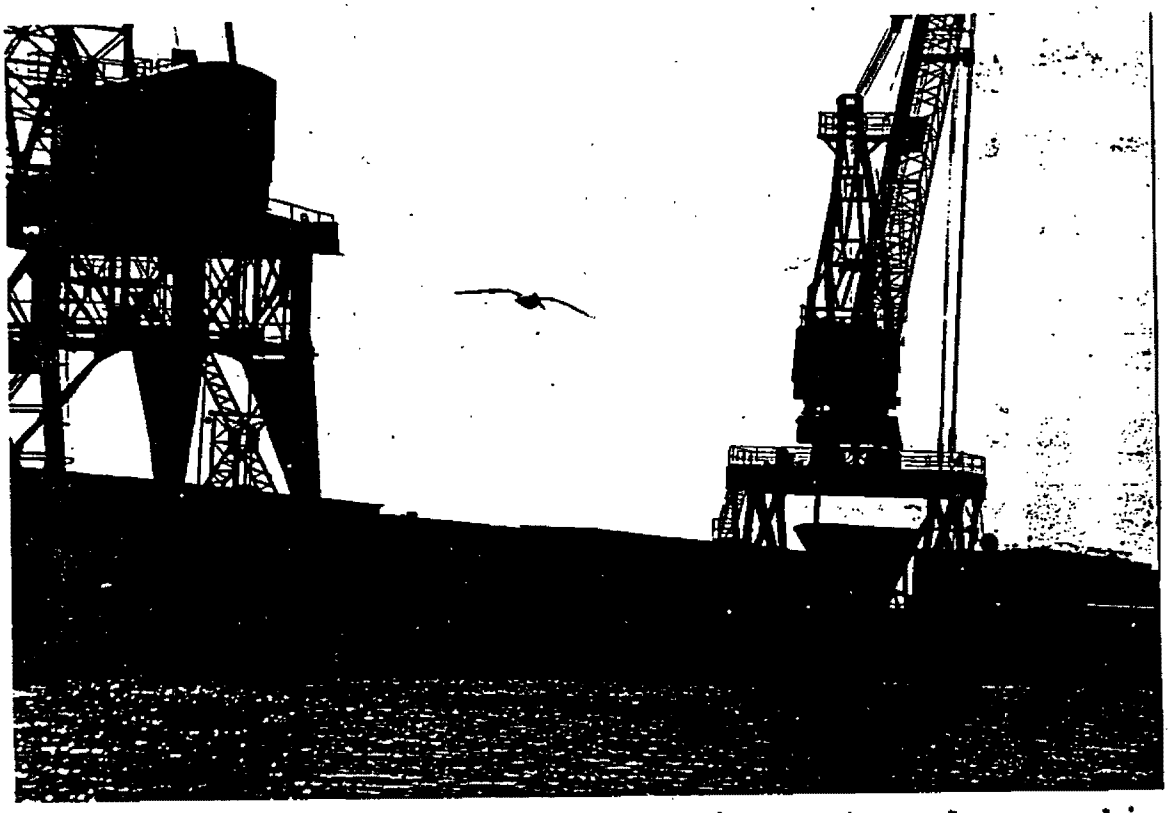
12. Gull, Lauritzen Channel. 12/13/93.



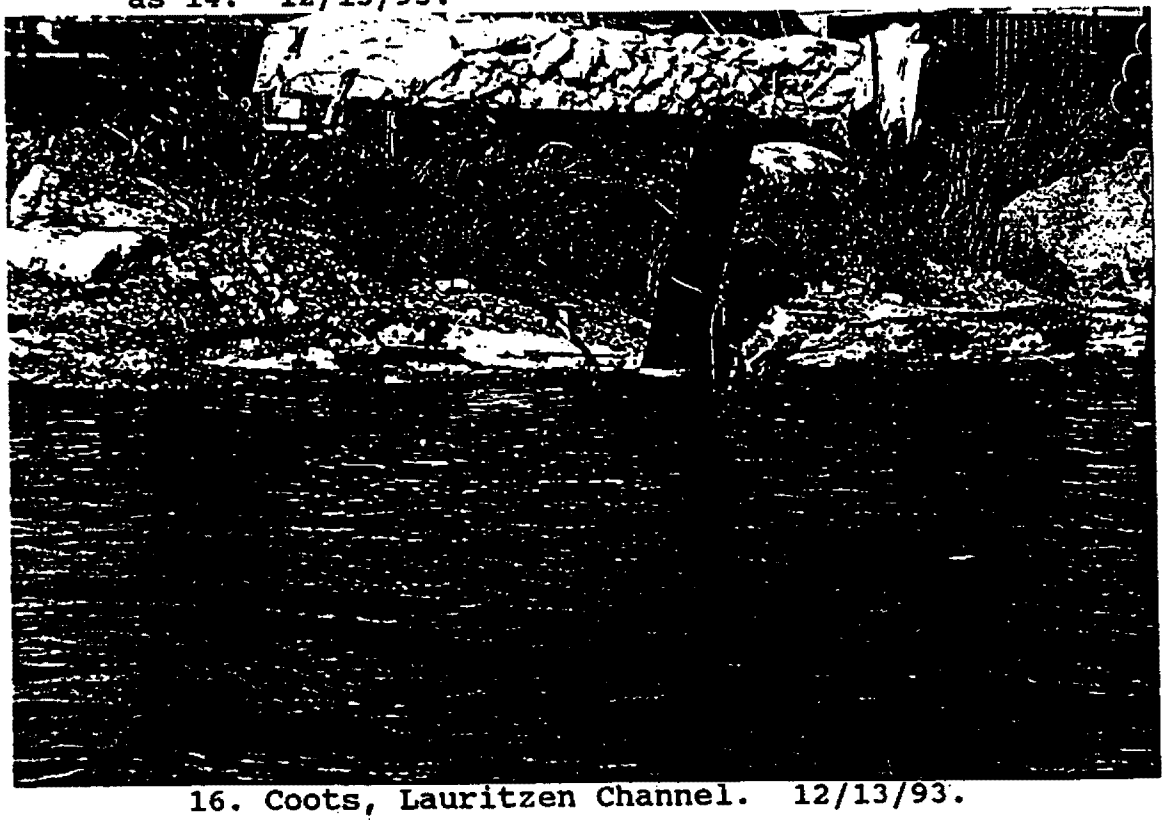
13. California brown pelican, Lauritzen Channel. 12/13/93.



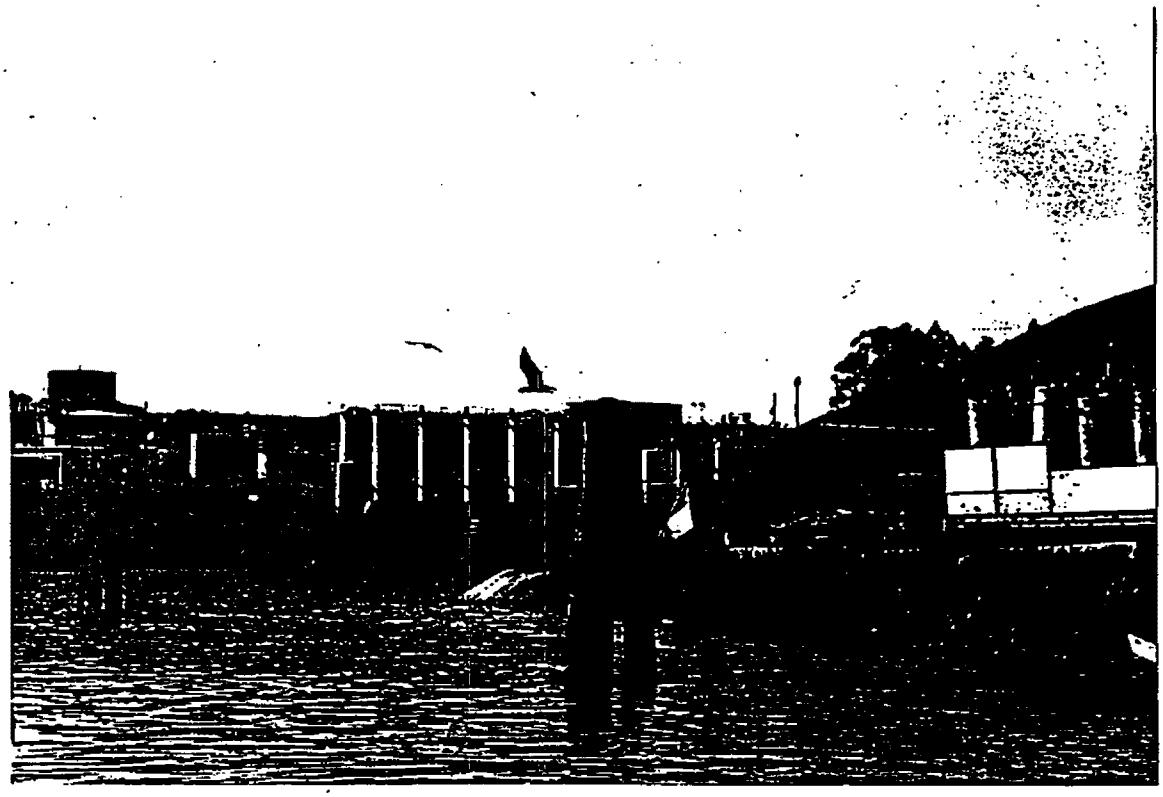
14. California brown pelican, Lauritzen Channel. 12/13/93.



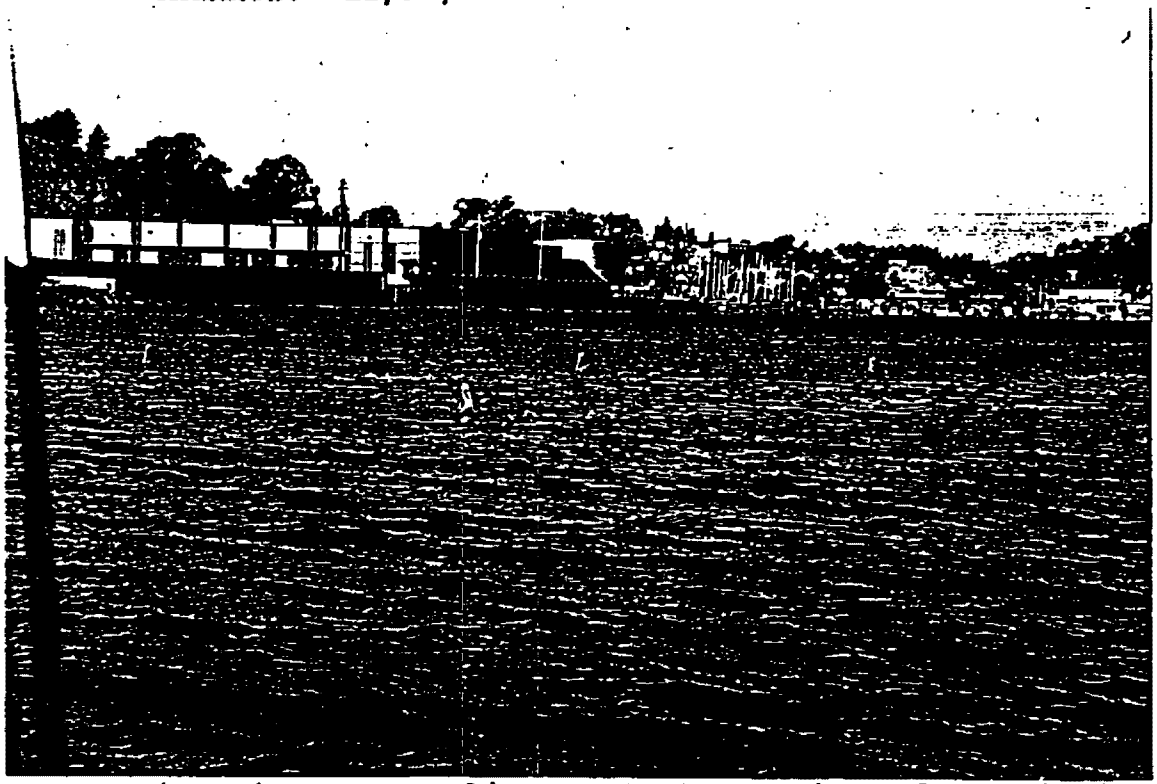
15. California brown pelican, Lauritzen Channel, same bird as 14. 12/13/93.



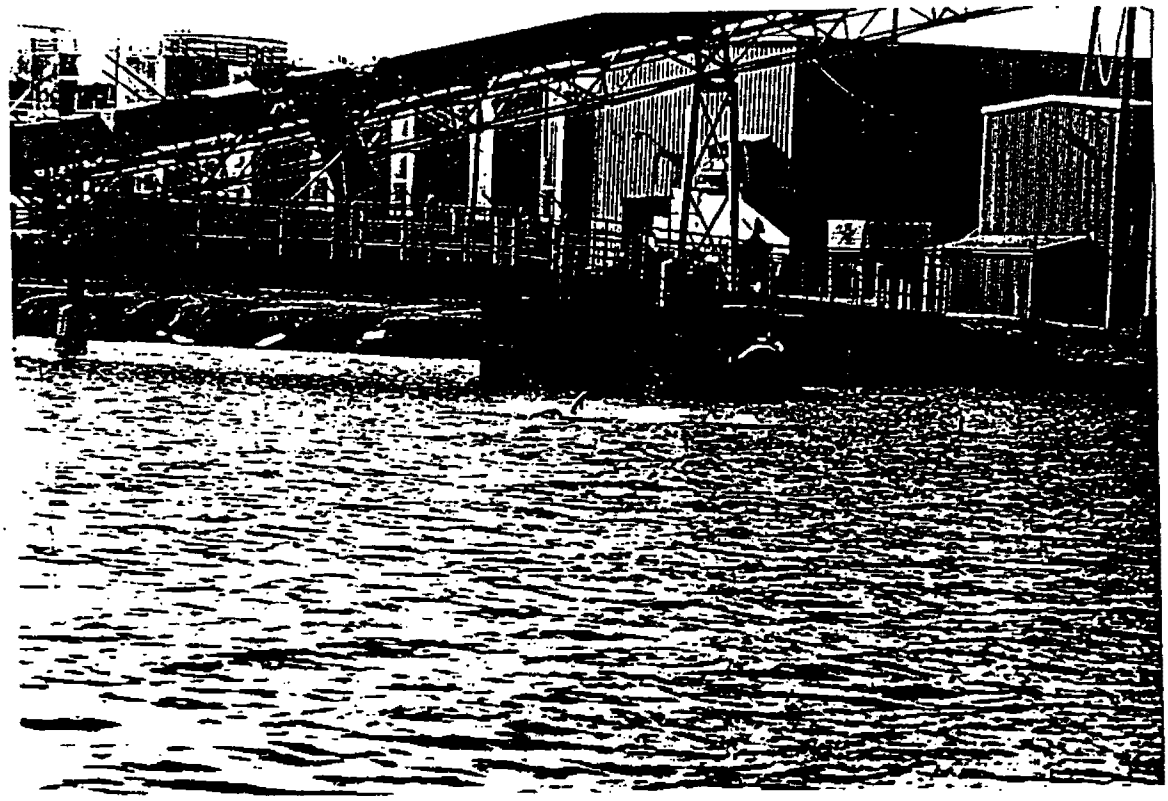
16. Coots, Lauritzen Channel. 12/13/93.



17. Cormorants, Lauritzen Channel, pelican, Santa Fe Channel. 12/13/93.

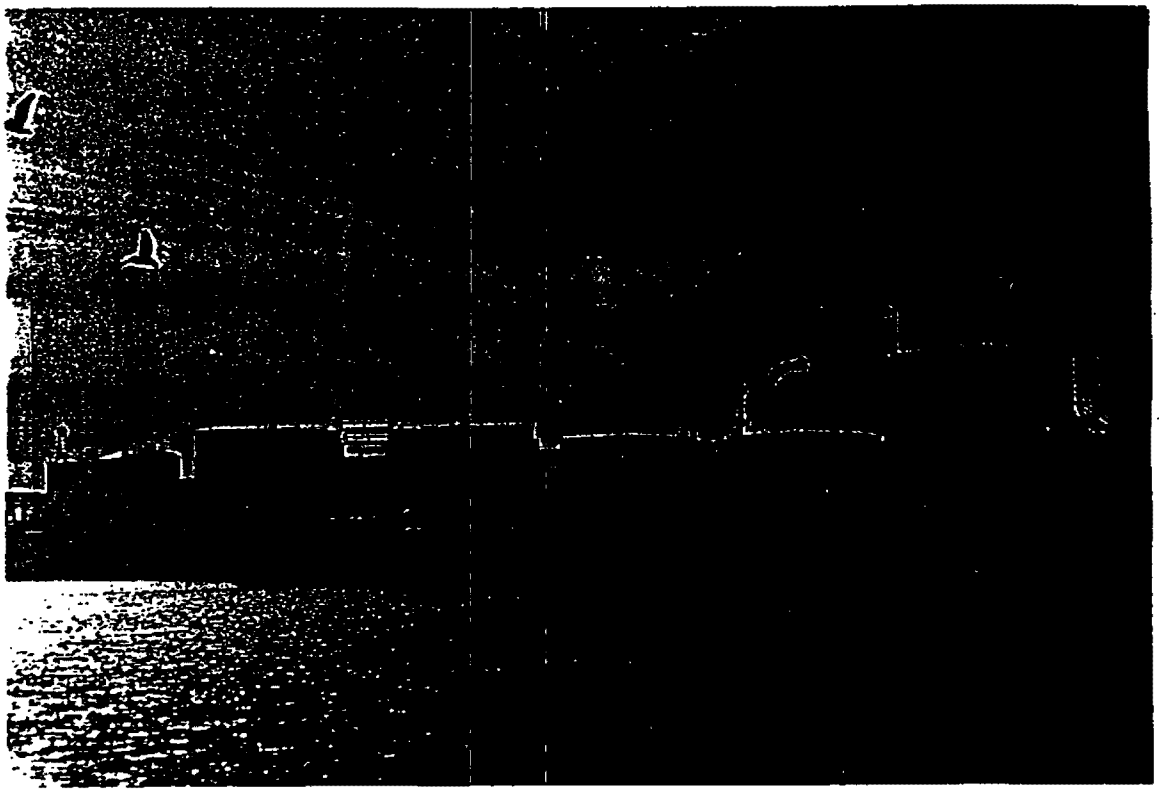


18. California brown pelicans, Santa Fe Channel. 12/13/93.

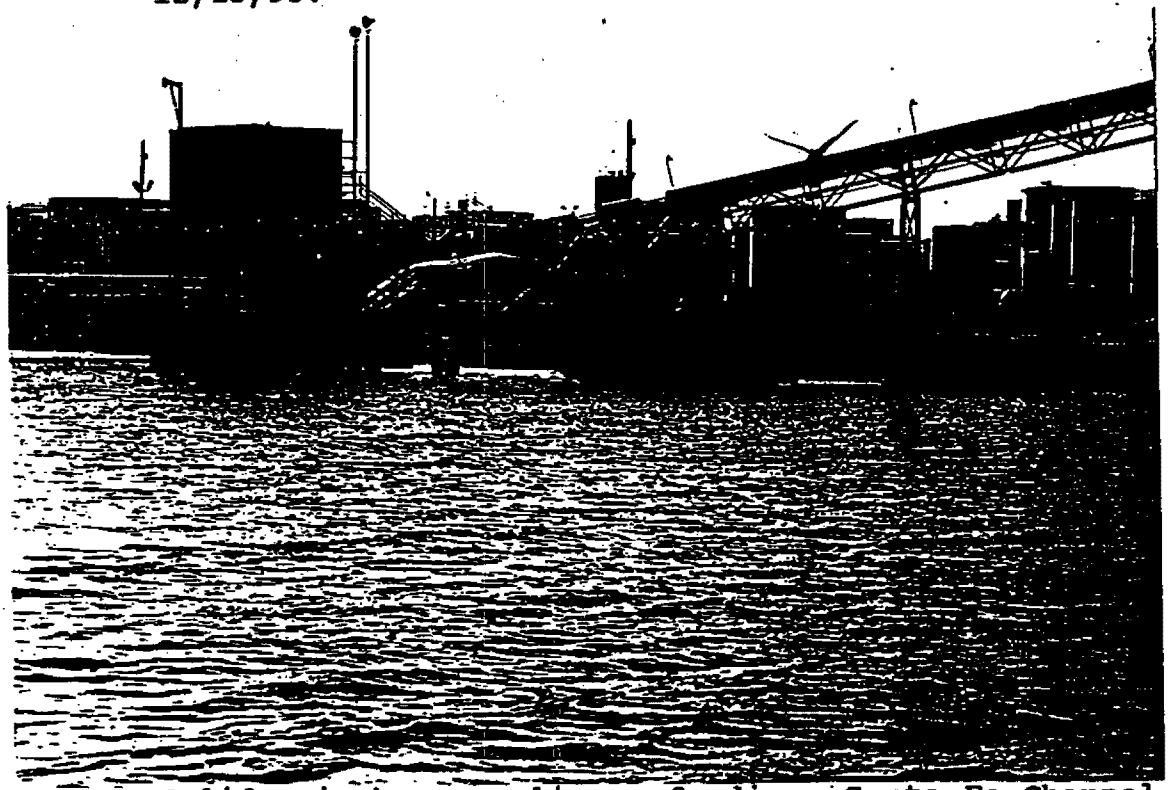


19. California brown pelican feeding, gull and cormorant, Santa Fe Channel. 12/13/93.

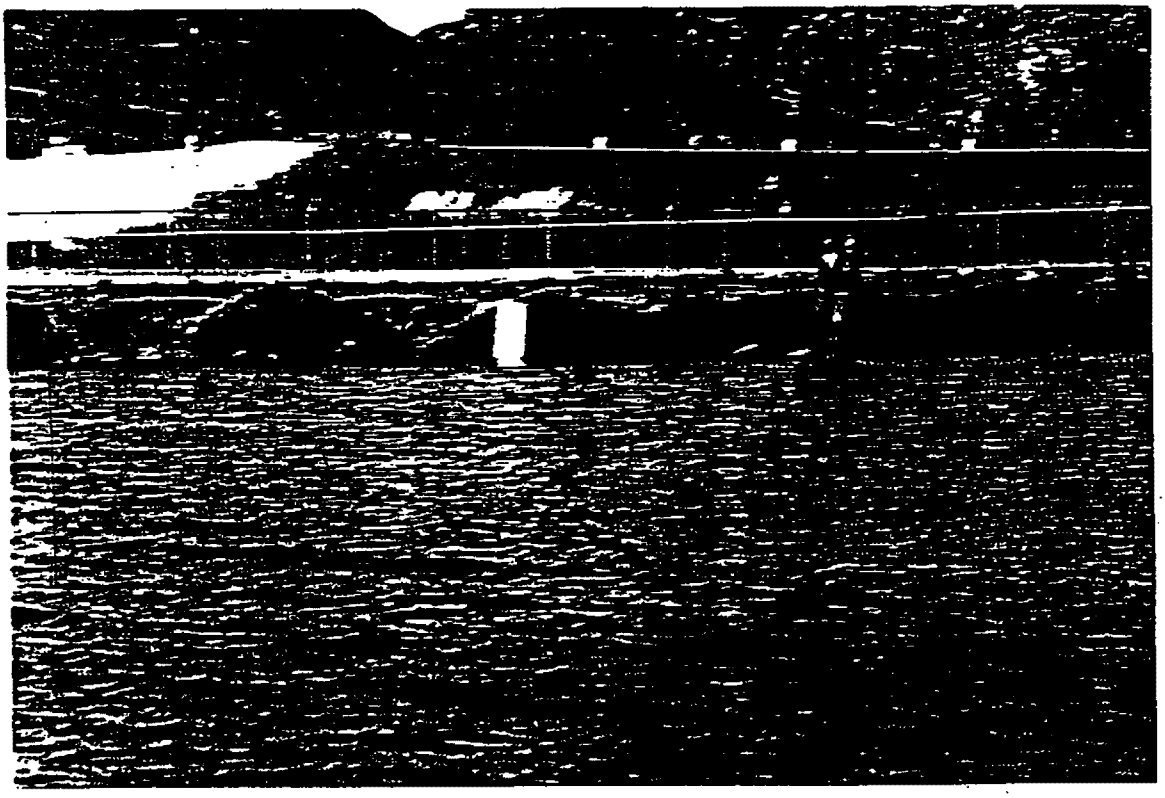
20. California brown pelicans, Santa Fe Channel. 12/13/93.



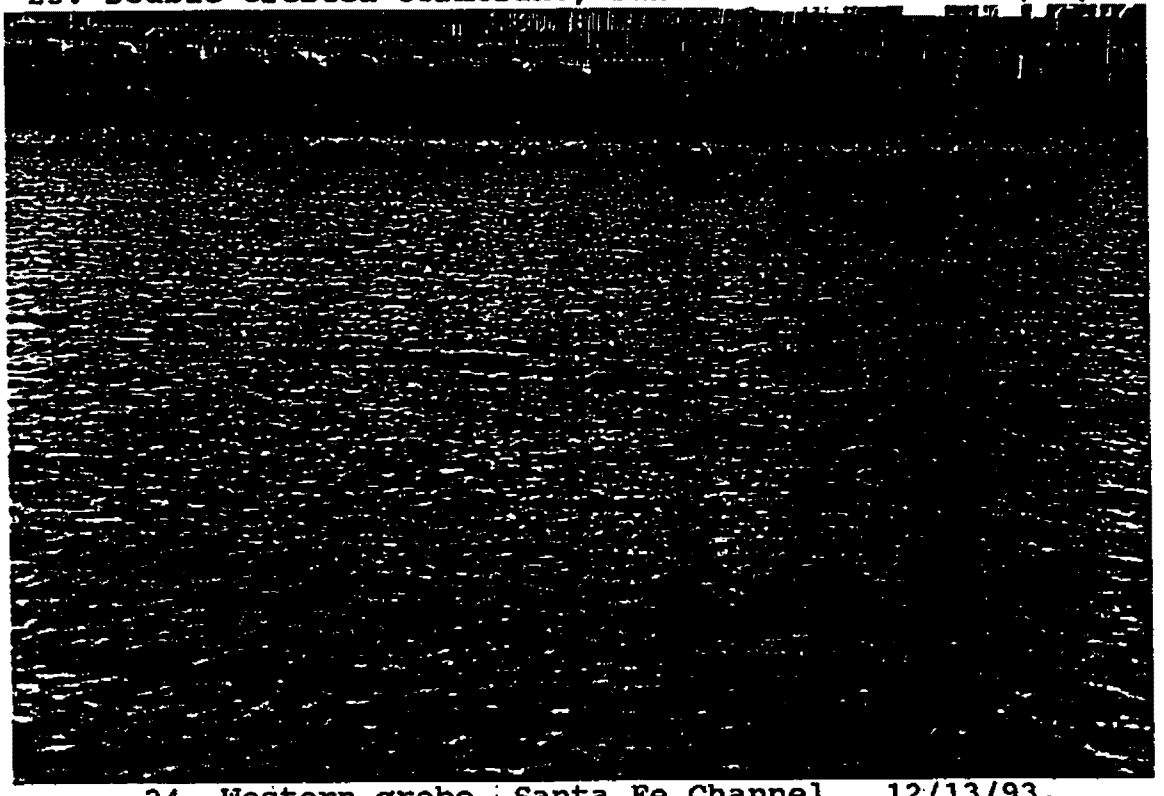
21. California brown pelicans feeding, Santa Fe Channel. 12/13/93.



22. California brown pelicans feeding, Santa Fe Channel. Most are same birds as in photo 21. 12/13/93.

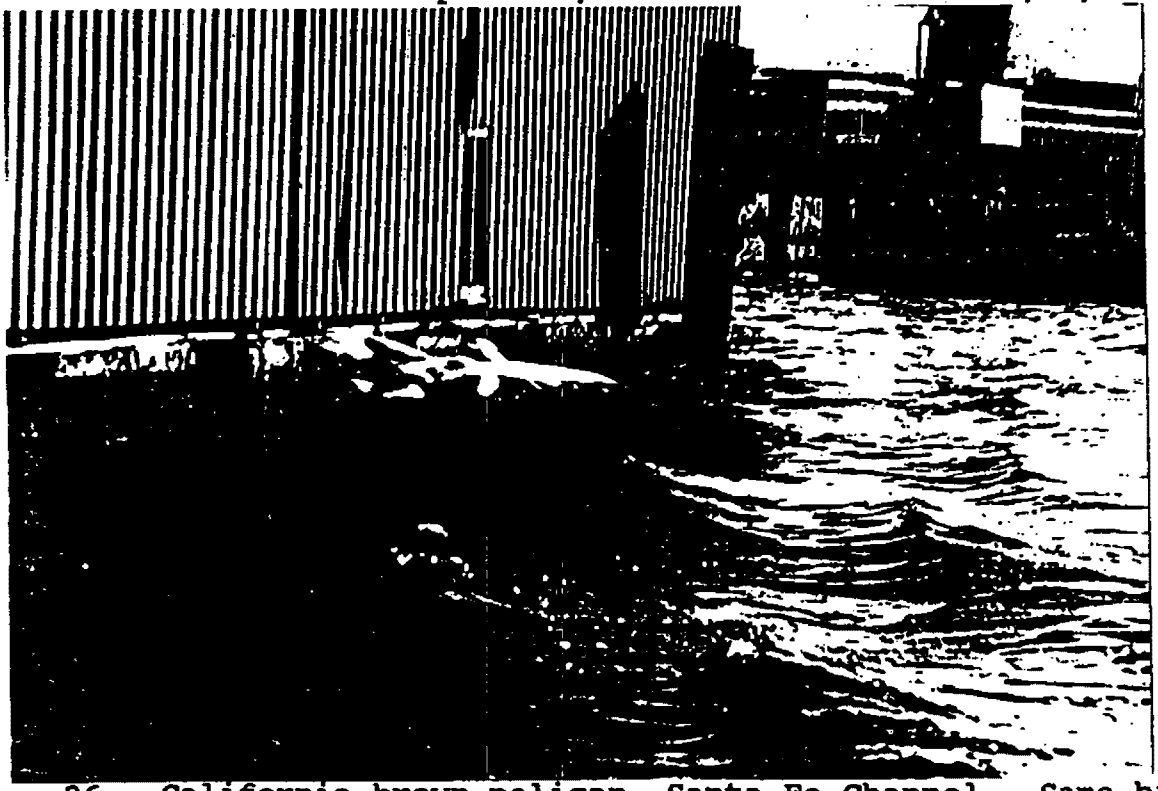


23. Double-crested cormorant, Santa Fe Channel. 12/13/93.

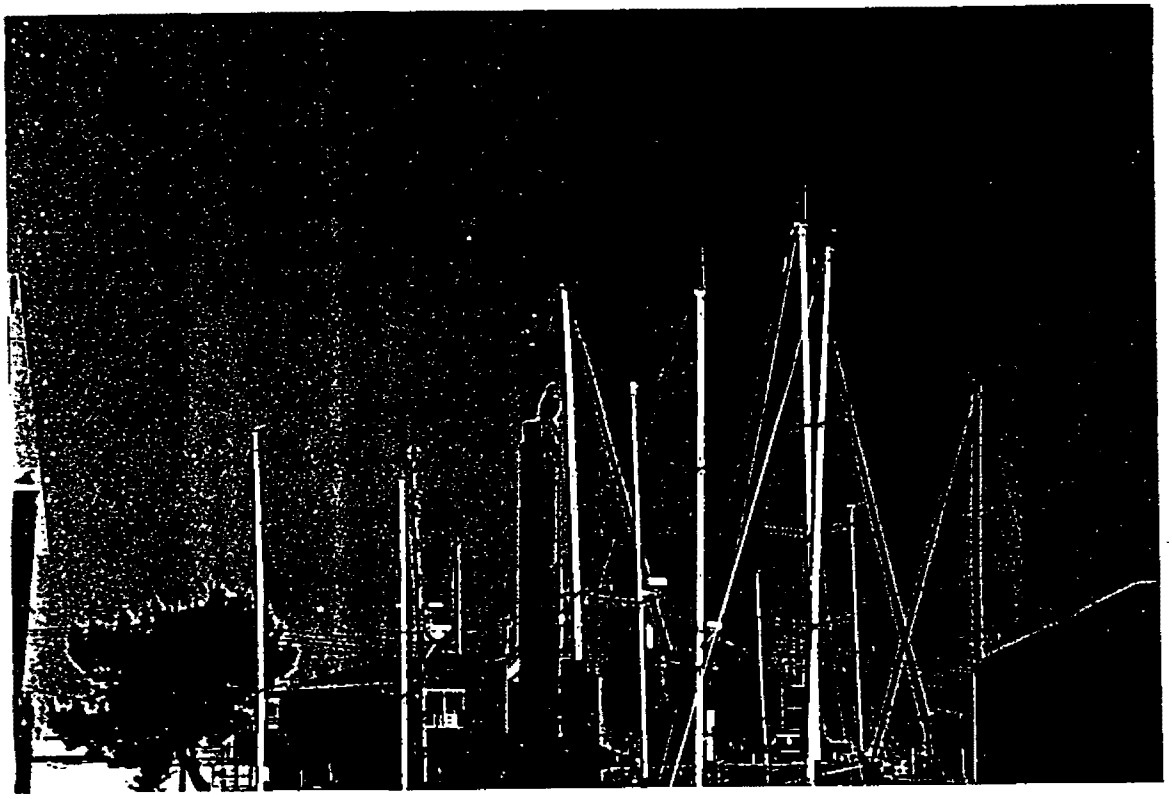


24. Western grebe, Santa Fe Channel. 12/13/93.

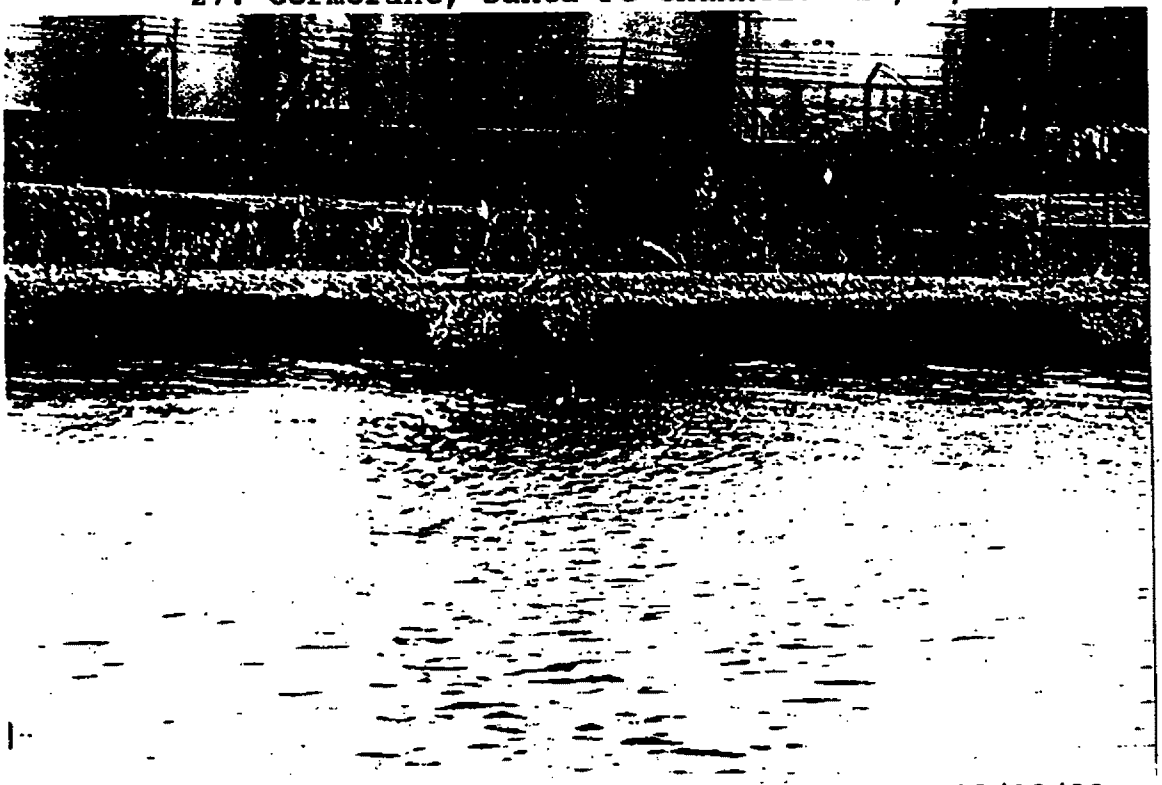
25. California brown pelican, Santa Fe Channel. 12/13/93.



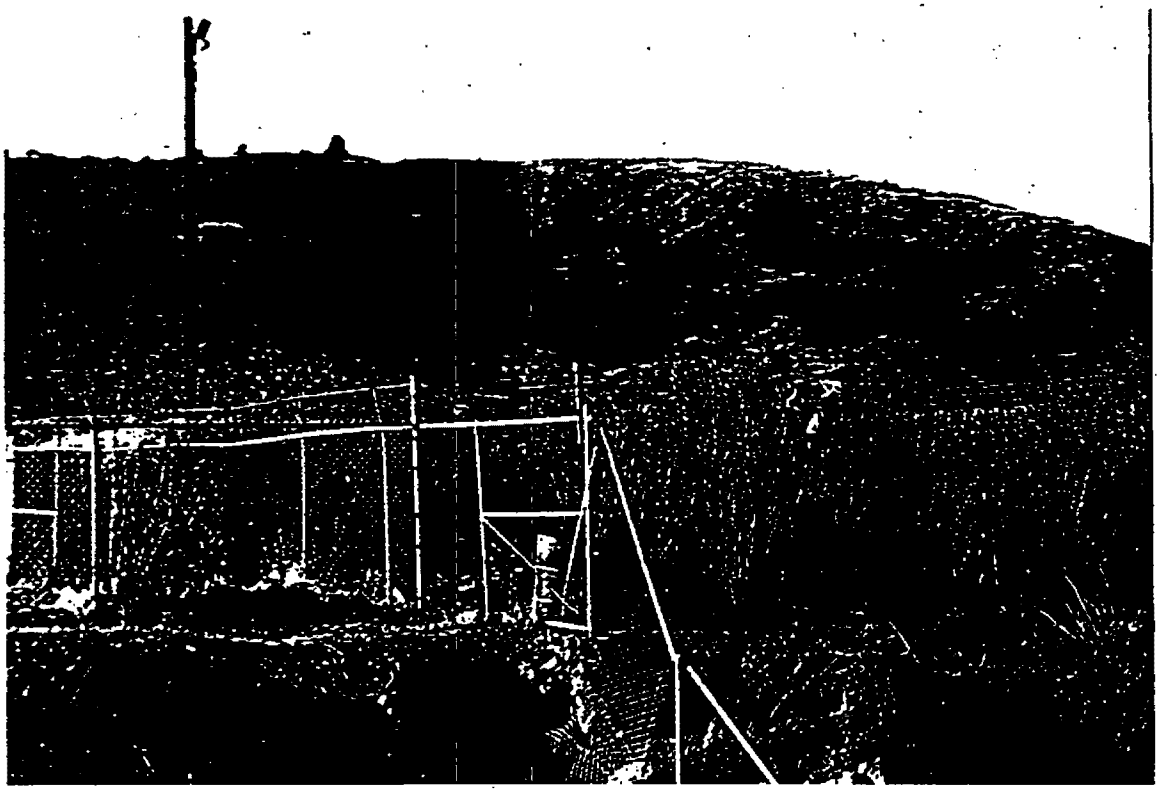
26. California brown pelican, Santa Fe Channel. Same bird as 25. 12/13/93.



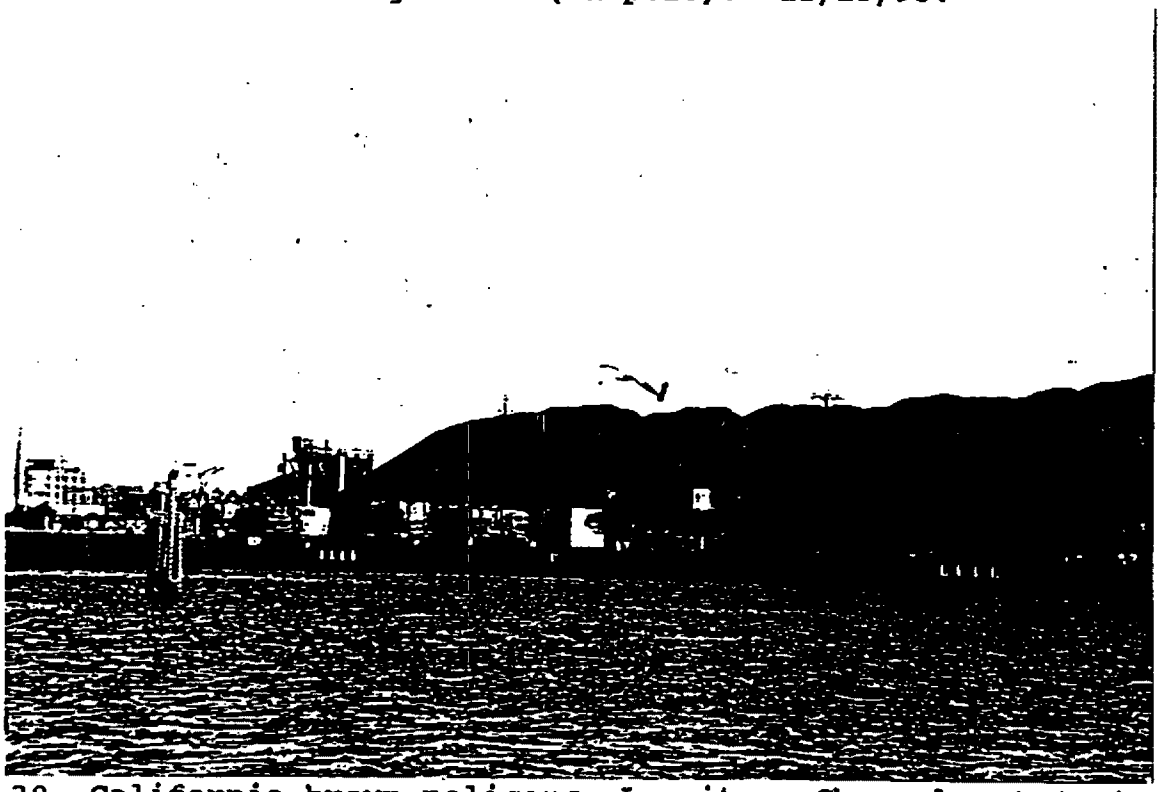
27. Cormorant, Santa Fé Channel. 12/13/93.



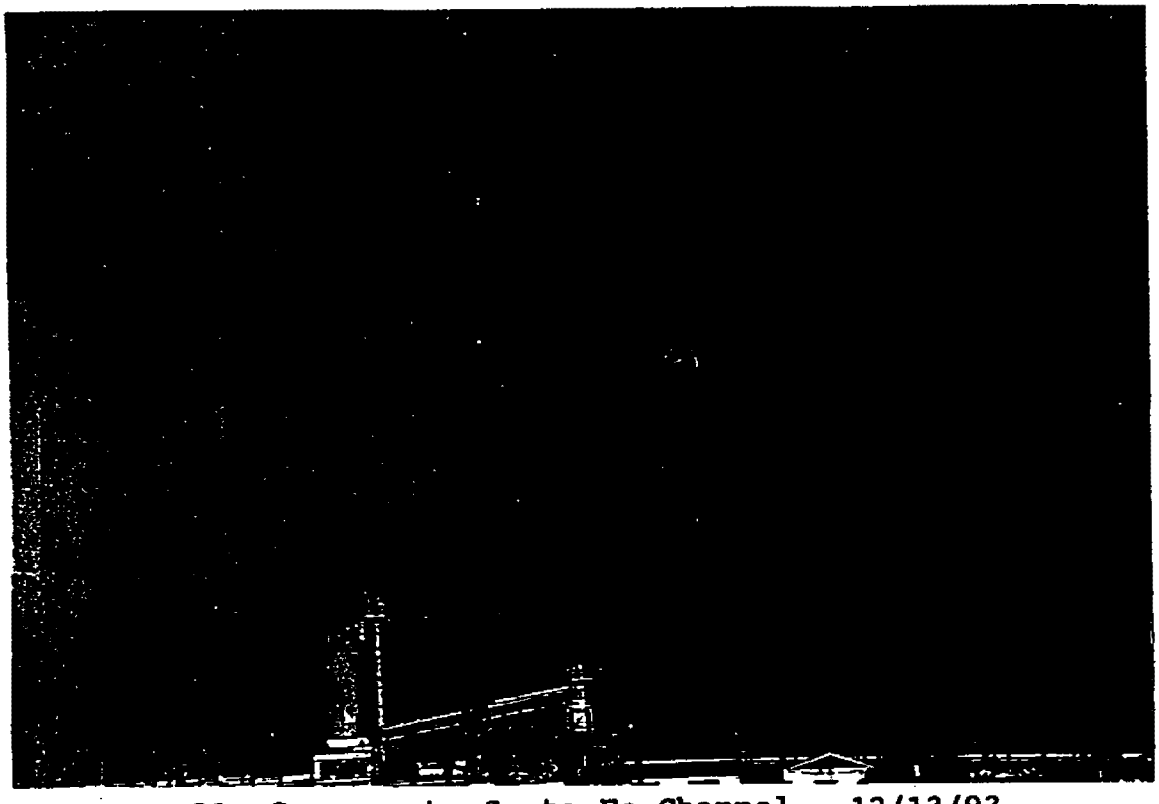
28. Grebes, Santa Fe Channel. 10:10 a.m., 12/13/93.



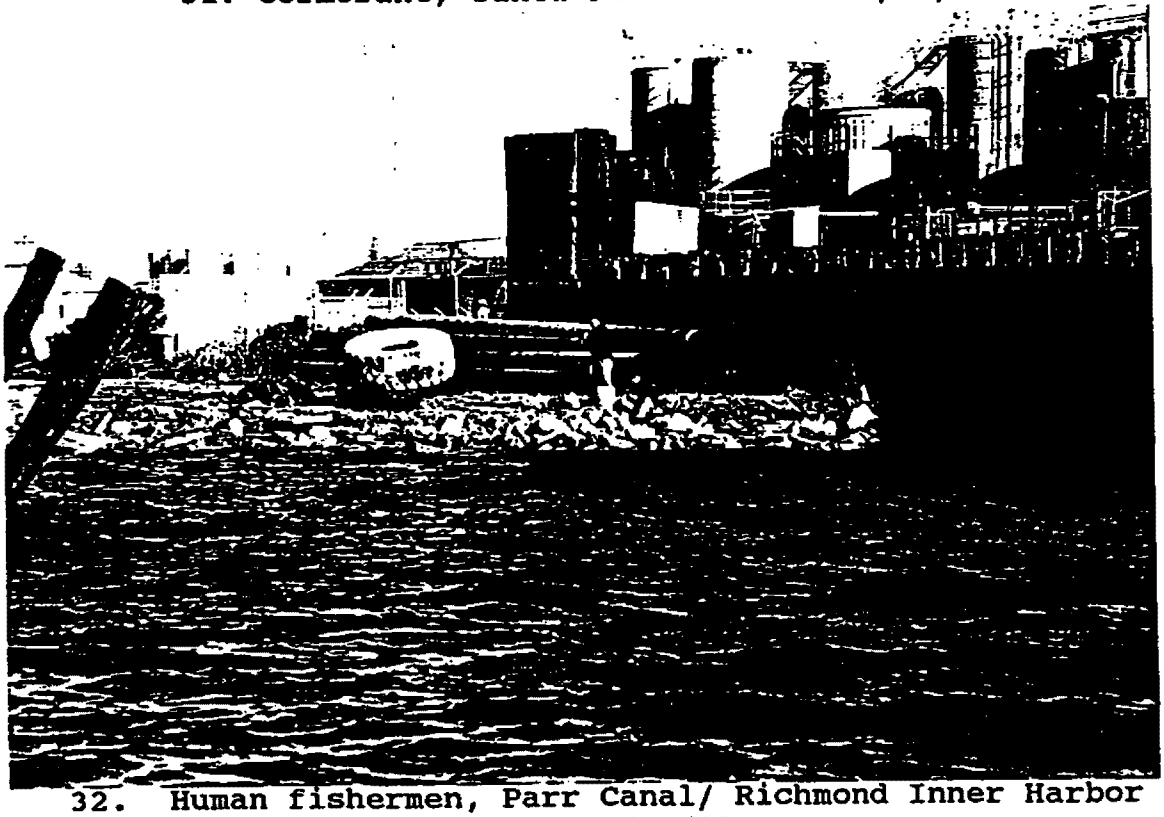
29. Kingfisher (on pole). 12/13/93.



30. California brown pelicans, Lauritzen Channel. 12/13/93.



31. Cormorant, Santa Fe Channel. 12/13/93.



32. Human fishermen, Parr Canal/ Richmond Inner Harbor Channel confluence. 12/13/93.

33. Sandpiper, Parr Canal. 12/13/93.

34. Grebes, Richmond Inner Harbor Channel/ Santa Fe Channel confluence. Group of approximately 14 birds. 10:30 a.m., 12/13/93.

APPENDIX 6-1. Chain-of-custody procedures.

All Superfund site samples collected were maintained under tight chain-of-custody procedures consistent with the National Enforcement Investigation Center 1978 guidelines as revised in 1986 (NEIC, 1986). Any sample collected was logged on Chain-of-Custody forms supplied by U.S. EPA Region IX, and identified by an eight digit sample number explained elsewhere in this report. In addition to the Chain-of-Custody forms, bound notebooks were prenumbered and assigned to personnel working on the project. Notebooks were individually pre-numbered 400, 401, and up to 449. Each page within those notebooks were pre-numbered using a six digit number starting with the notebook number followed by the page number. For example, page one in notebook 400 was numbered 400001. For the purpose of this report, we have split specific custody procedures into two (2) main categories (Field Operations and Laboratory Operations).

FIELD OPERATIONS

All field operations were coordinated by the U.S. EPA Field Coordinator, David T. Specht, and all water, sediment, and biological samples taken remained in the direct custody of the field crew doing the sampling. Once the sampling was completed, the samples, and one copy of the appropriate chain-of custody form, were placed into shipping containers and officially sealed with official U.S. EPA seals signed by the field coordinator, or alternate, and dated. Shipping containers were delivered to Newport, OR by the United Parcel Service (UPS) overnight service and/or trucked back by the sampling crew. All sampling and shipping operations were documented in the field notebooks of the sampling crew.

LABORATORY OPERATIONS

Upon arrival at the U.S. EPA laboratory in Newport, Oregon, the shipments were accepted by the sample custodian, Robert C. Randall, and placed into custody in the Superfund Laboratory (L-118). Only the sample custodian, his alternate - Kathy Sercu, and the Facilities Manager - David Sweitzer had official access to this space. The space was located in the laboratory wing of the Pacific Ecosystems Branch (PEB) Laboratory and contained a walk-in controlled temperature room, maintained at 4°C, and a chest freezer. For the record, Robert C. Randall was assigned to other duties as of October, 1992 and David T. Specht assumed the duties of sample custodian as of December 28, 1992.

To maintain strict custody of all Superfund samples, the laboratory wing was also locked and it remained locked after the first Superfund sample arrived. Persons on the Pacific Ecosystems Branch Telephone List had key access to the laboratory wing.

Other persons needing to enter the space were required to sign in at the reception desk and remain under escort of an authorized person.

Custody for all samples that arrived at Newport was accepted by the sample custodian who logged them in and documented the shipping containers and samples received. Documentation included:

* A description of the shipping container, how it was officially sealed, and if the seal/s were intact

* Who broke the seal/s on the shipping container and the date that they broke it.

* A list of items in each shipping container including a description of the samples, the sample numbers, the log sheets, and a description of the packing materials used.

Sample custody was transferred to others within the PEB laboratory by having them sign for the samples (in the custody log). All sample operations were documented in the sample analysis log of the persons doing the work. After the analyses were completed, the samples and custody of them was returned to the sample custodian who documented that in the custody log.

Samples shipped to other sites for analysis were shipped officially sealed and with completed custody transfer forms. Person/s receiving those samples were asked to accept and maintain official custody of them. Custody for samples returned was accepted by the sample custodian using procedures described above.

COMPLIANCE WITH CHAIN-OF-CUSTODY PROCEDURES

Neither the original sample custodian, Robert C. Randall, nor his replacement, David T. Specht, noted any violations of chain-of-custody with the field or laboratory samples.

APPENDIX 6-2. Detailed field notes, station descriptions.

The following list is excerpted from the sampling plan and annotated to reflect the actual field activity.

II. U.S. EPA SAMPLING PLAN, UNITED HECKATHORN/LAURITZEN CANAL

U.S. EPA Station #	Levine-Fricke Station #	Approximate Geographic Location
1	~~LC-5~~	Lauritzen Canal, west side, across from former Bldg 1 complex, ~50' offshore
	(Changed in the field to LC-2, former EPA alternate station #10; mid-channel, 20-23' depth)	
2	~~LC-11~~	Lauritzen Canal, west side, across from former Bldg 2 complex, ~50' offshore, ~250' south of EPA stn #1
	(Changed in the field to LC-5, former EPA primary station #1, ~20-25' off west shore; ~20' depth)	
3	LC-14	Lauritzen Canal, west side, across from former Bldg 4, ~20-25' off west shore, ~500' south of EPA stn #2;
4	LC-17	Lauritzen Canal, west side, across from pile-supported dock, ~20-30' off west shore, ~750' south of EPA stn #2, near mouth of canal
5	SFC-9	Santa Fe Channel, at westerly widening of boat basin, ~25-50 N of south shore
6	SFC-11	Santa Fe Channel, ~20-30' off center of pier at head of Wharf St., south shore
7	RI-1-C-41	Head of Richmond Channel, at mouth of Santa Fe Channel, ~150-200'E of west shore
8	RI-1-C-33	Midway along Richmond Channel, ~800' S of EPA Stn #7, ~150-200' E of west shore
9	~~RI-1-C-24~~	At mouth of Richmond Channel, ~1000' S of EPA Stn 8, ~150-200' E of west shore, ~at west shore mouth of Richmond Channel at Point Potrero
	(Changed in the field to the opposite (east) side of the channel, ~100 SSE of red nun #16, in 10-12' depth, out of the ship channel; the present RI-1-C-24 location became alternate station *-009.)	

Trawls:

200 Lauritzen Canal
Starting at the N end of Lauritzen Canal, proceeding southerly to its junction with Santa Fe Channel (~ EPA Stn #4)

(Changed in the field, starting approximately 1/2 the distance between station 1 and 2, proceeding southerly towards station 2, where the cod-end of the net snagged on the bottom and tore off. The cod end (~6-7') was recovered by gaffing a floating line attached to and trailing the cod-end. A substantial sample was recovered intact from the salvaged net. Due to damage to the gear, no subsequent sampling was attempted.)

300 Santa Fe Channel
Starting at the NW boat basin (EPA Stn #13), proceeding SE to its junction with the head end of the Richmond Harbor Channel (EPA Stn #7)

(Changed in the field to encompass station 6 to station 5.)

400 Richmond Harbor
Starting at the N end of the Richmond Channel Harbor at EPA Stn #7, proceeding southerly to EPA Stn #9

(Changed in the field to ~5 minute tows encompassing station 8.)

Mussel samples:
500-510-520 Santa Fe Channel mussels, to include Lauritzen Canal and Richmond Harbor Channel (Mytilus galloprovincialis)

Tissue residues in field-S1CHT-501-505 mussels, SFCh
collected mussels -511-515 ", LCh
-521-525 ", RHCh

Overlying water at mussel S1CHO-501 high-tide overlying water
collection site -502 mid-tide, SFCh
-503low-tide

S1CHO-511high-tide " "
-512mid-tide, LCh
-513low-tide

S1CHO-521high-tide " "
-522mid-tide, RHCh
-523low-tide

Idaho Flat Yaquina Bay, OR 1Bulk Sed (TOC, IW), gr sz
Control + incidental infauna

Sediment Toxicity: S1CES-005 -006
-007
(See Lamberson, 1991)

Interstitial waterS1CEI-005
Chemistry -006
-007
(See Lamberson, 1991)

S1CCS-001 Purging sediment added to contaminated sets
-002 Purging sediment added to controls

Eohaustorius Yaquina Bay, OR 1 Bulk Sed (TOC, IW), gr sz
Heaven + incidental infauna

Sediment Toxicity: S1CES-001
-002
-003

(See Lamberson, 1991)
Interstitial water SICEI-001
chemistry -002
-003
(See Lamberson, 1991)

ALTERNATE STATIONS:

*009	RI-1-C-24	At mouth of Richmond Channel, ~1000' S of EPA Stn 8, ~150-200' E of west shore, ~at west shore mouth of Richmond Channel at Point Potrero
~~*10~~	~~LC-2~~	~~Lauritzen Canal, ~mid-channel, aligned with N end of Bldg 1 complex, ~100' E of west shore~~

(Changed in the field to primary station #1)

~~*11~~	~~LC-4~~	~~Lauritzen Canal, aligned with ~center of Bldg 1 complex, ~35' W of east shore off abandoned pilings~~

(Station not sampled. Considered to be superfluous to primary stations in the area.)

~~*12~~	~~LC-7~~	~~Lauritzen Canal, aligned with ~center of Bldg 2 complex, ~20' W of east shore off abandoned pilings~~

(Station not sampled. Considered to be superfluous to primary stations in the area.)

*13	SFC-3	Santa Fe Channel, at westerly head of boat basin, ~50 SE of north shore piers
*14	SFC-15	Santa Fe Channel, ~400' E of 2-36"CMP, ~300'N south shore, ~1000' E of EPA Stn# 6
*15	SFC-16	Center of Santa Fe Channel, S of west shore mouth of Lauritzen Canal
*16	SFC-18	Center of Santa Fe Channel, ~250' E of mouth of Lauritzen Canal
*17	RI-1-C-37	North end of Richmond Harbor Channel, ~1200' S of Santa Fe Channel mouth, ~500' E of west shore
*18	RI-1-C-28	North end of Richmond Harbor Channel, ~2000'S of Santa Fe Channel mouth, ~500' E of west shore
*19	RI-1-C-21	Main Richmond Harbor Channel, ~1000' N of Brooks Island, due south of most westerly slip of Point Potrero facility, ~200' S of dredged channel
*20	RI-1-C-10	Main Richmond Harbor Channel, ~1650' E of Point Richmond, ~1000' S of pier, ~mid-channel
*21	RI-1-C-3	Main Richmond Harbor Channel, ~4000' W of Point Richmond, ~3500' E of Chevron Wharf, on N side of dredged channel

(*21B: station *21 resampled after station *22 because of insufficient benthos sample; the first sample, out of the incompletely filled grab, yielded a variety of organisims more typical of a relatively undisturbed station, i.e., brittle stars, amphipods, etc. It was deemed desirable to obtain a measured 10 cm core for benthos analysis. The station was automatically relocated by Loran-C coupled to the vessels' autopilot

control mechanism
(Station 21 Loran C lat 37°54.85'N, long 122°24.11'W
E of green #3 off Point Richmond
W of green #5 off Point Richmond.)
Upon receipt in Newport, the Sample Coordinator changed this number to *23)

*22 South of and parallel with the west end of the breakwater extending westerly from Brooks Island. Aligned with range markers for outer channel.

*23 (See Station *21 above) February, 1991 cruise:

Crabtrap, trotline locations:

F-1 Heavy rope trailing into water from piling 15 m N of N end of ferry, Lauritzen Canal, within ~30 m of Stn 1; mussels, attached to rope ~2 m below surface, or ~halfway to the bottom (estimated @ ~13').

F-4 Santa Fe Channel, S end Boathouse, crabpot "C".

F-21 Santa Fe Channel, S end Boathouse, crabpot "C".

F-22 Richmond Harbor Channel crabpot, off S end Richmond Terminal #3 pier, E side of channel.

F-24 Lauritzen Canal, N end ferry rudder, ~ 0.5 m below surface (mussels).

F-25 Lauritzen Canal, Nylon rope hanging from piling ~40 m N of N end of ferry, ~halfway between Stn 2 & Stn 1, mussels collected from -1.5 -> -4 m down rope.

F-26 Lauritzen Canal, ferry rudder, black flakes from surface of rudder.

F-30 Santa Fe Channel Boathouse, 2nd slip, (same location as October '91 mussel samples; ~0.25 m below surface.

F-33 Santa Fe Channel Boathouse, crabpot

F-34 Richmond Harbor Channel (Terminal 3 pier) crabpot

F-35 Richmond Harbor Channel, Buoy red nun #16, anchor chain, ~1 m depth, mussels.

February Trawl lines:

F-2, F-3 Trawl lines in Lauritzen Canal, in the vicinity of Stn 1, ~50 m in length.

F-5-10, F-12-14 Trawl lines in Santa Fe Channel, in the vicinity of Stns. 5-6, ~200 m in length.

F-15-18 Trawl lines in Richmond Harbor Channel, in the vicinity of Stn 8, ~200 m.

F-19-20 Trawl lines in Lauritzen Canal, in the vicinity of Stn 1, ~50 m.

F-23 Trawl line in Lauritzen Canal, in the vicinity of Stn 1, ~50 m.

F-27 Trawl line in Lauritzen Canal, in the vicinity of Stn 2 -> Stn 1, ~50 m.

F-28 Trawl line in Lauritzen Canal, in the vicinity of Stn 2 -> Stn 1, ~50 m.

F-29 Trawl line in Lauritzen Canal, in the vicinity of Stn 2 -> Stn 1, ~50 m.

F-31 Trawl line in Santa Fe Channel, over Stns. 5, 6, ~200 m.

F-32 Trawl line in Richmond Harbor Channel, over Stn 8, ~200 m.

Overlying water samples:

S2AHO-001-004 Water column samples at Lauritzen Canal mussel collection site, N ferry rudder, within ~1 m of stn.; ~0.5 below surface.

S2AHO-005-008 Water column samples at Santa Fe Channel Boathouse, 2nd slip, within ~1 m of stn.; ~0.5 below surface.

S2AHO-009
-0010
-0012 Water column samples at Richmond Harbor Channel, Buoy red nun #16, within ~1 m of stn.; ~0.5 below surface.

APPENDIX 6-3. Conversion of "F" sample numbers for water, mussel and trawl samples in February, 1992 to standard numbering protocol.

The temporary designations (F- numbers) by David Young assigned in the field for mussel and trawl samples are converted to the original scheme.

F-1(LC) = S2ADT20_ Lauritzen Channel; submerged rope attached to outer piling, ~15m N of N end ferry rudder, ~2m off bottom; 02/06/92@09:30

F-2(LC) = S2ADT21_ Lauritzen Channel; 2nd trawl, ~5m off line of outer pilings; proceeding N ~50m; 02/06/92@09:54

F-3(LC) = S2ADT22_ Lauritzen Channel; 3rd trawl, same area; 02/06/92@10:10

F-4(SFCB) = S2ADT50_ Santa Fe Ch.; contents of crabpot "C" off boathouse mussel station.; 02/06/92@10:58

(F-5 -> F-10, ALL FROM 1ST TRAWL):
F-5(SFCH) = S2ADT30_
F-6(SFCH) = S2ADT30_
F-7(SFCH) = S2ADT30_
F-8(SFCH) = S2ADT30_
F-9(SFCH) = S2ADT30_
F-10(SFCH) = S2ADT30_
(F-11 -> F-14, ALL FROM 2ND TRAWL):
F-11(SFCH) = S2ADT32_
F-12(SFCH) = S2ADT32_
F-13(SFCH) = S2ADT32_
F-14(SFCH) = S2ADT32_
(F-15 -> F-18, ALL FROM SAME TRAWL):
F-15(RCH) = S2ADT40_
F-16(RCH) = S2ADT40_
F-17(RCH) = S2ADT40_
F-18(RCH) = S2ADT40_
(F-19, 20 FROM SAME TRAWL):
F-19(LC) = S2ADT23_
F-20(LC) = S2ADT23_
F-21(SFCP) = S2ADT33_
F-22(RCH) = S2ADT44_
F-23(LC) = S2ADT24_
F-24(LC) = S2ADT51_
F-25(LC) = S2ADT53_
F-26(LC) = S2ADT51_
F-27(LC) = S2ADT25_
F-28(LC) = S2ADT26_
F-29(LC) = S2ADT27_
F-30(SFCB) = S2ADT50_
F-31(SFCH) = S2ADT34_

APPENDIX 6-3 (cont'd.). Conversion of "F" sample numbers for water, mussel and trawl samples in February, 1992 to standard numbering protocol.

F-32(RCH) = S2ADT45_
F-33(SFCB) = S2ADT50_
F-34(RCH) = S2ADT46_
F-35(RCH) = S2ADT52_

Overlying water samples were taken on 2/7/92 at the previous mussel stations (S1CHT5__):

Lauritzen Channel Ferry: (@ North end rudder)

S2AHO001
S2AHO002
S2AHO003
S2AHO004 (BLANK)

Santa Fe Channel: (@ Boathouse dock)

S2AHO005
S2AHO006 (BLANK)
S2AHO007
S2AHO008

Richmond Harbor Channel: (@ Buoy 16)

S2AHO009
S2AHO010
S2AHO011 (BLANK)
S2AHO012

****
LC = LAURITZEN Channel TRAWLS
SFCH = SAN FRANCISCO CHANNEL TRAWLS
SFCB = SAN FRANCISCO BOATHOUSE
SFCP = SAN FRANCISCO BOATHOUSE CRABPOT
RCH = RICHMOND CHANNEL TRAWLS

APPENDIX 6-4. Detailed field sampling procedures.

The following data was collected and recorded in the field log at each station:

1) Station name and number - Chain-of-custody numbers
2) Station coordinates, dead reckoning landmarks, Loran C coordinates if available
3) Weather conditions, sea state, etc.
4) Time on station
5) Time off station
6) Time start grab
7) Time to finish grab
8) Station depth
9) Grab depth
10) Wash out (yes, no, comments)
11) Color, odor, unusual contents, general remarks on character of sediment
12) Temperature of sediment, overlying water, salinity of surface water, E_h of sediment at 1 & 5 cm depth
13) Presence of birds or marine mammals in the sampling area, notes on behavior (i.e., feeding, etc.)

Sequence of procedures:

1) Record field data 1-4 above -
2) Winch operator deploys van Veen grab with assistance of 1 grab handler - record station depth and time of grab -
3) Grab retrieved on-board by 2 grab handlers, placed on grab stand -
4) Open flaps, accept or reject grab on basis of contents depth -
5) Insert E_h probe and thermometer to 1 cm depth, wiggle probe, read mv at ~30 seconds, temperature when stable; push probe and thermometer down to ~5 cm, wiggle, read mv at ~30 seconds, temperature when stable - remove probe and thermometer and rinse with deionized water -

6.1) Insert appropriate cores for given station, i.e., 1 ea. 4 cm glass corer, 1 ea 4 cm plastic corer, 2 ea. 7.6 cm glass corers, 3 ea 7.6 cm plastic corers, all to 10 cm depth., and 1 ea. modified syringe for AVS, to depth of device (~5 cm).

6.2) "4 cm" glass corer was fabricated of borosilicate glass tubing, has an actual i.d. of ~4.1 cm, o.d. of ~4.45 cm, and was used to take the sample for **sediment chemistry**, including TOC [S1CBSxxx, later subdivided at the laboratory to include S1CASxxx];

6.3) The "4 cm" plastic corer was fabricated of ~1 mm wall translucent plastic pipe with an i.d. of 3.6 cm and o.d. of ~3.85 cm, and was used to take the sample for **sediment grain size** [S1CJSxxx];

6.4) The "7.6 cm" glass corers were fabricated of borosilicate glass tubing, have an actual i.d. of ~7.6 cm and o.d. of ~8.0 cm., and were used to take the sample for **Laboratory sediment bioaccumulation** [S1CCSxxx] and **sediment toxicity** [S1CESxxx];

6.5) The "7.6 cm" plastic corers were fabricated from NSF Schedule 40 PVC pipe, are ~8.0 cm i.d., and were used to take the **Benthic Community Analyses [macrobenthos] sample** [S1CFSxxx];

6.6) The "modified syringe" was fabricated by cutting the distal end off of a 10 cc BD9604 "Plastipak" plastic syringe, and was used to take the sample for **AVS analysis** [S1CISxxx].)

Sediment sample containers were placed in ice chests (cooled with gel-ice) aboard deck, which were off-loaded at the dock after sealing with glass filament reinforced tape and chain-of-custody seals. One member of the sampling crew was detailed to deliver the sealed containers to the local UPS office each afternoon to transport the day's samples to the Newport EPA laboratory by overnight delivery. These containers were received, logged in and placed in tempera-

ture-controlled rooms at 4°C, and custody of individual samples or sample sets was transferred to appropriate individual investigators upon demand to the sample custodian (see Appendix 6-2 for Chain-of-custody procedures in the field and the laboratory).

7) Insert AVS sampler beside corers, push barrel into sediment ~10 cm, to second "click" of seal, while holding plunger still; remove AVS corer, invert, wipe clean with towel, cover exposed end with plastic wrap and secure with rubber band. Place apparatus in zip-loc bag, seal and label, place in cooler with dry ice.
8) Insert gloved hand to bottom of corer tubes, remove corer from grab, insert extruder in bottom of corer, expel overlying water if any, and extrude 10 cm of core into sample container, bucket for benthos; after all corers have been removed, dump residual grab contents into sieve to obtain incidental macrobenthos.
9) Take and record miscellaneous data, i.e., items 6-12 (salinity, odor of sample, color, texture, anomalous contents, etc.) -
10) Provide plastic cores to sieving station for macrobenthos, which are washed through the sieve with running seawater, picked and contents placed in labeled jars, preserved with formalin. Jars are gently shaken to distribute formalin. Cores are rinsed with seawater.
11) Dump and wash residual of grab into square coarse sieve to obtain incidental infauna. Rinse grab with seawater. Rinse glass cores with ethanol, allow to dry by evaporation.
12) Record time to finish processing of grab, and start of next grab cycle.
13) Record all Chain-of-custody information for samples taken at this station.
14) If vessel is not anchored, confirm location before dropping grab for start of next cycle.

APPENDIX 6-5. Procedure for cleanup of tissue and sediment samples.

This procedure is intended to remove biogenic material from sample extracts that have been previously cleaned up using the Bond Elut technique. The eluant, 50% methylene chloride/40% hexane/10% isooctane will elute the PCBs, PAHs, DDT, dieldrin, and most other organochlorine pesticides while leaving the biogenic compounds on the column. Samples should be in isooctane and should be 1 mL or less in volume.

Materials

EM Science Silica Gel 60 (70-230 mesh) or equivalent, activated 130°C for 8h
Baked-out (325°C, 8 h) Pasteur pipets
solvent rinsed glass wool
50% methylene chloride/40% hexane/10% isooctane, Omnisolv grade or better
Champagne funnel

Procedure

1. Place a small piece of glass wool into a Pasteur pipet, pushing it down until it reaches the point where the pipet begins to narrow. Using the champagne funnel add 4 cm of silica gel to the column. Tap the column lightly so that the top of the silica gel layer is level.

2. Rinse the column with 4 mL of hexane. Discard the rinse.

3. Place an 8 mL vial (with teflon septa cap) under the end of the column to collect the cleaned up sample extract.

4. Apply the sample extract to the column being careful not to disturb the surface of the silica gel. After the sample has moved completely onto the column rinse the sample vial three times with the 50% methylene chloride/40% hexane/10% isooctane solvent. Apply each rinse to the column.

5. Apply 6 mL of 50% methylene chloride/40% hexane/10% isooctane to the column.

6. After the flow from the column has stopped cap the vial and store the extract in the refrigerator or concentrate the sample under a flow of nitrogen, spike with the recovery standard, transfer to a GC vial and store the sample in the freezer.

APPENDIX 6-6. Determination of total and bound organics in water.

The procedure is designed to determine the concentrations of freely dissolved, and dissolved organic matter (DOM)-associated concentrations of neutral non-polar organic pollutants. The procedure is modeled after that of Landrum (ES&T, 18, 187-192, 1984). The procedure is based on the observation that freely dissolved compounds will adsorb to a C-18 solid phase surface if they come into contact with it. The DOM-associated compounds will not. If the contact time is too long, previously DOM-associated compounds can adsorb to the C-18 surface because these two forms are in equilibrium.

Equal volumes of a water sample are treated in different ways. One volume is passed through a C-18-containing cartridge, the other is not. Both volumes are then liquid/liquid extracted following addition of surrogate compounds. The extractable compounds in the SPE-treated sample fraction are considered to be bound to the DOM, and the compounds from the other fraction constitute the total extractable concentrations (freely dissolved **plus** DOM-associated).

Due to low and variable recoveries of the freely dissolved compounds from the SPE column, this fraction is not recovered. Instead, the concentrations of freely dissolved compounds are determined by the difference between the total and the bound concentrations. Our and Landrum's experiences with this procedure suggest that the recovery of the freely dissolved portion of the sample from the column is only moderately efficient, often about 50-80%.

MATERIALS:

Eckman Slough IW (filtered)
timer
25 mL graduated pipet
C-18 columns (200 mg, 3 mL, #1210-2025)
filtered bay seawater (building tap)

PREPARATION:

1. Mark an IW collection tube at the 23 mL, 33 mL, 43 mL, and 53 mL levels. This will allow you to determine the volume available for the Total (TOT) and Bound (BND) fractions, and DOC. Once the samples are received, compare their volumes to the marked tube. Equal volumes will go to the TOT and BND fractions, 1 and 2. If under 23 mL is found, no DOC will be taken. Samples between 23 and 33 mL, 33 and 43 mL, and 43 and 53 mL will have 10 mL, 15 mL, and 20 mL samples taken for both fractions. Sample volumes exceeding 53 mL will have 25 mL subsamples taken. DOC samples (if requested) will be 3 mL, and will be placed in narrow tubes.

2. Prepare Bond Elut columns (C-18, 200 mg, 3 mL, #1210-2025) by rinsing them with clean filtered seawater, SW. Mount several columns at once into the Vac Elut box. To ensure contact of all surfaces of the cartridge with the rinsing SW, initially fill each column with SW, attach plastic connector and plastic 50 mL reservoir. Add ~40 mL of SW to each reservoir and pull through the columns at a vacuum of ~11 inches of Hg. When through, continue to pull air for ~1 minute, store in zip-lock bag.

3. Obtain IW blank matrix for spiking: IW from zero-spiked sediment from toxicity testing experiments (PI provided), or filtered Eckman Slough IW for use with field-collected sediment IW. 25 mL and 40 mL of Eckman IW is placed into 40-mL flat-bottomed vials for QAMB and QAMS, respectively. The QAMS vial will be marked at the 25 mL level, spiked (following SOP 4.08, step 2 with vortexing) with target compounds (solution C), equilibrated for one half hour, and given to the IW collectors.

For field samples there will be 10 mL fractions taken for TOT and BND from the QAMB and QAMS (remaining following QAMS-SIP sampling). A 10 mL TOT-only sample will be taken from the QAMS-SIP sample. If requested, take a DOC from the QAMB **only** (others contain solvent!).

Different volumes for these QC samples may be specified for processing with toxicity testing samples.

4. STDMIX-1,-2 are mixtures of solutions (surrogate, recovery [with, -1 or without, -2] target spiking) that are used during a daily session of processing samples. They are prepared to verify the concentrations of these solutions on the days they were used. Because of the dissimilar solvent types used they must first be combined with the universal solvent toluene before they are mutually dissolved in isooctane, the solvent used on the GC/MS. The goal is to determine these concentrations with a minimum of manipulations. STDMIX-1 and STDMIX-2 will be prepared on extraction and blowdown days, respectively. For these IW samples the STDMIXs will use larger volumes than were used in the samples because solvent evaporation needs to be reduced or eliminated! See protocol for instructions.

SAMPLE PROCESSING:

1. Place up to 4, labeled 40-mL vials for the BND fraction (2) into the wooden block in the cylindrical Vac Elut chamber, making sure needles are directed into each vial.

With all openings of the Vac Elut closed with red plugs, engage vacuum pump, close black knob and pump down to the **appropriate vacuum***.

(* Pull the sample through the column at the optimum rate of

12 ml/min. This rate tends to vary with IW viscosity (vacuum of 10 inches of Hg will work with QC samples, higher, 11-13, will be needed for "real" samples. This is why the time is recorded so that vacuum can be adjusted for following samples. For real samples start with 11 inches. If a sample is clearly passing too slow or fast, adjustments can be made in the vacuum, by using the knurled brass knob).

[12 mL/min: 10 mL = 50 sec, 15 mL = 1 min, 15 sec,
20 mL = 1 min,40 sec, 25 mL = 2 min,5 sec]

2. Shake sample vigorously (recording color, turbidity, etc.); using a 25-mL graduated pipet with bulb, draw up the predetermined volume and dispense it into the vial to contain the first fraction (TOT), **immediately** using the same pipet, draw up the volume for the second, BND fraction.

Quickly exchange the plug over the correct vial with a cleaned column, immediately begin transferring the sample to the column (do not use reservoir) and begin timing. Stop timer when sample has entered the C-18 bed and then add ~0.5 ml of filtered seawater from Pasteur pipet to the column as soon as the sample has gone through. Do not allow the column to go dry before adding this rinse. Release the vacuum as soon as the rinse has been pulled through by using the black knob. You need not turn off pump! As soon as vacuum is zero, replace used column with plug and reengage vacuum with black knob. Record passage time on benchsheet margin. Dispense 3 mL DOC sample using the same pipet.

Between sample groups the Vac Elut manifold is cleaned by taking out the wooden block and replacing it with the standard manifold in the rinse position. With the vacuum on, use the tip of a glass syringe barrel as a wick, and direct a stream of methanol from a squirt bottle into the Luer fitting of the needles that were used. Attach the barrel and direct approximately 1/2 mL of methanol into it. When the interior of each needle is rinsed remove the lid and swab the exterior of the needle with a methanol-wetted Kimwipe.

SURROGATE SPIKING/EXTRACTION/BLOWDOWN:

1. Add surrogate solution (water miscible, standard A) to the vials containing the TOT and BND fractions, and the QC samples following SOP 4.08 step 3 with 15 second vortexing followed by a half hour equilibration.

2. Extract the samples, reduce the extract volume, and add the recovery standard B to the specified final combined volume (25 or 50 μL) following SOP 4.08 steps 4-9. For example, if final

combined volume is 25 μL and the volume of standard B that is needed is 15 μL, initially reduce the volume to below 25 μL, add B, mix and transfer to gc vial insert or conical gc vial.

APPENDIX 6-7. Homogenization of bivalve tissue using liquid nitrogen.

This procedure is designed for the rapid homogenization of tissue samples. It is appropriate for use with samples of approximately 5 grams or less. Larger samples should be done in 5 gram or smaller pieces and combined. As of Dec. 1991 it has been demonstrated on clams and mussels (shucked) but should do well on other tissues. The first time this SOP is followed it should be done with someone experienced in the procedure.

MATERIALS AND EQUIPMENT

top-loading balance (3-decimal places)
spatula
mortar and pestle(s)
400 ml beaker
foam cooler, small
liquid nitrogen (coordinate with Bruce Boese)
glass vials (40 ml or appropriate size)
poly weigh boats
acetone
gloves: latex or vinyl are fine, one heavy glove is nice
paper towels

1. Cover clean mortar and pestle(s) with aluminum foil and place in a freezer to cool. Thaw specimens if frozen.

2. Wipe dry and weigh each bivalve to three decimal places on a tared weighing boat.

3. Rinse a scalpel with acetone in the hood, cut the adductor muscles and pry the shell open. Remove byssal threads if present and discard, tease mantle, gonads and all other tissues from the shell. Holding the tissue in the shell, drain off and discard the pallial fluid. Scrape the dissected tissue into a tared weighing boat and record the weight to 3 decimal places. It will likely be easier to dissect half of the animal at a time from the shell, cutting the two halves apart near the shell hinge.

4. Open the valve and direct a stream of liquid nitrogen into a 400 ml beaker placed inside a foam cooler. Place a stainless steel spatula in the beaker to chill it. Half fill a pre-chilled mortar with liquid nitrogen from the beaker, place the pestle in the mortar and allow them both to become completely chilled, adding more liquid nitrogen if necessary. After the mortar and pestle are thoroughly chilled, add additional liquid nitrogen so that a pool of nitrogen is present in the mortar.

5. Drop the dissected tissue into the <u>middle</u> of the pool of nitrogen, wait a few seconds, and press the pestle straight down on the tissue. This should flatten out the freezing tissue.

Grind the sample with the mortar until it is roughly uniform in tiny pieces (like sand) and fully homogenized. Add more nitrogen if necessary until the process is complete. Note: if the tissue sample is a large one, cut it into chunks that are approximately 5 grams or less in size and homogenize one chunk at a time.

6. Scrape the homogenized sample into pre-labeled, pre-weighed (to 3 decimal places), acetone rinsed vials with the chilled spatula. Place the cap back on the vial but DO NOT TIGHTEN THE CAP UNTIL THE SAMPLE HAS PARTIALLY THAWED. IF A VIAL AT CRYOGENIC TEMPERATURES IS SEALED AND ALLOWED TO WARM, IT CAN EXPLODE -SENDING GLASS SHARDS ACROSS THE ROOM AND POSSIBLY RESULTING IN INJURY!

7. Wipe out the mortar and wipe off the pestle with a clean paper towel or Kim-wipe. It is unpractical to use a newly cleaned and chilled mortar and pestle with each sample so thought must be given to an appropriate sample homogenization order and grouping. Little cross-contamination is likely if the analyte concentrations in the samples are not vastly different and the mortar and pestle are kept frozen so that the frozen sample particles are easily wiped out between homogenizations.

8. After the tissue has been allowed to partially thaw, seal and weigh the vial to 3 decimal places. Obtain the sample weight by difference. Place the sample in the freezer pending further analysis.

SAFETY ASPECTS

Liquid nitrogen can be safely handled if care is taken to avoid direct exposure, the nitrogen containing beaker is only briefly handled with fingertips and light gloves, and super-cooled instruments such as the mortar are only briefly and lightly held or heavy gloves used. Most dangerous is tightening the cap on a sample immediately after tissue transfer, creating a bomb. Acetone exposure should be avoided.

APPENDIX 6-8. Gas chromatography - mass spectrometry.

This procedure describes the standard procedure for analyzing environmental sample extracts by GC/MS in the selected-ion monitoring mode.

Materials

HP 5890 II Gas Chromatograph
HP 5970B Mass Spectrometer
HP 1000 computer with RTE-A Rev. F and AQUARIUS software

Method

A. Daily system checks

1. Replace the septum if necessary.

2. Check the inlet, oven and interface temperatures.

3. Check the MS vacuum. The vacuum must be below 5.0×10^{-5} prior to operation of the mass spectrometer. Operation of the mass spectrometer at higher pressures may result in damage to the electron multiplier.

3. Perform autotune, using PFTBA as the calibrant, following the manufacturers instructions. Use the FORCED TUNE option. A hard copy of the autotune report and the spectrum should be filed for future reference.

4. Check for the presence of leaks using the MS. Use Autotune conditions and scan m/z's 18, 28, 32 with the PFTBA vial closed. The m/z 32 peak should be less than 1000 abundance units prior to beginning an analysis. File a hard copy of the leak check results with the autotune report.

5. Inject 3 μL of isooctane to check syringe cleanliness and to check for carryover. Clean syringe or elevate column temperature to the highest operating temperature for a brief period (<30 min) if peaks are observed for this injection.

6. Inject the lowest calibration standard for the analysis to be performed. Check the peak shapes and sensitivity. Loss of sensitivity for the low standard may require additional manual tuning, cleanup of the insert, or installation of a new column. If DDT breakdown is not a concern a highly contaminated sample extract or high level standard may be injected in an attempt to remove active sites in the system.

7. Inject the highest calibration standard to be used for the analysis. Check the peak shapes, especially checking for peaks which may have saturated the detector. Check the resolution

between phenanthrene and anthracene or a similar pair of compounds whose peak maximums are 8 scans or less apart.

8. If DDT is to be analyzed, inject 1 μL of 1,000 ng/mL 4,4'-DDT in isooctane to check for DDT breakdown. If the % breakdown defined in equation 1 is greater than 20%, clean or replace the inlet insert or remove a portion of the front end of the GC column.

$$\% \text{ Breakdown DDT} = \frac{(\text{Area DDD} + \text{Area DDE} * 100)}{(\text{Area DDD} + \text{Area DDE} + \text{Area DDT})} \quad (1)$$

B. Sample analysis

1. Create a GC/MS sample sequence using BEDIT. The sequence name should follow the convention SE_---.

2. The following general order should be used when analyzing samples:

a. Initial Calibration Standards (low to high with a solvent blank following the high level standard)
b. SRM or "traceable" solution to assess calibration accuracy
c. DDT degradation check (if necessary)
d. 5 samples
e. calibration standard
f. solvent blank
g. 5 samples
h. repeat d-f until all samples are analyzed

To reduce the possible effect of carryover it may be necessary to run a solvent blank after every standard and/or group blanks (QAB, QAMB) and low level samples together. A DDT degradation check should be included after every 10 samples if DDT is a target analyte.

C. Data Reduction

1. All SIM data files should be background subtracted using the SFX command.

2. Quantitation and peak detection should be carried out by using the AQUARIUS software routines on the HP 1000. Make sure that the ID files and calibration files are correct (RT's, ISTD concentrations) prior to using the QT command. WARNING: ALL MANUAL INTEGRATIONS ARE LOST IF THE DATA FILES HAVE TO BE RE-QUANTED FOR ANY REASON. THE CORRECT ID AND CAL FILES MUST BE USED THE FIRST TIME. DO NOT CHANGE THE ID OR CAL FILE PRIOR TO PRINTING OF THE CLP FORMS. THIS WILL CAUSE A FAILURE IN CLP FORM GENERATION.

2A. An alternative calibration may be obtained by using an EXCEL spreadsheet macro. The spreadsheet allows for the use of a power regression and may be easier when the list of compounds is less than 15. Use of the EXCEL calibration requires the result be calculated in the final FOCCALC spreadsheet not the quant reports. In this case the quant reports and CLP forms are to be regarded as reporting the peak areas, RT's , and ion ratios only, not the actual analyte concentrations.

3. Regardless of the calibration method chosen analytes are quantitated using the method of internal standards and the results are calculated from the peak areas for the internal standard and analyte, the response factor (or slope, intercept) from the calibration curve, and the amount of internal standard spiked into the sample (on a sample concentration basis).

4. The calibration standards run with the samples should be used for calibration.

D. Peak Identification

1. All peaks obtained from a SIM analysis must elute within 0.01 RRT units of the expected RRT where RRT = RT target/ RT surrogate IS. Any peak obtained by the data system but failing this criterion is reported as "failed RRT".

2. The characteristic (or qualitative) masses must have ion ratios within 20% of the mean ion ratios obtained from the calibration standards during the sample set analysis. Alternatively, the "Q" value must be greater than 80 or greater than that obtained from the calibration standards. Caution should be used when using the "Q" as this value is not updated following manual integration. Compound results may be reported with different qualifying codes depending upon whether 1 or more of the qualifying ratios were acceptable.

3. The peak area of the quant ion must be greater than three times the signal to noise. The peak area corresponding to three times the signal to noise may be estimated as described in Appendix A or by use of the RPN SNR (signal to noise ratio) function available in the RTE-A operating system.

E. Quality Assurance

1. All peaks in the samples and standards should be checked to verify proper integration. Peaks improperly integrated will be manually integrated.

2. The results for the procedural (QAB) and matrix (QAMB) blanks will be checked for the presence of analytes of interfering peaks.

3. The results for SRM or "traceable" solutions will be checked to assess calibration accuracy. The results for these solutions should normally fall between 80 and 120 % of their true values.

4. The entry of all hand entered data will be verified.

5. The carryover from high level standards will be monitored.

6. The recoveries of surrogate standards in the sample must be greater than 20%.

7. The range of surrogate recoveries in a sample must be less than 25%; a disproportionately high or low recoveries indicative of enrichment discrimination and may mean the use of one or more of the surrogates for purposes of quantitation has been compromised.

8. If any analyte concentration exceeds the upper limit of the calibration range the analysis must be repeated after appropriate dilution of the sample.

9. Documentation of autotune, calibration, surrogate recoveries," q" values (or ion ratios) and RRTs must accompany each set of analyses.

F. Data Backup

1. All data files will be backed up on tape as soon as practical after an analysis is completed.

2. Backup tapes of quant files will be made in time intervals that minimize potential data loss. Backups will be made weekly depending on instrument usage.

G. Instrument Maintenance

1. Maintenance schedules are given in the instrument manuals.

H. Troubleshooting

1. Refer to the GC/MS instrument log book, data system log book, instrument manuals, or senior chemistry personnel when troubleshooting problems.

I. Signal to noise ratio determination

The signal to noise ratio is measured in a matrix blank to obtain a minimum peak area for use in estimating a method detection limit. The instrumental detection limit may be obtained by measuring the signal to noise in a solvent blank.

1. Use the SNR command to determine the noise over a retention

time range approximately equal to the peak width. The noise should be measured in a region that is close to the peak of interest (usually within 0.5 minutes) and that does not contain any obvious peaks.

2. Convert the noise reported by SNR (in abundance units) to a minimum peak height corresponding to 3 S/N.

A. Adjust the noise (baseline standard deviation) for the time interval over which the noise was estimated using equation 1 to determine the RMS noise.

$$RMS = [(SNR\ noise) * 5] / d \qquad (1)$$

Table I. Values of d[1]

number of scans	d
7	2.704
8	2.847
9	2.970
10	3.078
11	3.173
12	3.258
13	3.336
14	3.407
15	3.472

B. The minimum peak height equals 3 * RMS noise.

3. Convert the minimum height to a minimum area measurement.

A. There are a number of ways to do this one of which utilizes the INT command to report both the peak height and the peak area of compounds in the lowest calibration standard.

4. Estimate the method detection limits by calculating the concentrations that would result from targets present at the minimum peak areas in the QAMB. Use the reported ISTD areas for calculation.

5. The procedure file SFE_03 utilizes the above method to estimate minimum peak areas. The following G-global values may be passed to SFE_03; 1G = data name, 2G = compound name, 3g = ion#, 4G = SIM group #, 5G = noise start time, 6G = noise stop time, 7G = d, 8G = area/height ratio.

References:

[1] Skoog, D. A. 1976. Fundamentals of Analytical Chemistry. 3rd Edition. Holt, Rhinehart & Winston, NY. p 59-60.

APPENDIX 7-1. Field data measurements, United Heckathorn/Lauritzen Channel.

SAMPLE NUMBER	STA #	GRA #	SALIN PPT @SED	SED. TEMP @1CM	SED. TEMP @5CM	SED. mvEH @1CM	SED. mvEH @5CM	SED. ODOR	SED. COLOR	DEPTH SED.IN GRAB,CM	OVLYG DEPTH FEET	SAMPL DATE	EH STDZ mv A	EH STDZ mv B	MISC. COMMENTS
S1C__-011	1	1		19.0	19.0	-56	-129	ANOXIC	BLACK	~10	23	10/09/91	214	276	BLACK IMMED BELOW SURFACE
S1C__-012	1	2	30.0	19.0	19.0	-197	-194	NO DATA	NO DATA	~10	23	"	214	276	31 PPT @ SURF; WORM TUBES
S1C__-013	1	3		19.0	19.0	-93	-173	NO DATA	NO DATA	10	23	"	214	276	CHEM TRIPLICATE
S1C__-014	1	4		20.0	19.0	-169	-173	NO DATA	NO DATA	18	22	"	214	276	LIVE MACOMA NASUTA
S1C__-015	1	5		19.0	19.0	-133	-150	OILY	OILSHEEN	18	23	"	214	276	BIG WORM OUTSIDE OF CORES
S1C__-016	1	6		19.0	19.0	-140	-140	NO DATA	NO DATA	>10	20	"	214	276	WORM TUBES @ SURFACE
S1C__-017	1	7		19.0	19.0	-129	-144	NO DATA	NO DATA	18	20	"	214	276	BEST GRAB OF THE DAY
S1C__-018	1	8		19.0	19.0	-51	-151	SL OILY	GRY GRN	18	20	"	214	276	WORM TUBES FIL. ALG, AMPHIP.@ SURF
S1C__-021	2	1		18.5	18.5	85	-105	NO DATA	FIL ALG	>10	20	10/10/91	207	271	SM WRM TUB, ALG, LG WRMS WSURF 30PPT
S1C__-022	2	2		18.5	18.5	-110	-135	NO DATA	NO DATA	>10	20	"	207	271	WORMS, CLAMS, MUSSELS @ SURF
S1C__-023	2	3		18.5	18.5	22	-143	NO DATA	NO DATA	18	20	"	207	271	WORMS, CLAMS, MUSSELS @ SURF
S1C__-024	2	4		18.5	18.5	3	-107	NO DATA	NO DATA	18	20	"	207	271	WORMS, CLAMS, MUSSELS; LOTS @ SURF
S1C__-025	2	5	30.0	18.5	18.5	-104	-135	NO DATA	GRY BRN	>10	20	"	207	271	WORM TUBES, FIL. ALG.; SURF 30 PPT
S1C__-031	3	1		19.0	19.0	-68	-110	~H2S	NO DATA	>10	34	10/09/91	214	276	WORM TUBES; ~H2S@BOTTOM, OILY SHEEN
S1C__-032	3	2		19.0	19.0	-113	-135	NO DATA	GRY GRN	18	33	"	214	276	"PUDDING" CONSISTENCY
S1C__-033	3	3		19.0	19.0	-141	-154	NO DATA	GRY GRN	>10	33	"	214	276	GRAVELY TOWARDS BOTTOM
S1C__-034	3	4		19.0	19.0	-132	-150	NO DATA	GRY GRN	>10	33	"	214	276	OILY SHEEN @ SURFACE
S1C__-035	3	5		19.0	19.0	-153	-176	NO DATA	GRY GRN	18	34	"	214	276	"SANDY PUDDING"; OILY SHEEN
S1C__-041	4	1		18.0	18.0	100	-99	NO H2S	BRN GRY	~15	38	"	214	276	NOT ANOXIC; 30PPT @WSURF
S1C__-042	4	2		18.0	18.0	-51	-126	NO DATA	NO DATA	>10	37	"	214	276	
S1C__-043	4	3	30.0	18.0	18.0	-115	-126	NO DATA	GRY	18	38	"	214	276	NOT ANOXIC; LIVE MACOMA; 30PPT SURF
S1C__-044	4	4		19.0	19.0	-131	-133	NO DATA	GRY	18	38	"	214	276	GRY TO BOTTOM
S1C__-045	4	5	30.0	19.0	19.0	147	124	NO DATA	NO DATA	>10	37	"	214	276	(Eh PROB. NEG., RECORDING ERROR)
S1C__-051	5	1		19.0	19.0	-143	-161	NO DATA	NO DATA	>10	35	10/08/91	232	294	
S1C__-052	5	2		19.0	18.5	-100	-137	~H2S	GRY BRN	>10	34	"	232	294	SM CLAMS, WORMS
S1C__-053	5	3	31.0	19.0	19.0	-138	-155	NO DATA	NO DATA	>10	33	"	232	294	LG WRMS, AMPHIPODS; 31PPT@SURF
S1C__-054	5	4		18.5	18.5	-142	-156	NO H2S	GRY	18	34	"	232	294	NOT ANOXIC TO BOTTOM; LIVE MACOMA
S1C__-055	5	5		18.5	18.5	-101	-101	NO DATA	NO DATA	>10	~34	"	232	294	
S1C__-061	6	1		18.0	18.0	-37	-104	PETR.CREO	GRY BRN	8-9	34	"	232	294	OILY SHEEN, 2"WD SPLNT,SM WRM TUBES
S1C__-062	6	2		19.0	19.0	-87	-140	H2S,CREOSO	IL SHEEN	18	34	"	232	294	ANOXIC@1-3MM,SPLINTERS,PETROLEUM
S1C__-063	6	3		19.0	19.0	-139	-160	PETR.CREO	IL SHEEN	~10	33	"	232	294	LARGE SPLINTERS, OILY SHEEN
S1C__-064	6	4		19.0	19.0	-108	-130	H2S	BRN	~10	33	"	232	294	SM WORM TUBES, ANOXIC ~MM DOWN
S1C__-065	6	5		19.0	19.0	-52	-102	H2Sstrong		18	30	"	232	294	H2S VERY STRONG, PETROLEUM
S1C__-071	7	1		18.0	18.5	-54	-103	NO DATA	GRY	>10	27	"	232	294	ULVA, MANY SM WORM TUBES, SHELLS
S1C__-072	7	2		18.5	18.5	162	466	NO DATA	GRY GRN	~18	27	"	232	294	(FAULTY CONN., SPURIOUS EH DATA)
S1C__-073	7	3		18.5	18.5	90	-53	NO DATA	GRY GRN	~15	28	"	232	294	SED CHARCOAL BLACK BELOW SURF
S1C__-074	7	4		18.0	18.0	-117	-134	NO DATA	NO DATA	~10	31	"	232	294	CORES FROM MIDDLE;CRAB,ANENOME
S1C__-075	7	5		18.0	18.0	-53	-72	NO DATA	NO DATA	~10	~30	"	232	294	
S1C__-081	8	1		19.0	19.0	-125	-130	NO DATA	GRY GRN	10	36	10/07/91	235	293	V SMALL WORM TUBES, GRAVEL
S1C__-082	8	2		18.5	18.5	-123	-140	NO DATA	GRY GRN	10	36	"	235	293	V SM THIN RED WORMS ON SIEVE
S1C__-083	8	3		18.5	18.5	-125	-141	PETR SHN	GRY GRN	10	36	"	235	293	GRAVEL BOTTM OF GRAB
S1C__-084	8	4		18.5	18.5	-128	-94	NO DATA	GRY GRN	10	36	"	235	293	EH RECHECKED
S1C__-085	8	5		18.5	18.5	-128	-150	NO DATA	GRY GRN	10	36	"	235	293	GRAB ~25' FARTHER SOUTH

APPENDIX 7-1 (cont'd.). Field data measurements, United Heckathorn/Lauritzen Channel..

SAMPLE NUMBER	STA #	GRA #	SALIN PPT @SED	SED. TEMP @1CM	SED. TEMP @5CM	SED. mvEH @1CM	SED. mvEH @5CM	SED. ODOR	SED. COLOR	DEPTH SED.IN GRAB,CM	OVLYG DEPTH FEET	SAMPL DATE	EH STDZ mv A	EH STDZ mv B	MISC. COMMENTS
S1C__-091	9	1		18.0	18.0	141	-21	NO DATA	FIL ALG	9.5	10	10/07/91	235	293	LOTS OF SHELL
S1C__092	9	2		18.0	18.0	-23	-119	NO DATA	GRYGRN	10	10	10/07/91	235	293	CRABS ON SURFACE
S1C__-093	9	3		18.0	18.0	63	-71	NO DATA	NO DATA	10	10	10/07/91	235	293	
S1C__-094	9	4	30.0	18.0	NO DA	144	-55	NO DATA	NO DATA	10	12	10/08/91	232	294	STN REOCCUPIED
S1C__-095	9	5	30.0	18.0	18.0	96	57	NO DATA	NO DATA	10	12	10/08/91	232	294	STN REOCCUPIED; 30PPT @ WSURF
S1C__-009009	1			18.0	NO DA	43	-44	NO ODOR	GRY	18	41	10/07/91	STN REJ		DEPAUPERATE
S1C__-110	10														STATION NOT SAMPLED
S1C__-111	11														STATION NOT SAMPLED
S1C__-112	12														STATION NOT SAMPLED
S1C__-113	13			NO DANO DATA				NO DATA	GRY GRN	>10	34	10/10/91			
S1C__-114	14			NO DANO DATA				NO DATA	NO DATA	>10	34	10/10/91			
S1C__-115	15			NO DANO DATA				NO DATA	GRY	20	NO DAT	10/10/91			2 CM SED SQUISHED THRU SCREEN
S1C__-116	16			NO DANO DATA				NO ODOR	GRY	18	38	10/10/91			
S1C__-117	17			NO DANO DATA				NO ODOR	GRYBRNGRN	18	NO DAT	10/10/91			
S1C__-118	18			NO DANO DATA				NO DATA	NO DATA	21	40	10/10/91			3 CM SQUISHED THRU SCREEN
S1C__-119	19			NO DANO DATA				NO DATA	GRY	>10	NO DAT	10/09/91			SURF 18C, pH8.05; SURF 30 PPT
S1C__-120	20			NO DANO DATA				NO ODOR	GRY	18	32	10/09/91			SED GRY TO BOTTOM; SURF 30PPT
S1C__-121	21			NO DANO DATA				NO DATA	SANDCLAY	>10	30	10/10/91			3CM SND OVRGRYBRN CLY BRTL STR AMPH
S1C__-122	22			NO DANO DATA				NO DATA	NO DATA	INSUFF	20	10/10/91			BENTHOS INSUFF
S1C__-123	23			NO DANO DATA				NO DATA	NO DATA	>10	30	10/10/91			MEASURED BENTHOS CORES
S1CKS-001	9	6	30.0					NO DATA	NO DATA	10	12	10/08/91			BIOASSAY
S1CKS-011	1							NO DATA	NO DATA		~23	10/09/91			BIOASSAY
S1CKS-021	2			~18.5	~18.5			NO DATA	NO DATA	>10	~20	10/10/91			BIOASSAY
S1CKS-031	3							NO DATA	NO DATA	~10	33	10/09/91			UNEVEN FILL
S1CKS-041	4							NO DATA	NO DATA	<10	38	10/09/91			UNEVEN FILL; BIOASSAY
S1CKS-051	5			NO DANO DATA				NO DATA	NO DATA	L~10/R~	34	10/08/91			GRAB IMPACTING ON SLOPE
S1CKS-061	6			NO DANO DATA				NO DATA	NO DATA	>10	30	10/08/91			SPLINTERS, BIOASSAY
S1CKS-071	7			NO DANO DATA				NO DATA	NO DATA	>10	38	10/08/91			BIOASSAY, INSUFF VOL FOR REG SAMPLE
S1CKS-081	8			NO DANO DATA				NO DATA	NO DATA	10	~36	10/08/91			BIOASSAY
S1CBS-010	1	3		19.0	19.0	-93	-173	NO DATA	NO DATA	10	23	10/09/91	214	276	CHEM TRIPLICATE
S1CBS-013	1	3		19.0	19.0	-93	-173	NO DATA	NO DATA	10	23	10/09/91	214	276	CHEM TRIPLICATE
S1CBS-019	1	3		19.0	19.0	-93	-173	NO DATA	NO DATA	9	23	10/09/91	214	276	CHEM TRIPLICATE
S1CBS-050	5	1		19.0	19.0	-143	-161	NO DATA	NO DATA	>10	35	10/08/91	232	294	CHEM TRIPLICATE
S1CBS-051	5	1		19.0	19.0	-143	-161	NO DATA	NO DATA	>10	35	10/08/91	232	294	CHEM TRIPLICATE
S1CBS-059	5	1		19.0	19.0	-143	-161	NO DATA	NO DATA	>10	35	10/08/91	232	294	CHEM TRIPLICATE
S1CBS-070	7	1		18.0	18.5	-54	-103	ALG	GRY	>10	27	10/08/91	232	294	CHEM TRIPLICATE
S1CBS-071	7	1		18.0	18.5	-54	-103	ALG	GRY	>10	27	10/08/91	232	294	CHEM TRIPLICATE
S1CBS-079	7	1		18.0	18.5	-54	-103	ALG	GRY	>10	27	10/08/91	232	294	CHEM TRIPLICATE
S1CCS-010	1			19.0	19.0	NO DANO	DANO	DATA	NO DATA	<10	20	10/09/91			BIOACCUMULATION TEST

APPENDIX 7-1 (cont'd.). Field data measurements, United Heckathorn/Lauritzen Channel.

SAMPLE NUMBER	STA #	TOW #	OVLYG DEPTH FEET	SAMPL DATE	MISC. COMMENTS
S1C__-2XX200	7		10/10/91	TOW N->S,~100M STG FROM BETW STN 1,2	17:54 - 18:01; ~2.5 KNOTS
S1C__-3XX300	5		10/10/91	TOW NW->SE, ACROSS STN 5,6, ~250M	17:39->17:45; ~2.5 KNOTS; 2 M ON BOT
	6		10/10/91	TOW NW->SE, ACROSS STN 5,6, ~250M	17:54->18:01; ~2.5 KNOTS
S1C__-4XX400	1		10/10/91	PRACTICE DRILL: PARALLEL TO SHORE	15:36->~15:40; DIDN'T HIT BOTTOM
	2		10/10/91	TOW N->S, PAR.TO STN 8,~100'OFFSHORE	15:50->15:52 (2 MIN ON BOTTOM)
	3		10/10/91	TOW N->S, PAR.TO STN 8,~100'OFFSHORE	1607->16:12 ON BOTTOM;ON DECK@16:19
	4		10/10/91	TOW N->S, PAR.TO STN 8,~100'OFFSHORE	

APPENDIX 7-1 (cont'd.). Field data measurements, United Heckathorn/Lauritzen Channel.

SAMPLE NUMBER	STA #	SALIN GRA #	PPT @SED	OVLYG DEPTH METERS	SAMPL DATE	SAMPLE TIME PDT	MISC. COMMENTS
S1CHT-501SFC				.1-.5M	10/07/91	12:50	ATTACHED TO STYROFOAM FLOAT
S1CHT-502SFC				.1-.5M	10/07/91	"	ATTACHED TO STYROFOAM FLOAT
S1CHT-503SFC				.1-.5M	10/07/91	"	ATTACHED TO STYROFOAM FLOAT
S1CHT-504SFC				.1-.5M	10/07/91	"	ATTACHED TO STYROFOAM FLOAT
S1CHT-505SFC				.1-.5M	10/07/91	13:20	ATTACHED TO STYROFOAM FLOAT
S1CHO-501SFC		HIGH TIDE		0.5M	10/07/91	13:28	TAKEN WITHIN 1-2 M OF MUSSEL SITE
S1CHT-511LCF				.1-.5M	10/07/91	13:35	NORTH END FYBOAT RUDDER
S1CHT-512LCF				.1-.5M	10/07/91	"	NORTH END FYBOAT RUDDER
S1CHT-513LCF				.1-.5M	10/07/91	"	NORTH END FYBOAT RUDDER
S1CHT-514LCF				.1-.5M	10/07/91	"	NORTH END FYBOAT RUDDER
S1CHT-515LCF				.1-.5M	10/07/91	13:55	NORTH END FYBOAT RUDDER
S1CHO-511LCF		HIGH TIDE		0.5M	10/07/91	14:03	TAKEN WITHIN 1-2 M OF MUSSEL SITE
S1CHT-521RCH				.1-.5M	10/07/91	15:00	ATTACHED TO BUOY 16 ANCHOR LINE
S1CHT-522RCH				.1-.5M	10/07/91	"	ATTACHED TO BUOY 16 ANCHOR LINE
S1CHT-523RCH				.1-.5M	10/07/91	"	ATTACHED TO BUOY 16 ANCHOR LINE
S1CHT-524RCH				.1-.5M	10/07/91	"	ATTACHED TO BUOY 16 ANCHOR LINE
S1CHT-525RCH				.1-.5M	10/07/91	15:13	ATTACHED TO BUOY 16 ANCHOR LINE
S1CHO-521RCH		HIGH TIDE		0.5M	10/07/91	15:16	TAKEN WITHIN 1-2 M OF MUSSEL SITE
S1CHO-502SFC		MID TIDE		0.5M	10/08/91	11:30	TAKEN WITHIN 1-2 M OF MUSSEL SITE
S1CHO-512LCF		MID TIDE		0.5M	10/08/91	11:45	TAKEN WITHIN 1-2 M OF MUSSEL SITE
S1CHO-522RCH		MID TIDE		0.5M	10/08/91	12:18	TAKEN WITHIN 1-2 M OF MUSSEL SITE
S1CHO-503SFC		LOW TIDE		0.5M	10/09/91	07:18	TAKEN WITHIN 1-2 M OF MUSSEL SITE
S1CHO-513LCF		LOW TIDE		0.5M	10/09/91	07:30	TAKEN WITHIN 1-2 M OF MUSSEL SITE
S1CHO-523RCH		LOW TIDE		0.5M	10/09/91	07:50	TAKEN WITHIN 1-2 M OF MUSSEL SITE

APPENDIX 8-1. Field sediment and interstitial water concentrations (sediment = µg/kg dry; interstitial water = ng/l).

SAMPLE # reg,proj,mat,st,grab	STA	GRAB	ASEP	SCREP	TOC: N.A.O. gOC/gdry DOC: mgOC/L	T=total B=bound	2,4'-DDE CONC	CODE	4,4'DDE CONC	CODE	2,4'-DDD CONC	CODE	4,4'-DDD CONC	CODE	2,4'-DDT CONC	CODE	4,4'-DDT CONC	CODE	Σ DDT DDT	dieldrin CONC	CODE
408 ave S1CAS011	1	1	3	1	0.019	0.4	208.6		3213.6		17196.1	DRA	68690.2		975.5		77137.6		167421.5	2679.2	
407 0 1 S1CBI011	1	1	1	1	9.1	T	8.4		101.9		984.3		4142.1		94.2	NC3	279.3		5610.1	630.2	
407 ave 02 S1CBI011R	1	1	3	1		B	1.3		21.7		91.2		340.2		18.4		64.3		537.2	15.3	
408 10 S1CAS012	1	2	1	1	0.0235	0.5	129.8	L	1809.9		7840.5		46026.7		2737.6		31033.5		89577.9	801.5	L
407 0 1 S1CBI012	1	2	1	1	9.4	T	3.3	NC,L	56.6		474.4		2498.3		47.1	NC3	151.8		3231.5	226.6	
407 0 2 S1CBI012	1	2	1	1		B	0.8	L,NC3	9.4		34.0		148.2		7.5	NC3	21.4		221.3	0.0<DL	
408 ave S1CAS010 R	1	3	3	1	0.0237	0.4	58.3		873.6		1945.5		14231.3		1027.9		17811.6		35948.1	201.2	
no interstitial water																					
408 11 S1CAS013	1	3	1	1	0.0242	0.4	56.3	L	857.4		2143.2		13692.5		1179.4		16504.6		34433.4	193.1	L
407 0 1 S1CBI013	1	3	1	1	5.6	T	1.4	L	26.0		237.6		1268.5		53.2		189.4		1776.1	52.0	
407 0 2 S1CBI013	1	3	1	1		B	0.0<DL		3.9	L	12.5		52.5		8.8		34.2		111.8	0.0<DL	
408 10 S1CAS019	1	3	1	1	0.0244	0.4	84.4	NC3,L	915.2		2389.7		15484.1		175.7	L	10321.7		29370.8	153.2	L
407 0 1 S1CBI019	1	3	1	1	5.1	T	1.8	L	20.7		140.1		722.1		7.6		52.6		945.0	47.2	
407 0 2 S1CBI019	1	3	1	1		B	0.0<DL		2.2	L	6.5	L	19.2		0.0<DL		59.2	NC3	87.1	99.5	NC3
408 10 S1CAS014	1	4	1	1	0.0241	0.4	108.3	L	1161.5		3245.7		19216.4		10965.2		31265.4		65962.5	342.2	L
407 0 1 S1CBI014	1	4	1	1	6.4	T	3.0	L	42.0		254.3		1368.7		8.8	NC3	41.5	NC	1718.3	103.4	
407 0 2 S1CBI014	1	4	1	1		B	0.0<DL,L,NC		4.7	NC3,L	12.5		57.5		0.0<DL		0.0<DL		74.7	0.0<DL	
408 10 S1CAS015	1	5	1	1	0.0272	0.4	65.0	L	1023.7		2730.3		16122.8		1298.8		15917.1		37157.6	289.3	NC,L
407 0 1 S1CBI015	1	5	1	1	5.4	T	2.4	L	36.3		278.3		1230.3		0.0<DL		142.4		1689.7	79.4	
407 0 2 S1CBI015	1	5	1	1		B	1.1	L	15.5	L	52.5		208.7		0.0<DL		75.6		353.3	14.7	NC3
408 10 S1CAS016	1	6	1	1	0.0263	0.4	54.0	L	985.1		2509.1		17330.4		186.6	L	26381.1		47446.2	251.6	NC3,L
407 0 1 S1CBI016	1	6	1	1	17.4	T	2.6	L	32.3		119.0		617.4		0.0<DL		41.6		812.9	48.5	
407 0 2 S1CBI016	1	6	1	1		B	0.0<DL		4.4	NC,L	11.1		47.6	NC	0.0<DL		0.0	<DL,L	63.1	0.0<DL	
408 10 S1CAS017	1	7	1	1	0.0247	0.4	72.4	L	967.1		2915.1		19632.0		1391.0		19685.9		44663.5	298.5	L
407 ave 01 S1CBI017R	1	7	3	1	9.7	T	3.0		51.8		210.0		1116.3		27.0		77.2		1485.3	79.3	
407 0 2 S1CBI017	1	7	1	1		B	0.0<DL		5.4	L	14.3		61.1		1.7	NC3,L	6.0	NC,L	88.5	0.0<DL,NC3,	
408 10 S1CAS018	1	8	1	1	0.0218	0.5	138.5	L	2251.4		12163.0		60327.9		2019.9		59514.1	DRA	136414.7	1139.2	
407 0 1 S1CBI018	1	8	1	1	6.7	T	38.6		462.4		3420.0		3760.0		612.1		1868.5		10161.7	634.1	
407 0 2 S1CBI018	1	8	1	1		B	1.5	L	21.6		101.8		446.2		12.0	NC	40.6	NC	623.7	17.0	
408 10 S1CAS021	2	1	1	1	0.0176	0.5	46.0	L	656.1		2556.0		14432.0		133.1	NC3,L	8898.6		26721.8	391.5	L
407 0 1 S1CBI021	2	1	1	1	7.1	T	6.0	L	68.5		527.2		2696.2		253.5		856.4		4407.8	177.5	
407 0 2 S1CBI021	2	1	1	1		B	1.2	NC3,L	10.0		28.6		143.9		6.4	NC3	8.9	NC,L	198.9	0.0	
408 10 S1CAS022	2	2	1	1	0.0185	0.5	42.5	NC3,L	913.5		3077.3		18915.4		798.1	L	26735.4		50482.2	1119.8	NC
407 0 1 S1CBI022	2	2	1	1	6.1	T	5.7	L	70.7		283.6		1505.9		11.5	NC3	38.4		1915.8	256.8	
407 0 2 S1CBI022	2	2	1	1		B	0.0<DL,NC3,		8.5		30.2		140.4		0.0<DL		0.0	<DL,L	179.1	0.0<DL,NC3,	
408 10 S1CAS023	2	3	1	1	0.0205	0.5	53.9	L	812.7		3053.6		19772.1		314.0	NC,L	13250.1		37256.3	477.2	L
407 0 1 S1CBI023	2	3	1	1	5.9	T	3.0	NC3,L	52.0		218.1		1100.9		41.0		144.2		1559.2	151.9	
407 0 2 S1CBI023	2	3	1	1		B	0.0	<DL	4.7	L	14.5		58.0		0.0<DL		0.0<DL,NC,L		77.3	0.0<DL	
408 11 S1CAS024	2	4	1	1	0.0119	0.5	0.0<DL,NC3,		711.0		1430.4		8598.6		163.4	NC,L	30671.2		41574.6	167.5	NC3,L
407 0 1 S1CBI024	2	4	1	1	6.8	T	5.2	L	72.6		281.5		1249.8		0.0<DL		0.0	<DL,L	1609.1	109.9	NC
407 0 2 S1CBI024	2	4	1	1		B	0.0	<DL	10.5		26.0		102.6		0.0<DL		134.1	NC3	273.2	0.0<DL	

APPENDIX 8-1 (cont'd.). Field sediment and interstitial water concentrations.

SAMPLE # req,proj,mat,st,grab	STA	GRAB	ASEP	SCREP	TOC: N.A.O. gOC/gdry DOC: mgOC/L	T=total B=bound	2,4'-DDE CONC	CODE	4,4'DDE CONC	CODE	2,4'-DDD CONC	CODE	4,4'-DDD CONC	CODE	2,4'-DDT CONC	CODE	4,4'-DDT CONC	CODE	Σ DDT DDT	dieldrin CONC	CODE
408 10 S1CAS025	2	5	1	1	0.0208	0.4	60.2	L	1223.2		2658.1		16847.4		4063.5		58255.4		83107.8	483.8	L
407 0 1 S1CBI025	2	5	1	1	9	T	0.0<DL	NC3,L	24.5		180.7		864.9		2.3		14.1		1086.6	143.0	NC3
407 0 2 S1CBI025	2	5	1	1		B	0.0<DL		4.0		14.2		55.3		1.3	NC3,L	11.5		86.2	0.0<DL	NC3,
408 10 S1CAS031	3	1	1	1	0.0179	0.5	42.2	L	656.1		3201.7		17430.8		60.3	NC3,L	6686.1		28077.0	627.0	L
407 0 1 S1CBI031	3	1	1	1	8.1	T	2.6	L,NC3	38.9		142.0		708.4		0.0<DL		169.6	NC3	1061.6	80.4	
407 0 2 S1CBI031	3	1	1	1		B	0.0<DL		6.1	L	15.3		53.5		0.0<DL		16.7	NC3	91.6	21.8	NC3
408 10 S1CAS032	3	2	1	1	0.0191	0.4	0.0<DL	NC3,	437.6		1441.2		9169.8		60.3	NC3,L	4626.7		15735.7	224.2	NC,L
407 0 1 S1CBI032	3	2	1	1	5	T	1.4	L,NC3	13.7		73.0		340.7		0.0<DL		13.4	NC3	442.1	79.9	
407 0 2 S1CBI032	3	2	1	1		B	0.0<DL		2.2	L,NC3	6.3	NC,L	20.8	NC	0.0<DL		51.5	NC3	80.8	71.0	NC3
408 10 S1CAS033	3	3	1	1	0.0166	0.4	42.2	L	462.7		1537.6		9934.8		0.0<DL	NC,L	4611.4		16588.6	277.0	L
407 0 1 S1CBI033	3	3	1	1	5.9	T	2.4	L,NC3	22.6		98.4		534.1		0.0<DL		0.0<DL		657.5	81.3	
407 0 2 S1CBI033	3	3	1	1		B	0.0<DL		2.5	L	6.6	NC,L	26.3		0.0<DL		9.5	NC3	44.8	9.0	NC3
408 10 S1CAS034	3	4	1	1	0.0165	0.4	0.0	<DL,L	721.6		1700.6		9750.7		142.4	NC3,L	9620.1		21935.3	259.2	L
407 0 1 S1CBI034	3	4	1	1	12.9	T	2.6	L,NC	30.0		183.2		1062.9		123.2		386.8		1788.7	88.9	
407 0 2 S1CBI034	3	4	1	1		B	0.0<DL		6.8	L	17.4	NC	70.1		0.0<DL		9.5	NC3	103.7	0.0<DL	
408 10 S1CAS035	3	5	1	1	0.0165	0.4	59.5	L	938.5		4548.4		30257.9		483.9	L	11399.1		47687.3	821.6	L
407 0 1 S1CBI035	3	5	1	1	7.7	T	0.0<DL		22.9		167.8		746.7		0.0<DL		0.0<DL	L,NC	937.4	177.6	
407 0 2 S1CBI035	3	5	1	1		B	0.0<DL		1.4	L	5.8	L	17.0		0.0<DL		94.6	NC3	118.8	143.6	
408 10 S1CAS041	4	1	1	1	0.0131	0.5	67.7	L	169.3		311.4		1702.9		0.0<DL	NC3,	673.8		2925.0	36.4	CRA, <DL,L
407 0 1 S1CBI041	4	1	1	1	9.2	T	0.0<DL		5.0	L	15.6		74.0	NC	0.0<DL		11.9		106.4	17.3	
407 0 2 S1CBI041	4	1	1	1		B	0.0<DL		0.0<DL		0.0<DL		2.9	L,NC3	0.0<DL		5.2	NC3,L	8.1	0.0<DL	
408 10 S1CAS042	4	2	1	1	0.0144	0.4	0.0	<DL,L	100.2	L	204.5	NC,L	1494.0		0.0<DL	NC3,	1190.4	NC	2989.1	37.3	CRA, <DL,L
407 0 1 S1CBI042	4	2	1	1	9.5	T	7.5	L	11.7		28.1		112.7		11.0	NC	10.5		181.5	19.8	
407 0 2 S1CBI042	4	2	1	1		B	0.0<DL		1.3	L	2.5	L	8.2	L	<DL	<DL	6.8		18.7	0.0<DL	
408 10 S1CAS043	4	3	1	1	0.0142	0.4	0.0	<DL,L	96.0	L	185.7	L	1157.5		0.0	<DL	L	1011.9	2451.1	32.1	CRA, <DL,L
407 0 1 S1CBI043	4	3	1	1	7.8	T	0.0<DL	L,NC	5.0	L	16.0	NC	80.8		0.0<DL		6.3	NC	108.2	13.0	
407 0 2 S1CBI043	4	3	1	1		B	0.0<DL	CO		L	0.0<DL	L,NC	CO	L	0.0<DL	CO		NC	NA	0.0<DL	
408 10 S1CAS044	4	4	1	1	0.0158	0.4	0.0	<DL,L	79.6	NC3,L	197.5	NC,L	1512.5		0.0	<DL,L	728.0	NC	2517.6	38.0	CRA, <DL,L
407 0 1 S1CBI044	4	4	1	1	8.7	T	7.2	L	9.9		18.0		60.6		10.5		9.0		115.2	18.4	NC
407 0 2 S1CBI044	4	4	1	1		B	0.0		0.0<DL		1.8	L,NC3	1.7	NC,L	0.0<DL		74.0	NC3	77.5	2.4	L,NC
408 10 S1CAS045	4	5	1	1	0.0155	0.4	0.0	<DL,L	101.9	L	190.3	L	1747.3		0.0	<DL,L	799.8		2839.3	34.6	CRA, <DL,L
407 0 1 S1CBI045	4	5	1	1	10.2	T	0.0<DL		3.0	NC,L	12.3		56.5		8.7	NC3	21.3		101.7	15.8	L,NC
407 0 2 S1CBI045	4	5	1	1		B	0.0<DL		174.7		0.0<DL		928.1	NC3	0.0<DL		1629.7	NC3	2732.4	0.0<DL	
408 ave S1CAS050 R	5	1	3	1	0.0156	0.4	2.9		45.7		42.7		236.3		7.2		46.2		381.1	7.9	
407 0 1 S1CBI050	5	1	1	1	13.4	T	0.8	L,NC3	5.3	NC3,L	4.8	NC,L	CO	NC3	0.0<DL		402.6	NC3	413.5	0.0<DL	
407 0 2 S1CBI050	5	1	1	1		B	2.0	L	2.1	L	2.5	L	0.0		0.0<DL		0.0<DL		6.6	0.0<DL	
408 ave S1CAS051	5	1	1	3	0.0137	0.4	3.1		52.8		44.5		253.8		56.3		117.3		527.9	7.7	
407 0 1 S1CBI051	5	1	1	1	228.6	T	0.0<DL		1.7	L	1.5	L,NC3	7.2	NC,L	0.0<DL		0.0<DL		10.3	0.0<DL	L,NC
407 0 2 S1CBI051	5	1	1	1		B	0.0<DL		0.0	>DL,L	0.8	L	0.0		0.0<DL		2.0	L,NC3	2.9	2.3	L,NC3
408 10 S1CAS059	5	1	1	1	0.0148	0.4	2.2	L	38.4		33.7		190.4		1.8	L,NC3,I	49.3		315.7	8.3	
407 0 1 S1CBI059	5	1	1	1	11.6	T	0.7	L,NC3	1.7	L	3.1	L	9.4	L	0.0<DL		4.0	L	18.9	1.9	L,NC3
407 0 2 S1CBI059	5	1	1	1		B	0.0<DL		0.4	L,NC3	0.0<DL		0.0		0.0<DL		13.8		14.3	0.0<DL	

APPENDIX 8-1 (cont'd.). Field sediment and interstitial water concentrations.

SAMPLE # req,proj,mat,st,grab	S T A	G A S C TOC: R N.A.O. gOC/gdry A R R R DOC: B EPEPEP mgOC/L	T=total B=bound	2,4'-DDE CONC	CODE	4,4'DDE CONC	CODE	2,4'-DDD CONC	CODE	4,4'-DDD CONC	CODE	2,4'-DDT CONC	CODE	4,4'-DDT CONC	CODE	Σ DDT DDT	dieldrin CONC	CODE
408 10 S1CAS052	5	2 1 1 1 0.0148	0.4	3.4		62.3		46.2		292.8		0.0	NPA	74.8	NC	479.4	4.7	NC3
407 0 1 S1CBI052	5	2 1 1 11.6	T	0.0	DL	4.6	L	4.1	L	14.5		0.0	<DL	7.6	NC	30.7	3.8	L,NC3
407 0 2 S1CBI052	5	2 1 1	B	0.0	<DL,L	1.0	L	0.0	<DL,L,NC	2.5	L,NC3	0.0	<DL	6.7		10.3	3.7	L,NC3
408 10 S1CAS053	5	3 1 1 1 0.0149	0.4	2.7		47.5		38.6		232.7		0.0	NPA	65.2		386.6	7.6	
407 0 1 S1CBI053	5	3 1 1 12.9	T	0.0	<DL	3.3	L	3.8	NC	15.6		0.0	<DL	10.4	NC3	33.1	0.0	<DL
407 0 2 S1CBI053	5	3 1 1	B	0.0	<DL	0.0	<DL	0.0	<DL,L,NC	0.0		1.8	L,NC3	75.5	NC3	77.4	3.5	L
408 10 S1CAS054	5	4 1 1 1 0.0147	0.4	2.7		42.4		37.0		230.5		4.0	L,NC3	153.9		470.5	1.5	L,NC3
407 0 1 S1CBI054	5	4 1 1 8.3	T	0.0	<DL	1.5	L	2.1	L	6.3	L	0.0	<DL	10.7	NC3	20.6	0.0	<DL
407 0 2 S1CBI054	5	4 1 1	B	0.0	<DL	0.0	<DL	0.0	<DL,L,NC	1.2	L,NC	0.0	<DL,L,NC	5.8	L	7.0	0.0	<DL
408 10 S1CAS055	5	5 1 1 1 0.0152	0.4	2.1	L	33.8		38.8		223.7		0.0	NPA	55.4		353.7	6.0	
407 0 1 S1CBI055	5	5 1 1 6.8	T	CO	CO	CO		CO		CO		CO		CO		NA	CO	
407 0 2 S1CBI055	5	5 1 1	B	CO	CO	CO		CO		CO		CO	NC	CO		NA	CO	
408 10 S1CAS061	6	1 1 1 1 0.0205	0.5	41.5		786.8	DRA	320.0		1573.5	DRA	269.5		1560.4	DRA	4551.6	51.8	
407 01 S1CBI061	6	1 1 1 1 8.1	T	0.0	<DL	99.2	NC	0.0	<DL	353.6	NC3	0.0	<DL	205.7		658.5	0.0	<DL
407 02 S1CBI061	6	1 1 1 1	B	0.0	<DL	11.1		3.4	L,NC3	43.7	NC	0.0	<DL	16.1		74.2	0.0	<DL
408 11 S1CAS062	6	2 1 1 1 0.0266	0.4	16.9		381.3		116.8		707.3		11.8	L,NC3,I	124.4	NC3	1358.5	47.4	
407 01 S1CBI062	6	2 1 1 1 10.2	T	0.0	<DL	103.8		24.9	NC3	184.7	NC	0.0	<DL	93.7		407.0	0.0	<DL
407 02 S1CBI062	6	2 1 1 1	B	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	0.0	<DL
408 10 S1CAS063	6	3 1 1 1 0.0397	0.4	33.0		280.6	DRA	241.3		484.6	DRA	48.4	NC,I	374.0	I,NC3	1462.0	107.1	NC3
407 01 S1CBI063	6	3 1 1 1 13.4	T	0.0	<DL	39.6	NC3	9.2	L,NC3	111.4	NC3	0.0	<DL	0.0	<DL	160.3	0.0	<DL
407 02 S1CBI063	6	3 1 1 1	B	0.0	<DL	11.5	NC3	4.5	L,NC3	18.5		0.0	<DL	48.3	NC3	82.8	0.0	<DL
408 10 S1CAS064	6	4 1 1 1 0.0331	0.4	41.3		698.5	DRA	272.5		977.9	DRA	0.0	NPA	169.7	I,NC3	2159.9	106.4	NC3
407 01 S1CBI064	6	4 1 1 1 16.9	T	11.0	NC3	256.1		55.2		387.0		0.0	<DL	25.6	NC	734.9	0.0	<DL
407 02 S1CBI064	6	4 1 1 1	B	2.6	NC3,L	74.1		15.8	NC3	112.6		0.0	<DL	0.0	<DL	205.2	0.0	<DL
408 10 S1CAS065	6	5 1 1 1 0.029	0.5	38.4		758.3	DRA	240.7		1022.5	DRA	0.0	NPA	132.4	I,NC3	2192.2	79.4	
407 01 S1CBI065	6	5 1 1 1 27.4	T	2.8	NC3,L	92.9		20.9		136.1		0.0	<DL	101.5		354.2	8.9	
407 02 S1CBI065	6	5 1 1 1	B	0.6	NC3,L	10.9		3.4	L	21.5		0.0	<DL	11.3	NC	47.7	0.0	<DL
408 ave S1CAS070 R	7	1 3 1 1 0.0102	0.6	2.4		30.0		41.0		242.7		6.1		310.9		633.1	8.2	
407 01 S1CBI070	7	1 1 1 14.9	T	0.5	L,NC3	4.2	L,NC3	4.8	L	26.1		0.0	<DL	4.2	NC3,L	39.8	0.0	<DL
407 02 S1CBI070	7	1 1 1	B	0.0	<DL	1.7	L,NC3	1.0	L	5.3	L	0.0	<DL	0.0	<DL	8.0	0.0	<DL
408 ave S1CAS071	7	1 1 3 1 0.0106	0.6	2.8		33.6		44.0		271.6		19.1		116.3		487.3	11.4	
407 01 S1CBI071	7	1 1 1 9	T	0.0	<DL	1.3	L,NC3	2.9	L	18.3		0.0	<DL	4.4	NC3,L	26.9	1.5	L,NC3
407 02 S1CBI071	7	1 1 1	B	0.0	<DL <DL			1.4	L,NC3	1.0	L	0.0	<DL	4.9	L	7.4	0.0	<DL
408 10 S1CAS079	7	1 1 1 1 0.0091	0.6	2.7		29.7		34.2		232.2		0.0	NPA	118.1		416.9	9.0	
407 01 S1CBI079	7	1 1 1 15.9	T	0.0	<DL	2.4	L	3.8	L	24.1		0.0		0.0	<DL	30.2	2.4	L
407 02 S1CBI079	7	1 1 1	B	0.0	<DL	0.8	L	1.4	L,NC3	4.8	NC,L	0.0		0.0	<DL	6.9	0.0	<DL
408 10 S1CAS072	7	2 1 1 1 0.0112	0.6	2.0		24.5		27.1		168.2		0.0	NPA	34.1		256.0	6.1	NC3
407 01 S1CBI072	7	2 1 1 1 8.1	T	0.0	<DL	2.9	L	3.8	L	21.1	NC	0.0		6.6	L	34.4	0.0	<DL
407 02 S1CBI072	7	2 1 1 1	B	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0		4.9	L	4.9	0.0	<DL
408 10 S1CAS073	7	3 1 1 1 0.01	0.6	2.8		30.5		46.3		291.4		0.0	NPA	52.6		423.5	12.4	NC3
407 01 S1CBI073	7	3 1 1 1 27	T	0.0	<DL	1.1	L	3.0	L,NC	14.8		0.0		2.5	L	21.4	1.6	L,NC3
407 02 S1CBI073	7	3 1 1 1	B	0.0	<DL	0.0	<DL	1.2	L,NC	2.7	NC3,L	0.0		2.8	L,NC3	6.6	0.0	<DL

APPENDIX 8-1 (cont'd.). Field sediment and interstitial water concentrations.

SAMPLE # reg,proj,mat,st,grabT.	STA	GRAB	A N.A.O. DOC:	S C TOC: gOC/gdry T=total B=bound	2,4'-DDE CONC	CODE	4,4'DDE CONC	CODE	2,4'-DDD CONC	CODE	4,4'-DDD CONC	CODE	2,4'-DDT CONC	CODE	4,4'-DDT CONC	CODE	Σ DDT DDT	dieldrin CONC	CODE
408 10 S1CAS074	7	4 1 1 1	0.0115	0.5	1.6	L	22.2		27.9		173.6		0.0	NPA	67.2		292.4	6.3	
407 01 S1CBI074	7	4 1 1 1	63.4	T	0.0<DL	CO			2.2	L CO			0.0	CO		NA		0.0<DL	
407 02 S1CBI074	7	4 1 1 1		B	0.0<DL	CO		L,NC3	0.8	L,NC3 CO		NC3,L	0.0	CO		L NA		0.0<DL	
408 10 S1CAS075	7	5 1 1 1	0.0112	0.5	2.0	L	28.8		34.1		215.4		0.0	NPA	76.6		357.0	13.0	NC
407 0 1 S1CBI075	7	5 1 1	39.5	T	0.0<DL		1.3	L	2.6	L	8.4	L	0.0		13.4	NC3	25.7	0.3L,NC3,<D	
407 0 2 S1CBI075	7	5 1 1		B	0.6	L,NC3 CO		L	1.4	L,NC3 CO			2.5	NC CO		NA		0.0<DL,L,NC	
408 10 S1CAS081	8	1 1 1 1	0.0121	0.6	0.7	L	8.8		8.3		47.8		2.0	L,NC3	10.9		78.4	2.5	NC3
407 01 S1CBI081	8	1 1 1 1	12	T	0.0<DL		0.0<DL		2.3	L	3.6	L	0.0<DL		1.9	L,NC	7.8	0.0<DL	
407 02 S1CBI081	8	1 1 1 1		B	4.4	NC3,L	4.2	L,NC3	8.4		10.0		5.7	NC,L	4.3	NC3,L	37.0	10.4	NC3
408 10 S1CAS082	8	2 1 1 1	0.0127	0.5	0.4	L,I	13.0	NC	8.0		46.3	NC	0.0	NPA	23.8	I,NC3	91.5	0.0	NPA
407 01 S1CBI082	8	2 1 1 1	10.9	T	0.0<DL		0.0<DL		0.0<DL		4.9	NC3,L	0.0		0.0<DL		4.9	0.0<DL	
407 02 S1CBI082	8	2 1 1 1		B	0.0<DL		0.0<DL		0.0<DL		0.0<DL		0.0		0.0<DL		0.0	0.0<DL	
408 10 S1CAS083	8	3 1 1 1	0.0113	0.5	0.6	L,NC3	8.4		9.4		47.3		0.0	NPA	11.0		76.8	2.2	L,NC3
407 01 S1CBI083	8	3 1 1 1	9.6	T	0.7	L,NC3	7.7		1.5	L,NC3	0.0<DL		0.0		4.5	L,NC3	14.3	0.0<DL	
407 02 S1CBI083	8	3 1 1 1		B	0.0<DL		0.0>DL,L,NC		0.8	L,NC3	0.0<DL		0.0		0.0<DL		0.8	0.0<DL	
408 10 S1CAS084	8	4 1 1 1	0.0117	0.5	0.8	L,NC3	8.4		8.3		45.9		0.0	NPA	31.1		94.5	1.6	L,NC3
407 01 S1CBI084	8	4 1 1 1	8.3	T	0.0<DL	CO		L	1.2	L CO			0.0	CO		NA		0.0<DL	
407 02 S1CBI084	8	4 1 1 1		B	0.0<DL	CO			0.7	L,NC3 CO			0.0	CO		NC3,L NA		0.0<DL	
408 10 S1CAS085	8	5 1 1 1	0.0112	0.5	0.6	L,I	8.7		7.1		43.9		0.0	NPA	10.9	NC3	71.2	2.3	L,NC3
407 01 S1CBI085	8	5 1 1 1	15.6	T	0.0<DL		0.0>DL,L,NC		0.0<DL		0.0<DL		0.0		0.0<DL		0.0	0.0<DL	
407 02 S1CBI085	8	5 1 1 1		B	0.0<DL		0.0<DL		1.6	L,NC3	0.0<DL		0.0		0.0<DL		1.6	0.0<DL	
408 10 S1CAS009	9	1 1 1 1	0.0122	0.5	0.3	L,NC3	3.7		2.8	NC3	14.0		0.0	NPA	3.4	L,NC3	24.0	0.0	NPA
407 ave 01 S1CBI009R	9	1 3 1 1	17.9	T	0.0		1.2		1.4		5.9		0.0		6.8		15.3	0.0	
407 02 S1CBI009	9	1 1 1 1		B	0.0<DL		0.0<DL		0.0<DL		0.0<DL		0.0		20.6	NC3,L	20.6	0.0<DL	
408 10 S1CAS091	9	1 1 1 1	0.0081	0.6	0.3	L,I,NC3	2.7		1.9		8.1		0.0	NPA	3.5	L,NC3	16.4	0.0	NPA
407 01 S1CBI091 lost 02 sample	9	1 1 1 1	9.9	T	0.0<DL		0.0<DL		0.0<DL		0.0<DL		0.0		0.0<DL 0.0		0.0	0.0<DL	
408 10 S1CAS092	9	2 1 1 1	0.0085	0.7	0.0<DL		1.4	L,NC3	0.8		3.6	NC	0.0	NPA	2.3	L,NC3	8.1	0.0	NPA
407 01 S1CBI092	9	2 1 1 1	10.9	T	0.0<DL		0.0<DL		0.0<DL		0.0<DL		0.0		0.0<DL,NC3,		0.0	0.0<DL	
407 02 S1CBI092	9	2 1 1 1		B	0.0<DL		0.0<DL		0.0<DL		0.0<DL		0.0		0.0<DL		0.0	0.0<DL	
408 10 S1CAS093	9	3 1 1 1	0.0086	0.7	0.0<DL		1.5	L	1.4	L,NC3	4.8		0.0	NPA	1.8	L,NC	9.5	0.0	NPA
407 01 S1CBI093	9	3 1 1 1	18.8	T	0.0<DL		0.0<DL		0.0<DL		0.0<DL		0.0		5.5	L	5.5	0.0<DL	
407 02 S1CBI093	9	3 1 1 1		B	0.0<DL		0.0<DL		0.0<DL,L,NC		0.0<DL		0.0		0.0	<DL,L	0.0	0.0<DL	
408 10 S1CAS094	9	4 1 1 1	0.0094	0.7	0.2	L	1.5	L	1.2	L,NC3	5.2		0.0	NPA	3.9	L	12.0	0.0	NPA
407 01 S1CBI094	9	4 1 1 1	9.3	T	0.0<DL		0.0<DL		0.0<DL		0.0<DL		0.0		86.2	NC3	86.2	0.0<DL	
407 02 S1CBI094	9	4 1 1 1		B	0.0CO	CO		CO		CO		CO		NC,L CO		NC NA	CO		
408 10 S1CAS095	9	5 1 1 1	0.0088	0.7	0.0<DL		3.0	NC3	1.0	L,NC3	4.7		0.0	NPA	3.4	L,NC3	12.0	3.2	NC3
407 01 S1CBI095	9	5 1 1 1	12.4	T	2.0	NC3,L	3.8	L	3.0	L,NC	5.4	NC3,L	1.7	NC3,L	0.0<DL		15.8	0.0<DL	
407 02 S1CBI095	9	5 1 1 1		B	0.0<DL		0.0<DL		1.2	L,NC	0.0<DL		0.0		3.9	NC3,L	5.1	1.6	L
408 10 S1CAS113	13	1 1 1 1	0.0155	0.5	4.2		48.3		61.8		375.5		0.0	NPA	66.2		556.0	16.9	
407 0 1 S1CBI113	13	1 1 1	5.7	T	0.5	L	1.9	L	4.6	L	19.3		0.0		0.0<DL		26.3	0.0<DL	
407 0 2 S1CBI113	13	1 1 1		B	0.0<DL		0.0<DL		0.0,<DL,C3,		0.0<DL		0.0		5.5	NC3,L	5.5	0.0<DL	

APPENDIX 8-1 (cont'd.). Field sediment and interstitial water concentrations.

SAMPLE #	S T A	G A S C TOC: R N.A.O. gOC/gdry R R DOC:	T=total	2,4'-DDE		4,4'DDE		2,4'-DDD		4,4'-DDD		2,4'-DDT		4,4'-DDT		Σ DDT	dieldrin	
reg.proj.mat.st.grabT.	B	EPEPEP mgOC/L	B=bound	CONC	CODE	CONC	CODE	CONC	CODE	CONC	CODE	CONC	CODE	CONC	CODE	DDT	CONC	CODE
408 10 S1CAS114	14	1 1 1 1 0.0151	0.5	3.1		53.0		78.8		506.4		0.0	NPA	88.2		729.6	16.9	
407 0 1 S1CBI114	14	1 1 1 7.6	T	0.0		2.8	L	4.2	L	20.1		0.0		0.0	<DL	27.1	0.0	<DL
407 0 2 S1CBI114	14	1 1 1	B	0.5	NC,L	6.8	L	0.0	<DL,NC3,	0.0	<DL	0.0		0.0	<DL	7.3	0.0	<DL
408 10 S1CAS115	15	1 1 1 1 0.0146	0.5	2.2		30.7		51.4		341.5		4.6	L	92.2		522.5	9.4	
407 0 1 S1CBI115	15	1 1 1 8.9	T	0.0	<DL	1.5	L	3.7	NC,L	16.6		0.0		0.0	<DL	21.8	0.0	<DL
407 0 2 S1CBI115	15	1 1 1	B	0.0	<DL,NC3,	2.1	NC3,L	0.0	<DL	0.0	<DL	0.0		0.0	<DL	2.1	0.0	<DL
408 10 S1CAS116	16	1 1 1 1 0.0135	0.5	2.6		39.7		48.0		342.2		3.8	L	259.4		695.7	11.1	NC3
407 0 1 S1CBI116	16	1 1 1 29.8	T	0.4	NC3,L	1.9	L	2.8	L	11.6		0.0		0.0	<DL	16.6	0.0	<DL
407 0 2 S1CBI116	16	1 1 1	B	0.0	<DL,NC3,	1.5	L	5.0	L	24.4		0.0		3.1	NC3,L	34.0	0.0	<DL
408 10 S1CAS117	17	1 1 1 1 0.0134	0.5	0.6	L	9.6		11.4		72.0		0.0	NPA	38.0		131.6	3.0	
407 0 1 S1CBI117	17	1 1 1 8	T	0.0	<DL CO		L	0.9	NC3,L CO		L	0.0	CO		NA		0.0	<DL
407 0 2 S1CBI117	17	1 1 1	B	0.0	<DL	0.0	<DL	0.0	<DL,NC3,	0.0	<DL,NC3,	0.0		0.0	<DL	0.0	0.0	<DL
408 10 S1CAS118	18	1 1 1 1 0.0125	0.5	0.6	L,NC3	4.7		3.7		24.4		0.0	NPA	5.1	L	38.4	0.0	NPA
407 0 1 S1CBI118	18	1 1 1 16.6	T	0.0	<DL	0.3	L	1.0	NC3,L	1.7	L	0.0		39.2	NC3	42.1	0.0	<DL,NC3,
407 0 2 S1CBI118	18	1 1 1	B	0.4	NC3,L	0.0	<DL	0.8	NC,L	1.4	NC,L	0.0		7.1	NC3	9.6	0.0	<DL
408 10 S1CAS119	19	1 1 1 1 0.0114	0.6	0.4	L	3.0		1.6	L,NC3	6.8		1.5	L,NC	3.5	L,NC3	16.6	2.4	L,NC3
407 0 1 S1CBI119	19	1 1 1 8.8	T	0.0	<DL,NC3,	0.0	<DL	0.9	NC3,L	1.6	NC,L	0.0		3.1	NC3,L	5.6	0.0	<DL
407 0 2 S1CBI119	19	1 1 1	B	0.0	<DL,NC3,	0.8	L	0.6	L	2.0	NC,L	0.0		2.1	NC3,L	5.5	0.0	<DL,NC3,
408 10 S1CAS120	20	1 1 1 1 0.011	0.5	0.4	L	2.6	NC3	1.4	NC	6.1		0.0	NPA	4.2	L	14.6	1.9	L,NC
no interstitial water																		
408 10 S1CAS121	21	1 1 1 1 0.0105	0.7	0.3	L	2.8		2.3	NC3	9.7		0.0	NPA	3.4	L	18.5	0.0	NPA
407 0 1 S1CBI121	21	1 1 1 6.6	T CO	CO	CO		CO		CO		CO		CO		CO	CO		
407 0 2 S1CBI121	21	1 1 1	B	4.9	L	5.1	L	4.4	NC,L	7.1	L	6.4	L	6.7	NC,L	34.6	0.0	<DL
408 10 S1CAS122	22	1 1 1 1 0.0076	0.7	0.3	L,NC3	1.2	L,NC3	1.6	NC3	4.9		0.0	NPA	3.2	L,NC3	11.2	0.0	NPA
407 0 1 S1CBI122	22	1 1 1 5.3	T	0.0	<DL CO		NC3,L	0.0	<DL,NC,LCO			0.0	CO		CO		0.0	<DL
407 0 2 S1CBI122	22	1 1 1	B	0.8	NC3,L	0.0	<DL	10.6		0.0	<DL	0.0		12.4		23.8	0.0	<DL
408 10 S1CAS123	23	1 1 1 1 0.0102	0.6	0.4	L,NC3	2.5		1.5	L,NC3	5.0		0.0	NPA	5.2	L	14.6	3.4	NC3
407 0 1 S1CBI123	23	1 1 1 5.6	T CO	CO	CO		CO		CO		CO		CO		CO	CO		
407 0 2 S1CBI123	23	1 1 1	B	20.2		20.8		19.9		23.7		20.5		25.3		130.4	21.0	L

APPENDIX 8-2. Bioassay sediment and interstitial water concentrations; (sediment = µg/kg dry; interstitial water = ng/l).

SAMPLE # req,proj,mat,st,grab	STA	GRAB	AEP	SREP	CREP	TOC: N.A.O. gOC/gdry DOC: mgOC/L	T=total B=bound	2,4'-DDE CONC	2,4'-DDE CODE	4,4'DDE CONC	4,4'DDE CODE	2,4'-DDD CONC	2,4'-DDD CODE	4,4'-DDD CONC	4,4'-DDD CODE	2,4'-DDT CONC	2,4'-DDT CODE	4,4'-DDT CONC	4,4'-DDT CODE	Σ DDT DDT	dieldrin CONC	dieldrin CODE
420 10 S1CES011	1	1	1	1		0.0224	0.5	438.5		6142.6		39423.3		203333.0	DRA	18581.1		152839.6	DRA	420758.0	6337.4	NC
419 0 1 S1CB1011	1	1	1	1		10.1	T	554.6		5182.7	DRA	36825.1	DRA	134868.6	DRA	2326.6	DRA	6444.5	DRA	186202.0	5422.6	DRA
419 0 2 S1CB1011	1	1	1	1			B	11.1	NC	112.3		935.4		3008.3		0.0	<DL	57.9		4125.0	148.3	
420 10 S1CES013	1	3	1	1		0.024	0.5	56.3	L	832.2		1859.3		12218.7		192.3	NC3	7900.3		23059.0	197.9	L
419 0 1 S1CB1013	1	3	1	1		7.1	T	3.7	NC3,L	77.1		519.3		2678.9		192.1		637.4		4108.4	59.9	
419 0 2 S1CB1013	1	3	1	1			B	0.0	<DL	5.9	L	15.5		65.0		0.0	<DL	9.1	NC	95.5	0.0	<DL
420 10 S1CES015	1	5	1	1		0.027	0.5	77.8	NC3,L	978.9		1982.8	NC	15198.0		965.4		13888.7		33091.6	414.0	L
419 0 1 S1CB1015	1	5	1	1		7.9	T	7.3	L	84.1		770.8		3659.6		66.3		217.7		4805.8	470.7	
419 0 2 S1CB1015	1	5	1	1			B	0.0	<DL	8.9		50.5		225.0		0.0	<DL	9.1	NC3	293.5	0.0	<DL
420 12 S1CES021	2	1	1	1		0.0184	0.6	63.9	L	869.9		2088.4	NC	14585.6		2981.9		24821.4		45411.0	487.2	NC3,L
419 0 1 S1CB1021	2	1	1	1		6.3	T	3.5	NC3,L	58.6		327.9		1567.6		29.3	NC	80.3		2067.3	179.7	
419 0 2 S1CB1021	2	1	1	1			B	1.9	NC3,L	17.3		35.5		152.2		5.5	L	21.9		234.2	0.0	<DL
420 10 S1CES022	2	2	1	1		0.0213	0.5	73.5	L	1254.2		3323.0		27898.0		1366.6		55928.6	DRA	89843.8	914.1	
419 0 1 S1CB1022	2	2	1	1		8	T	6.8	NC3,L	98.0		549.5		2148.5		60.2		215.7		3078.6	314.3	
419 0 2 S1CB1022	2	2	1	1			B	0.0	<DL	9.2		40.2		139.4		0.0	<DL	7.0	NC	195.8	0.0	<DL
420 10 S1CES023	2	3	1	1		0.0221	0.6	64.9	L	852.4		2480.5	NC	19994.9		895.0		15012.1		39299.8	645.7	L
419 0 1 S1CB1023	2	3	1	1		7.3	T	3.3	NC,L	47.0		230.8		1081.7		28.6		107.1		1498.5	164.9	
419 0 2 S1CB1023	2	3	1	1			B	0.0	<DL	7.0	L	21.0		83.8		0.0	<DL	0.0	<DL	111.8	0.0	<DL
420 10 S1CES032	3	2	1	1		0.0179	0.5	52.9	NC3,L	408.3		975.4	NC	5804.2		1850.7		6375.7		15467.1	295.4	L
419 0 1 S1CB1032	3	2	1	1		19.5	T	0.9	L	12.5		76.9		396.0		0.0	<DL	4.4	NC3,L	490.7	68.1	
419 0 2 S1CB1032	3	2	1	1			B	0.0	<DL	2.6	L	5.4	NC,L	28.2		0.0	<DL	0.0	<DL	36.2	0.0	<DL
420 10 S1CES033	3	3	1	1		0.0169	0.6	51.7	NC,L	594.2		2221.4		14617.7		1014.9		9312.5		27812.3	473.5	L
419 0 1 S1CB1033	3	3	1	1		6.8	T	2.5	NC3,L	32.3		228.8		1235.9		6.2	NC3,L	22.3		1528.1	129.5	
419 0 2 S1CB1033	3	3	1	1			B	0.0	<DL	3.4	L	17.1		75.9		0.0	<DL	0.0	<DL	96.4	0.0	<DL
420 10 S1CES034	3	4	1	1		0.0169	0.6	45.8	NC3,L	537.1		1617.1	NC	10546.1		273.7		6241.7		19261.3	402.2	L
419 0 1 S1CB1034	3	4	1	1		9.9	T	2.6	L	46.1		246.4		1316.5		13.1	NC	50.9		1675.6	142.4	
419 0 2 S1CB1034	3	4	1	1			B	0.0	<DL	4.6	L	19.8		89.8		0.0	<DL	4.1	NC3,L	118.3	0.0	<DL
420 ave S1CES041R	4	1	3	1		0.015	0.6	0.0		102.1		202.7		1058.9		0.0		786.1		2149.8	49.9	CRA, L
419 0 1 S1CB1041	4	1	1	1		6.8	T	0.0	<DL	6.3	L	18.4		88.1		0.0	<DL	5.1	NC3,L	117.9	15.5	
419 0 2 S1CB1041	4	1	1	1			B	0.0	<DL	0.0	<DL	2.2	NC,L	10.7	NC3	0.0	<DL	0.0	<DL	12.9	0.0	<DL
420 ave S1CES042	4	2	1	3		0.0167	0.6	0.0		122.6		205.9		1082.6		74.1		617.0		2102.1	38.1	CRA, L
419 0 1 S1CB1042	4	2	1	1		8	T	0.0	<DL	5.8	L	16.7		84.2		0.0	<DL	0.0	<DL	106.6	12.3	NC
419 0 2 S1CB1042	4	2	1	1			B	0.0	<DL	0.0	<DL	1.4	NC3,L	5.6	NC,L	0.0	<DL	0.0	<DL	7.0	0.0	<DL
420 10 S1CES044	4	4	1	1		0.0162	0.5	0.0		115.8		208.0		1311.4		0.0		1479.3		3114.5	37.0	CRA, L, NC
419 0 1 S1CB1044	4	4	1	1		10.1	T	0.0	<DL	7.2	L	16.3		79.0		0.0	<DL	7.4	L	109.8	13.1	
419 0 2 S1CB1044	4	4	1	1			B	0.0	<DL	1.8	NC3,L	1.7	NC,L	4.6	NC,L	0.0	<DL	0.0	<DL	8.2	0.0	<DL
420 10 S1CES051	5	1	1	1		0.0141	0.5	2.4		38.0		27.1		172.8		2.0		27.6	NC	269.9	4.1	NC3,L
419 ave S1CB1051R	5	1	3	1		13.1	T	3.6		4.8		2.8		13.4		0.0	<DL	4.2	NC3,L	28.8	6.0	
419 0 2 S1CB1051	5	1	1	1			B	0.0	<DL	0.0	<DL	0.0		0.0	<DL	0.0	<DL	6.9	L	6.9	0.0	<DL
420 10 S1CES054	5	4	1	1		0.0146	0.5	1.7	NC3	33.3		34.6		210.1		5.4	NC3	33.5	NC	318.7	6.2	L
419 0 1 S1CB1054	5	4	1	1		6.9	T	0.0	<DL	2.5	L	2.9	NC,L	10.2		0.0	<DL	0.0	<DL	15.6	0.0	<DL
419 0 2 S1CB1054	5	4	1	1			B	0.0	<DL	0.0		0.0		0.0	<DL	0.0	<DL	120.7	NC3	120.7	0.0	<DL

APPENDIX 8-2 (cont'd.). Bioassay sediment and interstitial water concentrations; (sediment = µg/kg dry; interstitial water = ng/l).

SAMPLE #	S T A	G R A B	A R E B	S C R E P	C R E P	TOC: N.A.O. gOC/gdry DOC: mgOC/L	T=total B=bound	2,4'-DDE CONC	2,4'-DDE CODE	4,4'DDE CONC	4,4'DDE CODE	2,4'-DDD CONC	2,4'-DDD CODE	4,4'-DDD CONC	4,4'-DDD CODE	2,4'-DDT CONC	2,4'-DDT CODE	4,4'-DDT CONC	4,4'-DDT CODE	Σ DDT DDT	dieldrin CONC	dieldrin CODE
reg,proj,mat,st,grab																						
420 12 S1CES055	5	5	1	1		0.0148	0.5	2.4		34.1		30.7	NC	199.3		2.6		24.6		293.7	5.5	NC3,L
419 0 1 S1CBI055	5	5	1	1		7.9	T	0.0	<DL	1.9	NC3,L	2.7	NC3,L	10.4	NC	0.0	<DL	0.0	<DL	14.9	0.0	
419 0 2 S1CBI055	5	5	1	1			B	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	0.0	<DL
420 10 S1CES062	6	2	1	1		0.0257	0.5	19.9		367.3		147.0		638.1	DRA	10.6		32.9		1215.8	36.8	
419 0 1 S1CBI062	6	2	1	1		14.4	T	0.0	<DL	169.2		35.8		197.5	NC	19.9	NC3	211.2	NC3	633.6	0.0	
419 0 2 S1CBI062	6	2	1	1			B	0.0		8.4		2.2	L	8.9	L	0.0	<DL	0.0		19.5	0.0	<DL
420 10 S1CES063	6	3	1	1		0.03	0.5	38.3		746.9	DRA	257.6		1042.0	DRA	13.5	NC	95.6	NC	2193.9	73.5	NC3
419 0 1 S1CBI063	6	3	1	1		10.9	T	0.0	<DL	208.6		52.6		307.8		22.3	NC3	11.6	NC3	602.9	0.0	<DL
419 0 2 S1CBI063	6	3	1	1			B	0.0	<DL	4.9	L	1.6	NC3,L	8.1	NC3,L	0.0	<DL	7.0	NC3,L	21.5	0.0	<DL
420 10 S1CES065	6	5	1	1		0.0322	0.5	40.7		1031.8	DRA	306.1		1071.3	DRA	13.1	NC3	59.4		2522.4	91.8	
419 0 1 S1CBI065	6	5	1	1		8.4	T	22.0		396.4		82.9		462.1		35.7	NC3	16.3	NC3	1015.3	0.0	<DL
419 0 2 S1CBI065	6	5	1	1			B	0.0		16.9		4.0	NC3,L	18.3		0.0	<DL	3.9	NC3,L	43.0	0.0	<DL
420 10 S1CES073	7	3	1	1		0.0106	0.6	2.8		29.1		45.6		320.7		1.1	NC	115.4		514.6	8.3	
419 0 1 S1CBI073	7	3	1	1		7.1	T	0.0	<DL	0.0	<DL	5.0	NC3,L	16.7		0.0	<DL	5.6	L	27.3	0.0	<DL
419 0 2 S1CBI073	7	3	1	1			B	0.0		0.0		0.0		2.6	NC3,L	0.0	<DL	5.9	L	8.5	0.0	
420 10 S1CES074	7	4	1	1		0.012	0.6	1.4		19.7		21.2		135.5		3.0	NC3	43.7		224.5	4.9	NC3,L
419 0 1 S1CBI074	7	4	1	1		15.5	T	0.0	<DL	0.0	<DL	3.2	NC3,L	9.3		0.0	<DL	84.9	NC3	97.4	0.0	<DL
419 0 2 S1CBI074	7	4	1	1			B	0.0	<DL	0.0		0.0		2.8	NC3,L	0.0	<DL	81.0	NC3	83.7	0.0	<DL
420 10 S1CES075	7	5	1	1		0.0125	0.6	1.8		24.5		37.6		225.9		1.1	NC3	35.7	NC	326.5	5.7	NC3,L
419 0 1 S1CBI075	7	5	1	1		11.5	T	0.0		1.9	NC3,L	3.6	NC,L	13.8	NC	0.0	<DL	0.0	<DL	19.3	0.0	<DL
419 0 2 S1CBI075	7	5	1	1			B	0.0	<DL	0.0		0.0		0.0	<DL	0.0	<DL	0.0	<DL	0.0	0.0	<DL
420 10 S1CES081	8	1	1	1		0.0118	0.6	0.6	L	7.6		6.8		32.8		1.9	NC3	4.1	NC3	53.7	0.0	
419 0 1 S1CBI081	8	1	1	1		9.2	T	0.0	<DL	0.0	<DL	0.0		3.1	NC3,L	0.0	<DL	18.4	NC3	21.5	0.0	
419 0 2 S1CBI081	8	1	1	1			B	0.0	<DL	0.0	<DL	0.0		0.0	<DL	0.0	<DL	4.0	NC3,L	4.0	0.0	<DL
420 10 S1CES082	8	2	1	1		0.0128	0.6	0.0		7.4		5.2		26.4	NC	0.7	L	5.4	NC3	45.0	0.0	
419 0 1 S1CBI082	8	2	1	1		10.1	T	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	0.0	<DL
419 0 2 S1CBI082	8	2	1	1			B	0.0		0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	0.0	
420 10 S1CES084	8	4	1	1		0.012	0.6	0.6	L	7.1		5.5	NC	28.0		0.0		0.7	L	42.0	0.0	
419 0 1 S1CBI084	8	4	1	1		6.7	T	0.0	<DL	0.0	<DL	1.1	NC,L	3.7	L	0.0	<DL	3.6	NC3,L	8.4	0.0	
419 0 2 S1CBI084	8	4	1	1			B	0.0		0.0	<DL	0.0		0.0	<DL	0.0	<DL	33.2	NC3	33.2	0.0	<DL
420 10 S1CES091	9	1	1	1		0.0092	0.7	0.0		2.6		2.1	NC	9.0		0.0		0.0		13.6	1.8	NC3,L
419 0 1 S1CBI091	9	1	1	1		4.6	T	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	5.7	NC3,L	5.7	0.0	<DL
419 0 2 S1CBI091	9	1	1	1			B	0.0		0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0		0.0	0.0	<DL
420 10 S1CES092	9	2	1	1		0.0082	0.7	0.0		1.4		1.1	NC3	3.6	NC	0.0		0.0		6.1	0.0	
419 0 1 S1CBI092	9	2	1	1		5.8	T	0.0		0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	<DL	0.0	0.0	
419 0 2 S1CBI092	9	2	1	1			B	0.0	<DL	0.0		0.0	)	0.0	<DL	0.0	<DL	2.6	L	2.6	0.0	
420 10 S1CES095	9	5	1	1		0.0092	0.7	0.0		1.5		0.9	NC3	3.7	NC	0.0		2.3	NC	8.4	0.0	
419 0 1 S1CBI095	9	5	1	1		23.4	T	0.0	<DL	0.0	<DL	0.0		0.0	<DL	0.0	<DL	3.3	NC3,L	3.3	0.0	<DL
419 0 2 S1CBI095	9	5	1	1			B	0.0	<DL	0.0		0.0		0.0	<DL	0.0	<DL	2.6	NC3,L	2.6	0.0	<DL

APPENDIX 8-3A. Estimated sediment concentrations of probable compounds -Lauritzen Channel (* = internal standards).

Sediment: S1CES021

COMPOUND NAME	CAS #	SCAN PROB %	BASE PEAK ION	BASE PK ABUND SCAN	nG/G CONC. SIM	nG/G CONC. SCAN	SIM Q VALUE
4,4' - DDT	50293	99	235	252773	16473	1502	1,M,FT
4,4' - DDD	72548	99	235	226344	9061	1345	M,FT
2,4' - DDD	-	-	235	202154	2501	1201	95
BENZO b FLUORANTHENE	-	-	252	89417	1700	2336	98,M
BENZO - a - PYRENE	50328	83	252	181678	1274	4747	98
PYRENE	129000	94	202	252859	1235	1502	96,FT
CHRYSENE	218019	86	228	165924	1016	1387	99,FT
BENZO - e - PYRENE	192972	79	252	151897	1015	3969	98
2,4' - DDT	789026	93	235	57201	1002	340	M
BENZO k FLUORANTHENE	207089	94	252	119452	901	3121	M
4,4' - DDE	72559	99	246	71326	845	424	96
*d-12 - PERYLENE	0	62	264	85545	845	2235	
*d8 - NAPHTHALENE	434	96	136	154071	845	845	97
*d - 10 ACENAPHTHENE	435	69	164	96400	845	845	97
*d-10 FLUORANTHENE			212	164945	845	845	94
*d-12 CHRYSENE	141628	86	240	117261	845	845	99
*d-12 BENZO b FLUORANTHENE	-	-	264	37505	845	845	93
*d12 - BENZO (ghi) PERYLENE	-	-	288	47790	845	845	98
FLUORANTHENE	206440	95	202	182754	728	1086	96
INDENO [1,2,3-cd] PYRENE	193395	75	276	48873	709	1002	81
BENZO [ghi] PERYLENE	191242	81	276	53528	626	1098	99,M
DIELDRIN	60571	95	79	30538	511	181	97
BENZO [a] ANTHRACENE	56553	81	228	58645	440	348	78
PERYLENE	198550	81	252	70109	329	1832	97,M
ANTHRACENE	120127	94	178	96765	328	984	98,M
PHENANTHRENE	85018	96	178	91758	290	933	97
DIBENZ [ah] ANTHRACENE	53703	67	278		227	0	98
2 - CH3 - NAPHTHALENE	91576	88	142	22530	190	143	98,M
NAPHTHALENE	91203	96	128	27133	154	173	97,M
ACENAPHTHYLENE	166	88	152	30325	146	193	98,M
2,6 - DI - CH3 - NAPHTHALENE	-	-	156	9038	86	57	95,M
FLUORENE	86737	83	166	12069	79	123	97,M
1 - CH3 - NAPHTHALENE			142		77	0	97,M
2,4'-DDE			246		55		99
1- CH3 - PHENANTHRENE			192		45	0	31,M
2,3,5 - TRI - CH3 NAPHTHALENE			170		40	0	82,M
BIPHENYL	92524	92	154	7756	34	49	99,M
MIREX			272		32		43,NS,IP
ACENAPHTHENE			154		31	0	84,M
cis-CHLORDANE			375		18		85
PCB - 052 TETRA	-	-	292	5794	17	59	58,M,FQ
PCB - 153 HEXA					15	0	99,M
HEPTACHLOR			100		13		74,NS,IP
PCB - 101 PENTA	-	-	326		12	0	82,M,FQ
trans-NONACHLOR			409		10		94
ALDRIN			66		9		62,M,NS
PCB - 138 HEXA					9	0	96,M
PCB - 118 PENTA					9	0	52,M,FQ
PCB - 095 PENTA	-	-	326		8	0	1,M,FQ
PCB - 110 PENTA					7	0	79,IP
PCB - 149 HEXA					6	0	74,M
PCB - 105 PENTA					6	0	29,FQ,NS
PCB - 180 HEPTA					5	0	92,M
PCB - 099 PENTA					5	0	13,M,NS
PHTHALATE ESTER, bis(2-ETHYL HEXYL)	603112		149		3	0	29,NS

APPENDIX 8-3A (cont'd.). Estimated sediment concentrations of probable compounds -Lauritzen Channel (* = internal standards).

Sediment: S1CES021 COMPOUND NAME	CAS #	SCAN PROB %	BASE PEAK ION	BASE PK ABUND SCAN	nG/G CONC. SIM	nG/G CONC. SCAN	SIM Q VALUE
PCB - 141 HEXA					3	0	33,M
PCB - 174 HEPTA					3	0	95,M
PCB - 170 + 190 HEPTA					2	0	33,M
PCB - 136 HEXA			360		2	0	0,M,FQ
PCB - 066 TETRA	-	-	292		1	0	68,M,NS
PCB - 176 HEPTA					1	0	86
PCB - 183 HEPTA					1	0	93
1-(1-PROPYN-1-yl)-2-(PROPYN-1-yl) BENZENE	93769052	31	153	39740		1038	
BENZENE , 1,1'-(2-CHLOROETHYLIDENE)BIS[4-CHLO	1022226	99	212	117301		697	
NO MATCH	-	-	153	31454		645	
TRIPHENYLENE , 2-METHYL	1705846	83	242	68099		569	
BENZO [c] PHENANTHRENE	195197	45	228	64623		540	
1-ACETYL-6-(ACETYLAMINO)-5-METHYL-1H-INDAZOL	75844303	45	231	25238		518	
BENZENE , 1,1'-(2-CHLOROETHYLIDENE)BIS[4-CHLO	2642800	93	235	86710		515	
BENZO [ghi] FLUORANTHENE	203123	76	226	71014		422	
BENZO [b] CHRYSENE	214175	85	278	20410		419	
NO MATCH	-	-	73	40337		410	
9,10-ANTHRACENEDIONE , 1,5 -DIMETHOXY-	6448904	42	268	15512		405	
CYCLO HEXA SILOXANE , DODECA METHYL	540976	88	73	55917		356	
CANNABISPIRADIENONE	71135790	60	242	41833		350	
BENZ [j] ACEANTHRYLENE , 3-METHYL	3343100	92	266	16879		346	
BENZO [c] PHENANTHRENE , 5- METHYL	652040	69	242	34857		291	
BENZENE , 1,1'-ETHENYLIDENEBIS[4-CHLORO-]	2642811	89	192	27851		283	
7-PHENACYLIDENE-7,8-DIHYDRO-6(1H)-PTERIDINO	86475094	52	266	13117		269	
BENZ [j] ACEANTHRYLENE , 3-METHYL	3343100	94	266	12980		266	
HEXA DECANOIC ACID , METHYL ESTER	112390	96	74	26163		266	
N-METHYL-DIBENZ (E,G) ISO INDOLE	59788157	52	231	12676		260	
1,2-BENZO PERYLENE	0	65	276	11973		246	
PYRENE , 1-METHYL-	2381217	83	216	39912		237	
NAPHTHALENE , 2- PHENYL	612942	64	204	22704		231	
11H- BENZO[b]FLUORENE	243174	70	216	38627		229	
1,1',3-TRIMETHYL-4,5'-BIPYRIMIDINE-2,2',4',6(1H,1'H,	79888861	27	264	10753		221	
11-HYDROXY-7,8,9,10-TETRAHYDROBENZO[b]XANT	83344789	60	266	10437		214	
NO MATCH	-	-	73	20984		213	
1,1' : 3',1''-TERPHENYL-	92068	64	234	31573		188	
BENZO[b]NAPHTHO[2,3-d]FURAN	243425	83	218	28583		170	
DIBENZO FURAN	132649	86	168	16602		169	
INDENO [1,2,3-cd] PYRENE	193395	68	276	8208		168	
2,2' - BI NAPHTHALENE	612782	71	254	20113		168	
BENZENE , 1,1'-ETHENYLIDENEBIS[4-CHLORO-3-AMINO-2-PHENYL-4,7-DIMETHYL-4,5-DIHYDRO-(3H	68194569	25	254	19071		159	
PENTA DECANOIC ACID , METHYL ESTER	7132641	83	74	15412		157	
2,2' - BI NAPHTHALENE	612782	48	240	18535		155	
TRIPHENYLENE	217594	87	228	25536		152	
BENZ [a] ANTHRACENE, 12-METHYL	2422799	40	242	16678		139	
1,2 : 3,4- DIBENZOANTHRACENE	215587	71	278	6677		137	
BENZ [j] ACEANTHRYLENE , 3-METHYL	3343100	30	266	6234		128	
PYRENE , 1-METHYL-	2381217	62	216	20536		122	
BENZ [j] ACEANTHRYLENE , 3-METHYL	3343100	60	266	5646		116	
N-METHYL-DIBENZ(E,G) ISO INDOLE	59788157	30	231	5487		113	
BENZO(a)THIOPHENO(2,3,4,5-def) PHENANTHRENE	0	59	258	4247		111	
6- AMONO BENZ ANTHRONE	0	25	245	5229		107	

APPENDIX 8-3A (cont'd.). Estimated sediment concentrations of probable compounds -Lauritzen Channel (* = internal standards).

Sediment: S1CES021 COMPOUND NAME	CAS #	SCAN PROB %	BASE PEAK ION	BASE PK ABUND SCAN	nG/G CONC. SIM	nG/G CONC. SCAN	SIM Q VALUE
N-DEUTERIO - .ALPHA.-NITROBENYLIDENE PHENY	59473846	43	242	12643		106	
BENZO [a] ANTHRACENE , 7,12-DIMETHYL	57976	59	256	12417		104	
POOR MATCH (ALKANE ?)	70412474	35	123	10079		102	
TETRA DECANOIC ACID , METHYL ESTER	124107	77	74	9875		100	
BENZ [j] ACEANTHRYLENE , 3-METHYL	3343100	60	266	4760		98	
BENZENE , 1,1'-(2-CHLOROETHYLIDENE)BIS[4-CHLO	2642800	89	235	16405		97	
3-(4-NITROPHENYLMETHYL)-6-NITROISOBENZOFU	85164416	31	178	16301		97	
1,1' : 2',1''-TERPHENYL-	84151	68	230	16014		95	
BENZENE , 1,1'-ETHENYLIDENEBIS[4-CHLORO-]	2642811	86	178	8991		91	
EICOSANE , 10-METHYL-	54833237	74	57	8774		89	
PYRENE , 1,3- DIMETHYL	64401214	75	230	14520		86	
1,5-NAPHTHYRIDIN-4-AMINE	27392683	29	145	7916		80	
BENZENE , 1-CHLORO-2-[2-CHLORO-1-(4-CHLORO....	14835940	92	212	7826		80	
BENZ [a] ANTHRACENE, 12-METHYL	2422799	41	209	8816		74	
HEPTANE , 3-ETHYL -5- METHYL	52896909	67	57	10598		67	
ANTHRACENE , 2-CHLORO-	17135783	64	206	6475		66	
ALKANE	-	-	57	6468		66	
PENTA DECANAL	0	72	57	10981		65	
ALKANE	-	-	57	10195		65	
14- ISO-20-EPIDEHYDRO PSEUDO ASPIDO SPERMID	91740124	37	280	3127		64	
CHRYSENE , 5-METHYL	3697243	72	242	7582		63	
2,5,9-TETRADECATRIENE , 3,12-DIETHYL	74685873	58	123	6142		62	
4-ACETYL AMINO-3-PHENYL-QUINOLINE-2(1H)-ONE	91622638	38	278	2913		60	
BENZO [2,1-b:3,4-b'] BISBENZOFURAN	222231	43	258	7107		59	
p,p' - DDD	72548	94	235	9546		57	
9- OCTA DECYNE	35365594	59	55	5576		57	
NAPHTHALENE , 1,3- DIMETHYL	575417	96	156	8677		55	
PENTACENE , 6,13-DIHYDRO	0	25	280	2613		54	
NO MATCH	-	-	64	5176		53	
ANDROSTANE , (5.BETA.)-	438233	47	245	5129		52	
BENZO[b]NAPHTHO [1,2-d]THIOPHENE	205436	54	234	8477		50	
3-ETHYL-1-METHYL NAPHTHALENE	17179418	67	155	4925		50	
PYRENE , 1,3- DIMETHYL-	64401214	64	230	8360		50	
PCB , 2,4,6- TRI CHLORO	35693926	96	256	4725		48	
BENZENE , 1-CHLORO--2-[2,2'-DICHLORO-1-(4-CHLO	3424826	71	246	8036		48	
NAPHTHALENE , 1,6,7- TRI METHYL	2245387	78	170	4450		45	
L- TYROSINE , N-(TRIFLUOROACETYL)-,BUTYL ESTE	5282984	30	95	4432		45	
NAPHTHALENE , 1,7- DIMETHYL	575371	96	156	7080		45	
2,3- BENZO CARBAZOLE	0	41	217	5387		45	
BENZO [c] PHENANTHRENE , 5,8-DIMETHYL	54986639	66	256	5167		43	
NO MATCH	-	-	165	4010		41	
ETHYL DIBENZOTHIOPHENE-1-CARBOXYLATE	34724715	45	256	4661		39	
BENZO [c] PHENANTHRENE , 5,8-DIMETHYL	54986639	67	256	4578		38	
N-N-PROPYL-3D2-CYCLOHEXENE-1,2-DICARBOXIMID	54815188	59	196	3422		35	
1H-INDENE , 1-(DIPHENYLMETHYLENE)-	13245904	28	284	1675		34	
NO MATCH	-	-	145	3289		33	
NO MATCH	-	-	95	3251		33	
ALKANE	-	-	123	4733		30	
N-(1-CYCLO HEXYL ETHYL) ANILINE	58008176	43	120	3475		29	
NO MATCH	-	-	141	1381		28	
OXIDOHIMACHALENE	64825849	36	161	4681		28	
BENZOIC ACID , 4-CYANO- , 3-METHOXYPHENYL ES	53327110	43	253	3145		26	
NAPHTHALENE , 1,6,7- TRI METHYL	2245387	68	170	2517		26	

APPENDIX 8-3A (cont'd.). Estimated sediment concentrations of probable compounds -Lauritzen Channel.

Sediment: S1CES021

COMPOUND NAME	CAS #	SCAN PROB %	BASE PEAK ION	BASE PK ABUND SCAN	nG/G CONC. SIM	nG/G CONC. SCAN	SIM Q VALUE
ALKANE	-	-	57	3794		24	
ALKANE	-	-	71	3727		24	
PCB , 2,2',3,4 - TETRA CHLORO	52663599	90	292	2298		23	
1H-INDENE , OCTAHYDRO-2,2,3,3,7,7-HEXA METHYL	54832836	59	193	3615		23	
PHENOL , 4-(2-PHENYLETHENYL)-, (E)-	6554989	59	181	2213		22	
NO MATCH	-	-	55	2112		21	
BENZENE , 2,4-DICHLORO-1-(2-CHLOROETHENYL)-	45892475	95	208	3118		20	
THIAZOLO[5,4-d]PYRIMIDINE , 5,7-DICHLORO	13479884	41	170	1907		19	
2,4-DIMETHOXY-2',4',6'-TRIMETHYLBIPHENYL	72968980	52	256	2258		19	
4-BENZYL PYRIDINE-3-CARBONITRILE	67839638	48	193	2922		19	
NAPHTHALENE , 1,2- DIMETHYL	573988	91	156	2244		14	
ALKANE	-	-	55	1243		13	
BENZOFURAN , 3-CH3-2-(1-METHYLETHENYL)-	23911582	43	158	1584		9	

APPENDIX 8-3B. Estimated sediment concentrations of probable compounds - Santa Fe Channel (* = internal standard).

Sediment: S1CES063

COMPOUND NAME	CAS #	SCAN PROB %	BASE PEAK ION	BASE PK ABUND SCAN	nG/G CONC. SIM	nG/G CONC. SCAN	SIM Q VALUE
PHENANTHRENE					19555	0	FT
FLUORANTHENE	258	93	202	250859	17648	12515	FT
PYRENE	252	94	202	255324	16548	12740	FT
BENZO [a] ANTHRACENE	56553	15	228	55744	14851	2777	FT
NAPHTHALENE	91203	95	128	240432	14539	7799	76
FLUORENE	86737	56	166	129318	10967	11515	92 FT
ACENAPHTHENE	83329	40	154	228639	10165	20376	FT
CHRYSENE	218019	31	228		10073	0	FT
1 - CH3 - NAPHTHALENE	90120	89	142	215246	9660	6982	98
BENZO b FLUORANTHENE	-	-	252		7448	0	FT
2 - CH3 - NAPHTHALENE	91576	88	142	175227	6917	5696	97
BENZO - a - PYRENE	50328	87	252	219354	5740	6288	FT
ANTHRACENE					4701	0	FT
BENZO - e - PYRENE		71	252	97581	4134	2797	M
1- CH3 - PHENANTHRENE	832699	66	192	144649	3473	631	99 NS
INDENO [1,2,3-cd] PYRENE	193395	64	276	28528	3049	1102	M
BENZO [ghi] PERYLENE	191242	69	276	33864	2274	1307	M
2,3,5 - (CH3)3 NAPHTHALENE (match = 1,4,6 - (CH3)3)	2131422	89	170	52944	2187	4716	77
BENZO k FLUORANTHENE	292	-	252		2050	0	FT
2,6 DI - CH3 - NAPHTHALENE -- (1,5 DI- CH3) = 99	571619	99	156	86279	1704	2797	93
BIPHENYL	92524	95	154	169949	1676	5512	M
PERYLENE	198550	-	252	137579	1506	3940	M
4,4' - DDD	354	92	235	50932	1345	2552	99
*d8-NAPHTHALENE	434	88	136	30176	980	980	M
*d-10 ACENAPHTHENE	435	-	164	11000	980	980	92
*d-10 FLUORANTHENE			212	19630	980	980	90
*d-12 BENZO b FLUORANTHENE	0	-	264	34233	980	980	M
*d12-BENZO (ghi) PERYLENE	-	-	288	25438	980	980	97
DIBENZO [ah] ANTHRACENE	53703	64	278	10219	892	388	M
4,4' - DDE	348	96	246	26245	808	1307	94
ACENAPHTHYLENE	208968	95	152	36172	558	1184	95
PHTHALATE ESTER, bis(2-ETHYL HEXYL)	603112	-	149		365	0	M
2,4' - DDD	-	-	235		237	0	89
PHENANTHRENE , 2,5-DIMETHYL	3674666	73	206	7383	174	367	
HEPTACHLOR	-	-	100		129	0	FQ
DIELDRIN					122	0	84,FQ
*d-8 4,4' - DDE			254		94	0	
PCB - 118 PENTA			326		62	0	M
PCB - 153 HEXA			360		60	0	M
2 , 4' -- DDE	-	-	246		52	0	92,NS
*PCB - 030 TRI	-	-	186		49	0	
*PCB - 065 TETRA					49	0	
PCB - 052 TETRA			292		39	0	M
PCB - 138 HEXA			360		31	0	M
PCB - 099 PENTA			326		29	0	M
PCB - 110 PENTA			326		22	0	M
PCB - 149 HEXA			360		20	0	M
PCB - 180 HEPTA			394		20	0	M
PCB - 044 TETRA					18	0	M
PCB - 105 PENTA			326		15	0	M
PCB - 174 HEPTA			394		11	0	M
PCB - 136 HEXA			360		11	0	M
PCB - 141 HEXA			360		9	0	M
PCB - 183 HEPTA			394		6	0	M
PCB - 176 HEPTA			394		6	0	M
PCB - 170 + 190 HEPTA			394		4	0	M
DIBENZOFURAN , 4-METHYL	7320538	89	182	225151	0	20049	
DIBENZOFURAN , 4-METHYL	7320538	89	182	178537	0	15905	
HEPTA COSANE	593497	88	57	241430	0	12046	
DOCOSANE , 9- BUTYL	55282149	84	57	240032	0	11985	

APPENDIX 8-3B (cont'd.). Estimated sediment concentrations of probable compounds -Santa Fe Channel.

Sediment: S1CES063 CAS # COMPOUND NAME	SCAN PROB	BASE PEAK %	BASE PK ABUND ION	nG/G CONC. SCAN	nG/G CONC. SIM	SIM Q SCAN	VALUE
HEPTA DECANE , 9- OCTYL	7225641	79	57	236180	0	11801	
TETRA COSANE	646311	83	57	235406	0	11760	
BENZO[b]NAPHTHO[2,3-d]FURAN	243425	52	218	227760	0	11372	
OCTA DECANE	593453	70	57	224950	0	11229	
1,1'-BIPHENYL , 2-METHYL	643583	69	168	120474	0	10739	
1,1'-BIPHENYL , 4-METHYL	644086	47	165	112068	0	9984	
PENTA DECANE , 2,6,10,14- (CH3)4	1921706	89	57	96112	0	8555	
HEPTA COSANE	593497	79	57	163589	0	8167	
BENZO[b]NAPHTHO[2,1-d] THIOPHENE	239350	89	234	162477	0	8105	
TRIPHENYLENE	217594	82	228	155552	0	7758	
OCTA DECANOIC ACID , BUTYL ESTER	123955	67	56	153526	0	7656	
PYRENE , 1 - CH3	2381217	70	216	150957	0	7534	
TRI PHENYLENE , 2-METHYL	1705846	83	242	149265	0	7452	
BENZO [c] PHENANTHRENE	195197	47	228	144726	0	7227	
BENZENE , 1,1' - METHYLENE BIS	101815	71	168	72594	0	6472	
CYCLO PENTA [cd] PYRENE	27208373	41	226	129572	0	6472	
PYRENE , 1 - CH3	2381217	63	216	127941	0	6390	
PYRENE , 1 - CH3	2381217	71	216	116179	0	5798	
PYRENE , 1 - CH3	2381217	59	215	109869	0	5492	
HEXATRIACONTANE	630068	76	57	189245	0	5410	
BENZENE , CYCLOPENTADIENE ETC..	2320323	67	153	59027	0	5267	
PHENANTHRENE , 4,5-DIMETHYL	3674699	89	206	58819	0	5247	
2-PHENYL-2-(p-TOLYL) PROPANE	0	78	195	55401	0	4941	
HEXA DECANE	544763	87	57	98894	0	4941	
UNDECANE , 3,6- DI-METHYL	17301289	69	57	55264	0	4920	
DIBENZOFURAN , 4-METHYL	7320538	71	182	54073	0	4818	
1,3 DI - CH3 - NAPHTHALENE	575417	97	156	145219	0	4716	
BENZO [c] PHENANTHRENE	195197	31	228	93608	0	4675	
PHENANTHRO[4,5-bcd]THIOPHENE	30796920	40	208	92571	0	4614	
PYRENE , 4,5-DIHYDRO	6628984	42	204	88891	0	4430	
BENZ [a] ANTHRACENE , 7- METHYL	2541697	73	242	87724	0	4390	
NO MATCH -- HUGE PEAK (AB = 1,000,000)			76	1000000	0	4364	
PHENANTHRENE , 9-ETHYL	3674757	92	191	47667	0	4247	
PHENANTHRENE , 2,5-DIMETHYL	3674666	97	206	42636	0	3797	
UNKNOWN			252	123050	0	3532	
11 H - BENZO [a] FLUORENE	2381217	84	216	68192	0	3410	
HEXA TRIA CONTANE	630068	83	57	119302	0	3410	
2,2' -BI NAPHTHALENE	612782	49	254	55733	0	2777	
NAPHTHALENE , 1,4,6- TRI CH3	2131422	80	155	30276	0	2695	
1 - ETHYL - NAPHTHALENE	1127760	96	141	82628	0	2675	
BENZENE , 1,1'-(1,2 -ETHENEDIYL)-, BIS	103300	49	57	30128	0	2675	
1,2 DI - CH3 - NAPHTHALENE	573988	97	156	70852	0	2307	
HEXA TRIA CONTANE	630068	85	57	75360	0	2164	
CHRYSENE , 3- METHYL	3351313	68	242	39396	0	1960	
1- HYDROXY CH3 PYRENE	0	21	218	38956	0	1940	
2,3 DI - CH3 - NAPHTHALENE	581408	92	156	52588	0	1715	
2-ETHYL- 4,4'- BI PYRIDINE	0	25	240	34474	0	1715	
N - CH3 - DIBENZ(E,G) ISO INDOLE	59788157	52	231	56675	0	1613	
2,7 DI - CH3 - NAPHTHALENE	582161	96	156	46920	0	1531	
PHENANTHRENE , 2,7-DIMETHYL	1576698	92	206	16488	0	1470	
CHRYSENE , 5-METHYL	3697243	53	242	28547	0	1429	
HEXA DECANE , 2,6,10,14 - TETRA METHYL	638368	89	57	38625	0	1245	
HEPTA COSANE	593497	89	57	41229	0	1184	
4- ISO PROPYL ANTI PYRENE	479925	25	215	22241	0	1102	
NAPHTHACENE , 5,12- di HYDRO	959024	73	229	21993	0	1102	
BENZ [a] ANTHRACENE , 7- METHYL	2541697	80	242	21971	0	1102	
1,4-ETHENO ANTHRACENE , 1,4- DI HYDRO	27765964	67	204	249751	0	1090	
N-CH3-DIBENZ(E,G) ISO INDOLE	59788157	52	231	37141	0	1062	
1,1' -BI NAPHTHALENE	604535	67	254	19750	0	980	
BENZO (A) PYRENE -- 4,5 OXIDE	0	37	268	34458	0	980	
1,2 - BENZO PERYLENE	0	62	276	25622	0	980	
DIBENZO [ah] ANTHRACENE	53703	71	278	22394	0	857	

APPENDIX 8-3B (cont'd.). Estimated sediment concentrations of probable compounds -Santa Fe Channel.

Sediment: S1CES063 CAS # COMPOUND NAME	SCAN PROB	BASE PEAK %	BASE PK ABUND ION	nG/G CONC. SCAN	nG/G CONC. SIM	SIM Q SCAN	VALUE
1,7 DI - CH3 - NAPHTHALENE	575371	95	156	25271	0	817	
1,1':2' , 1''- TERPHENYL	84151	40	215	15208	0	755	
ANTHRACENE , 2-METHYL	613127	78	192	171661	0	749	
BENZ [j] ACEANTHRYLENE, 3- CH3	93007	52	266	26018	0	735	
2 - ETHYL - NAPHTHALENE	939275	66	141	22098	0	715	
ANTHRACENE , 1-METHYL	610480	63	192	154585	0	675	
POOR MATCH			252	22816	0	653	
ANTHRACENE , 9-METHYL (1 - CH3 - PHENAN)	779022	20	192	146913	0	641	
BENZ [j] ACEANTHRYLENE, 3- CH3	93007	82	266	22292	0	633	
2- (3-METHYL BENZOYL) BENZOIC ACID	2159377	34	119	11027	0	551	
ALKANE -- NO MATCH	-	-	57	14377	0	551	
PYRENE , 1,3 (CH3)2	64401214	58	230	10353	0	510	
1,2' - BI NAPHTHALENE	4325740	73	254	10342	0	510	
DELTA.4a, 10b,.....ETC	67938889	52		17798	0	510	
2,2' -BI INDOLYL	40899998	32	232	9748	0	490	
BI PYRIMIDINE TETRONE .....ETC.	62880879	70	278	12043	0	470	
NO MATCH	-	-	95	8913	0	449	
BENZENE , 1,1'-METHYLENEBIS[4-ISO CYANO-	956627	52	218	8436	0	429	
PYRENE , 1,3- (CH3)2	40899998	26	230	8737	0	429	
2,2' -DI DEUTERIO - 1,1' -BI NAPHTHALENE	0	60	254	7832	0	388	
9H - FLUORENE , 3- CH3	2523399	91	165	86236	0	376	
9H - CARBAZOLE	86748	83	167	70452	0	307	
NO MATCH	-	-	119	6145	0	306	
POOR MATCH		25	69	6273	0	306	
2,3-DIHYDRO FLUORANTHENE	30339878	47	203	70155	0	306	
BENZO [1,2-b: 4,3-b'] DI THIOPHENE, 1-PHENYL-	16587589	52	266	8989	0	265	
BENZ [j] ACEANTHRYLENE, 3- CH3	93007	42	266	8868	0	245	
DIBENZOTHIOPHENE	132650	56	184	51508	0	225	
DIBENZOTHIOPHENE , 3 - METHYL	16587523	95	198	43840	0	191	
??	65753799	30	210	40886	0	178	
METHYL - FLUORENE	0	58	165	39200	0	171	
DIBENZO THIOPHENE	132650	69	184	39205	0	171	
NO MATCH			119	5612	0	163	
9H - FLUORENE , 2-CH3	1430973	81	165	35084	0	153	
(Z) -3-PHENYL-2-PROPENAL ((Z) -CINNAMALDEHYDE	57194691	48	131	4152	0	122	
9H - FLUORENE , 2,3- (CH3)2	4612639	86	179	25874	0	113	
9H - FLUORENE , 2,3- (CH3)2	4612639	48	179	19747	0	86	
9H - FLUORENE , 2 - METHOXY	2523468	42	196	17044	0	74	
3,6 - DIMETHYL DIBENZO THIOPHENE	31613044	13	208	16121	0	70	
1,1-bis (p-TOLYL) ETHANE	0	43	195	14221	0	62	
BENZENE, PENTACHLOROMETHOXY-	1825214	25	133	2228	0	61	
??	88441218	27	205	7852	0	34	
NO MATCH	-	-	179	7408	0	32	
NO MATCH	-	-	57	1988	0	9	

APPENDIX 8-3C. Estimated sediment concentrations of probable compounds -Richmond Inner Harbor Channel (* = internal standard).

Sediment: S1CES095 CAS # COMPOUND NAME	SCAN PROB	BASE PEAK %	BASE PK ABUND ION	nG/G CONC. SCAN	nG/G CONC. SIM	SIM Q SCAN	VALUE
*d-8 NAPHTHALENE	434	96	136	86074	695	695	98
*d-10 ACENAPHTHENE	435	94	164	70954	695	695	97
*d-10 FLUORANTHENE			212	249623	695	695	97
*d-12 BENZO b FLUORANTHENE	0	-	264	163728	695	695	99
*d-12 BENZO (ghi) PERYLENE	N/A		288	61583	695	695	97
PYRENE	252	94	202	73304	157.6	204.1	97
FLUORANTHENE	258	93	202	46471	116.7	129.4	98
BENZO - a - PYRENE	50328	95	252	24979	107.5	106.0	98
BENZO [ghi] PERYLENE	191242	72	276	12873	101.2	145.3	99
INDENO [1,2,3-cd] PYRENE	303	75	276	9814	88.8	110.8	97
BENZO b FLUORANTHENE	-	-	252	13780	83.8	58.5	98,M
CHRYSENE	274	89	228	18017	71.5	50.2	97
BENZO k FLUORANTHENE	292	-	252	11770	70.5	50.0	9,M
BENZO - e - PYRENE	192972	92	252	18024	69.5	76.5	97
*d-8 4,4' - DDE	52663624	81	326	7941	67	67	
BENZO [a] ANTHRACENE	56553	84	228	19906	61.7	55.4	15,M
PHENANTHRENE	203	94	178	21439	41.4	210.0	98
PERYLENE	198550	81	252	11424	41.1	48.5	95,M
*PCB - 030 TRI			186		35	35	80
*PCB - 065 TETRA			292		35	35	86,M
PHTHALATE ESTER, bis(2-ETHYL HEXYL)	603112	59	149	15451	30.1	43.0	98,M
NAPHTHALENE		-	128	2491	24.1	20.1	67,M
2 - CH3 - NAPHTHALENE			142	734	12.5	5.9	89,M
ACENAPHTHYLENE			152	1939	10.2	19.0	94,M
ANTHRACENE	209		178	4370	10.1	42.8	84
DIBENZ [ah] ANTHRACENE			278		9.0	0.0	89,M
1 - CH3 - NAPHTHALENE			142		7.6	0.0	89,M
ACENAPHTHENE			154	607	7.1	5.9	97
1- CH3 - PHENANTHRENE			192	2110	6.7	20.7	86,M
FLUORENE			166		5.7	0.0	99
4,4' - DDD			235		4.9	0.0	79,M,
2,6 - DI - CH3 - NAPHTHALENE			156		4.4	0.0	77,M
BIPHENYL			154		3.5	0.0	87,M
DIELDRIN			79		3.1	0.0	1,M,NS
2,3,5 - TRI - CH3 NAPHTHALENE			170		3.0	0.0	M
PCB-153 HEXA			79		2.9		M,NS
HEPTACHLOR EPOXIDE			353		2.3		91,M,NS
4,4' - DDE			246		2.3	0.0	16,M
4,4' - DDT			235		2.2		M,NS
HEPTACHLOR			100		2.0		21,M,NS
PCB-149 HEXA			360		1.5		M,FQ
PCB-138 HEXA			360		1.1		M,FQ
cis - CHLORDANE			375		1.0		99,M
2,4 - DDD			235		0.8		38,M,NS
ALKANE			57	48476	0.0	547.1	
CYCLOHEXASILOXANE , DODECAMETHYL	540976	52	73	65156	0.0	526.1	
HEXA DECANOIC ACID, METHYL ESTER	112390	94	74	46171	0.0	452.2	
TETRA DECANOIC ACID , METHYL ESTER	124107	95	74	36115	0.0	353.7	
HEPTA DECANE, 2,6,10,15- TETRAMETHYL	54833486	88	57	65711	0.0	278.9	
DI ETHYL PHTHALATE ESTER	266	86	149	24405	0.0	239.0	
9-OCTA DECANOIC ACID (Z)-, METHYL ESTER	112629	71	55	22781	0.0	223.1	
NONA COSANE (MW=408) C29 H60	630035	86	57	38545	0.0	163.6	
TETRA SILOXANE, (CH3)10 (R.T.=d-12 CHRYSENE)	141628	79	73	55397	0.0	154.2	
HEPTACOSANE (MW=380) C27 H56	593497	99	57	46957	0.0	130.7	
HENE ICOSANOIC ACID, METHYL ESTER	6064900	86	74	12805	0.0	125.4	
OCTA COSANE (MW=394) C28 H58	630024	86	57	43695	0.0	121.7	
PENTA DECANOIC ACID, METHYL ESTER	7132641	98	74	12256	0.0	120.0	
NONA DECANE	629925	29	57	12218	0.0	119.7	
TRIA CONTANE (MW=422) C30 H62	630068	73	57	27996	0.0	118.8	
DOCOSANE, 11-BUTYL-	13475768	95	57	40905	0.0	113.9	
HEXA TRIA CONTANE ?? (MW=506) C36 H74	630068		57	21391	0.0	90.8	
EICOSANE (MW=282) C20 H42	112958	81	57	30445	0.0	84.8	

APPENDIX 8-3C (cont'd.). Estimated sediment concentrations of probable compounds -Richmond Inner Harbor Channel.

Sediment: S1CES095 CAS # COMPOUND NAME	SCAN PROB	BASE PEAK %	BASE PK ABUND ION	nG/G CONC. SCAN	nG/G CONC. SIM	SIM Q SCAN	VALUE
ALKANE			57	8533	0.0	83.6	
17-OCTADECENE-14-yn-1-ol	18202283	55	79	7134	0.0	69.9	
MW = 472 C20 H40 O5 Si4	37148655	28	73	6662	0.0	65.3	
PENTA DECANE (MW=212) C15 H32	629629	67	57	21909	0.0	61.0	
ALKANE			57	6176	0.0	60.5	
ALKANE			57	5882	0.0	57.6	
NO MATCH			69	5868	0.0	57.5	
NO MATCH			79	5555	0.0	54.4	
ALKANE			67	4217	0.0	41.3	
1,1'-BIPHENYL, 2,2',3,4- TETRA CHLORO	52663599	99	292	4125	0.0	40.4	
1,1' - BIPHENYL, 2,4,6-TRICHLORO	35693926	99	256	4064	0.0	39.8	
NO MATCH			55	3972	0.0	38.9	
ALKANE			56	13559	0.0	37.8	
ALKANE			56	13292	0.0	37.0	
1-CYANOMETHYL-7-CH3-DIBENZOTHIOPHENE	88113977	28	237	12698	0.0	35.4	
NO MATCH			74	3534	0.0	34.6	
ALKANE			57	11383	0.0	31.7	
ALKANE			55	11310	0.0	31.5	
N-(2-ACETOXYETHYL)-1-ISOQUINOLONE	68152216	43	231	7080	0.0	30.1	
ALKANE			57	3047	0.0	29.8	
NO MATCH			55	2983	0.0	29.2	
ALKANE			57	3559	0.0	28.7	
ALKANE			57	10220	0.0	28.5	
ALKANE			57	3144	0.0	25.4	
ALKANE			57	9021	0.0	25.1	
PHTHALATE ESTER			74	2544	0.0	24.9	
1H - INDENE, 2,3 (OH)2, 1,1,3 (CH3)3, 3 PHENYL	3910358	68	221	2520	0.0	24.7	
ALKANE			57	3045	0.0	24.6	
ALKANE			57	2432	0.0	23.8	
NO MATCH			79	2399	0.0	23.5	
ALKANE			57	2056	0.0	20.1	
ALKANE			74	1991	0.0	19.5	
ALKANE			58	2308	0.0	18.6	
ALKANE			57	1852	0.0	18.1	
1,1'-BIPHENYL, 2,,2',3,3',5,5',6,6'- OCTA CHLORO	2136994	99	430	6205	0.0	17.3	
ALKANE			55	1644	0.0	16.1	
NO MATCH			191	1598	0.0	15.7	
BENZO[ghi]FLUORANTHENE	203123	42	226	5616	0.0	15.6	
ALKANE			57	1732	0.0	14.0	
DIBENZO [A,K] PYRENE	0	59	302	4466	0.0	12.4	
ALKANE			57	1105	0.0	10.8	
N-METHYL-DIBENZ(E,G)ISOINDOLE	59788157	45	231	2540	0.0	10.8	
ALKANE			177	848	0.0	8.3	
ALKANE			57	922	0.0	7.4	
ACETLYLENE DICARBONIC ACID, DIMETHYL ESTER	85111406	20	385	321	0.0	3.6	
NO MATCH			151	848	0.0	3.6	

APPENDIX 8-3D. Recovery of compounds from NTIS reference material, SRM 1941, Organics in Marine Sediments (quantified in scan mode using extracted ions).

COMPOUND	REPORTED conc ng/g	RECOVERY %
PERYLENE	399	71
ANTHRACENE	194	52
PHENANTHRENE	573	54
BENZO b FLUORANTHENE	610	154
BENZO k FLUORANTHENE	422	130
BENZO - a - PYRENE	543	97
PYRENE	968	90
CHRYSENE and TRIPHENYLENE	554	250
BENZO - e - PYRENE	453	131
FLUORANTHENE	1072	81
BENZO [a] ANTHRACENE	517	191
INDENO[1,2,3-CD]PYRENE	549	109
BENZO[GHI]PERYLENE	459	116
4,4' - DDT	1.07	4238
4,4' - DDD	9.99	103
4,4' - DDE	9.35	138
DIELDRIN	0.61	2317
CIS-CHLORDANE	1.98	73
TRANS-NONACHLOR	0.94	61
PCB - 052 TETRA	9.94	416
PCB - 153 HEXA	21.1	189
PCB - 101 PENTA	21.1	164
PCB - 138 HEXA	23.9	85
PCB - 118 PENTA	14.6	114
PCB - 105 PENTA	5.53	85
PCB - 180 HEPTA	13.7	103
PCB - 066 TETRA	21.5	19

APPENDIX 8-4A. Overlying water concentrations.

Overlying water samples taken at the study site in October 1991 (S1CH0) and February 1992 (S2AH0). Values are in ng/l. LC=Lauritzen Channel, RCH=Richmond Harbor Channel, SFCH=Santa Fe Channel, L=Below calibration curve, NC=Failed first ion check, NC3=Failed first and second ion check, NPA=No peak area, <DL=Below detection limit, ERR=Unknown or unspecified error, CO=Excess carry over, value not reliable, I=Interfering Peak, Fails RRT=Fails relative retention time, DRA=Diluted and reanalyzed.

Sample #	Site	TOC	24DDE	Code	44DDE	Code	24DDD	Code	44DDD	Code	24DDT	Code	44DDT	Code	ΣDDT	DIELDRIN	Code
S1CH0511	LC		0.12	L,NC3	0.89		3.42		14.03		1.17		2.30		21.92	10.46	NC3
S1CH0512	LC	1.7	0.24	L	2.40		8.70		34.15		4.22		10.10		59.81	18.65	
S1CH0513	LC	3.6	0.00	<DL	2.38		11.55		36.93		5.61		11.95		68.41	34.44	
S1CH0501	SFCH	1.7	0.00	<DL	0.27	L	0.86		3.74		0.00	<DL	0.25	L,NC3	5.12	3.07	NC,L
S1CH0502	SFCH	1.9	0.00	<DL	0.40	L	1.33		6.74		0.34	L,NC3	0.51	L	9.33	2.85	L
S1CH0503	SFCH		1.29	NC3	1.30		2.04		5.14		1.90		2.22	NC3	13.89	0.00	<DL
S1CH0521	RCH	1.7	0.00	FAILRRT	0.00	<DL	0.11	L,NC3	0.22	L,NC	0.00	<DL	FAIL RRT		0.33	0.00	<DL
S1CH0522	RCH	1.6	0.00	<DL	0.00	<DL	0.00	<DL	0.00	<DL	0.00	<DL	0.00	<DL	0.00	0.00	<DL
S1CH0523	RCH	0.6	0.00	<DL	0.11	L	0.30	L,NC	0.86	L	0.00	<DL	0.23	L	1.50	0.00	<DL
Detection Limit			0.06		0.06		0.10		0.10		0.22		0.17		1.02		
S2AH0001	LC	1.3	0.20		1.83		9.09		26.88		4.89		10.68		53.56	15.74	
S2AH0002	LC	1.6	0.20	L	1.79		8.53		25.66		4.51	NC	9.22		49.91	14.83	
S2AH0003	LC	1.3	0.20	L	1.61		8.14		24.82		4.14		8.32		47.23	13.30	
S2AH0004[1]	LC	1.0	0.00	FAIL RRT	0.00	FAILRRT	0.11	L,NC3	0.00	<DL	0.00	<DL	0.22	L,NC	0.33	0.00	<DL
S2AH0005	SFCH	1.2	0.00	DL	0.30	L	3.64		4.51		0.52	L,NC3	0.85	L	9.82	1.74	L,NC
S2AH0006[1]	SFCH	1.6	0.00	FAIL RRT	0.00	FAILRRT	0.14	L,NC3	0.00	<DL	0.00	<DL	0.00	<DL	0.14	0.00	<DL
S2AH0007	SFCH	1.1	0.00	<DL	0.29	L	1.21	L,NC	3.92		0.47	L,NC3	0.74	L,NC	6.63	1.67	L
S2AH0008	SFCH	1.3	0.00	FAIL RRT	0.28	L,NC3	1.33	L	3.96		0.36	L	0.74	L	6.66	1.68	L
S2AH0009	RCH	1.3	0.00	FAIL RRT	0.09	L,NC3	0.23	L,NC	0.71	L	0.00	<DL	0.43	L,NC	1.46	0.00	<DL
S2AH0010	RCH	1.3	0.00	FAIL RRT	0.08	L,NC3	0.51	L	0.88	L,NC3	0.00	<DL	0.00	FAIL RRT	1.47	0.00	<DL
S2AH0011[1]	RCH	0.9	0.00	FAIL RRT	0.00	<DL	0.16	L,NC3	0.00	<DL	0.00	<DL	0.00	<DL	0.16	0.00	<DL
S2AH0012	RCH		0.00	FAIL RRT	0.06	L,NC3	0.56	L,NC3	0.64	L,NC	0.00	<DL	0.00	FAIL RRT	1.27	0.00	FAIL RRT
Detection Limit			0.06		0.06		0.09		0.1		0.22		0.17				

[1] Field blank

APPENDIX 8-4B. Mytilus taxonomy.

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Message date: April 20, 1994

Message to: David T. Specht, EPA Office of Research and Development

FAX number: 1-503-867-4049

Pages including this one: 4

Dear David,

Finally I have examined all of the *Mytilus* specimens that you sent last June! I am in agreement with your initial assessment that most specimens are *Mytilus galloprovincialis*, with a slight possibility of a couple of *Mytilus trossulus*, or hybrids. For your purposes, I suspect it is fine to call them all *M. galloprovincialis*.

For my identifications I have primarily referred to: Seed, Raymond. 1992. Systematics evolution and distribution of mussels belonging to the genus *Mytilus*: an overview. *American Malacologica Bulletin* 9(2): 123-137.

I have attached to this fax, the approach we have taken in my forthcoming book on west coast bivalves (to be published in late 1994 or early 1995). You may cite this if you wish as: Coan, E.V., P. H. Scott, and F. R. Bernard. In Prep. *Bivalve Seashells of Western North American*. Santa Barbara Museum of Natural History, Santa Barbara, California. I would be happy to send you the mytilid portion of the bibliography if you are intrested.

Please accept my sincere apologies for taking nearly a year to get back to you with an answer. I must admit I dread working with these impossible mussels, and yours were no exception to the vexing problems in the group.

Feel free to contact me if you have further questions. I will be out of town from 22 April - 2 May.

Cordially,

Paul Scott

Genus Mytilus Linnaeus, 1758

Mytilus Linnaeus, 1758. Type species (SD Anton, 1839; ICZN Opinions 94, 1926, & 333, 1955): *M. edulis* Linnaeus, 1758. Recent, northeastern Atlantic.

Eumytilus Ihering, 1900. Type species (M?): *Mytilus edulis* Linnaeus, 1758.

Crassimytilus Scarlato & Starobogatov, 1979. Type species (OD): *Mytilus coruscus* A. A. Gould, 1861. Recent, northwestern Pacific.

Pacifimytilus Kafanov, 1984. Type species (OD): *Mytilus californianus* Conrad, 1837. Recent, northeastern Pacific.

Shell inequilateral, subtriangular and inflated. Periostracum thick, usually black or dark brown. Umbones terminal. Sculpture absent, or of low radial ribs that are divaricate in some species, sometimes with concentric undulations. Hinge with small ventral dysodont teeth and obscure tubercles, with marginal crenulations in some. Ligament external, attached to a nymph.

A temperate and cold water cosmopolitan genus, due to the very wide dispersal of members of the *Mytilus edulis* complex, in part as a result of shipping activity the past 500 years. Morphologically, the group is remarkably consistent, but several additional generic units have been proposed, one or two of which may be valuable at the subgeneric level after more careful evaluation. Additionally, *Mytilus californianus* Conrad, 1837, has sometimes been assigned to *Crenomytilus* Soot-Ryen, 1955, but absence of marginal crenulations on the pitted pseudonymphal ridge show it to be a *Mytilus, s.s.* The name is derived form the diminutive of the Latin *mya*, a sea mussel; the gender is masculine.

There has been much recent study of the species of the *Mytilus edulis* complex -- the smooth-shelled *Mytilus* -- from various points of view. At least three distinct taxa are involved. In the well-studied case of *M. edulis* and *M. galloprovincialis* in Europe, although the two taxa hybridize, and the hybrids are fertile, apparently larvae of hybrids are less viable, and this, together with differences in breeding times, are sufficient to maintain species integrity.

In one of the many unanswered questions about this complex, *M. edulis* itself has not yet been introduced into the northern Pacific by shipping or other means, although it is now present on both coasts of South America. Live specimens can be seen for sale in fish markets on the West Coast, imported from the western Atlantic. Perhaps it is just a matter of time before the taxonomic situation becomes still more complicated! *Mytilus edulis*, which is also illustrated here (Plate 15: xxxx), is narrow and elongate, rounded dorsally, somewhat blunt anteriorly, with a narrower ventral surface. The hinge is dark blue and curved. The anterior adductor muscle scar is large, and the posterior byssal retractor scar is narrow and elongate.

The native smooth mytilid of the North Pacific is *Mytilus trossulus*, which also occurs across the Arctic and in northern Europe. *Mytilus galloprovincialis* has been introduced into the eastern Pacific. The two form hybrids in central California, and it remains to be seen how distinct the species will remain. Populations in the Pacific north of Washington and in Asia have yet to be studied in detail, so there are many unresolved questions, and it is not easy to assign some material to a particular species.

Mytilus galloprovincialis Lamarck, 1819 Plate 15: 1a,b,c,

MEDITERRANAN MUSSEL

Mytilus galloprovincialis Lamarck, 1819: 126; *M. edulis diegensis* Coe, 1945a: 28; ?*M. (Mytilus) edulis zhirmunskii* Scarlato & Starobogatov, 1979a: 108; plus various synonyms in other parts of the world.

Triangular, expanded dorsally; anterior end pointed. Ventral margin broad laterally. Hinge light in color, straight. Anterior adductor muscle scar tiny; posterior byssal retractor scar broad. Length to 150 mm.

From approximately Westport, Mendocino Co., California (39.6°N) (McDonald & Koehn, 1988), to Manzanillo, Colima, Mexico (19.1°N) [LACM], but occurrences south of central Baja California are probably sporadic and do not represent breeding populations, in the intertidal zone - 5 m. Forming hybrids with *M. trossulus* in central California. Preliminary information suggests that populations may now be living in Puget Sound and the Strait of Georgia, British Columbia. Native to the Mediterranean, western France, and Britain and Ireland. Also introduced in other parts of the world, including South Africa, Australia, New Zealand, and Asia from Japan and Korea south to Hong Kong. Its introduction in Asia probably occurred long before that in the eastern Pacific.

Literature: J. H. MacDonald & Koehn (1988), Seed (1992). See also literature listed below.

Mytilus trossulus A. A. Gould, 1850 Plate 15: xxxxx

FOOLISH MUSSEL

Mytilus trossulus A. A. Gould, 1850c: 344; *M. glomeratus* A. A. Gould, 1851: 92; A. A. Gould, 1853: 402; *M. pedroanus* Conrad, 1855b: 15; *M. edulis "normalis"* Carpenter, 1857b: 197; *M. e. latissimus* Carpenter, 1857b: 197; *M. septentrionalis* Clessin, 1887: 58; *M. ficus* Dall, 1909a: 113; *M. (Mytilus) edulis kussakini* Scarlato & Starobogatov, 1979a: 109; *M. edulis declivis* Petrov, 1982: 77.

Outline variable, frequently expanded dorsally, as in *M. galloprovincialis*. Ventral surface relatively wide laterally; anterior end pointed. Hinge plate dark to light in color, curved. Anterior adductor muscle scar relatively small; posterior byssal retractor muscle scar relatively long and narrow. Length to 90 mm.

Some guesswork is involved in the listing of synonyms, based on the assumption that *Mytilus galloprovincialis* was introduced into the eastern Pacific after 1900. Clearly, more study is necessary.

In the Arctic from Banks Island, Northwest Territories (72.0°N), west to Point Barrow, Alaska (71.4°N) [LACM], throughout the Bering Sea, south to central California, and probably sporadically to southern California (about 33.5°N). From central California southward, forming hybrids with *M. galloprovincialis*. Eastward throughout the Aleutian Islands to Siberia, probably northern Japan. Also with disjunct populations in the Baltic Sea and northeastern Canada. In the intertidal zone, on rocks and pilings. Records from as far back as the Miocene in western North America are probably this species.

Additional study is needed concerning the relationships of *Mytilus coruscus* A. A. Gould, 1861, described from Japan and made the type species of *Crassimytilus* Scarlato & Starobogatov, 1979. Whereas the subgeneric unit may not be useful, it is undoubtedly a distinct species of the *M. edulis* group. It grows to a very large size, and specimens may be very flattened. *Mytilus crassitesta* Lischke, 1868, is apparently a synonym. There has been some confusion in the literature between this species and the very different *Crenomytilus grayanus* (Dunker, 1853).

Literature: J. H. MacDonald & Koehn (1988), N. L. MacGinitie (1959: 157), E. J. Moore (1983: 61), Scarlato (1981: 245-246), Seed (1992). See also literature listed after next species.

Literature on the *Mytilus edulis* species group

There may be more literature on *"Mytilus edulis"* than on any other mollusk. Because it is has only recently become understood that more than one species is involved, we cannot be certain in many cases which species was studied. Indeed, in some cases, more than one species may have been included in a single study. There would be little point in trying to divide the literature into species. The categories are selected here to aid in finding desired literature, but of course these categories overlap.

General: Field (1922), Haderlie & D. P. Abbott (1980: 361-363), Gosling (1992), Lubinsky (1980: 22-24), Lutz (1977a), E. J. Moore (1983: 61), Mossop (1922), Rankin (1918), Soot-Ryen (1955: 19-22), K. M. White (1937).

Anatomy & Morphology: Carriker (1979), Craig & Hallam (1963), Faussek (1897a), Fernandes & Seed (1983), Flemming (1883), Fritz et al. (1991), Giusti (1970), Kafanov & Romeiko (1987), Kellogg (1915: 652-656), Kessel (1940), La Course & Northrop (1978, 1981), Lutz & Hidu (1978, 1979), Mitton (1977), Rosen et al. (1978), Seed (1968), Soot-Ryen (1927), Theisen (1982), Trueman (1950), Waldron et al. (1976).

Ecology: B. L. Bayne & Widdows (1978), Coe (1946, 1948, 1953, 1956), D. A. Hancock (1965), Harger (1968, 1970a-c, 1972a-c), Harger & Landenberger (1971), Levinton & Lassen (1978a), C. L. Newcombe (1935a), Paine (1976), Paris (1960), Paul et al. (1978), Petraitis (1978), K. P. Rao (1953b), Raubenheimer & Cook (1990), Reish (1964a, b), Ross & Goodman (1974), Seed (1969a, b, 1980a), Suchanek (1981), van Winkle (1970).

Genetics, Relationships & Distribution: Ahmad & Beardmore (1976), Ahmad & Sparks (1970), Barsott & Meluzzi (1968), Beaumont et al. (1989), Boyer (1974), Crespo et al. (1990), Gosling (1984), Gosling & Wilkins (1981, 1985), W. S. Grant & Cherry (1985), A. N. Hodgson & R. T. F. Bernard (1986a, b), Hoeh et al. (1991), Ieyama & Inaba (1974), Knudsen (1980), Koehn (1975), Koehn & Mitton (1972), Koehn et al. (1973, 1976, 1984), Lessen & Turano (1978), Levinton & Koehn (1976), Levinton & Lassen (1978b), Levinton & Suchanek (1978), Lubinsky (1958), J. H. MacDonald & Koehn (1988), J. A. MacDonald et al. (1991), Milkman & Koehn (1977), Mitton & Koehn (1973), Newkirk (1980), Sarver (1988), Scarlato & Statobogatov (1979a), Seed (1978b, 1990, 1992), Skibinski & Beardmore (1979), Skibinski et al. (1978a, b, 1980, 1983), A. G. Smith (1944), Theisen (1978), Vlinîlî & Hvilson (1991), Varvio et al. (1988), N. P. Wilkins et al. (1983).

Parasites: F. R. Bernard (1969), Bradley & Siebert (1978), S. A. Campbell (1970), Giusti (1967), Pauley et al. (1966).

Physiology: B. F. Brown & R. C. Campbell (1972), Carr & Reish (1978), Coe (1945b), Coleman & Trueman (1971), Davenport (1983), Dodd (1965), Fankboner et al. (1978), Gaffney & Diehl (1986), Gilles (1972), Jokumsen & Fyhn (1982), JÖrgensen (1982), La Course &

Northrop (1983), R. F. Lee et al. (1972), Malone & Dodd (1967), J. M. Martin & Stephenson (1990), J. M. Martin et al. (1975), Pickens (1965), Price (1983), K. P. Rao (1954), Read (1962), Read & Cunningham (1967), Reish & Ayers (1968), Rossi & Reish (1976).

Reproduction & Growth: Barrois (1879a), Battle (1932), Board (1983), Breeze et al. (1963), R. A. Brown et al. (1976), Coe (1945b), Davalos (1983), De Schweinitz & Lutz (1976), Emmett et al. (1987), Gosling & Wilkins (1985), Hines (1979), Jamieson (1992), Jamieson & Heritage (1988), Jesperson & Olsen (1982), JÖrgensen (1976, 1981), Kniprath (1978), D. R. Moore & Reish (1969), Nelson (1928), Page (1988), Prieur (1983), Richards (1946), Seed (1969a, b), Strathmann (1987: 322-325), B. R. Wilson & Hodgkin (1967).

APPENDIX 8-5. Long-term kinetic exposure with Macoma nasuta.

Tissue residue values are in µg/kg on a dry tissue weight basis. L=Below calibration curve, NC=Failed first ion check, NC3=Failed first and second ion check, NPA=No peak area, <DL=Below detection limit, ERR=Unknown or unspecified error, CO=Excess carry over, value not reliable, I=Interfering Peak, Fails RRT=Fails relative retention time, DRA=Diluted and reanalyzed.

Sample #	Site	Time	Lipid	Dry/Wet	22DDE	Code	44DDE	Code	24DDD	Code	44DDD	Code	24DDT	Code	44DDT	Code	ΣDDT	DIELDRIN	Code
S1CCT001	0	0	0.057	0.197	<DL		4.03	L	<DL		<DL		<DL	NPA	<DL		4.03	<DL	
S1CCT002	1	1	0.064	0.186	2.47	L	52.00		203.89		1147.18		<DL	NPA	57.91	NC	1463.45	54.87	NC
S1CCT003	1	2	0.048	0.206	7.27	L	151.76		534.50		2944.45		12.63	NC,L	293.71		3944.32	123.90	
S1CCT004	1	3	0.046	0.197	12.67	L	304.07		891.32		5039.23		8.37	NC,L	221.37		6477.03	195.28	
S1CCT005	1	4	0.054	0.201	18.05	L	411.62		1169.54		8044.09	DRA	13.95	NC,L	225.45		9882.70	262.74	
S1CCT006	1	10	0.055	0.190	24.77	L	617.06		1536.14		8126.64	DRA	17.44	NC,L	291.55		10613.60	354.51	
S1CCT056	1	28	0.051	0.218	31.48		1335.53		2393.73		27905.58	DRA	30.01		150.66		31847.00	730.36	
S1CCT057	0	28	0.057	0.238	CO	NC3,L	CO		CO		CO		CO		CO		NC3,L CO	CO	L
S1CCT058	1	42	0.057	0.200	23.60	L	1620.25		3031.71		42455.38	DRA	57.46	NC	405.19		47593.59	1002.66	
S1CCT059	1	60	0.036	0.204	71.27		2546.32		3916.59		84915.77	DRA	51.93		648.35		92150.23	1232.37	
S1CCT060	1	90	0.035	0.198	78.42		2951.67		4752.00		85202.34	DRA	<DL		1634.60		94619.03	1264.32	
S1CCT061	0	90	0.049	0.209	CO	NC,L	CO		CO		CO		CO		CO		CO	CO	
Detection Limit Estimate							0.75		1.00		2.50		3.30		5.00		15.00	16.81	

APPENDIX 8-6. 28-Day bioaccumulation test results with Macoma nasuta.

Tissue residues values are in μg/kg on a dry tissue weight basis. L=Below calibration curve, NC=Failed first ion check, NC3=Failed first and second ion check, NPA=No peak area, <DL=Below detection limit, ERR=Unknown or unspecified error, CO=Excess carry over, value not reliable, I=Interfering Peak, Fails RRT=Fails relative retention time, DRA=Diluted and reanalyzed.

Sample #	Site	Grab	Lipid	Dry/Wet	24DDE	Code	44DDE	Code	24DDD	Code	44DDD	Code	24DDT	Code	44DDT	Code	ΣDDT	DIELDRIN	Code
S1CCT007	1	1	0.045	0.204	114.98		1943.41		6706.26		21682.74	DRA	<DL		336.35		30783.74	2013.56	
S1CCT008	1	2	0.036	0.233	20.45	L	1436.74		2583.57		12720.85	DRA	<DL		348.97		17110.59	822.40	
S1CCT009	1	3	0.053	0.216	25.28	L	1039.18		1612.86		11430.97		<DL		316.84		14425.13	392.81	
S1CCT010	1	4	0.049	0.209	14.34	L	696.70		1331.09		9918.31	DRA	13.68	NC,L	220.84		12194.97	387.03	
S1CCT011	1	5	0.032	0.218	29.54	L	1259.64		1632.75		11761.68	DRA	<DL		362.48		15046.09	367.63	
S1CCT012	2	1	0.050	0.190	110.26		2120.74		6207.05		22925.65	DRA	<DL		1163.57		32527.27	1366.37	
S1CCT013	2	2	0.029	0.208	9.26	L	872.90		1307.96		9020.78	DRA	<DL		441.26		11652.17	811.07	
S1CCT014	2	3	0.057	0.192	45.79	L	1564.07		3966.62		23245.76	DRA	50.63	NC	426.36		29299.23	1621.67	
S1CCT015	2	4	0.047	0.189	9.88	NC3,L	797.61		1323.67		7202.71	DRA	<DL		457.79		9791.67	546.73	
S1CCT016	2	5	0.049	0.174	10.94	NC3	1467.51		1340.46		11203.72	DRA	5.32	NC,L	591.95		14619.90	565.88	
S1CCT017	3	1	0.054	0.164	6.05	L	1094.61		1008.85	NC	11931.09		<DL		159.37		14199.98	443.30	
S1CCT018	3	2	0.043	0.204	31.21	L	1889.76		2308.15		15728.52	DRA	12.68	NC,L	348.60		20318.92	726.99	
S1CCT019	3	3	0.047	0.204	84.55		1677.59		5386.73		44065.01	DRA	54.91	NC,L	979.19		52247.98	1478.63	
S1CCT020	3	4	0.044	0.191	47.52	L	1374.64		2375.92		13068.18	DRA	<DL	NC	153.77	NC	17020.04	680.12	
S1CCT021	3	5	0.049	0.214	18.95	L	1310.53		2068.18		17120.26	DRA	32.46	NC,L	865.15		21415.53	765.94	
S1CCT022	4	1	0.038	0.204	8.43	NC3,L	290.64		192.06		2361.79		<DL		68.03	NC	2920.94	86.65	
S1CCT023	4	2	0.048	0.204	8.85	NC3,L	308.04		262.69		2490.63		11.45	NC,L	217.81	NC	3299.49	99.14	NC
S1CCT024	4	3	0.054	0.198	ERR		247.76		190.58	NC3	1868.74		4.78	NC3,L	80.07	NC	2391.92	<DL	
S1CCT025	4	4	0.060	0.185	10.19	L	227.46		257.37		2061.66		<DL		127.15	NC	2683.84	109.57	
S1CCT026	4	5	0.050	0.195	6.87	NC3,L	295.02		287.16		2285.92		17.21	NC,L	122.14	NC	3014.32	120.20	
S1CCT027	5	1	0.048	0.205	1.29	NC3,L	127.79		31.98	L	338.61		<DL		27.32	NC3,L	526.99	22.52	NC3,L
S1CCT029	5	3	0.055	0.218	<DL		77.05		24.28	L	194.31		<DL		<DL	NC3,L	295.65	<DL	
S1CCT030	5	4	0.045	0.186	33.31	L	105.21		47.88		214.13		31.81	L	41.57	NC	473.91	14.74	NC,L
S1CCT031	5	5	0.048	0.197	3.46	L	108.31		129.06		664.91		<DL		27.10	NC3,L	932.84	38.75	
S1CCT032	6	1	0.047	0.198	<DL		257.00		43.54	L	375.55		<DL		<DL	NC3,L	676.09	<DL	
S1CCT033	6	3	0.045	0.183	5.83	NC3,L	232.54		52.57	NC3	307.12		3.72	NC3,L	51.27	NC3	653.04	42.34	NC3
S1CCT034	6	4	0.048	0.176	20.01	L	260.25		88.81		337.86		<DL		38.11	NC3,L	745.04	56.65	
S1CCT035	6	5	0.048	0.193	11.56	L	383.26		72.27	L	427.41		<DL		40.23	NC3,L	934.72	63.18	L
S1CCT036	7	1	0.064	0.202	1.50	L	60.34		28.48		275.59		<DL		36.28	NC3	402.19	17.80	L
S1CCT037	7	2	0.055	0.166	<DL		34.65	L	8.09	NC3,L	83.63	L	<DL		<DL		126.37	<DL	
S1CCT038	7	3	0.050	0.221	36.73		106.95		121.96		338.77		26.89		63.26	NC	694.56	50.90	
S1CCT039	7	4	0.064	0.193	4.15	NC3,L	60.16		42.64	NC3	317.16		6.51	NC3,L	22.39	NC3,L	453.01	<DL	
S1CCT040	7	5	0.066	0.182	CO	L	CO		CO		CO		CO	L	CO	NC3	CO	CO	L
S1CCT041	8	1	0.070	0.203	2.16	L	26.85	L	<DL		50.55	L	<DL		25.98	NC3,L	105.55	<DL	
S1CCT042	8	2	0.064	0.227	CO	L	CO	L	CO	L	CO		CO		CO	NC3	CO	CO	
S1CCT043	8	3	0.040	0.202	3.15	NC3,L	35.51	L	12.52	L	47.22	L	<DL		<DL	NC3,L	98.40	<DL	
S1CCT044	8	4	0.046	0.182	<DL		29.26	L	8.71	L	71.35	L	<DL		20.14	NC3,L	129.46	<DL	
S1CCT045	8	5	0.061	0.166	<DL		23.07	L	<DL		41.88	L	<DL		<DL		64.95	<DL	

APPENDIX 8-6(cont'd.). 28-Day bioaccumulation test results with Macoma nasuta.

Sample #	Site	Grab	Lipid	Dry/Wet	24DDE	Code	44DDE	Code	24DDD	Code	44DDD	Code	24DDT	Code	44DDT	Code	ΣDDT	DIELDRIN	Code
S1CCT046	9	1	0.030	0.212	CO	L	CO	L	CO	L	CO	L	CO	L	CO	L	CO	CO	L
S1CCT047	9	2	0.047	0.201	2.12	NC3,L	7.91	L	2.84	NC3,L	8.48	L	<DL		<DL		21.35	<DL	
S1CCT048	9	3	0.051	0.200	<DL		7.97	NC3,L	<DL		10.54	NC3,L	<DL		7.14	NC3,L	25.65	<DL	
S1CCT049	9	4	0.040	0.209	<DL		10.50	NC3,L	<DL		13.26	NC,L	<DL		11.14	NC3,L	34.90	ERR	I,NC3
S1CCT050	9	5	0.047	0.207	CO		CO		CO	NC	CO		CO		CO		CO	CO	
S1CCT051	0	1	0.048	0.252	2.08	L	5.28	L	<DL		2.12	NC3,L	<DL		<DL		9.48	<DL	
S1CCT052	0	1	0.053	0.173	<DL		3.63	NC3,L	<DL		2.68	NC3,L	<DL		<DL		6.31	<DL	
S1CCT053	0	1	0.064	0.274	<DL		4.45	L	<DL		<DL		<DL		11.67	NC3,L	16.12	<DL	
S1CCT054	0	1	0.054	0.225	CO	L	CO	L	CO	L	CO	NC3,L	CO	NC3,L	CO	NC3,L	CO	CO	
S1CCT055	0	1	0.055	0.229	<DL		6.27	NC3,L	<DL		<DL		<DL		<DL		6.27	12.71	NC3,L
Detection Limit Estimate					0.75		1.00		2.50		3.30		5.00		15.00			16.81	

APPENDIX 8-7. Field infauna tissue residues.

Tissue residues values are in µg/kg on a dry tissue weight basis. Samples labeled S2ADT were collected by trawl rather than grab sampler, thus collection locations for these samples are less specific. LC=Lauritzen Channel, SFCH=Santa Fe Channel, L=Below calibration curve, NC=Failed first ion check, NC3=Failed first and second ion check, NPA=No peak area, <DL=Below detection limit, ERR=Unknown or unspecified error, CO=Excess carry over, value not reliable, I=Interfering Peak, Fails RRT=Fails relative retention time, DRA=Diluted and reanalyzed.

Sample #	Site	Species Name	Lipid	Dry/Wet	24DDE	Code	44DDE	Code	24DDD	Code	44DDD	Code	24DDT	Code	44DDT	Code	ΣDDT	DIELDRIN	Code
S1CDT012	1	Macoma nasuta	0.063	0.076	61.60		1693.96		2661.46		19408.87		60.11		1017.39		24903.40	822.74	
S1CDT022	2	M. nasuta	0.045	0.077	95.05		3345.12		4298.73		34805.90		1639.47		1483.46		45667.74	2550.45	
S1CDT030	3	M. nasuta	0.051	0.080	61.95	NC3	2662.43		2825.17		23252.72		37.46	NC3	839.14		29678.87	747.87	
S1CDT032	3	M. nasuta	0.057	0.057	<DL		1737.90		<DL		15217.10		<DL		595.67		17550.67	632.14	
S1CDT041	4	M. nasuta	0.075	0.071	38.86	NC3	1901.00		810.34		12770.97		<DL		223.97	L,NC3	15745.15	311.89	L,NC3
S1CDT052	5	M. nasuta	0.058	0.151	<DL		422.12		75.82		1041.60		<DL		152.67	NC3	1692.21	50.56	L,NC3
S1CDT082	8	M. nasuta	0.041	0.117	<DL		150.14		38.54	NC3	490.47	NC	<DL		39.76	L,NC3	718.91	<DL	
S1CHT042	4	M. nasuta	0.070	0.109	<DL		475.90		222.18		3207.17		<DL	NC3	170.83	L,NC3	4076.08	95.69	L,NC3
S2ADT212	LC	M. nasuta	-	0.104	165.31		4116.11		6317.35		58568.90		530.37		3954.49		73652.52	1928.04	
S1CDT015	1	Musculus sen.	0.088	0.075	114.54		1808.26		2882.68		14335.62		190.29	NC3	742.97		20074.37	409.14	
S1CDT025	2	M. senhousia	0.078	0.079	83.01		1397.78		2314.84		9914.43		260.80		1115.40		15086.26	642.71	
S1CDT035	3	M. senhousia	0.098	0.108	77.36		1196.49		1725.16		8687.58		84.29	NC3	607.54	NC	12378.42	269.32	
S2ADT235	LC	M. senhousia	0.072	0.0500	105.08		1535.83		1509.18		8956.46		466.00		1600.48		14173.04	213.32	L
S2ADT245	LC	M. senhousia	0.070	0.0580	93.35		1276.69		1494.44		8025.57		494.36		1456.72		12841.14	299.37	L,NC
S2ADT255	LC	M. senhousia	0.070	0.0740	70.72		1016.88		1259.85		6727.66		268.62		1021.96		10365.70	187.96	L
S2ADT265	LC	M. senhousia	0.067	0.0640	89.20		1463.91		1308.79		7779.29		397.56		1907.20		12945.95	140.41	
S1CDT016	1	Pachycerianthus	0.069	0.164	122.03		3521.08		2046.88		5815.44		433.65		1174.31		13113.39	370.55	
S1CDT026	2	P. fimbriatus	0.046	0.231	810.25		3870.77		18552.47		47497.15		100974.27	DRA234	117.26	DRA	405822.18	2180.97	
S1CDT036	3	P. fimbriatus	0.078	0.170	37.13		771.94		872.48		3032.93	NC	68.90	NC	299.76	NC,I	5083.14	226.39	
S1CDT046	4	P. fimbriatus	0.131	0.200	30.45		822.40		512.56		1333.41		156.92		300.74		3156.48	122.60	
S1CDT056	5	P. fimbriatus	0.011	0.399	1.26		32.83		12.23		81.82		<DL		21.49		149.64	<DL	
S1CDT113	13	P. fimbriatus	0.074	0.087	<DL		204.56		84.52		297.36		19.71	L,NC3	83.12	L,NC3	689.26	<DL	
S1CDT114	14	P. fimbriatus	0.081	0.104	10.22	L	274.81		105.72		325.45	NC	20.20	L,NC3	131.54	L,NC8	867.96	46.47	L,NC3
S2ADT304	SFCH	Potamocorbula	0.091	0.0670	<DL	NPA	111.06		64.19	L	355.56	L	<DL	NPA	<DL	L,<DL	530.81	ERR	
S2ADT344	SFCH	P. amurensis	0.090	0.1040	<DL	NPA	87.64		60.50	NC	279.08	L,NC	36.29		97.55	L,NC	561.05	64.94	L,NC3
S1CDT094	9	P. amurensis	0.054	0.100	<DL		22.82	L	7.03	L,NC3	29.49	L,NC3	<DL		<DL		59.34	<DL	
S1CDT023	2	Tapes japonica	0.120	0.086	25.97	L,NC	3369.14		1333.63		4721.92		814.47		1285.48		8550.61	675.51	
S1CDT033	3	T. japonica	0.111	0.116	11.56	L	288.79		493.48		2615.76		19.69	L	227.20		3656.48	156.42	L
S2ADT213	LC	T. japonica	0.084	0.0970	21.74		277.89		462.38		2594.69		105.57		832.50		4294.77	185.77	
S2ADT223	LC	T. japonica	0.090	0.0790	29.09	L,MI	274.68		463.03		2469.91		82.18	NC3	920.87		4239.76	206.95	L
S2ADT233	LC	T. japonica	0.078	0.0740	18.07	L,NC	252.34		396.51		2521.52		72.67	NC3	566.97		3828.10	103.98	L,NC
S2ADT263	LC	T. japonica	0.058	0.0853	11.44		219.68		356.15		1954.37		68.76		433.19		3043.58	123.31	
S2ADT273	LC	T. japonica	0.069	0.0740	20.53		244.10		366.65		2036.76		68.77		420.58		3157.39	137.58	

Species name abbreviations:
Macoma nasuta = "bent-nose" clam
Musculus senhousia = "mud" mussel
Pachycerianthus fimbriatus = burrowing anenome
Potamocorbula amurensis = "Asian" clam
Tapes japonica = Manila clam

APPENDIX 8-8. Tissue residues in field-collected mussels.

Tissue residue values are in µg/kg on a dry tissue weight basis. Lipid values are in proportions of dry weight. Samples labeled S1CHT and S2AHT were collected in Oct. 91 and Feb. 92 respectively. #Comp=number of individuals in sample, LC=Lauritzen Channel, SFCB=Santa Fe Channel Boathouse, SFCH=Santa Fe Channel, RCH=Richmond Harbor Channel L=Below calibration curve, NC=Failed first ion check, NC3=Failed first and second ion check, NPA=No peak area, <DL=Below detection limit, ERR=Unknown or unspecified error, CO=Excess carry over, value not reliable, I=Interfering Peak, Fails RRT=Fails relative retention time, DRA=Diluted and reanalyzed.

Sample #	Species	Site	Lipid	Dry/Wet	24DDE	Codes	44DDE	Codes	24DDD	Codes	Codes	44DDD	Codes	24DDT	Codes	44DDT	Codes	SUM DDT	DIELDRIN	#Comp
S1CHT501	M. gal.	SFCH	0.101	0.212	18.71		290.88		232.07			1314.59		27.66		204.87		2088.77	131.67	1
S1CHT502	M. gal.	SFCH	0.118	0.194	23.85	L	356.52		282.52			1493.58		34.90	L	219.07	NC	2410.44	166.01	1
S1CHT503	M. gal.	SFCH	0.109	0.195	21.97	L	352.78		276.72			1541.49		42.67	L	242.37		2478.00	109.10	1
S1CHT504	M. gal.	SFCH	0.131	0.204	29.95		409.31		308.95			1694.70		53.52		240.98	NC	2737.41	199.06	1
S1CHT505	M. gal.	SFCH	0.115	0.184	36.63		468.75		527.87			2211.95		82.78		339.51	NC	3667.48	178.03	1
S1CHT511	M. gal.	LC	0.050	0.168	74.05		1350.96		1609.00			6096.55		605.48	NC	2394.00		12130.05	548.84	1
S1CHT512	M. gal.	LC	0.070	0.144	174.54		2828.04		2959.14			9135.80		726.94		1900.72		17725.18	606.87	1
S1CHT513	M. gal.	LC	0.119	0.224	276.14		3500.39		5206.34			23767.02	DRA	3681.85		9654.36	DRA	46086.10	1693.16	1
S1CHT514	M. gal.	LC	0.106	0.137	111.42		2171.48		1932.06			8144.84		1027.67		3384.97		16772.44	732.89	1
S1CHT515	M. gal.	LC	0.064	0.175	274.02		4038.27		5080.91			29347.68	DRA	2563.00		5432.13		46736.02	1162.73	1
S1CHT521	M. gal.	RCH	0.140	0.187	7.35	L	82.02		42.79	NC,L	NC,L	232.84		<DL		43.69	NC,L	408.69	15.87	1
S1CHT522	M. gal.	RCH	0.082	0.205	3.87	NC,L	64.60		27.58	NC,L		130.61		<DL		29.17	NC,L	255.83	<DL	1
S1CHT523	M. gal.	RCH	0.098	0.194	5.77	NC,L	52.15		19.20	L		98.47		<DL		30.10	NC,L	205.70	<DL	1
S1CHT524	M. gal.	RCH	0.081	0.143	2.04	NC,L	34.99	L	12.90	NC,L	L	55.99	L	<DL		24.06	NC,L	129.97	39.17	1
S1CHT525	M. gal.	RCH	0.081	0.146	1.76	NC,L	22.07	L	9.68	NC,L		28.54	L	<DL		16.58	NC,L	78.64	<DL	1
608103-1	M. cal.	Control	0.057	0.174	<DL		6.14	L	<DL			<DL		<DL		<DL		6.14	<DL	1
608103-2	M. cal.	Control	0.088	0.181	<DL		8.40	L	<DL			<DL		<DL		<DL		8.40	<DL	1
608103-3	M. cal.	Control	0.049	0.186	<DL		8.50	L	<DL			<DL		<DL		<DL		8.50	<DL	1
608103-4	M. cal.	Control	0.052	0.172	<DL		6.96	L	<DL			<DL		<DL		<DL		6.96	<DL	1
608103-5	M. cal.	Control	0.048	0.175	<DL		9.11	L	<DL			<DL		<DL		<DL		9.11	<DL	1
Detection Limit					1.19		1.78		3.56			3.56		4.04		8.31			10.09	
S2AHT501	M. gal.	SFCB	0.094	0.0800	12.45		230.29		151.50	0.00		675.05		97.58		239.02		1405.89	81.24	5
S2AHT502	M. gal.	SFCB	0.086	0.1560	40.97		492.23		265.67		NC3	1215.63		222.58		524.93		2762.01	91.83	1
S2AHT503	M. gal.	SFCB	0.083	0.0680	<DL	NPA	198.01		161.07		L	739.21		134.46		324.60	NC	1557.33	53.40	1
S2AHT504	M. gal.	SFCB	0.095	0.1220	18.21		184.36		151.85			678.78		123.16		272.11	NC	1428.46	79.40	1
S2AHT505	M. gal.	SFCB	0.054	0.0850	7.10		111.15		75.78		NC3	306.37		38.51		97.69	NC	636.61	43.60	1
S2AHT511	M. gal.	LC	0.054	0.0830	46.01		424.29		447.27			1336.90		754.65		1130.10		4139.23	225.83	1
S2AHT512	M. gal.	LC	0.074	0.0900	141.59		1723.23		1098.00			4062.47		2338.69		3902.07		13266.05	253.25	1
S2AHT513	M. gal.	LC	0.059	0.0980	52.02		899.68		746.01			2227.50		1343.93		2171.69		7440.82	265.55	1
S2AHT514	M. gal.	LC	0.049	0.0900	46.54		465.28		481.71			1556.18		860.18		1484.92		4894.79	217.99	1
S2AHT515	M. gal.	LC	0.034	0.0820	74.49		669.30		963.40			2884.00		1289.16		2337.14		8217.49	450.62	1
S2AHT521	M. gal.	RCH	0.084	0.1240	4.54		75.75		44.55			181.53		27.15		75.41	NC	408.93	25.61	1
S2AHT522	M. gal.	RCH	0.063	0.1070	2.15		48.46		25.07		L	102.69		23.08		59.00	NC	260.45	18.83	1
S2AHT523	M. gal.	RCH	0.051	0.0900	2.84		42.66		30.33		L,NC3	117.25		28.66		72.84	NC	294.57	117.16	1
S2AHT524	M. gal.	RCH	0.123	0.1150	6.20	MI	79.73		56.05			214.95		37.04		106.03	NC	500.00	79.63	1
S2AHT525	M. gal.	RCH	0.093	0.1250	2.55		54.40		25.16		L,NC3	107.14		27.27		72.55	NC	289.07	12.50	1
Detection Limit					0.47		0.32		0.56			2.35		1.36		2.05			2.56	

M. gal. = Mytilus galloprovincialis; M. cal. = Mytilus californianus; see Section 8.4.2.

APPENDIX 8-9. Tissue residues in trawl-collected fishes and megabenthos.

Pollutant values are µg/kg dry weight. Lipid values are proportions of dry weight. #Comp=number of individuals in sample, LC=Lauritzen Channel, SFCH=Santa Fe Channel, RCH=Richmond Harbor Channel, L=Below calibration curve, NC=Failed first ion check, NC3=Failed 1st and 2nd ion check, NPA=No peak area, <DL=Below detection limit, I=Interfering Peak, DRA=Diluted and reanalyzed. M=Muscle, W=Whole organism, TM=Tail Muscle, EM=Embryo Mass

Sample #	Site	Species	Lipid	Dry/Wet	24DDE	CODE	44DDE	CODE	24DDD	CODE	44DDD	CODE	24DDT	CODE	44DDT	CODE	SUM DDT	DIELDRIN	CODE	#Comp	Tissue
S1CHT201	LC	Cymat. aggr	0.091	0.229	44.63		3750.54		1385.39		5796.55	DRA	566.69		1609.20		13153.00	903.69		1	M
S1CHT202	LC	Cymat. aggr	0.096	0.222	26.66		1102.48		264.49		1580.63		213.70		391.40		3579.36	522.94		1	M
S1CHT203	LC	Cymat. aggr	0.107	0.227	33.50		2238.59		1052.66		3970.62	DRA	265.62		817.81		8378.81	548.40		1	M
S1CHT204	LC	Cymat. aggr	0.076	0.217	19.83		2444.50		410.92		3481.14		102.14		336.26		6794.78	258.23		1	M
S1CHT205	LC	Cymat. aggr	0.106	0.217	40.41		2697.80		1128.13		7993.74		189.94		750.72		12800.74	447.83		1	M
S1CHT221	LC	Anchoveta	0.092	0.195	<DL		100.79		45.42		247.56		29.40		77.54		500.70	78.54	NC3	1	W
S1CHT231	LC	Morone saxi	0.064	0.203	6.53	NC3	248.65		57.19		282.37		59.01		94.95	NC	748.70	52.27		1	M
S1CHT241	LC	Morone saxi	0.067	0.197	12.35	NC3	887.67		189.16		1231.97		67.87		231.61		2620.62	204.92	NC3	1	M
S1CHT251	LC	Cancer ant.	0.062	0.152	<DL		1914.81		52.87		310.33		<DL		68.77		2346.78	73.25	NC	1	M
S1CHT301	SFCH	Cymat. aggr	0.139	0.221	10.15		765.47		138.42		1013.99		46.60		175.86		2150.50	84.72		1	M
S1CHT321	SFCH	Gobiid fish	0.152	0.213	15.61		1055.04		74.25		2382.78		14.19	NC3	214.81	NC	3756.68	144.67		1	W
S1CHT322	SFCH	Gobiid fish	0.131	0.219	11.60		874.76		70.81		2416.03		13.69		196.84	NC	3583.73	168.52		1	W
S1CHT323	SFCH	Gobiid fish	0.130	0.214	8.20		699.21		44.63		1649.34		10.33	NC3	141.13	NC	2552.84	96.43		1	W
S1CHT324	SFCH	Gobiid fish	0.118	0.216	6.95		665.90		42.03		1634.34		3.87	NC3	141.89	NC	2494.97	100.04		1	W
S1CHT325	SFCH	Gobiid fish	0.139	0.220	11.29		822.41		67.13		2216.83		13.35	NC3	188.49	NC	3319.51	129.14		1	W
S1CHT331	SFCH	Anchoveta	0.092	0.206	17.10		859.14		198.82		1675.26		129.66		377.24		3257.22	83.54		5	W
S1CHT401	RCH	Cymat. aggr	0.098	0.216	<DL		28.07		7.61		35.62	NC	<DL	NC3	6.35	L,NC3	77.65	<DL		1	M
S1CHT402	RCH	Cymat. aggr	0.073	0.202	2.46		39.65		8.52		54.36		4.20		11.96		121.15	<DL		1	M
S1CHT403	RCH	Cymat. aggr	0.102	0.217	13.48		79.78		28.63		111.80		14.13	NC3	20.52	NC	268.34	26.09	NC3	1	M
S1CHT404	RCH	Cymat. aggr	0.057	0.206	<DL		23.04		4.57	NC	15.73	NC	<DL		<DL		43.35	<DL		1	M
S1CHT405	RCH	Cymat. aggr	0.057	0.208	<DL		22.85		<DL		21.77		<DL		<DL		44.62	<DL		1	M
S1CHT431	RCH	Anchoveta	0.089	0.207	8.09		310.16		41.79		339.68		24.30	NC	80.27	NC	804.30	20.22	NC3	5	W
S1CHT432	RCH	Genyon. lin	0.090	0.212	2.48		64.11		10.26		113.22		<DL		16.68		206.75	20.87		3	M
S1CHT441	RCH	Crangon fr.	0.066	0.241	1861.60		1837.91		1402.78		1555.09		1345.86		1079.32		9082.56	287.54		10	TM
S1CHT451	RCH	Gobiid fish	0.037	0.202	13.04		53.81		11.49		87.85		11.58		23.15		200.91	<DL		1	M
S2AHT211	LC	Gobiid fish	0.105	0.214	22.45		11107.70		147.15	0.00	11585.69		110.64		2267.29		25235.91	947.55		1	W
S2AHT221	LC	Crangon fr.		0.245	<DL	NPA	294.63		<DL	L,NC,<D	<DL	L,NC,	<DL	NPA	<DL	NPA	294.63	<DL	NPA	3	TM
S2AHT251	LC	Crangon fr.	0.060	0.255	<DL	NPA	312.49		<DL	NPA	<DL	NPA	<DL	NPA	<DL	NPA	312.49	<DL	L,NC3,<DL	2	TM
S2AHT301	SFCH	Crangon fr.	0.064	0.230	<DL	NC,<DL	83.34		5.27	NC	57.		<DL	NC3,<DL	10.45	L	156.51	<DL	NPA	10	TM
S2AHT303	SFCH	Parophrys v	0.083	0.228	<DL	NC3,<DL	192.97		64.57	NC3	423.81	NC3	24.49	NC3	192.52	NC3	898.37	<DL	L,<DL	2	W
S2AHT321	SFCH	Crangon fr.	0.079	0.225	<DL	NPA	106.54		6.75	NC	66.16		<DL	<DL	15.13	NC3	194.58	27.02	NC3	4	W
S2AHT323	SFCH	Platich. st	0.049	0.286	4.09		150.75		21.59		303.47		4.84	NC3	20.65	NC	505.39	25.61	NC3	1	W
S2AHT341	SFCH	Cymat. aggr	0.083	0.208	4.69		380.90		87.91		704.76		31.47		97.76		1307.49	68.43		1	M
S2AHT343	SFCH	Crangon fr.	0.059	0.219	2.73	NC	56.30		4.11	NC	52.24		<DL	NPA	<DL	NPA	115.38	<DL	NPA	10	TM
S2AHT345	SFCH	Cithar. sti	0.083	0.222	97.50		2825.74	DR	299.02		4444.74		404.10		1626.48		9697.59	101.12	DRA	1	W
S2AHT401	RCH	Gobiid fish	0.056	0.208	2.81		77.41		12.67		336.45	NC	<DL	NC3,<DL	26.84	NC	456.18	26.59		2	W
S2AHT402	RCH	Crangon fr.	0.051	0.312	<DL	NPA	12.85		<DL	NPA	<DL	NPA	<DL	NPA	<DL	NPA	12.85	<DL	NPA	60	TM
S2AHT451	RCH	Gobiid fish	0.060	0.189	<DL	NPA	109.29		19.34		473.86	NC	<DL	NPA	58.62	NC	661.10	62.80		1	M
S2AHT452	RCH	Crangon fr.	0.065	0.225	<DL	NPA	30.78		<DL	NPA	ERR		<DL	NPA	<DL	NPA	30.78	25.51		10	TM
S2AHT454	RCH	Cithar. stig.		0.222	5.04		62.55		15.26	NC	126.20	NC	26.01	NC	35.79	NC	270.85	<DL	L,NC3,<DL	4	W
S2AHT461	RCH	Cancer ant.	0.039	0.165	0.68		108.22		<DL	0.00	<DL		<DL		<DL		108.90	10.73		1	M
S2AHT463	RCH	Cancer ant.	0.042	0.195	<DL	NPA	48.26		1.45	NC3	<DL	L,NC3	<DL	NPA	<DL	L,NC,<DL	49.71	<DL	NPA	1	M
S2AHT464	RCH	Cancer ant.	0.047	0.226	<DL	NPA	117.54		1.42	NC3	23.60	NC3	<DL	NPA	<DL	L,NC3,<DL	142.56	31.63	NC3	1	M
S2AHT465	RCH	Cancer ant.	0.040	0.184	<DL	NPA	140.82		1.97		<DL	L,NC3,<DL	<DL	NC,<DL	<DL	NPA	142.79	11.50	L,NC3	1	M
S2AHT466	RCH	Cancer ant.	0.038	0.214	<DL	NPA	123.92		0.90	0.00	22.35		1.05		2.56		150.78	9.23		1	M
S2AHT481	RCH	Cancer ant.	0.101	0.158	3.41		1310.96		13.47		95.58	NC	5.60		9.41	L,NC3	1438.43	5.46	L,NC3	1	E
S2AHT482	RCH	Cancer ant.	0.271	0.463	2.18		396.76		7.59		57.58	NC	2.52	NC3	4.61	L,NC	471.24	8.70		1	E

Anchoveta = probably Engraulis mordax, northern anchovy; Cancer ant. = Cancer antennarius, rock crab; Cithar. stig = Citharichthys stigmaeus, speckled sanddab; Crangon fr. = Crangon franciscorum, bay shrimp; Cymat. aggr. = Cymatogaster aggregatus, shiner surfperch; Genyon. lin. = Genyonemus lineatus, white croaker; Gobiid fish, probably Lepidogobius lepidus, bay goby; Morone saxi. = Morone saxitalis, striped bass; Parophrys v. = Parophrys vetulus, Engish sole; Platich. st. = Platichthys stellatus, starry flounder.

APPENDIX 8-10. Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S1CHT 201 repl 1, 2, & 3	S1CHT 211 repl 1, 2, & 3	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	Whole Body
TISSUE TYPE	Skinned fillets	Dissection residua	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	93.666
TISSUE WEIGHT @ DISSECTION, G WT WT	27.055	66.611	na
PERCENT OF GROSS WET WEIGHT	28.9%	71.1%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	27.055	61.442	na
NUMBER INDIVIDUALS IN COMPOSITE	5	5	5
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	318; $\bar{x}$ = 276 269; rsd = 242; 13.8%	2,731; $\bar{x}$ = 7,483 4,596; rsd = 15,121; 88.9%	7,759
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	26,263; $\bar{x}$ = 23,206 22,716; rsd = 20,640; 12.3%	193,705; $\bar{x}$ =199,655 203,896; rsd = 201,365; 2.7%	222,862
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	9,844; $\bar{x}$ = 8,572 7,904; rsd = 7,969; 12.9%	97,185; $\bar{x}$ = 96,697 95,787; rsd = 97,119; 0.8%	105,269
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	41,826; $\bar{x}$ = 35,877 32,790; rsd = 33,015; 0.1%	550,273; $\bar{x}$ =549,852 552,671; rsd = 546,610; 0.6%	585,729
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	4,028; $\bar{x}$ = 3,506 3,167; rsd = 3,324; 13.1%	42,298; $\bar{x}$ = 35,304 33,039; rsd = 30,574; 17.5%	38,810
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	11,503; $\bar{x}$ = 9,957 9,109; rsd = 9,259; 13.5%	137,485; $\bar{x}$ =133,022 131,956; rsd = 129,625; 3.1%	142,979
SUM ALL DDT CONGENERS, NG	81,395	1,022,013	1,103,408
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	3,009	15,343	11,780
PORTION WEIGHT. @ DISSECTION X CONC. DIELDRIN,NG/PORTION	6,456; $\bar{x}$ = 5,592 4,957; rsd = 5,362; 13.9%	59,417; $\bar{x}$ = 60,083 60,350; rsd = 60,483; 1.0%	65,675
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	207	902	701

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S1CHT 202	S1CHT 212	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	WHOLE BODY
TISSUE TYPE	Skinned fillets	Dissection residua	na
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	14.789
TISSUE WEIGHT @ DISSECTION, G WT WT	5.011	9.778	na
PERCENT OF GROSS WET WEIGHT	33.9%	66.1%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	5.011	9.217	na
NUMBER INDIVIDUALS IN COMPOSITE	1	1	1
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	30	117	147
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	1,226	11,206	12,432
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	294	3,941	4,235
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	1,758	24,230	25,988
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	238	3,403	3,640
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	435	7,324	7,759
SUM ALL DDT CONGENERS, NG	3,982	50,220	54,202
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	795	5,136	3,665
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	582	5,867	6,449
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	116	600	436

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S1CHT 203	S1CHT 213	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	WHOLE BODY
TISSUE TYPE	Skinned fillets	Dissection residua	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	19.497
TISSUE WEIGHT @ DISSECTION, G WT WT	6.408	13.089	na
PERCENT OF GROSS WET WEIGHT	32.9%	67.1%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	6.408	12.367	na
NUMBER INDIVIDUALS IN COMPOSITE	1	1	1
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	49	223	271
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	3,256	24,490	27,746
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	1,531	5,707	7,238
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	5,776	41,859	47,634
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	386	1,623	2,009
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	1,190	6,584	7,773
SUM ALL DDT CONGENERS, NG	12,188	80,484	92,672
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	1,902	6,149	4,753
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	798	2,644	3,442
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	125	202	177

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S1CHT 204	S1CHT 214	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	WHOLE BODY
TISSUE TYPE	Skinned fillets	Dissection residua	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	10.992
TISSUE WEIGHT @ DISSECTION, G WT WT	3.782	7.050	na
PERCENT OF GROSS WET WEIGHT	35.2%	64.1%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	3.872	6.704	na
NUMBER INDIVIDUALS IN COMPOSITE	1	1	1
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	17	247	263
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	2,054	15,031	17,085
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	345	8,890	9,235
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	2,925	50,203	53,128
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	86	2,383	2,469
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	283	8,932	9,215
SUM ALL DDT CONGENERS, NG	5,709	85,686	91,395
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	1,474	12,154	8,315
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	217	3,525	3,742
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	56	500	340

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S1CHT 205	S1CHT 215	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	Whole Body
TISSUE TYPE	Skinned fillets	Dissection residua	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	17.13
TISSUE WEIGHT @ DISSECTION, G WT WT	5.823	11.307	na
PERCENT OF GROSS WET WEIGHT	34.0%	66.0%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	5.823	10.633	na
NUMBER INDIVIDUALS IN COMPOSITE	1	1	1
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	51	441	492
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	3,409	26,877	30,286
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	1,425	14,925	16,351
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	10,101	83,242	93,343
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	240	2,748	2,988
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	949	12,913	13,681
SUM ALL DDT CONGENERS, NG	16,715	141,145	157,320
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	2,778	12,483	9,184
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	566	4,738	5,304
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	97	419	310

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S2AHT 341	S2AHT 342	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	WHOLE BODY
TISSUE TYPE	Skinned fillets	Dissection residua	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	16.083
TISSUE WEIGHT @ DISSECTION, G WT WT	5.985	10.098	na
PERCENT OF GROSS WET WEIGHT	37.2%	62.8%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	5.985	9.293	na
NUMBER INDIVIDUALS IN COMPOSITE	1	1	1
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	6	50	56
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	474	2,949	3,423
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	109	909	1,018
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	877	7,866	8,744
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	39	273	312
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	122	1,050	1,172
SUM ALL DDT CONGENERS, NG	1,628	13,097	14,725
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	272	1,297	916
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	85	555	641
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	14	55	40

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S1CHT 401	S1CHT 411	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	WHOLE BODY
TISSUE TYPE	Skinned fillets	Dissection residua	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	18.794
TISSUE WEIGHT @ DISSECTION, G WT WT	6.801	11.993	na
PERCENT OF GROSS WET WEIGHT	36.2%	63.8%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	6.801	11.140	na
NUMBER INDIVIDUALS IN COMPOSITE	1	1	1
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	0	24	24
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	41	180	221
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	11	48	59
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	52	588	640
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	0	24	24
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	9	36	45
SUM ALL DDT CONGENERS, NG	114	899	1,013
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	17	75	54
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	0	0	0
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	0	0	0

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S1CHT 402	S1CHT 412	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	WHOLE BODY
TISSUE TYPE	Skinned fillets	Dissection residua	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	8.264
TISSUE WEIGHT @ DISSECTION, G WT WT	2.732	5.532	na
PERCENT OF GROSS WET WEIGHT	33.1%	66.9%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	2.732	4.837	na
NUMBER INDIVIDUALS IN COMPOSITE	1	1	1
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	1	11	12
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	22	111	133
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	5	28	32
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	30	503	533
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	2	17	19
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	7	39	45
SUM ALL DDT CONGENERS, NG	67	708	775
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	25	128	94
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	0	0	0
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	0	0	0

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S1CHT 403	S1CHT 413	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	WHOLE BODY
TISSUE TYPE	Skinned fillets	Dissection residua	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	31.382
TISSUE WEIGHT @ DISSECTION, G WT WT	9.731	21.650	na
PERCENT OF GROSS WET WEIGHT	31.0%	69.0%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	9.731	20.213	na
NUMBER INDIVIDUALS IN COMPOSITE	1	1	1
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	29	43	72
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	168	844	1,013
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	60	281	342
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	236	3,529	3,765
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	30	152	181
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	43	217	260
SUM ALL DDT CONGENERS, NG	567	5,066	5,633
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	58	234	179
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	55	325	380
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	6	15	12

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S1CHT 404	S1CHT 414	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	WHOLE BODY
TISSUE TYPE	Skinned fillets	Dissection residua	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	11.941
TISSUE WEIGHT @ DISSECTION, G WT WT	3.696	8.245	na
PERCENT OF GROSS WET WEIGHT	31.0%	69.0%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	3.696	7.046	na
NUMBER INDIVIDUALS IN COMPOSITE	1	1	1
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	0	8	8
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	18	214	232
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	3	33	36
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	12	635	647
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	0	16	16
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	0	33	33
SUM ALL DDT CONGENERS, NG	33	940	973
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	9	114	81
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	0	0	0
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	0	0	0

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S1CHT 405	S1CHT 415	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Cymatogaster aggregata	Cymatogaster aggregata	WHOLE BODY
TISSUE TYPE	Skinned fillets	Dissection residua	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	11.893
TISSUE WEIGHT @ DISSECTION, G WT WT	3.957	7.936	na
PERCENT OF GROSS WET WEIGHT	33.3%	66.7%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	3,957	6.728	na
NUMBER INDIVIDUALS IN COMPOSITE	1	1	1
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	0	8	8
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	19	214	233
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	0	32	32
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	18	714	732
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	0	32	32
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	0	40	40
SUM ALL DDT CONGENERS, NG	36	1,040	1,076
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	9	131	91
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	0	0	0
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	0	0	0

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S2AHT 221	S2AHT 222	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Crangon franciscorum	Crangon franciscorum	WHOLE BODY
TISSUE TYPE	tail muscle	Dissection residua (estim.)*	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	1.013
TISSUE WEIGHT @ DISSECTION, G WT WT	0.441	0.576*	na
PERCENT OF GROSS WET WEIGHT	43.5%	56.9%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	0.441	0.516*	na
NUMBER INDIVIDUALS IN COMPOSITE	3	3	3
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	0	0	0
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	32	167	199
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	0	13	13
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	0	94	94
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	0	0	0
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	0	10	10
SUM ALL DDT CONGENERS, NG	32	285	317
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	73	375	313
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	0	24	24
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	0	24	24

* = estimated from dissection length and whole weight data by proportion from undissected individuals remaining in sample.

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S2AHT 251	S2AHT 252	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Crangon franciscorum	Crangon franciscorum	WHOLE BODY
TISSUE TYPE	tail muscle	Dissection residua (estim.)*	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	0.919
TISSUE WEIGHT @ DISSECTION, G WT WT	0.342	0.576*	na
PERCENT OF GROSS WET WEIGHT	37.2%	62.7%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	0.342	0.516*	na
NUMBER INDIVIDUALS IN COMPOSITE	2	2	2
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	0	0	0
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	27	170	197
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	0	9	9
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	0	65	65
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	0	0	0
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	0	11	11
SUM ALL DDT CONGENERS, NG	27	254	281
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	79	441	306
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	0	0	0
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	0	0	0

* = estimated from dissection length and whole weight data by proportion from undissected individuals remaining in sample.

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S2AHT 301	S2AHT 302	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Crangon franciscorum	Crangon franciscorum	WHOLE BODY
TISSUE TYPE	tail muscle	Dissection residua (estim.)*	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	12.362
TISSUE WEIGHT @ DISSECTION, G WT WT	5.685	7.536*	na
PERCENT OF GROSS WET WEIGHT	46.0%	61.0%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	5.685	6.560*	na
NUMBER INDIVIDUALS IN COMPOSITE	10	10	10
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	0	15	15
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	109	354	463
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	7	23	29
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	75	528	603
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	0	38	38
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	14	68	81
SUM ALL DDT CONGENERS, NG	204	1,025	1,229
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	36	156	99
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	0	38	38
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	0	6	3

* = estimated from dissection length and whole weight data by proportion from undissected individuals remaining in sample.

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S2AHT 321	S2AHT 322	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Crangon franciscorum	Crangon franciscorum	WHOLE BODY
TISSUE TYPE	tail muscle	Dissection residua (estim.)*	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	13.889
TISSUE WEIGHT @ DISSECTION, G WT WT	6.015	7.973*	na
PERCENT OF GROSS WET WEIGHT	43.3%	57.4%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	6.015	7.119*	na
NUMBER INDIVIDUALS IN COMPOSITE	10	10	10
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	0	16	16
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	144	423	567
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	9	16	25
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	90	917	1,006
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	0	56	56
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	20	80	100
SUM ALL DDT CONGENERS, NG	263	1,507	1,770
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	44	189	127
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	37	0	37
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	6	0	3

* = estimated from dissection length and whole weight data by proportion from undissected individuals remaining in sample.

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S2AHT 343	S2AHT 344	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Crangon franciscorum	Crangon franciscorum	WHOLE BODY
TISSUE TYPE	tail muscle	Dissection residua (estim.)*	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	15.986
TISSUE WEIGHT @ DISSECTION, G WT WT	7.436	8.963*	na
PERCENT OF GROSS WET WEIGHT	46.5%	56.1%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	7.436	7.905*	na
NUMBER INDIVIDUALS IN COMPOSITE	10	10	10
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	4	9	13
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	92	296	387
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	7	9	16
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	85	905	990
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	0	63	63
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	0	54	54
SUM ALL DDT CONGENERS, NG	188	1,335	1,523
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	25	149	95
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN, NG/PORTION	0	72	72
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	0	8	5

* = estimated from dissection length and whole weight data by proportion from undissected individuals remaining in sample.

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S2AHT 402	S2AHT 403	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Crangon franciscorum	Crangon franciscorum	WHOLE BODY
TISSUE TYPE	tail muscle	Dissection residua (estim.)*	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	46.072
TISSUE WEIGHT @ DISSECTION, G WT WT	19.073	26.350*	na
PERCENT OF GROSS WET WEIGHT	41.4%	57.2%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	19.073	22.510*	na
NUMBER INDIVIDUALS IN COMPOSITE	60	60	60
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	0	0	0
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	76	158	235
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	0	0	0
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	0	1,001	1,001
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	0	0	0
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	0	0	0
SUM ALL DDT CONGENERS, NG	77	1,159	1,236
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	2	44	27
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	0	0	0
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	0	0	0

* = estimated from dissection length and whole weight data by proportion from undissected individuals remaining in sample.

APPENDIX 8-10 (Cont'd.). Method for Calculation of Whole-Body Tissue Residues From Tissue and Remainder Portion Concentrations.

SAMPLE NUMBER	S2AHT 452	S2AHT 453	RECONSTI-TUTED SUM
TISSUE SOURCE (species)	Crangon franciscorum	Crangon franciscorum	WHOLE BODY
TISSUE TYPE	tail muscle	Dissection residua (estim.)*	Whole
ORIG. GROSS WT., GRAMS WET WEIGHT	na	na	13.374
TISSUE WEIGHT @ DISSECTION, G WT WT	6.095	6.943*	na
PERCENT OF GROSS WET WEIGHT	45.6%	51.9%	100.0%
TISSUE SAMPLE, G WET WEIGHT @ PREP	6.095	6.504*	na
NUMBER INDIVIDUALS IN COMPOSITE	10	10	10
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDE, NG/PORTION	0	0	0
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDE, NG/PORTION	28	49	76
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDD, NG/PORTION	0	0	0
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDD, NG/PORTION	0	201	201
PORTION WEIGHT @ DISSECTION X CONC. 2,4'DDT, NG/PORTION	0	14	14
PORTION WEIGHT @ DISSECTION X CONC. 4,4'DDT, NG/PORTION	0	0	0
SUM ALL DDT CONGENERS, NG	27	264	291
SUM CONCENTRATION DDT CONGENERS (SUM/GR WET WT) = NG/G WET WEIGHT	4	38	22
PORTION WEIGHT @ DISSECTION X CONC. DIELDRIN,NG/PORTION	35	0	35
CONCENTRATION DIELDRIN (SUM/GR WET WT) = NG/G WW	6	0	3

* = estimated from dissection length and whole weight data by proportion from undissected individuals remaining in sample.

APPENDIX 9-1. Benthic infauna data, >1.0 mm sieve size.

STATION 1

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/09/91

		GRAB 1	2	3	4	5	6	7	8	TOTAL	PCT.
21 2139	THEORA LUBRICA	1	-	4	19	36	34	3	6	103	33.55
18 2124	PSEUDOPOLYDORA PAUCIBRANCHIATA	6	1	3	18	15	6	1	21	71	23.13
18 1822	EXOGONE LOUREI	1	1	-	-	3	1	-	16	22	7.17
30 2166	GRANDIDIERELLA JAPONICA	-	2	-	3	4	9	-	-	18	5.86
19 2132	TUBIFICIDAE, GROUP 3	-	-	-	2	5	-	-	3	10	3.26
18 1800	CAPITELLA SPP	5	-	-	-	-	-	-	4	9	2.93
18 1835	CIRRATULUS CIRRATUS	1	-	-	-	-	-	-	8	9	2.93
19 2131	TUBIFICIDAE, GROUP 2	-	-	-	2	4	2	-	1	9	2.93
18 2125	HARMOTHOE IMBRICATA	1	-	-	-	3	1	1	2	8	2.61
18 2123	POLYDORA LIGNI	2	1	1	1	2	-	-	-	7	2.28
18 1684	SCHISTOMERINGOS LONGICORNIS	-	-	-	2	-	-	-	4	6	1.95
21 2164	MUSCULUS SENHOUSIA	-	-	-	-	-	-	1	5	6	1.95
18 2126	EUCHONE LIMNICOLA	1	-	-	2	1	1	-	-	5	1.63
18 2135	CIRRATULID SP A	-	-	1	-	1	1	-	-	3	0.98
21 1544	MACOMA NASUTA	-	-	1	-	-	-	1	1	3	0.98
10 1636	HETERONEMERTEA SPP	-	-	1	1	-	-	1	-	3	0.98
18 1757	POLYDORA SOCIALIS	-	-	1	-	1	-	-	-	2	0.65
19 2130	TUBIFICIDAE, GROUP 1	-	-	-	-	-	1	-	-	1	0.33
18 2128	HETEROMASTUS FILIFORMIS	-	1	-	-	-	-	-	-	1	0.33
18 1673	THARYX SPP	-	-	-	-	1	-	-	-	1	0.33
18 1719	LEITOSCOLOPLOS PUGETTENSIS	-	-	-	1	-	-	-	-	1	0.33
18 2137	ETEONE LIGHTI	-	-	1	-	-	-	-	-	1	0.33
10 1635	TUBULANUS SPP	-	-	-	-	-	-	1	-	1	0.33
18 2148	NEANTHES ACUMINATA	-	1	-	-	-	-	-	-	1	0.33
32 2151	ZEUXO NORMANI	-	-	-	-	-	1	-	-	1	0.33
5 2154	PACHYCERIANTHUS FIMBRIATUS	-	-	-	-	1	-	-	-	1	0.33
18 2160	COSSURA SP	-	-	-	1	-	-	-	-	1	0.33
18 2127	CIRRIFORMIA SPIRABRANCHA	-	-	1	-	-	-	-	-	1	0.33
18 1670	MEDIOMASTUS SPP	-	-	-	1	-	-	-	-	1	0.33
21 2167	TAPES JAPONICA	-	-	-	-	-	-	-	1	1	0.33
TOTAL		18	7	14	53	77	57	9	72	307	
Infaunal Index		56.25	75.00	77.78	64.00	58.79	63.08	90.00	63.79		
Biomass (g wet weight)											
Annelida		0.015	0.095	1.344	0.044	0.155	0.167	0.026	0.126		
Mollusca		0.017	0.000	0.516	0.230	0.629	0.431	0.320	2.869		
Crustacea		0.000	0.015	0.000	0.020	0.053	0.064	0.000	0.009		
Other		0.000	0.000	0.000	0.050	0.017	0.000	0.064	0.000		
Total biomass		0.032	0.110	1.860	0.344	0.854	0.662	0.410	3.004		

APPENDIX 9-1 (cont'd.). Benthic infauna data.

STATION 2

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/09/91

		GRAB 1	2	3	4	5	TOTAL	PCT.
18 2124	PSEUDOPOLYDORA PAUCIBRANCHIATA	34	55	11	10	24	134	17.84
18 1822	EXOGONE LOUREI	3	77	23	3	11	117	15.58
21 2139	THEORA LUBRICA	22	25	9	32	16	104	13.85
32 2151	ZEUXO NORMANI	8	19	11	1	5	44	5.86
18 2160	COSSURA SP	3	27	9	-	-	39	5.19
19 2131	TUBIFICIDAE, GROUP 2	2	30	3	-	3	38	5.06
19 2132	TUBIFICIDAE, GROUP 3	2	27	5	-	3	37	4.93
21 2164	MUSCULUS SENHOUSIA	1	17	6	1	6	31	4.13
18 1684	SCHISTOMERINGOS LONGICORNIS	4	21	-	2	-	27	3.60
18 2123	POLYDORA LIGNI	5	13	2	2	2	24	3.20
30 2166	GRANDIDIERELLA JAPONICA	11	4	-	3	6	24	3.20
18 2127	CIRRIFORMIA SPIRABRANCHA	2	13	2	1	5	23	3.06
18 2126	EUCHONE LIMNICOLA	15	2	1	2	1	21	2.80
19 2130	TUBIFICIDAE, GROUP 1	-	15	2	2	-	19	2.53
18 2125	HARMOTHOE IMBRICATA	1	3	2	2	-	8	1.07
21 2167	TAPES JAPONICA	-	6	-	2	-	8	1.07
18 1719	LEITOSCOLOPLOS PUGETTENSIS	1	2	-	2	2	7	0.93
18 1800	CAPITELLA SPP	-	7	-	-	-	7	0.93
18 2135	CIRRATULID SP A	-	5	-	1	-	6	0.80
18 2136	ASYCHIS ELONGATA	-	1	-	2	2	5	0.67
18 1673	THARYX SPP	2	2	-	1	-	5	0.67
26 2038	HARPACTICOID COPEPOD	-	3	1	-	-	4	0.53
18 2128	HETEROMASTUS FILIFORMIS	1	2	1	-	-	4	0.53
5 1643	ANTHOZOA UNID	-	2	-	-	-	2	0.27
18 2162	SPHAEROSYLLIS SP A	-	-	2	-	-	2	0.27
18 1835	CIRRATULUS CIRRATUS	1	1	-	-	-	2	0.27
30 2173	CAPRELLA MUTICA	-	-	-	-	2	2	0.27
18 1692	PODARKEOPSIS GLABRA	1	-	-	-	-	1	0.13
18 1667	NOTOMASTUS TENUIS	-	1	-	-	-	1	0.13
18 2137	ETEONE LIGHTI	-	1	-	-	-	1	0.13
18 2129	CIRRATULIDAE, UNID	1	-	-	-	-	1	0.13
20 2020	PHILINE SP A	-	-	1	-	-	1	0.13
18 1670	MEDIOMASTUS SPP	-	-	1	-	-	1	0.13
18 2149	LUMBRINERIS TETRAURA	-	1	-	-	-	1	0.13
TOTAL		120	382	92	69	88	751	
Infaunal Index		75.18	54.77	69.73	68.97	71.38		
Biomass (g wet weight)								
Annelida		0.152	1.457	0.165	0.237	0.289		
Mollusca		0.943	15.992	2.999	1.414	3.515		
Crustacea		0.047	0.051	0.020	0.017	0.045		
Other		0.000	0.032	0.000	0.000	0.000		
Total biomass		1.142	17.532	3.184	1.668	3.849		

APPENDIX 9-1 (cont'd.). Benthic infauna data.

STATION 3

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/09/91

		GRAB 1	2	3	4	5	TOTAL	PCT.
21 2139	THEORA LUBRICA	16	15	21	31	29	112	32.09
18 2124	PSEUDOPOLYDORA PAUCIBRANCHIATA	44	1	2	1	11	59	16.91
18 2126	EUCHONE LIMNICOLA	11	1	3	3	2	20	5.73
18 1822	EXOGONE LOUREI	1	10	1	-	7	19	5.44
18 2160	COSSURA SP	6	6	3	2	-	17	4.87
18 2127	CIRRIFORMIA SPIRABRANCHA	-	5	4	3	3	15	4.30
18 2135	CIRRATULID SP A	1	4	1	4	3	13	3.72
19 2130	TUBIFICIDAE, GROUP 1	2	1	-	4	4	11	3.15
21 2164	MUSCULUS SENHOUSIA	-	7	1	-	3	11	3.15
19 2131	TUBIFICIDAE, GROUP 2	1	3	-	4	2	10	2.87
18 1684	SCHISTOMERINGOS LONGICORNIS	2	2	2	1	-	7	2.01
18 2123	POLYDORA LIGNI	3	-	1	1	2	7	2.01
32 2151	ZEUXO NORMANI	-	4	-	-	2	6	1.72
18 2125	HARMOTHOE IMBRICATA	3	2	-	-	-	5	1.43
19 2132	TUBIFICIDAE, GROUP 3	2	-	-	2	1	5	1.43
30 2166	GRANDIDIERELLA JAPONICA	4	1	-	-	-	5	1.43
18 1673	THARYX SPP	-	1	1	2	-	4	1.15
21 2171	MACOMA BALTHICA	-	2	-	1	-	3	0.86
18 1800	CAPITELLA SPP	-	-	-	-	2	2	0.57
18 2136	ASYCHIS ELONGATA	1	-	-	1	-	2	0.57
18 2148	NEANTHES ACUMINATA	-	1	-	-	1	2	0.57
20 2020	PHILINE SP A	-	2	-	-	-	2	0.57
18 1670	MEDIOMASTUS SPP	-	1	-	1	-	2	0.57
21 2167	TAPES JAPONICA	-	-	-	-	2	2	0.57
18 2150	OPHIODROMUS PUGETTENSIS	-	1	-	-	-	1	0.29
18 1719	LEITOSCOLOPLOS PUGETTENSIS	-	-	1	-	-	1	0.29
18 1692	PODARKEOPSIS GLABRA	-	-	-	1	-	1	0.29
18 2137	ETEONE LIGHTI	-	1	-	-	-	1	0.29
21 1544	MACOMA NASUTA	-	-	-	-	1	1	0.29
18 2134	STREBLOSPIO BENEDICTI	1	-	-	-	-	1	0.29
21 2169	POTAMOCORBULA AMURENSIS	-	1	-	-	-	1	0.29
18 2129	CIRRATULIDAE, UNID	-	-	-	-	1	1	0.29
TOTAL		98	72	41	62	76	349	
Infaunal Index		76.00	69.05	72.63	54.00	63.50		
Biomass (g wet weight)								
Annelida		0.069	0.125	0.105	0.120	0.065		
Mollusca		0.185	1.255	0.508	0.633	2.307		
Crustacea		0.029	0.021	0.000	0.000	0.020		
Other		0.000	0.000	0.000	0.000	0.000		
Total biomass		0.283	1.401	0.613	0.753	2.392		

APPENDIX 9-1 (cont'd.). Benthic infauna data.

STATION 4

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/09/91

		GRAB 1	2	3	4	5	TOTAL	PCT.
18 2160	COSSURA SP	19	47	8	15	41	130	28.70
21 2139	THEORA LUBRICA	35	30	31	16	9	121	26.71
18 2124	PSEUDOPOLYDORA PAUCIBRANCHIATA	31	14	46	18	1	110	24.28
18 2126	EUCHONE LIMNICOLA	3	1	2	5	-	11	2.43
18 1822	EXOGONE LOUREI	4	1	-	4	-	9	1.99
18 1684	SCHISTOMERINGOS LONGICORNIS	5	1	1	1	-	8	1.77
18 2123	POLYDORA LIGNI	3	1	1	3	-	8	1.77
18 2127	CIRRIFORMIA SPIRABRANCHA	3	1	3	1	-	8	1.77
21 2164	MUSCULUS SENHOUSIA	3	-	2	3	-	8	1.77
18 2135	CIRRATULID SP A	4	2	1	-	-	7	1.55
19 2130	TUBIFICIDAE, GROUP 1	2	-	-	1	2	5	1.10
18 1673	THARYX SPP	3	1	-	-	-	4	0.88
18 1800	CAPITELLA SPP	2	-	1	-	1	4	0.88
32 2151	ZEUXO NORMANI	-	4	-	-	-	4	0.88
30 2166	GRANDIDIERELLA JAPONICA	1	-	-	2	-	3	0.66
18 2125	HARMOTHOE IMBRICATA	-	-	-	2	-	2	0.44
18 1670	MEDIOMASTUS SPP	-	1	-	1	-	2	0.44
5 2138	ACTINARIA SPA	1	-	-	-	-	1	0.22
5 1643	ANTHOZOA UNID	-	1	-	-	-	1	0.22
18 1757	POLYDORA SOCIALIS	-	-	-	1	-	1	0.22
3 2152	CNIDARIA, UNID	-	1	-	-	-	1	0.22
10 1636	HETERONEMERTEA SPP	-	-	-	1	-	1	0.22
18 1719	LEITOSCOLOPLOS PUGETTENSIS	1	-	-	-	-	1	0.22
18 2137	ETEONE LIGHTI	1	-	-	-	-	1	0.22
21 2167	TAPES JAPONICA	1	-	-	-	-	1	0.22
30 2168	COROPHIUM HETEROCERATUM	-	-	1	-	-	1	0.22
TOTAL		122	106	97	74	54	453	
Infaunal Index		70.86	79.19	78.10	78.80	74.67		
Biomass (g wet weight)								
Annelida		0.292	0.271	0.117	0.084	0.089		
Mollusca		0.944	0.013	0.418	0.481	0.098		
Crustacea		0.015	0.018	0.036	0.023	0.000		
Other		0.039	0.002	0.000	0.000	0.000		
Total biomass		1.290	0.304	0.571	0.588	0.187		

APPENDIX 9-1 (cont'd.). Benthic infauna data.

STATION 5

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/08/91

		GRAB 1	2	3	4	5	TOTAL	PCT.
18 2160	COSSURA SP	10	7	17	30	39	103	39.46
21 2139	THEORA LUBRICA	20	26	22	24	8	100	38.31
18 2127	CIRRIFORMIA SPIRABRANCHA	3	2	-	2	6	13	4.98
18 2124	PSEUDOPOLYDORA PAUCIBRANCHIATA	5	1	-	-	2	8	3.07
18 1673	THARYX SPP	-	2	1	-	3	6	2.30
30 2168	COROPHIUM HETEROCERATUM	-	4	1	-	1	6	2.30
19 2132	TUBIFICIDAE, GROUP 3	-	1	1	-	3	5	1.92
18 1719	LEITOSCOLOPLOS PUGETTENSIS	-	-	-	2	2	4	1.53
18 2126	EUCHONE LIMNICOLA	-	1	1	-	1	3	1.15
18 2123	POLYDORA LIGNI	1	-	-	1	1	3	1.15
19 2130	TUBIFICIDAE, GROUP 1	-	-	-	1	1	2	0.77
18 2135	CIRRATULID SP A	-	2	-	-	-	2	0.77
18 1670	MEDIOMASTUS SPP	-	-	-	-	2	2	0.77
18 2153	AMAENA OCCIDENTALIS	-	1	-	-	-	1	0.38
5 2154	PACHYCERIANTHUS FIMBRIATUS	-	-	1	-	-	1	0.38
19 2131	TUBIFICIDAE, GROUP 2	-	-	1	-	-	1	0.38
20 2020	PHILINE SP A	1	-	-	-	-	1	0.38
TOTAL		40	47	45	60	69	261	
Infaunal Index		80.00	77.65	73.33	75.56	73.67		
Biomass (g wet weight)								
Annelida		0.337	0.204	0.033	0.271	0.491		
Mollusca		0.335	0.454	0.418	0.276	0.072		
Crustacea		0.000	0.021	0.017	0.000	0.011		
Other		0.000	0.007	2.357	0.000	0.000		
Total biomass		0.672	0.686	2.825	0.547	0.574		

APPENDIX 9-1 (cont'd.). Benthic infauna data.

STATION 6

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/08/91

		GRAB 1	2	3	4	5	TOTAL	PCT.
21 2139	THEORA LUBRICA	19	17	11	10	21	78	36.11
18 2124	PSEUDOPOLYDORA PAUCIBRANCHIATA	15	14	4	7	7	47	21.76
18 2126	EUCHONE LIMNICOLA	12	7	6	4	-	29	13.43
30 2168	COROPHIUM HETEROCERATUM	-	6	1	3	7	17	7.87
18 2123	POLYDORA LIGNI	6	2	1	-	-	9	4.17
18 2127	CIRRIFORMIA SPIRABRANCHA	2	-	3	2	2	9	4.17
18 2135	CIRRATULID SP A	1	2	1	1	-	5	2.31
18 1719	LEITOSCOLOPLOS PUGETTENSIS	1	1	1	1	-	4	1.85
20 2020	PHILINE SP A	1	2	-	-	-	3	1.39
18 1673	THARYX SPP	2	-	1	-	-	3	1.39
18 2160	COSSURA SP	1	2	-	-	-	3	1.39
19 2132	TUBIFICIDAE, GROUP 3	-	-	1	-	1	2	0.93
19 2131	TUBIFICIDAE, GROUP 2	-	-	-	-	1	1	0.46
19 2130	TUBIFICIDAE, GROUP 1	-	1	-	-	-	1	0.46
18 1684	SCHISTOMERINGOS LONGICORNIS	1	-	-	-	-	1	0.46
18 1710	NEPHTYS CAECOIDES	-	-	-	-	1	1	0.46
3 2152	CNIDARIA, UNID	-	1	-	-	-	1	0.46
18 1822	EXOGONE LOUREI	1	-	-	-	-	1	0.46
18 1800	CAPITELLA SPP	-	-	-	-	1	1	0.46
TOTAL		62	55	30	28	41	216	
Infaunal Index		82.86	80.69	80.00	82.67	61.54		
Biomass (g wet weight)								
Annelida		0.057	0.054	0.114	0.034	0.097		
Mollusca		0.279	0.378	0.196	0.339	0.601		
Crustacea		0.000	0.028	0.016	0.022	0.068		
Other		0.000	0.012	0.000	0.000	0.000		
Total biomass		0.336	0.472	0.326	0.395	0.766		

APPENDIX 9-1 (cont'd.). Benthic infauna data.

STATION 7

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/08/91

		GRAB 1	2	3	4	5	TOTAL	PCT.
18 2160	COSSURA SP	28	6	6	36	29	105	24.53
21 2139	THEORA LUBRICA	15	18	10	36	16	95	22.20
18 2124	PSEUDOPOLYDORA PAUCIBRANCHIATA	9	12	9	10	9	49	11.45
18 1822	EXOGONE LOUREI	1	1	-	30	1	33	7.71
30 2168	COROPHIUM HETEROCERATUM	7	6	1	4	8	26	6.07
18 2126	EUCHONE LIMNICOLA	15	3	1	2	3	24	5.61
18 2123	POLYDORA LIGNI	4	1	-	12	-	17	3.97
18 1670	MEDIOMASTUS SPP	2	-	-	2	5	9	2.10
18 1800	CAPITELLA SPP	2	-	-	3	2	7	1.64
18 2127	CIRRIFORMIA SPIRABRANCHA	-	-	-	6	-	6	1.40
18 1719	LEITOSCOLOPLOS PUGETTENSIS	1	2	2	-	-	5	1.17
21 2169	POTAMOCORBULA AMURENSIS	-	1	-	2	2	5	1.17
19 2130	TUBIFICIDAE, GROUP 1	1	-	-	2	1	4	0.93
19 2131	TUBIFICIDAE, GROUP 2	-	-	-	4	-	4	0.93
32 2151	ZEUXO NORMANI	-	1	-	2	1	4	0.93
26 2038	HARPACTICOID COPEPOD	-	-	-	4	-	4	0.93
18 1684	SCHISTOMERINGOS LONGICORNIS	1	-	-	1	1	3	0.70
18 2125	HARMOTHOE IMBRICATA	1	-	-	2	-	3	0.70
18 1817	AUTOLYTUS SP A	-	-	-	3	-	3	0.70
18 1942	EUPOLYMNA HETEROBRANCHIA	-	-	-	2	-	2	0.47
30 2157	AMPELISCA ABDITA	1	1	-	-	-	2	0.47
18 2162	SPHAEROSYLLIS SP A	-	-	-	2	-	2	0.47
21 2167	TAPES JAPONICA	-	-	-	1	1	2	0.47
20 2020	PHILINE SP A	-	-	2	-	-	2	0.47
5 2154	PACHYCERIANTHUS FIMBRIATUS	-	-	-	1	-	1	0.23
40 2155	AMPHIPHOLIS SQUAMATA	-	-	-	1	-	1	0.23
18 2156	ARENICOLIDAE SP A	-	-	-	-	1	1	0.23
5 2138	ACTINARIA SPA	-	-	-	1	-	1	0.23
21 1544	MACOMA NASUTA	-	-	-	1	-	1	0.23
10 1635	TUBULANUS SPP	-	1	-	-	-	1	0.23
21 2164	MUSCULUS SENHOUSIA	-	-	-	1	-	1	0.23
30 2166	GRANDIDIERELLA JAPONICA	1	-	-	-	-	1	0.23
18 2150	OPHIODROMUS PUGETTENSIS	-	-	-	1	-	1	0.23
18 2135	CIRRATULID SP A	-	1	-	-	-	1	0.23
19 2132	TUBIFICIDAE, GROUP 3	-	-	-	1	-	1	0.23
28 2175	NIPPOLEUCON HINUMENSIS	-	-	-	1	-	1	0.23
TOTAL		89	54	31	174	80	428	
Infaunal Index		79.38	80.00	76.67	73.45	73.60		
Biomass (g wet weight)								
Annelida		0.089	0.045	0.054	0.294	0.980		
Mollusca		0.091	0.252	0.091	0.212	0.141		
Crustacea		0.042	0.043	0.006	0.022	0.049		
Other		0.000	0.023	0.000	20.610	0.000		
Total biomass		0.222	0.363	0.151	21.138	1.170		

APPENDIX 9-1 (cont'd.). Benthic infauna data.

STATION 8

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/07/91

		GRAB 1	2	3	4	5	TOTAL
18 1793	NEPHTYS CORNUTA FRANCISCANA	-	-	-	-	1	1
18 2125	HARMOTHOE IMBRICATA	1	-	-	-	-	1
18 2126	EUCHONE LIMNICOLA	-	-	1	2	-	3
21 2139	THEORA LUBRICA	-	-	1	5	4	10
30 2168	COROPHIUM HETEROCERATUM	-	-	-	1	-	1
TOTAL		1	0	2	8	5	16

Infaunal Index not calclulated

Biomass (g wet weight)						
Annelida	0.008	0.000	0.016	0.007	0.016	
Mollusca	0.000	0.000	0.002	0.029	0.031	
Crustacea	0.000	0.000	0.000	0.007	0.000	
Other	0.000	0.000	0.000	0.000	0.000	
Total biomass	0.008	0.000	0.018	0.043	0.047	

APPENDIX 9-1 (cont'd.). Benthic infauna data.

STATION 9

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/07/91

		GRAB 1	2	3	4	5	TOTAL	PCT.
18 2126	EUCHONE LIMNICOLA	9	34	104	8	31	186	35.84
50 2140	PHORNONIS CF. PALLIDA	2	1	4	126	34	167	32.18
30 2157	AMPELISCA ABDITA	2	9	23	1	-	35	6.74
18 2124	PSEUDOPOLYDORA PAUCIBRANCHIATA	-	3	4	3	4	14	2.70
21 2167	TAPES JAPONICA	2	2	3	6	1	14	2.70
21 1893	CRYPTOMYA CALIFORNICA	-	2	2	-	6	10	1.93
21 2139	THEORA LUBRICA	2	3	4	1	-	10	1.93
18 2160	COSSURA SP	4	4	1	1	-	10	1.93
18 2161	GLYCINDE POLYGNATHA	3	3	1	-	2	9	1.73
18 2125	HARMOTHOE IMBRICATA	-	2	1	2	2	7	1.35
18 1822	EXOGONE LOUREI	3	3	-	-	1	7	1.35
30 2168	COROPHIUM HETEROCERATUM	1	3	1	1	1	7	1.35
18 2136	ASYCHIS ELONGATA	-	2	-	2	1	5	0.96
19 2132	TUBIFICIDAE, GROUP 3	1	1	-	2	-	4	0.77
32 2151	ZEUXO NORMANI	3	-	-	1	-	4	0.77
10 1636	HETERONEMERTEA SPP	-	-	-	1	3	4	0.77
21 2066	PELECYPOD, ERODED	3	1	-	-	-	4	0.77
18 1670	MEDIOMASTUS SPP	-	1	-	1	1	3	0.58
10 1635	TUBULANUS SPP	-	-	1	-	1	2	0.39
18 2153	AMAENA OCCIDENTALIS	-	-	1	-	1	2	0.39
18 1684	SCHISTOMERINGOS LONGICORNIS	-	2	-	-	-	2	0.39
5 2138	ACTINARIA SPA	2	-	-	-	-	2	0.39
30 2172	CAPRELLA INCISA	-	-	-	2	-	2	0.39
18 1800	CAPITELLA SPP	-	-	-	-	1	1	0.19
21 1544	MACOMA NASUTA	-	-	1	-	-	1	0.19
21 2164	MUSCULUS SENHOUSIA	-	-	-	-	1	1	0.19
30 2166	GRANDIDIERELLA JAPONICA	-	1	-	-	-	1	0.19
18 1757	POLYDORA SOCIALIS	-	1	-	-	-	1	0.19
5 1643	ANTHOZOA UNID	-	1	-	-	-	1	0.19
21 2169	POTAMOCORBULA AMURENSIS	-	-	-	-	1	1	0.19
18 2158	?BARANTOLLA SP	-	-	-	1	-	1	0.19
31 2174	HEMIGRAPSUS SP	-	-	-	-	1	1	0.19
TOTAL		37	79	151	159	93	519	
Infaunal Index		87.50	91.15	99.28	97.12	97.03		
Biomass (g wet weight)								
Annelida		0.105	0.268	0.436	0.854	0.438		
Mollusca		0.195	0.168	2.029	0.148	0.604		
Crustacea		0.016	0.023	0.027	0.027	0.038		
Other		0.675	0.007	0.114	0.000	0.139		
Total biomass		0.991	0.466	2.606	1.029	1.219		

APPENDIX 9-1 (cont'd.). Benthic infauna data.

STATION 22

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/09/91

			TOTAL	PCT.
30	2168	COROPHIUM HETEROCERATUM	25	39.68
30	2157	AMPELISCA ABDITA	10	15.87
21	1893	CRYPTOMYA CALIFORNICA	9	14.29
18	2126	EUCHONE LIMNICOLA	5	7.94
21	2165	GEMMA GEMMA	4	6.35
10	2142	PALEONEMERTEA SPP	2	3.17
10	1636	HETERONEMERTEA SPP	2	3.17
30	1511	PHOTIS BREVIPES	2	3.17
18	2143	EUSYLLIS SP	1	1.59
50	2140	PHORNONIS CF. PALLIDA	1	1.59
18	2125	HARMOTHOE IMBRICATA	1	1.59
18	1670	MEDIOMASTUS SPP	1	1.59
TOTAL			63	

Infaunal Index 96.84

Biomass (g wet weight)

Annelida	0.012
Mollusca	0.164
Crustacea	0.165
Other	0.028
Total biomass	0.369

APPENDIX 9-1 (cont'd.). Benthic infauna data.

STATION 23

United Heckathorn Superfund Study >=1.0 mm sieve size
COLLECTED 10/09/91

			TOTAL	PCT.
30	2168	COROPHIUM HETEROCERATUM	69	68.32
30	1511	PHOTIS BREVIPES	8	7.92
4	2147	TURBELLARIA SPP	5	4.95
18	2126	EUCHONE LIMNICOLA	4	3.96
21	1893	CRYPTOMYA CALIFORNICA	3	2.97
18	2144	ARMANDIA BREVIS	2	1.98
18	1684	SCHISTOMERINGOS LONGICORNIS	2	1.98
21	2139	THEORA LUBRICA	1	0.99
18	1697	PHYLLODOCE LONGIPES	1	0.99
18	2145	HALOSYDNA BREVITOSA	1	0.99
20	2146	TURBONILLA SPP	1	0.99
18	2136	ASYCHIS ELONGATA	1	0.99
21	2167	TAPES JAPONICA	1	0.99
18	1670	MEDIOMASTUS SPP	1	0.99
21	2169	POTAMOCORBULA AMURENSIS	1	0.99
TOTAL			101	

Infaunal Index 67.78

Biomass (g wet weight)

Annelida	0.340
Mollusca	0.102
Crustacea	0.417
Other	0.018
Total biomass	0.877

APPENDIX 9-2. Benthic infauna data, <1.0, >0.5 mm sieve size.

LAURITZEN CHANNEL - >= 0.5 MM SIEVE SIZE
FILE:tltzbio.dat

			STATION 1								
		1	2	3	4	5	6	7	8	TOTAL	PCT.
21 2139	THEORA LUBRICA	2	-	6	19	36	37	3	8	111	20.52
18 2124	PSEUDOPOLYDORA PAUCIBRA	6	2	4	18	16	8	1	21	76	14.05
19 2131	TUBIFICIDAE, GROUP 2	10	1	1	6	9	31	11	6	75	13.86
19 2132	TUBIFICIDAE, GROUP 3	-	-	-	7	15	12	8	7	49	9.06
30 2166	GRANDIDIERELLA JAPONICA	10	9	-	8	6	10	-	2	45	8.32
18 1684	SCHISTOMERINGOS LONGICO	1	2	2	7	1	11	1	12	37	6.84
18 1822	EXOGONE LOUREI	2	1	1	-	3	5	-	22	34	6.28
18 1800	CAPITELLA SPP	6	-	1	1	-	-	-	6	14	2.59
18 1835	CIRRATULUS CIRRATUS	1	-	-	-	-	1	-	10	12	2.22
18 2125	HARMOTHOE IMBRICATA	3	-	-	-	3	2	1	3	12	2.22
18 2126	EUCHONE LIMNICOLA	1	-	1	5	1	3	-	1	12	2.22
18 2123	POLYDORA LIGNI	2	2	1	1	3	-	-	-	9	1.66
18 2160	COSSURA SP	-	-	1	2	3	3	-	-	9	1.66
19 2130	TUBIFICIDAE, GROUP 1	3	-	1	-	-	3	1	-	8	1.48
21 2164	MUSCULUS SENHOUSIA	-	-	-	-	-	-	1	5	6	1.11
21 2167	TAPES JAPONICA	-	-	-	1	1	-	1	1	4	0.74
10 1636	HETERONEMERTEA SPP	-	-	1	1	-	-	1	-	3	0.55
18 2135	CIRRATULID SP A	-	-	1	-	1	1	-	-	3	0.55
21 1544	MACOMA NASUTA	-	-	1	-	-	-	1	1	3	0.55
32 2151	ZEUXO NORMANI	-	-	-	-	1	1	-	1	3	0.55
18 1719	LEITOSCOLOPLOS PUGETTEN	-	-	-	1	1	-	-	-	2	0.37
18 2148	NEANTHES ACUMINATA	-	1	-	-	-	1	-	-	2	0.37
18 1757	POLYDORA SOCIALIS	-	-	1	-	1	-	-	-	2	0.37
18 1673	THARYX SPP	-	-	-	-	1	-	1	-	2	0.37
18 2137	ETEONE LIGHTI	-	-	1	-	-	-	-	-	1	0.18
18 1670	MEDIOMASTUS SPP	-	-	-	1	-	-	-	-	1	0.18
5 2154	PACHYCERIANTHUS FIMBRIA	-	-	-	-	1	-	-	-	1	0.18
10 1635	TUBULANUS SPP	-	-	-	-	-	-	1	-	1	0.18
18 2162	SPHAEROSYLLIS SP A	1	-	-	-	-	-	-	-	1	0.18
5 2138	ACTINARIA SPA	-	-	-	-	-	-	-	1	1	0.18
18 2128	HETEROMASTUS FILIFORMIS	-	1	-	-	-	-	-	-	1	0.18
18 2127	CIRRIFORMIA SPIRABRANCH	-	-	1	-	-	-	-	-	1	0.18
TOTAL		48	19	25	78	103	129	32	107	541	

APPENDIX 9-2 (cont'd.). Benthic infauna data.

LAURITZEN CHANNEL - >= 0.5 MM SIEVE SIZE
FILE:tltzbio.dat

			STATION 2						
			1	2	3	4	5	TOTAL	PCT.
18	2124	PSEUDOPOLYDORA PAUCIBRA	49	58	14	15	26	162	12.85
18	1822	EXOGONE LOUREI	8	93	23	10	17	151	11.97
21	2139	THEORA LUBRICA	29	36	12	36	19	132	10.47
19	2131	TUBIFICIDAE, GROUP 2	11	64	8	21	27	131	10.39
19	2132	TUBIFICIDAE, GROUP 3	4	66	11	8	10	99	7.85
18	1684	SCHISTOMERINGOS LONGICO	11	45	8	13	13	90	7.14
18	2160	COSSURA SP	8	40	15	4	6	73	5.79
32	2151	ZEUXO NORMANI	9	23	15	6	13	66	5.23
30	2166	GRANDIDIERELLA JAPONICA	25	10	1	19	9	64	5.08
21	2164	MUSCULUS SENHOUSIA	2	29	9	1	7	48	3.81
18	2126	EUCHONE LIMNICOLA	22	5	1	9	4	41	3.25
18	2123	POLYDORA LIGNI	8	16	3	5	5	37	2.93
19	2130	TUBIFICIDAE, GROUP 1	2	25	3	5	2	37	2.93
18	2127	CIRRIFORMIA SPIRABRANCH	2	14	3	1	5	25	1.98
18	1800	CAPITELLA SPP	-	11	2	-	1	14	1.11
18	1673	THARYX SPP	5	5	-	1	1	12	0.95
21	2167	TAPES JAPONICA	-	9	-	2	-	11	0.87
18	2125	HARMOTHOE IMBRICATA	1	4	2	3	-	10	0.79
18	1719	LEITOSCOLOPLOS PUGETTEN	1	2	-	2	2	7	0.56
18	2135	CIRRATULID SP A	-	5	-	1	-	6	0.48
18	2136	ASYCHIS ELONGATA	-	1	-	2	2	5	0.40
18	2128	HETEROMASTUS FILIFORMIS	1	2	1	-	-	4	0.32
26	2038	HARPACTICOID COPEPOD	-	3	1	-	-	4	0.32
5	1643	ANTHOZOA UNID	1	2	-	-	1	4	0.32
18	2137	ETEONE LIGHTI	-	2	-	1	-	3	0.24
18	2162	SPHAEROSYLLIS SP A	-	-	2	-	1	3	0.24
18	1835	CIRRATULUS CIRRATUS	2	1	-	-	-	3	0.24
18	1670	MEDIOMASTUS SPP	-	1	1	-	-	2	0.16
18	2150	OPHIODROMUS PUGETTENSIS	-	1	-	-	1	2	0.16
30	2173	CAPRELLA MUTICA	-	-	-	-	2	2	0.16
18	1692	PODARKEOPSIS GLABRA	1	-	-	-	-	1	0.08
18	1757	POLYDORA SOCIALIS	-	-	-	1	-	1	0.08
18	1667	NOTOMASTUS TENUIS	-	1	-	-	-	1	0.08
20	2020	PHILINE SP A	-	-	1	-	-	1	0.08
18	1982	ARMANDIA BREVIS	-	1	-	-	-	1	0.08
18	1793	NEPHTYS CORNUTA FRANCIS	-	-	1	-	-	1	0.08
18	2129	CIRRATULIDAE, UNID	1	-	-	-	-	1	0.08
21	2169	POTAMOCORBULA AMURENSIS	1	-	-	-	-	1	0.08
18	2149	LUMBRINERIS TETRAURA	-	1	-	-	-	1	0.08
28	2175	NIPPOLEUCON HINUMENSIS	1	-	-	-	-	1	0.08
18	2176	BOCCARDIA PROBOSCIDEA	1	-	-	-	-	1	0.08
18	2177	CAULLERIELLA ALATA	-	-	-	-	1	1	0.08
23	2181	SARSIELLA SP A	1	-	-	-	-	1	0.08
TOTAL			207	576	137	166	175	1261	

APPENDIX 9-2 (cont'd.). Benthic infauna data.

LAURITZEN CHANNEL - >= 0.5 MM SIEVE SIZE
FILE:titzbio.dat

			STATION 3						
			1	2	3	4	5	TOTAL	PCT.
21	2139	THEORA LUBRICA	20	20	23	31	32	126	18.21
19	2131	TUBIFICIDAE, GROUP 2	4	22	9	28	15	78	11.27
18	2124	PSEUDOPOLYDORA PAUCIBRA	60	1	3	1	12	77	11.13
18	1684	SCHISTOMERINGOS LONGICO	7	14	12	13	9	55	7.95
18	2160	COSSURA SP	9	11	17	10	3	50	7.23
18	2126	EUCHONE LIMNICOLA	19	4	9	10	3	45	6.50
19	2132	TUBIFICIDAE, GROUP 3	6	5	5	15	3	34	4.91
19	2130	TUBIFICIDAE, GROUP 1	5	9	1	11	7	33	4.77
18	2135	CIRRATULID SP A	4	9	1	5	8	27	3.90
18	1822	EXOGONE LOUREI	2	12	4	1	7	26	3.76
21	2164	MUSCULUS SENHOUSIA	4	8	1	-	4	17	2.46
18	2127	CIRRIFORMIA SPIRABRANCH	-	5	5	3	3	16	2.31
18	1835	CIRRATULUS CIRRATUS	1	5	-	1	7	14	2.02
30	2166	GRANDIDIERELLA JAPONICA	13	1	-	-	-	14	2.02
18	2123	POLYDORA LIGNI	8	-	1	1	2	12	1.73
18	2125	HARMOTHOE IMBRICATA	5	4	-	-	1	10	1.45
18	1673	THARYX SPP	-	1	3	4	1	9	1.30
18	1800	CAPITELLA SPP	2	-	-	-	6	8	1.16
32	2151	ZEUXO NORMANI	1	4	-	-	2	7	1.01
5	1643	ANTHOZOA UNID	-	-	2	-	3	5	0.72
18	1719	LEITOSCOLOPLOS PUGETTEN	-	-	2	-	1	3	0.43
18	2137	ETEONE LIGHTI	-	1	1	1	-	3	0.43
21	2171	MACOMA BALTHICA	-	2	-	1	-	3	0.43
21	2066	PELECYPOD, ERODED	-	-	-	2	-	2	0.29
18	2136	ASYCHIS ELONGATA	1	-	-	1	-	2	0.29
18	2148	NEANTHES ACUMINATA	-	1	-	-	1	2	0.29
20	2020	PHILINE SP A	-	2	-	-	-	2	0.29
18	1670	MEDIOMASTUS SPP	-	1	-	1	-	2	0.29
21	2167	TAPES JAPONICA	-	-	-	-	2	2	0.29
18	2129	CIRRATULIDAE, UNID	-	-	-	-	1	1	0.14
18	1692	PODARKEOPSIS GLABRA	-	-	-	1	-	1	0.14
21	1544	MACOMA NASUTA	-	-	-	-	1	1	0.14
18	2049	SPIONIDAE, UNID	-	1	-	-	-	1	0.14
18	2150	OPHIODROMUS PUGETTENSIS	-	1	-	-	-	1	0.14
21	2169	POTAMOCORBULA AMURENSIS	-	1	-	-	-	1	0.14
18	2134	STREBLOSPIO BENEDICTI	1	-	-	-	-	1	0.14
28	2175	NIPPOLEUCON HINUMENSIS	-	-	-	1	-	1	0.14
TOTAL			172	145	99	142	134	692	

APPENDIX 9-2 (cont'd.). Benthic infauna data.

LAURITZEN CHANNEL - >= 0.5 MM SIEVE SIZE
FILE:tltzbio.dat

			STATION 4						
			1	2	3	4	5	TOTAL	PCT.
18	2160	COSSURA SP	29	63	24	24	56	196	27.22
21	2139	THEORA LUBRICA	49	34	40	23	10	156	21.67
18	2124	PSEUDOPOLYDORA PAUCIBRA	32	17	48	22	1	120	16.67
18	1684	SCHISTOMERINGOS LONGICO	21	1	10	3	-	35	4.86
18	2126	EUCHONE LIMNICOLA	7	1	8	8	4	28	3.89
18	1822	EXOGONE LOUREI	12	1	1	8	-	22	3.06
19	2130	TUBIFICIDAE, GROUP 1	7	1	7	2	5	22	3.06
18	1800	CAPITELLA SPP	7	-	5	4	2	18	2.50
19	2131	TUBIFICIDAE, GROUP 2	3	-	11	1	2	17	2.36
18	2135	CIRRATULID SP A	10	2	2	1	-	15	2.08
18	2123	POLYDORA LIGNI	3	1	4	6	-	14	1.94
21	2164	MUSCULUS SENHOUSIA	3	3	5	3	-	14	1.94
18	2127	CIRRIFORMIA SPIRABRANCH	3	1	3	3	-	10	1.39
18	1670	MEDIOMASTUS SPP	1	3	-	1	1	6	0.83
5	1643	ANTHOZOA UNID	3	1	1	1	-	6	0.83
30	2166	GRANDIDIERELLA JAPONICA	2	1	-	3	-	6	0.83
18	1673	THARYX SPP	3	1	-	-	1	5	0.69
19	2132	TUBIFICIDAE, GROUP 3	1	1	-	1	1	4	0.56
32	2151	ZEUXO NORMANI	-	4	-	-	-	4	0.56
18	2125	HARMOTHOE IMBRICATA	-	-	2	2	-	4	0.56
18	1719	LEITOSCOLOPLOS PUGETTEN	1	-	-	1	1	3	0.42
18	1835	CIRRATULUS CIRRATUS	2	-	-	-	-	2	0.28
18	1793	NEPHTYS CORNUTA FRANCIS	-	-	1	-	1	2	0.28
5	2138	ACTINARIA SPA	1	-	-	-	-	1	0.14
45	2028	SIPUNCULID UNID	-	-	-	1	-	1	0.14
18	2148	NEANTHES ACUMINATA	1	-	-	-	-	1	0.14
18	1757	POLYDORA SOCIALIS	-	-	-	1	-	1	0.14
3	2152	CNIDARIA, UNID	-	1	-	-	-	1	0.14
10	1636	HETERONEMERTEA SPP	-	-	-	1	-	1	0.14
18	1982	ARMANDIA BREVIS	-	-	1	-	-	1	0.14
18	2137	ETEONE LIGHTI	1	-	-	-	-	1	0.14
21	2167	TAPES JAPONICA	1	-	-	-	-	1	0.14
30	2168	COROPHIUM HETEROCERATUM	-	-	1	-	-	1	0.14
28	2175	NIPPOLEUCON HINUMENSIS	-	-	1	-	-	1	0.14
TOTAL			203	137	175	120	85	720	

APPENDIX 9-2 (cont'd.). Benthic infauna data.

LAURITZEN CHANNEL - >= 0.5 MM SIEVE SIZE
FILE:tltzbio.dat

			STATION 5						
			1	2	3	4	5	TOTAL	PCT.
18	2160	COSSURA SP	28	29	19	47	48	171	36.38
21	2139	THEORA LUBRICA	32	29	22	27	8	118	25.11
18	1673	THARYX SPP	6	8	1	5	8	28	5.96
18	2135	CIRRATULID SP A	1	11	4	-	2	18	3.83
18	2124	PSEUDOPOLYDORA PAUCIBRA	7	8	-	-	2	17	3.62
19	2131	TUBIFICIDAE, GROUP 2	3	4	2	2	6	17	3.62
19	2132	TUBIFICIDAE, GROUP 3	-	4	3	1	7	15	3.19
18	1670	MEDIOMASTUS SPP	6	3	-	2	3	14	2.98
18	2127	CIRRIFORMIA SPIRABRANCH	3	2	-	2	6	13	2.77
18	1719	LEITOSCOLOPLOS PUGETTEN	3	2	2	3	3	13	2.77
18	1684	SCHISTOMERINGOS LONGICO	-	12	1	-	-	13	2.77
19	2130	TUBIFICIDAE, GROUP 1	2	1	1	4	1	9	1.91
30	2168	COROPHIUM HETEROCERATUM	-	4	1	-	2	7	1.49
18	2126	EUCHONE LIMNICOLA	2	2	1	-	1	6	1.28
18	2123	POLYDORA LIGNI	2	1	-	1	1	5	1.06
18	1800	CAPITELLA SPP	1	-	-	-	-	1	0.21
18	2153	AMAENA OCCIDENTALIS	-	1	-	-	-	1	0.21
5	2154	PACHYCERIANTHUS FIMBRIA	-	-	1	-	-	1	0.21
20	2020	PHILINE SP A	1	-	-	-	-	1	0.21
18	1835	CIRRATULUS CIRRATUS	1	-	-	-	-	1	0.21
18	2178	MALMGRENIELLA MACGINITI	-	1	-	-	-	1	0.21
TOTAL			98	122	58	94	98	470	

APPENDIX 9-2 (cont'd.). Benthic infauna data.

LAURITZEN CHANNEL - >= 0.5 MM SIEVE SIZE
FILE:tltzbio.dat

			STATION 6						
			1	2	3	4	5	TOTAL	PCT.
21	2139	THEORA LUBRICA	32	21	11	11	23	98	26.42
18	2126	EUCHONE LIMNICOLA	20	10	10	15	10	65	17.52
18	2124	PSEUDOPOLYDORA PAUCIBRA	19	14	5	8	7	53	14.29
18	2160	COSSURA SP	16	21	2	6	-	45	12.13
30	2168	COROPHIUM HETEROCERATUM	-	7	4	7	16	34	9.16
18	2123	POLYDORA LIGNI	6	3	1	2	1	13	3.50
18	1684	SCHISTOMERINGOS LONGICO	4	2	3	4	-	13	3.50
18	2127	CIRRIFORMIA SPIRABRANCH	2	-	3	2	2	9	2.43
18	2135	CIRRATULID SP A	2	3	2	2	-	9	2.43
19	2132	TUBIFICIDAE, GROUP 3	1	1	2	-	2	6	1.62
18	1719	LEITOSCOLOPLOS PUGETTEN	1	2	1	1	-	5	1.35
18	1673	THARYX SPP	2	-	1	-	1	4	1.08
19	2131	TUBIFICIDAE, GROUP 2	1	-	1	-	1	3	0.81
20	2020	PHILINE SP A	1	2	-	-	-	3	0.81
19	2130	TUBIFICIDAE, GROUP 1	1	1	-	-	1	3	0.81
18	1710	NEPHTYS CAECOIDES	-	-	-	-	1	1	0.27
3	2152	CNIDARIA, UNID	-	1	-	-	-	1	0.27
30	2157	AMPELISCA ABDITA	-	-	-	-	1	1	0.27
18	1822	EXOGONE LOUREI	1	-	-	-	-	1	0.27
21	2164	MUSCULUS SENHOUSIA	1	-	-	-	-	1	0.27
18	1800	CAPITELLA SPP	-	-	-	-	1	1	0.27
21	2169	POTAMOCORBULA AMURENSIS	-	-	-	-	1	1	0.27
28	2175	NIPPOLEUCON HINUMENSIS	1	-	-	-	-	1	0.27
TOTAL			111	88	46	58	68	371	

APPENDIX 9-2 (cont'd.). Benthic infauna data.

LAURITZEN CHANNEL - >= 0.5 MM SIEVE SIZE
FILE:tltzbio.dat

			STATION 7					
			1	2	3	4	5	TOTAL	PCT.
18	2160	COSSURA SP	43	20	20	73	46	202	25.54
21	2139	THEORA LUBRICA	31	27	18	53	24	153	19.34
30	2168	COROPHIUM HETEROCERATUM	22	22	4	12	10	70	8.85
18	2124	PSEUDOPOLYDORA PAUCIBRA	10	14	11	10	10	55	6.95
18	2126	EUCHONE LIMNICOLA	21	9	6	6	11	53	6.70
18	1670	MEDIOMASTUS SPP	16	1	4	4	17	42	5.31
18	1800	CAPITELLA SPP	2	6	1	22	6	37	4.68
18	1822	EXOGONE LOUREI	1	2	-	31	2	36	4.55
18	2123	POLYDORA LIGNI	4	2	-	18	4	28	3.54
19	2131	TUBIFICIDAE, GROUP 2	3	1	-	6	3	13	1.64
19	2130	TUBIFICIDAE, GROUP 1	2	-	-	4	6	12	1.52
18	1684	SCHISTOMERINGOS LONGICO	1	3	1	3	2	10	1.26
18	1719	LEITOSCOLOPLOS PUGETTEN	1	3	3	-	2	9	1.14
18	2125	HARMOTHOE IMBRICATA	1	4	-	2	-	7	0.88
19	2132	TUBIFICIDAE, GROUP 3	2	1	1	2	1	7	0.88
18	2127	CIRRIFORMIA SPIRABRANCH	-	-	-	6	-	6	0.76
21	2169	POTAMOCORBULA AMURENSIS	-	2	-	2	2	6	0.76
32	2151	ZEUXO NORMANI	-	1	-	2	1	4	0.51
26	2038	HARPACTICOID COPEPOD	-	-	-	4	-	4	0.51
18	1817	AUTOLYTUS SP A	-	-	-	3	-	3	0.38
30	2157	AMPELISCA ABDITA	2	1	-	-	-	3	0.38
21	2167	TAPES JAPONICA	-	-	-	2	1	3	0.38
20	2020	PHILINE SP A	-	-	2	-	1	3	0.38
18	1942	EUPOLYMINA HETEROBRANCH	-	-	-	2	-	2	0.25
18	1793	NEPHTYS CORNUTA FRANCIS	-	-	-	1	1	2	0.25
18	2162	SPHAEROSYLLIS SP A	-	-	-	2	-	2	0.25
21	2164	MUSCULUS SENHOUSIA	-	-	-	2	-	2	0.25
30	2166	GRANDIDIERELLA JAPONICA	2	-	-	-	-	2	0.25
18	2135	CIRRATULID SP A	-	1	-	1	-	2	0.25
18	2156	ARENICOLIDAE SP A	-	-	-	-	1	1	0.13
10	1635	TUBULANUS SPP	-	1	-	-	-	1	0.13
21	1544	MACOMA NASUTA	-	-	-	1	-	1	0.13
18	2150	OPHIODROMUS PUGETTENSIS	-	-	-	1	-	1	0.13
5	2138	ACTINARIA SP A	-	-	-	1	-	1	0.13
5	2154	PACHYCERIANTHUS FIMBRIA	-	-	-	1	-	1	0.13
18	1835	CIRRATULUS CIRRATUS	-	1	-	-	-	1	0.13
5	1643	ANTHOZOA UNID	-	-	-	-	1	1	0.13
40	2155	AMPHIPHOLIS SQUAMATA	-	-	-	1	-	1	0.13
28	2175	NIPPOLEUCON HINUMENSIS	-	-	-	1	-	1	0.13
23	2181	SARSIELLA SP A	-	-	-	1	-	1	0.13
31	2182	CRAB ZOEA	-	-	-	1	-	1	0.13
30	2184	LISTRIELLA GOLETA	-	-	-	-	1	1	0.13
TOTAL			164	122	71	281	153	791	

APPENDIX 9-2 (cont'd.). Benthic infauna data.

LAURITZEN CHANNEL - >= 0.5 MM SIEVE SIZE
FILE:tltzbio.dat

			STATION 9						
			1	2	3	4	5	TOTAL	PCT.
18	2126	EUCHONE LIMNICOLA	13	42	151	22	48	276	28.93
50	2140	PHORNONIS CF. PALLIDA	5	1	4	132	39	181	18.97
30	2157	AMPELISCA ABDITA	4	15	57	1	-	77	8.07
18	1822	EXOGONE LOUREI	19	10	3	25	12	69	7.23
19	2132	TUBIFICIDAE, GROUP 3	11	10	4	18	7	50	5.24
18	2160	COSSURA SP	7	9	6	12	10	44	4.61
21	1893	CRYPTOMYA CALIFORNICA	-	11	4	-	13	28	2.94
18	1670	MEDIOMASTUS SPP	4	6	3	6	7	26	2.73
18	2161	GLYCINDE POLYGNATHA	5	3	4	-	8	20	2.10
21	2167	TAPES JAPONICA	6	2	5	6	1	20	2.10
18	2124	PSEUDOPOLYDORA PAUCIBRA	1	3	7	3	5	19	1.99
30	2168	COROPHIUM HETEROCERATUM	4	5	2	5	2	18	1.89
18	2125	HARMOTHOE IMBRICATA	-	4	1	3	3	11	1.15
21	2139	THEORA LUBRICA	2	4	4	1	-	11	1.15
23	2181	SARSIELLA SP A	-	5	3	-	1	9	0.94
18	1800	CAPITELLA SPP	-	-	1	-	6	7	0.73
32	2151	ZEUXO NORMANI	5	-	-	1	-	6	0.63
28	1600	EUDORELLA PACIFICA	2	-	2	1	1	6	0.63
18	2162	SPHAEROSYLLIS SP A	2	1	-	1	2	6	0.63
30	2180	DULICHIA RHABDOPLASTIS	1	4	1	-	-	6	0.63
1	1645	FORAMINIFERA UNID	-	-	1	-	5	6	0.63
18	2136	ASYCHIS ELONGATA	-	2	-	2	1	5	0.52
10	1636	HETERONEMERTEA SPP	-	-	-	1	4	5	0.52
21	2066	PELECYPOD, ERODED	3	1	-	-	-	4	0.42
5	1643	ANTHOZOA UNID	-	1	2	-	1	4	0.42
32	1599	LEPTOCHELIA DUBIA	-	-	-	1	2	3	0.31
21	2164	MUSCULUS SENHOUSIA	1	-	1	-	1	3	0.31
18	1719	LEITOSCOLOPLOS PUGETTEN	1	1	-	1	-	3	0.31
28	2175	NIPPOLEUCON HINUMENSIS	-	2	1	-	-	3	0.31
10	1635	TUBULANUS SPP	-	-	1	-	1	2	0.21
18	1684	SCHISTOMERINGOS LONGICO	-	2	-	-	-	2	0.21
18	1757	POLYDORA SOCIALIS	-	1	-	-	1	2	0.21
18	2153	AMAENA OCCIDENTALIS	-	-	1	-	1	2	0.21
30	2172	CAPRELLA INCISA	-	-	-	2	-	2	0.21
5	2138	ACTINARIA SP A	2	-	-	-	-	2	0.21
54	2179	LIGHTIELLA SERENDIPITA	-	-	-	-	2	2	0.21
19	2131	TUBIFICIDAE, GROUP 2	-	1	1	-	-	2	0.21
29	2185	PARANTHURA ELEGANS	-	-	-	2	-	2	0.21
18	2158	?BARANTOLLA SP	-	-	-	1	-	1	0.10
31	2174	HEMIGRAPSUS SP	-	-	-	-	1	1	0.10
30	2166	GRANDIDIERELLA JAPONICA	-	1	-	-	-	1	0.10
18	2178	MALMGRENIELLA MACGINITI	-	-	1	-	-	1	0.10
21	1544	MACOMA NASUTA	-	-	1	-	-	1	0.10
18	1793	NEPHTYS CORNUTA FRANCIS	-	-	1	-	-	1	0.10
18	2135	CIRRATULID SP A	1	-	-	-	-	1	0.10
30	2183	AOROIDES COLUMBIAE	-	-	-	1	-	1	0.10
21	2169	POTAMOCORBULA AMURENSIS	-	-	-	-	1	1	0.10
29	2186	MUNNA SP	-	-	-	1	-	1	0.10
TOTAL			99	147	273	249	186	954	

APPENDIX 9-2 (cont'd.). Benthic infauna data.

LAURITZEN CHANNEL - >= 0.5 MM SIEVE SIZE
FILE:tltzbio.dat

STATION 22

			TOTAL	PCT.
30	2168	COROPHIUM HETEROCERATUM	90	48.39
30	2157	AMPELISCA ABDITA	18	9.68
21	1893	CRYPTOMYA CALIFORNICA	17	9.14
18	2126	EUCHONE LIMNICOLA	14	7.53
18	1670	MEDIOMASTUS SPP	10	5.38
30	1511	PHOTIS BREVIPES	7	3.76
28	1600	EUDORELLA PACIFICA	4	2.15
10	2142	PALEONEMERTEA SPP	4	2.15
21	2165	GEMMA GEMMA	4	2.15
21	2169	POTAMOCORBULA AMURENSIS	4	2.15
10	1636	HETERONEMERTEA SPP	3	1.61
18	1982	ARMANDIA BREVIS	2	1.08
18	1719	LEITOSCOLOPLOS PUGETTEN	1	0.54
18	2143	EUSYLLIS SP	1	0.54
32	1599	LEPTOCHELIA DUBIA	1	0.54
18	2160	COSSURA SP	1	0.54
21	2164	MUSCULUS SENHOUSIA	1	0.54
50	2140	PHORONIS CF. PALLIDA	1	0.54
18	1684	SCHISTOMERINGOS LONGICO	1	0.54
18	2125	HARMOTHOE IMBRICATA	1	0.54
23	2181	SARSIELLA SP A	1	0.54
TOTAL			186	

APPENDIX 9-2 (cont'd.). Benthic infauna data.

LAURITZEN CHANNEL - >= 0.5 MM SIEVE SIZE
FILE:tltzbio.dat

STATION 23

			TOTAL	PCT.
30	2168	COROPHIUM HETEROCERATUM	199	66.78
30	1511	PHOTIS BREVIPES	24	8.05
28	1600	EUDORELLA PACIFICA	19	6.38
18	1670	MEDIOMASTUS SPP	10	3.36
21	1893	CRYPTOMYA CALIFORNICA	9	3.02
18	1982	ARMANDIA BREVIS	5	1.68
4	2147	TURBELLARIA SPP	5	1.68
18	2126	EUCHONE LIMNICOLA	4	1.34
21	2139	THEORA LUBRICA	4	1.34
18	1684	SCHISTOMERINGOS LONGICO	2	0.67
30	2157	AMPELISCA ABDITA	2	0.67
21	1538	MYSELLA TUMIDA	2	0.67
21	2170	CLINOCARDIUM NUTTALLI	2	0.67
18	2145	HALOSYDNA BREVITOSA	1	0.34
20	2146	TURBONILLA SPP	1	0.34
18	1761	PRIONOSPIO LIGHTI	1	0.34
40	1889	OPHIUROIDEA,UNID	1	0.34
18	2160	COSSURA SP	1	0.34
21	2165	GEMMA GEMMA	1	0.34
21	2167	TAPES JAPONICA	1	0.34
18	2136	ASYCHIS ELONGATA	1	0.34
21	2169	POTAMOCORBULA AMURENSIS	1	0.34
18	1697	PHYLLODOCE LONGIPES	1	0.34
23	2181	SARSIELLA SP A	1	0.34
TOTAL			298	

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