NOAA Technical Memorandum NOS OMA 55

CHLORDANE IN THE MARINE ENVIRONMENT OF THE UNITED STATES: REVIEW AND RESULTS FROM THE NATIONAL STATUS AND TRENDS PROGRAM

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National Ocean Service

Office of Oceanography and Marine Assessment
National Ocean Service
National Oceanic and Atmospheric Administration
U.S. Department of Commerce

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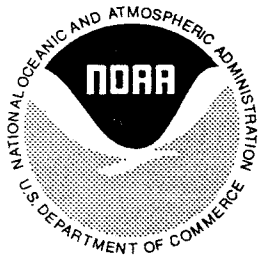
Gary Shigenaka



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EXECUTIVE SUMMARY

Chlordane is the common name for a complex technical mixture of chlorinated organic compounds employed over a span of five decades as an active ingredient in pesticide formulations. Although it saw initial application as an agricultural pesticide, the nature of its use broadened and chlordane became the pesticide of choice for structural protection against wood-boring insects as well as for other applications. Concerns about potential environmental and human health impacts prompted increasingly more restrictive regulations for permitted application of the compound, and currently in the United States, use has been virtually eliminated.

Much is unknown about the precise mechanisms of toxicity of chlordane. In acute exposures, it apparently acts as a poison of the central nervous system. Chronic exposures cause a number of subtle impacts reflected in altered blood chemistry and enzymatic activity. Chlordane is a known carcinogen in test animals, and is considered to be a probable human carcinogen.

Like other chlorinated organic pesticides, chlordane is very long lived in the environment and bioaccumulates in organisms. It is likely that a large reservoir of chlordane compounds remains in the environment. In addition, it is still exported from the United States for use in other nations and substantial evidence exists for long-range atmospheric transport of chlordane to regions far removed from the sites of initial application. For these reasons, chlordane remains a contaminant of concern in the marine environment.

A number of assessment programs have identified coastal and estuarine areas of the United States where chlordane has been found to be relatively elevated in either sediments or in resident biota. With only a few exceptions, these areas have been close to urban population centers. The National Status and Trends (NS&T) Program of NOAA is one of the major monitoring and assessment efforts by the Federal Government, and analysis of samples collected between 1984 and 1988 suggests that much of the nation's coastline is contaminated by chlordane to a relatively minor degree. Those regions that show higher concentrations in sediments and/or tissues of fish or bivalves are the urban northeastern Atlantic coast, some industrialized portions of the Gulf of Mexico coast, and the Southern California Bight region. An apparent anomaly is Choctawhatchee Bay, along the western Gulf of Mexico coast of Florida, which yielded some of the highest levels of chlordane compounds in both sediments and oysters found thus far by NS&T.

NS&T currently measures three constituents of the technical chlordane mixture: α-chlordane, *trans*-nonachlor, and heptachlor. These compounds occur in different proportions in sediments, bivalve tissue, stomach contents of bottom-feeding fish, and liver tissue of bottom-feeding fish. This probably reflects differences in metabolic breakdown of the parent compounds and accordingly, differences in biochemical metabolic pathways present in the organisms analyzed.

An extensive review of literature on chlordane is included in this report to provide both a reference on the subject as well as a context for consideration of NS&T and other results. However, comparison of results from different studies represents a substantially risky undertaking. Beyond the problems inherent in evaluating results obtained through different collection and analytical methodologies, chlordane investigations introduce other complicating factors. As examination of the reported results shows, researchers have reported concentrations of chlordane in various combinations of compounds and isomers (at least 50 constituents of technical chlordane have been reported), using at least three different reporting bases (*e.g.*, wet weight, dry weight, and lipid weight). The number of opportunities for direct comparison of results across different studies is rare, but inferences and qualified comparisons are possible and may provide some perspective for results such as those from NS&T.

Information needs for understanding the role of chlordane as a marine contaminant of concern are many. As mentioned previously, mechanisms of toxicity, particularly through chronic exposures to low concentrations, are poorly described and understood. Synergistic interactions with other contaminants of concern has only occasionally been studied. Information on physical processes as they relate to chlordane in the environment--for example, transport mechanisms, environmental chemistry, and abiotic degradation processes--would be helpful in determining fate and distribution. Fundamental data on use and distribution of chlordane, both past and present (*i.e.*, in nations where application is still permitted) is also surprisingly lacking. Even in the United States, reliability of information on past use from individual states has been marginal.

The limited temporal data that are available for the United States suggest that environmental concentrations of chlordane could be expected to decline, although there is a possibility that a residue "spike" due to permitted use of remaining stocks in the late 1980s may yet be manifested. The elimination of nearly all chlordane uses will likely result in the same kind of slow decrease seen for other chlorinated organics like DDT. There is no question, however, that chlordane will be found for many years to come by U.S. environmental monitoring and assessment programs, and that it will impact resources and users to a varying extent.

In an era when increased attention is being paid to environmental problems that are global in scope, the importance of better and more complete information may lie in its contribution to understanding potential impacts of chlordane and other compounds of human origin on distant and less resilient regions and ecosystems. Studies referenced in this review have documented the ubiquity of chlordanes in the Arctic and Antarctic, but the effects of such residues are not known.

ACKNOWLEDGMENTS

As is usually the case with acknowledgments, only a few of the people who contributed to the project can be listed here. This effort would not have been possible without the substantial and seemingly tireless assistance of NOAA Librarian Martha Thayer, who collected reference material from around the United States and elsewhere, and provided invaluable guidance for information gathering. Kitty MacCauley later assisted Ms. Thayer and I in these tasks. Dr. Alan Mearns guided me through voluminous amounts of recent and historical data in order to glean relevant information on chlordane compounds, and as always, was an enthusiastic reviewer of text, graphics, and ideas. Gunnar Lauenstein of NOAA, Ronald Sloan of the New York Department of Environmental Conservation, and Deirdre Murphy of the Maryland Department of the Environment reviewed the manuscript and provided valuable insights and criticisms. Many other federal, state, and local agency officials generously offered data and comments, and I am indebted to them for their substantial contributions. Finally, Charlene Swartzell organized and catalogued text, graphics, and references and edited the manuscript--highly formidable tasks that she performed in (mostly) good spirits.

INTRODUCTION

Chlordane is a member of the chlorinated organic pesticide group known as the cyclodienes. It was recognized as an effective insecticide in the 1940s, and like the more widely known DDT[1] gained widespread popularity between the end of World War II and the 1960s. However, like DDT, concerns about unintended and unanticipated effects of chlordane use arose during the late 1950s, and permitted uses became increasingly more restricted and regulated with growing awareness of possible adverse impacts on the environment and on human consumers. By 1983, the only remaining application of chlordane allowed by the U.S. Environmental Protection Agency (EPA) was for structural protection from termites. By 1988, chlordane use had, in effect, been eliminated.

Another characteristic that chlordane shares with other chlorinated pesticides like DDT is that of persistence in the environment. This trait was a desirable one for compounds employed to protect agricultural goods or building foundations against insect pests, yet this resistance also represented the basis for the environmental problems of toxic effects on nontarget species and accumulation in food webs. Therefore, although chlordane use in the United States is currently nonexistent, the compound and impacts from exposure to it can be expected to be observed in the environment into the future. In addition, chlordane continues to be used outside of the United States with varying degrees of restriction, especially in Third World nations. Volatilization and subsequent atmospheric transport and deposition processes will likely continue to contribute to global chlordane loads, affecting even those regions far removed from original application areas and discharge sources. As an ultimate sink for anthropogenic materials introduced into the global environment, the oceans can be expected to reflect contamination by chlordane compounds for years to come.

PURPOSE AND ORGANIZATION OF THIS REPORT

This report is intended to serve two broad purposes. First, recent results for marine environmental concentrations of chlordane from NOAA's National Status and Trends (NS&T) Program are summarized and interpreted. Second, research and monitoring results for chlordane from other sources are summarized for reference. It is hoped that the two complement each other, with the results from other studies providing a context for the NS&T results, and NS&T data providing a general point of reference for changes that may have occurred in extent of environmental contamination characterized by the other data.

This report is organized into several discrete sections which are intended to facilitate ease of use and reference. The first sections provide general information on chlordane chemistry, use, and production. These are followed by discussions of known toxic effects, primarily in traditional laboratory test animals used to generate results applicable to human health studies. Toxic effects in aquatic animals (with an emphasis on marine organisms) are summarized in the next section. Summaries and discussions of environmental residues comprise the remaining portions of the report. These are organized according to geography, with national and large regional studies presented first. This section focuses on current NS&T Program efforts, as these are the most comprehensive and complete results currently available. In subsequent chapters, the coast of the United States is divided into five subregions, and state and other studies focusing on areas within subregions are discussed under these groupings. In the following section, other chlordane investigations that have been undertaken in other parts of the world and in other environmental media are summarized in order to provide a broader context and a degree of perspective for the U.S. coastal and estuarine results. In the presentation of results from reviewed studies, emphasis is placed on the results themselves, and to a lesser extent, on interpretation of those results.

The final section of this report discusses general aspects of the results reviewed, such as any consistencies or inconsistencies suggested by the studies. Recommendations for further research and other relevant observations by the author are also included in this section. References cited, a list of species common and scientific names, and an overview of studies cited may be found in the appendices.

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[1] dichlorodiphenyltrichloroethane, in older nomenclature; the *o,p'*- isomer is currently specified as 1-chloro-2-[2,2,2-trichloro-1-(4-chlorophenyl)ethyl]-benzene

CHLORDANE

General Information

Chlordane is the common name for a synthetic organic compound used primarily as a broad-spectrum pesticide. It was the first of a group of chlorinated hydrocarbon insecticides called the cyclodienes which are characterized by cyclic structure and contain a methylene bridge (McEwen and Stephenson, 1979; Brooks, 1979a). Chlordane is considered to be a neurotoxin and acute exposure stimulates the control nervous system. Mice and rats exposed to high doses of chlordane develop tremors, a characteristic reaction to neurotoxins. However, the compound is less toxic than other cyclodiene pesticides (National Cancer Institute (NCI), 1977).

In its pure form, chlordane is a white crystalline solid with a chlorine odor. The more common industrial, or technical, product is a viscous, light yellow to brown-colored liquid with a cedar or camphor odor. It is virtually insoluble in water, but soluble in most organic solvents (International Programme on Chemical Safety (IPCS), 1984; U.S. EPA, 1986a).

Chlordane was first synthesized in 1944 by Julius Hyman of the Velsicol Corporation. The new compound was called "Velsicol 1068." The first patent describing synthesis of chlordane and other compounds of the cyclodiene group was taken out by Velsicol in December 1948. However, Hyman left Velsicol shortly after the first preparation of chlordane in Chicago and formed the Julius Hyman Company in Denver, Colorado. He continued work on the Diels-Alder reaction there, and entered into competition with his old company by producing a compound with the trade name of "Octa-Klor" that was very similar to 1068, but was produced under a different patent. Legal disputes arising from this series of events were resolved in favor of Velsicol in 1949 [National Research Council of Canada (NRCC), 1974; Brooks, 1979a].

Its insecticidal properties were initially described by Kearns, Ingle, and Metcalf (1945), who compared the relative toxicity of the new compound, then called "1068," to DDT and called it "exceptional" and "remarkable." In most tests on insect pests and larvae, chlordane was found to be several times more acutely toxic than DDT. The common name of "chlordane" was announced following conferences among groups from the U.S. Department of Agriculture, the Food and Drug Administration (FDA), and companies producing the compound. The term "technical chlordane" was proposed for the "commercially produced chemical containing 60 to 75 percent of chlordane, together with 25 to 40 percent of related compounds, occurring in the normal manufacturing processes, which are toxic to insects" (Roark, 1951). Commercial production and sales of chlordane began in 1947 (U.S. Tariff Commission, 1949).

Chlordane has been marketed under many trade names, including Aspon®, Belt®, CD 68®, Chlorindan®, Chlor Kil®, Chlortox, Corodane®, Cortilan-neu, Dowchlor®, ENT-9932, Gold Crest C-100, HCS 3260®, Kilex Lindane, Kypchlor®, M140®, Niran®, Octachlor®, Octa-Klor, Octaterr®, Ortho-Klor®, Penticklor, Prentox, Synklor®, Tat Chlor 4®, Termi-Ded®, Termide® (combination with heptachlor), Topichlor 20®, Toxichlor®, Velsicol 168, and Velsicol 1068® (International Agency for Research on Cancer (IARC), 1979 IPCS, 1984; Farm Chemicals Handbook, 1986; U.S. EPA, 1988). It has been manufactured and employed in a variety of forms, with concentrations of the technical compound ranging from 2 to 80 percent (Weiss, 1980). Oil solution formulations have been used almost exclusively for subterranean termite control, and emulsifiable concentrates, dusts, granules, and wettable powders for both termite control and certain agricultural applications (von Rümker *et al.*, 1974; Dick, 1982; Farm Chemicals Handbook, 1986).

Chlordane has the empirical formula $C_{10}H_6Cl_8$. Its CAS chemical designation is 1,2,4,5,6,7,8,8-octachloro-2,3,3*a*,4,7,7*a*-hexahydro-4,7-methano-1*H*-indene; its CAS registry number is 57-47-9. U.S. EPA (1984) reported CAS numbers of 57-74-98 for chlordane mixture, 5103-74-2 for the *cis*- isomer, and 5103-71-9 for the *trans*- isomer. The U.S. EPA assigns its own Pesticide Chemical Code (Shaughnessy) number of 058201. Figure 1 illustrates the physical structure of chlordane.

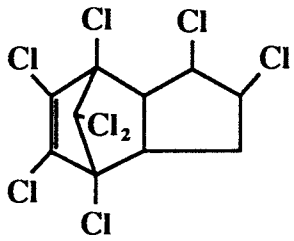

Figure 1. Chemical structure of chlordane (1,2,4,5,6,7,8,8-octachloro-3*a*, 4,7,7*a*-tetrahydro-4,7-methanoindane).

The term "chlordane" generally refers to a complex technical mixture of chlordane isomers, other chlorinated hydrocarbons, and by-products that are not separated in the manufacturing process. Technical chlordane is produced via the Diels-Alder condensation of cyclopentadiene and hexachlorocyclopentadiene, followed by a chlorination step (Morrison and Boyd, 1973; Kerkhoff, Otte, and de Boer, 1982). Details of the Diels-Alder reaction, in which an olefinic compound adds to the terminal carbon atoms of a conjugated 1,3-Diene system to give a cyclic structure, may be found elsewhere. Brooks (1979a) for example, contains a detailed discussion of the synthesis of chlordane compounds. The composition of technical chlordane has remained fairly constant since 1953 (Vettorazzi, 1981). However, there are at least 50 constituent compounds in technical chlordane (Miyazaki, Yamagishi, and Matsumoto, 1985), of which at least 14 have been identified chromatographically. Principal components are α-chlordane and γ-chlordane, heptachlor, and nonachlor. The complicated chemical makeup of chlordane mixtures has made characterization of parent compounds and analysis of residues difficult, and development of analytical capabilities to distinguish components has been relatively recent. Table 1 lists the approximate composition of technical chlordane, as specified by the Velsicol Chemical Corporation. However, it should be noted that this was a 1971 determination. As recently as 1985, the composition of technical chlordane had not been completely resolved (McEwen and Stephenson, 1979; IPCS, 1984; Miyazaki, Yamagishi, and Matsumoto, 1985), although Sovocool *et al.* (1977) performed partial or complete structure identifications on 45 individual constituents of technical chlordane using gas chromatography/mass spectrometry (GC-MS).

Table 1. Approximate percent composition of technical chlordane. Source: Velsicol Chemical Corporation, as reported in NRCC (1974).

Component	Approximate % Composition
$C_{10}H_7Cl_5$-Diels-Alder adduct from pentachlorocyclopentadiene and cyclopentadiene	2±1
Chlordene isomers in order of GLPC retention time	
chlordene	1±1
α–chlordene	7.5±2
β– and γ-chlordene (combined)	13±2
Heptachlor - $C_{10}H_5Cl_7$	10±3
Chlordane isomers	
cis (α) -chlordane	19±3
trans (γ) -chlordane	24±2
trans-nonachlor - $C_{10}H_5Cl_9$	7±3
Other constituents	
Hexachlorocyclopentadiene	maximum 1
Octachlorocyclopentadiene	1±1
$C_{10}H_{7\text{-}8}Cl_{6\text{-}7}$	8.5±2
Constituents with lower retention times than hexachlorocyclopentadiene	2±2
Constituents with higher GLPC retention times than *trans*-nonachlor	4±3

Heptachlor, the common name for a closely related cyclodiene product, is a technical chlordane constituent, and a primary manufactured pesticide unto itself. Heptachlor has the empirical formula $C_{10}H_5Cl_7$. It has been used primarily as an agricultural pesticide, although the extent of its use has been small, relative to chlordane. Heptachlor was introduced for agricultural applications by Velsicol around 1948 under the trade names "E3314" and "Velsicol 104," and widely used as a soil insecticide (Brooks, 1979a). The pure form of the compound is a white crystalline substance, but like chlordane has been much more prevalent in a technical mixture consisting of 73 percent heptachlor, 21 percent γ-chlordane, 5 percent nonachlor, and up to 2 percent other manufacturing impurities (U.S. EPA, 1976a).

Chlordane Metabolites

The three most common metabolites of technical chlordane are oxychlordane, heptachlor epoxide, and chlordene epoxide and result from epoxidation reactions of various components (Tashiro and Matsumura, 1977; NRCC, 1974; Brooks, 1979b; Musselman, 1979). These epoxides are intermediates in the biochemical detoxification process: oxychlordane derives from both the *cis*- and *trans*- isomers of pure chlordane (more readily from the latter); heptachlor epoxide, from heptachlor; and chlordene epoxide, from chlordane, chlordene, or heptachlor. Evidence exists that the epoxide intermediates are more toxic than the compounds from which they are derived.

Tashiro and Matsumura (1977) studied the metabolic pathways for *cis*- and *trans*-chlordane in rats. Their results are illustrated in Figure 2. The major route of metabolism for both *cis*- and *trans*-chlordane was found to be via dichlorochlordane and oxychlordane, which were subsequently degraded to a chlorochlordene and a chlordene epoxide. These were apparently stable and did not metabolize further. The results showed that the *trans*-isomer was the more degradable of the two chlordane isomers, although it was noted that the *cis*-isomer had been determined by others to be more readily excreted. A minor, but previously undemonstrated metabolic pathway determined by Tashiro and Matsumura was that for the formation of heptachlor from chlordanes.

Tashiro and Matsumura also found that metabolites of chlordane, such as heptachlor, dichlorochlordene, and oxychlordane were more toxic to mosquito larvae than were the parent compounds. Metabolites further down the pathway showed lesser degrees of toxicity.

Evidence referenced by Musselman (1979) indicated that epoxides of chlordane compounds have been found to accumulate to relatively high levels in tissues of both fresh- and saltwater fish at all trophic levels. In the present report, discussion focuses on parent compounds in order to simplify the already complex task of reviewing available information on chlordanes. However, some studies cited here have investigated occurrences of metabolites, and those results are included for consideration.

PRODUCTION AND USE

United States

The nature of chlordane use has changed in both qualitative and quantitative ways over the past 15 years. This may be attributed to increasingly more restrictive limitations on uses that have been imposed by regulatory agencies in the United States and other countries, as indications of adverse environmental and human health impacts have mounted.

The primary and largest U.S. manufacturer of chlordane has been the Velsicol Chemical Corporation plant in Marshall, Illinois. Velsicol has been the sole commercial producer of chlordane since about 1950 (U.S. EPA, 1976a). Other producers have included the J. Hyman Company of Denver, S.B. Penick & Co., and Chempar Chemical Co., both based in New York (NRCC, 1974; Weiss, 1980). Ingle (1965) classified the production history of chlordane as being comprised of two distinct periods: "Early Chlordane," from 1945 through 1950, during which two chlordane products were being produced (1068 chlordane, by the Velsicol Corporation, and Octa-Klor chlordane, by the Hyman Company); and "Later Chlordane," from 1951 on, during which chlordane was produced exclusively by Velsicol. The chemical composition of the former differed from that of the latter in that the early product lacked uniformity and contained variable amounts of hexachlorocyclopentadiene as an unreacted intermediate (NRCC, 1974). Ingle noted that "Early Chlordane" was more toxic to warm-blooded animals than "Later Chlordane" primarily due to the presence

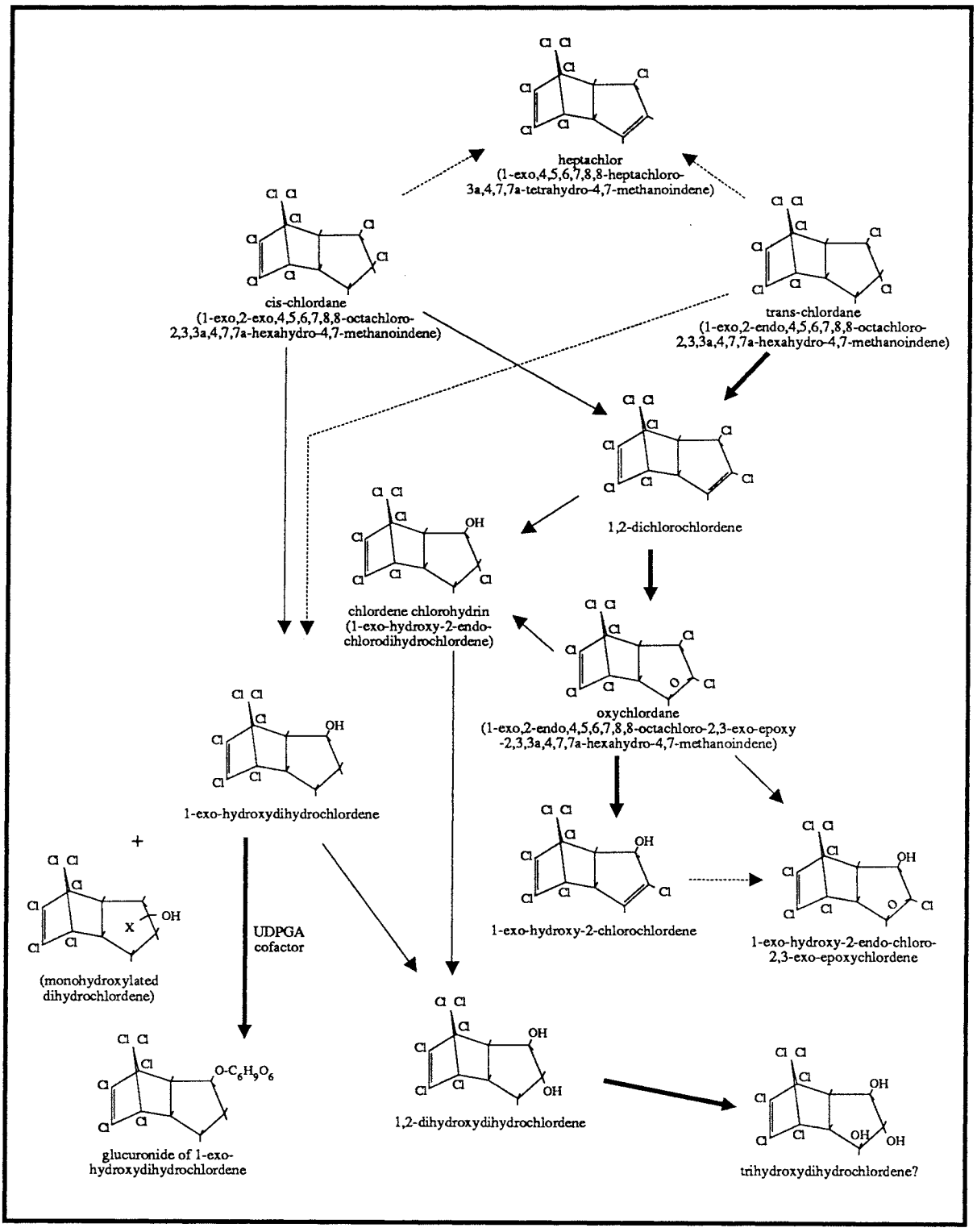

Figure 2. Metabolic pathways for chlordane isomers in rats. Solid lines indicate the active routes and broken lines represent weak metabolic routes. Thicker solid lines indicate more active pathways. Source: Tashiro and Matsumura (1977).

of hexachlorocyclopentadiene. The later formulations contained negligible amounts of the intermediate. Ingle also stated that Octa-Klor chlordane was more toxic than was Velsicol chlordane of either period.

While chemical production and distribution figures are proprietary information and not easily accessed, von Rümker *et al.* (1974) estimated 1972 chlordane production by Velsicol at its Marshall facility. Total plant production, essentially equivalent to total U.S. production, was estimated at 20 million pounds of active ingredients per year. Chlordane use as a percentage of total U.S. insecticidal application within each category was estimated by U.S. EPA (1976a). Table 2 details use by category and geographic region.

Table 2. Estimated chlordane use and distribution for 1972, based on domestic production by Velsicol Chemical Corporation. Values in millions of pounds per year active ingredients except as indicated. Source: von Rümker *et al.* (1974); U.S. EPA (1976a).

Region	Agricultural	Industrial, Commercial	Gov't Agencies[a]	Sub-Totals	Home & Garden[b]	Totals
Northeast	0.7	1.0		1.7		
Southeast	0.3	1.3		1.8		
North Central	1.7	1.6		3.4		
South Central	0.1	1.3		1.6		
Northwest	0.1	0.3		0.4		
Southwest	0.1	1.0		1.1		
(Export)				5.0		
Total	3.0	6.5	0.5	15.0	5.0	20.0
% of total U.S. insecticide use	1.4	30.2	7.1		22.9	5.5

[a] Too small to break down by region; included in subtotals.
[b] Geographic distribution not known.

In 1974, approximately 21 million pounds of chlordane were used in the United States Based on information provided by the Velsicol Chemical Corporation, the U.S. EPA estimated that commercial pest control accounted for 34.7 percent of the total; home, lawn and garden use, 29.9 percent; corn, 20.4 percent; turf, 5.9 percent; potatoes, 5.2 percent; tomatoes, 1.6 percent; ornamental shrubs, 1.2 percent strawberries, 0.8 percent; and other vegetables, 0.3 percent (U.S. EPA, 1976b). Different breakdowns of use-distribution were given in other reviews: In 1974, 22 percent agriculture, 68 percent termite control, and 10 percent home and garden and miscellaneous uses (U.S. EPA, 1976c); for 1976, 25 percent agricultural, 50 percent termite control, and 25 percent home and garden applications (U.S. EPA, 1976d).

In 1980, chlordane represented 80 percent (9.5 million pounds active ingredient) of the total pesticide usage for termite control. By 1985, chlordane use had declined substantially, but still accounted for 60 percent (3.0 to 3.5 million pounds) of termiticides used (U.S. EPA, 1986a). Agricultural applications of chlordane climbed from approximately 0.6 million pounds in 1964 and 1966, to 1.9 million pounds in 1971, to 3.0 million pounds in 1976 (U.S. EPA, 1976a). Figure 3 uses estimates of chlordane use to construct a partial time series. The estimated totals are portrayed in a temporal context, and illustrate an apparent decline following initial regulatory actions by the U.S. EPA beginning around 1977. It should be noted that the 1980 and 1985 estimates for total use are for termiticide applications only; because of regulatory restrictions, these would have constituted a substantial portion of the total use in 1980, while in 1985 would have been nearly equivalent to total use.

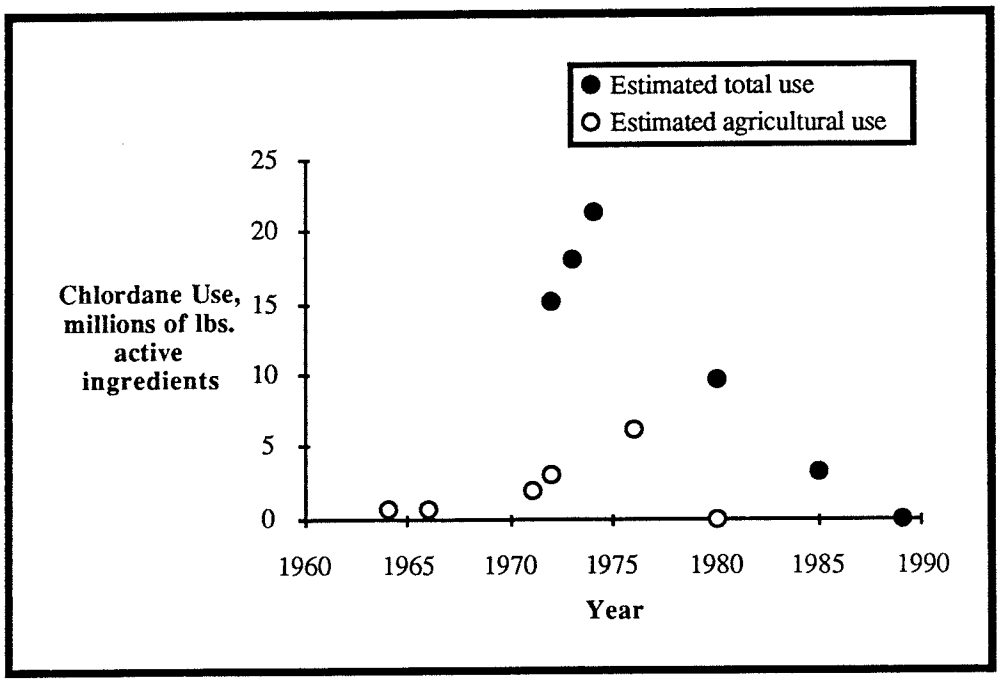

Figure 3. Temporal trend of recent chlordane use in the United States. Source: von Rümker *et al.* (1974); U.S. EPA (1976a); U.S. EPA (1986a); G. LaRocca, U.S. EPA, pers. comm.

The State of New Hampshire has required both private and commercial pesticide applicators to report pesticide usage information as a condition of license or permit renewal since 1965. This has been one of the few state programs with such rigorous reporting requirements, and is most likely the only state reporting provision that has been in existence for such a long period of time. A summary of the program and pesticide usage patterns from 1965 to 1988 was issued as McKay and Wiggin (1988). Chlordane (not specifically identified, but likely to be technical chlordane) was among those compounds whose usage was summarized. A year-by-year breakdown of chlordane use in New Hampshire was obtained from M.L. McKay, director of the Division of Pesticide Control and those results are shown below as Figure 4.

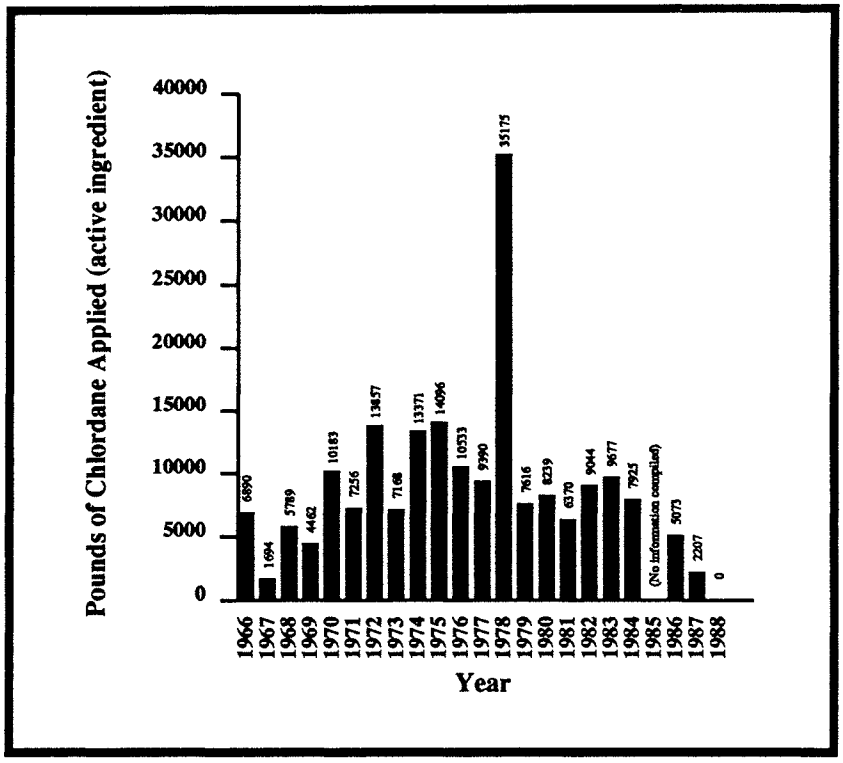

Figure 4. Temporal trend of chlordane usage in New Hampshire, 1966-1988. Source: Data supporting McKay and Wiggin (1988).

Figure 4 illustrates that use patterns in New Hampshire were variable over the two-decade period covered. Highest use totals were observed in the 1970s, while the 1980s reflected the increasingly restricted nature of chlordane use with steadily declining reported amounts. The New Hampshire Pesticide Control Board prohibited use of the compound in 1987.

The State of California has recorded patterns of use and use totals for chlordane since 1970. Figures for the period between 1970 and 1988 were obtained from the California Department of Food and Agriculture for this report and provide a number of interesting insights into use of chlordane in that state over the past two decades. Table 3 lists the commodities for which chlordane was employed in 1970 and amounts in pounds applied. This allows a detailed look at the distribution of applications prior to the imposition of significant regulatory restrictions in a state with both a large population and large-scale agricultural activities.

Table 3 shows that although chlordane was applied to a wide range of crops to control insect pests, by far its most significant use, even as early as 1970, was for structural protection. Statewide application for structural protection accounted for 83 percent of total usage in the year's totals depicted above.

Table 3. Chlordane use in California by specific application, 1970. Source: Pesticide use reports obtained from the California Department of Food and Agriculture (1990).

Application or Commodity	Pounds Applied
Almonds	24.00
Asparagus	1552.99
Avocados	35.00
Beans, dry	2035.35
Beets	69.00
Broccoli	102.00
Brussel sprouts	726.00
Cabbage	289.00
Cauliflower	156.00
Celery	318.00
Corn	1.07
Cotton	495.88
Citrus	3117.17
Fallow	2813.95
Grapes	908.00
Lemons	5109.79
Lettuce	1071.82
Melons	492.19
Nectarines	7.50
Nursery plantings	153.07
Onions	221.66
Oranges	1391.22
Ornamental plants	5.35
Onion seed	2.00
Peppers, bell	404.00
Rangeland	80.00
Residential control	77776.79
Strawberries	231.00
Sugar beets	455.00
Structural control	551207.70
Tomatoes	6299.83
Turf	133.20
Vector control	115.99
Weed	272.11
State highway	136.31
County road	4.81
Water resources	153.21
Federal agency	45.07
County Agricultural Commissioner	16.00
County or city park	1190.56
University of California	348.38
Governmental agency	1328.22
Not specified	22.41
TOTAL	**661,318.60**

Figure 5 illustrates the temporal trend of total use (chlordane and chlordane-related compounds) in California between 1970 and 1988.

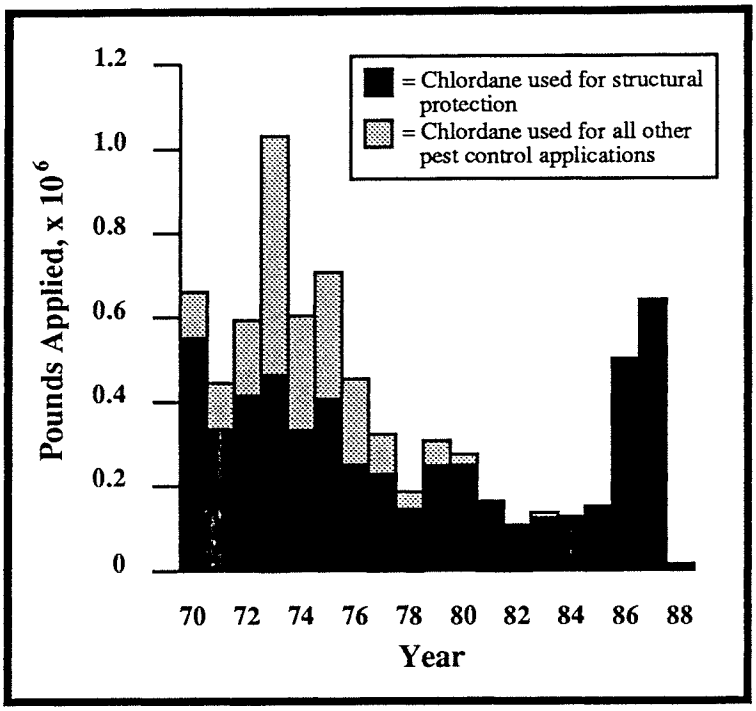

Figure 5. Temporal trend in use of chlordane and chlordane-related compounds in California, 1970-1988, showing percent contribution of applications for structural protection. Source: Pesticide use reports obtained from the California Department of Food and Agriculture (1990).

The trend portrayed in California use data differs from that in New Hampshire in that the amount of chlordane compounds applied began to increase sharply in the mid-1980s, after having declined and leveled off from peak amounts in the mid-1970s. This increasing trend ended abruptly in 1988, when nearly total elimination of permitted uses of chlordane was imposed by the U.S. EPA. Figure 5 also shows the percent contribution of structural and non-structural applications to the total for each year between 1970 and 1988. The effect of increasingly more restrictive regulations is apparent, with non-structural pesticide applications comprising a significant portion of total usage prior to 1978, but structural uses nearly equivalent to total use after 1980.

Heptachlor has been employed to a much lesser degree than chlordane. For example, in 1974, approximately 2 million pounds of heptachlor were used in the United States compared to 21 million pounds of chlordane. Heptachlor use patterns, supplied by the Velsicol Chemical Corporation and reproduced in U.S. EPA (1976a), are shown as Table 4.

Table 4. Heptachlor use in 1973 and 1974. Source: Velsicol Chemical Corporation, in U.S. EPA (1976a).

Use	1973	1974	1973	1974
Corn	1.13	1.19	57.4	58 0
Seed dressing	0.19	0.27	9.6	13.2
Pest control operators	0.61	0.55	31.0	26.8
Fire ants, miscellaneous	0.032	0.04	1.6	2.0
Total	1.97	2.05	100.0	100.0

In 1976, about 75 percent of heptachlor use was agricultural, with most of those applications targeted on corn crops. According to a 1971 U.S. Department of Agriculture survey, 91 percent of agricultural uses of heptachlor were related to corn. This was reflected in regional use distributions, which indicated that Corn Belt states employed the greatest amounts of the compound (U.S. EPA, 1976a)

In 1970, in an effort to assess urban contamination by use of persistent chemical pesticides, Carey, Wiersma, and Tai (1976) sampled and analyzed soils from 14 U.S. cities for residues of chlorinated pesticides, including chlordane and heptachlor (not specifically identified as to isomers or constituents). The cities ranged in population from less than 15,000 to nearly 2 million. Carey, Wiersma, and Tai found chlordane residues in all 14 cities, and heptachlor in 3 of 14. The frequencies of occurrence at the randomly selected sites within the cities varied considerably, however. Table 5 summarizes chlordane and heptachlor results by city.

Table 5. Chlordane and heptachlor residues in soils of 14 U.S. cities, 1970. Source: Carey, Wiersma, and Tai (1976); 1970 population information from Hoffman (1988).

Compound	City	1970 Population	Percent Positive Sites	Range of Detected Residues, ppm	Arith. Mean	Geom. Mean
Chlordane	Augusta, ME	21,945	11.1 (3/27)	0.21-0.27	0.03	0.0043
	Charleston, SC	66,945	25.9 (7/27)	1.01-1.35	0.12	0.0094
	Cheyenne, WY	41,254	5.3 (1/19)	8.99	0.47	-
	Grand Rapids, MI	197,649	43.5 (10/22)	0.15-6.58	0.71	0.0556
	Greenville, MS	39,648	7.1 (2/28)	0.37-1.40	0.06	0.0036
	Honolulu, HI	324,871	28.6 (6/21)	1.00-13.90	1.27	0.0406
	Memphis, TN	623,988	21.4 (6/28)	0.11-8.02	0.36	0.0138
	Mobile, AL	190,026	24.1 (7/29)	0.10-2.50	0.18	0.0157
	Philadelphia, PA	1,949,996	42.3 (11/26)	0.18-4.59	0.76	0.0705
	Portland, OR	379,967	12.0 (3/25)	0.40-0.59	0.06	0.0059
	Richmond, VA	249,332	18.5 (5/26)	0.18-6.42	0.51	0.0154
	Sikeston, MO	14,699	7.4 (2/27)	0.30-1.19	0.06	0.0036
	Sioux City, IA	85,925	18.2 (4/22)	0.30-3.00	0.24	0.0127
	Wilmington, DE	80,386	7.4 (2/27)	0.04-0.07	<0.01	0.0015
Heptachlor	Augusta, ME	21,945	Not detected			
	Charleston, SC	66,945	Not detected			
	Cheyenne, WY	41,254	Not detected			
	Grand Rapids, MI	197,649	4.3 (1/23)	0.13	0.01	-
	Greenville, MS	39,648	Not detected			
	Honolulu, HI	324,871	Not detected			
	Memphis, TN	623,988	3.6 (1/28)	0.23	0.01	-
	Mobile, AL	190,026	3.4 (1/29)	0.01	<0.01	-
	Philadelphia, PA	1,949,996	Not detected			
	Portland, OR	379,967	Not detected			
	Richmond, VA	249,332	Not detected			
	Sikeston, MO	14,699	Not detected			
	Sioux City, IA	85,925	Not detected			
	Wilmington, DE	80,386	Not detected			

Carey, Wiersma, and Tai also statistically compared mean residues in urban soils with those in cropland soils in the same state or agricultural region. Of the areas listed above, 12 of the 14 were examined (cropland data were not available for Portland, Oregon and Honolulu, Hawaii). In 3 of the 12 regions (Grand Rapids, Philadelphia, and in Richmond), statistically significant ($p<0.01$) differences were noted between urban and cropland chlordane concentrations. In all three cases urban concentrations were significantly greater than cropland concentrations.

Combining the above findings with 1970 population data (Hoffman, 1988) for the 14 cities studied shows significant correlations between chlordane concentrations measured in urban soils and populations in the regions. For example, using the nonparametric statistic, Spearmans rank correlation procedure (Zar, 1984), percent occurrence of chlordane residues was found to be significantly correlated to urban population with the Spearmans rank correlation coefficient r_S=0.66 and p=0.0173 and mean chlordane concentration for urban areas was found to be correlated with r_S=0.616 and p=0.0264. These results, with those of Carey, Wiersma, and Tai, suggest that chlordane is a contaminant more likely to be associated with urban regions than agricultural areas.

Other Nations

Production, distribution, and use information for pesticides in other nations is available to widely varying degrees and with similarly variable reliability. In general, it would appear that the industrialized nations of the West have restricted or banned the use of many persistent organochlorine compounds, while regulation of such pesticides is less evident in Third World or developing nations. As an example, the adverse ecological effects of DDT have been well documented for over 40 years, and its use has been banned in the United States and other western nations for nearly 30; yet, in countries such as Bangladesh, the compound is still widely employed for vector control. In fact, the only pesticide manufacturing plant in that country in 1984 was a facility for producing DDT (Staring, 1984). Foreign data for chlordane are not numerous, likely reflecting both more limited use compared to compounds like DDT and reporting inadequacies. Referring to chlordane and heptachlor in particular, Brooks (1979a) observed that "current patterns of usage in other countries are obscure."

In Canada, chlordane is regulated federally under the Pest Control Products Act, with individual provinces empowered to restrict uses more stringently. Historically, chlordane has been used in a variety of applications, including structural protection and pest control for ornamental plants, lawns, and some crops (IPCS, 1984). The first Canadian use restrictions for the compound occurred in 1970. By 1985, nearly all uses except those for subterranean termite control had been cancelled Further restrictions are currently under review (M. Edwards, Agriculture Canada, pers. comm., 2 December 1988) Table 6 summarizes the regulatory history of chlordane in Canada.

Table 6. Regulatory history for chlordane in Canada. Source: M. Edwards, Agriculture Canada, unpublished.

Date	Regulatory Action
2/23/70	Re-evaluation announced.
9/9/70	Minor revisions to use pattern, deletion of outdoor fogging for mosquito control and application to dogs.
11/13/75	Registrant comments requested on proposed revisions in use pattern.
9/29/76	Changes in use pattern deferred to 1/1/78.
1/21/77	Suspended most structural uses, mixtures, all foliar and corn applications.
2/3/78	Fertilizer regulations suspended.
1/1/79	Suspended chlordane in fertilizers.
10/31/83	Only carpenter ant, termite structural pest control uses acceptable after 1/1/85; suspensions to take effect 6/1/85.
12/2/85	All uses suspended 12/31/85, except essential restricted use for subterranean termites.

Chlordane is not produced in Europe, and has never been manufactured in Japan, although it has been imported for limited applications in control of termites and other structural pests (IARC, 1979; IPCS, 1984); Miyazaki, Yamagishi, and Matsumoto (1985) estimated use in Japan for control of termites and powder post beetles at 1,500 tons (682,000 pounds) per year. Use restrictions for chlordane have been in place in many countries for several years. For example, Germany banned its use on agricultural crops in May 1971 (Fuchs and Schröder, 1983). Japan ended termiticidal applications in September 1986 (Hirai and Tomokuni, 1989).

Chlordane has not been used in Norway since 1967 (Skåre *et al.*, 1985). Kerkhoff, Otte, and de Boer (1982) stated that use of chlordane in Sweden and Finland has been negligible in the past, and that in The Netherlands, present applications are not permitted. Despite these use restrictions, chlordane compounds have been detected in biota from these nations, a finding which will be discussed in greater detail in a subsequent section of this report.

In Australia, chlordane has been deregistered for use in any agricultural or horticultural application, but may still be purchased and used by licensed pest control officers. In May of 1987, organochlorine pesticide residues exceeding Australian standards were detected in export produce. Screening procedures were tightened, and it is possible that more extensive regulatory actions may result (Allender, 1989).

Staring (1984) summarized a large amount of information for supply, distribution, and use of pesticides in the Asia-Pacific region. However, of 20 nations discussed, chlordane was specifically mentioned for only 3. In India, chlordane import records between 1978 and 1982 suggested a declining trend. In 1978-79, 21 metric tons were imported; 1979-80, 12 tons; 1980-81, 14 tons; and 1981-82, 6 tons. In a discussion of methods for an acute toxicity test involving chlordane, Mani and Konar (1986) noted that chlordane was used in India for control of white ant and other insects, and further commented that they obtained Termex, a pesticide formulation containing 20 percent chlordane, from an Indian manufacturer, Rallis India Pvt. Ltd.

In Nepal, data for 1981 showed that 790 kilograms (kg), or 40 kg of chlordane active ingredient, were either imported or prepared in that country. Staring's discussion of Papua, New Guinea listed chlordane as a pesticide being used in 1982, but import or use figures were not given.

Soerjani (1988) listed chlordane and heptachlor as being among those pesticides whose uses are restricted in eight Asian countries (India, Indonesia, Malaysia, Pakistan, Thailand, the Phillipines, Sri Lanka, and Bangladesh) and banned by the Republic of Korea.

A report issued by the United Nations Industrial Development Organization (Maltby, 1980) summarized pesticide use in Latin America. The nations included in the report were Argentina, Bolivia, Brazil, Central America, Chile, Colombia, Ecuador, Mexico, Paraguay, Peru, Uruguay, and Venezuela. Although chlordane was not specifically listed in usage breakdowns, heptachlor was. The total Latin American use for heptachlor in 1978 was estimated to be 755 tons of active ingredient, with three nations--Argentina, Brazil, and Mexico--accounting for 91 percent of this total. Estimated use in those three countries was about equal, with 220, 210, and 255 tons being applied in Argentina, Brazil, and Mexico, respectively.

LePelley (1968), in a book on coffee pests, noted that emulsion or powder forms of chlordane have been applied to coffee plants in Venezuela to protect against ants attending mealybugs. In Nicaragua, tettigoniid (grasshopper) pests have been controlled by application of 10 percent chlordane dusted on the soil at a rate of 15 pounds per acre. Chlordane was also listed by LePelley as a tool for control of leaf-cutting ants in the Americas (2% oil emulsion). Chlordane injected with a syringe into nests was identified as a treatment for tree-boring ants in Africa. Emulsifiable concentrates at 0.3 percent were used in Brazil against leaf miner larvae, and, as noted above, in Venezuela at 0.37 percent (or 40% wettable powder) against ants and mealybugs.

The pattern that emerges from available information indicates that substantive use restrictions that have been enacted in other countries have occurred in industrialized Southeast Asian and Western nations. Consistent with use patterns of other organochlorine pesticides, such as the DDT family, chlordane use in developing nations appears to have been much less regulated or restricted. While Maltby (1980) estimated that heptachlor use in Latin America would decline by 24 percent in the 10-year period between 1978 and 1988, this decline is considerably less than that observed in the industrial nations over the same period.

REGULATORY ASPECTS

United States

In the United States, federal regulation of chlordane primarily falls under FIFRA (the Federal Insecticide, Fungicide, and Rodenticide Act, originally passed in 1947 by Congress; subsequent pesticide legislation and amendments, such as the Federal Environmental Pesticide Control Act of 1972, continue to be referred to as FIFRA). Waste materials resulting from the manufacture of chlordane are considered hazardous and are partially regulated under the Resource Conservation and Recovery Act of 1976 (RCRA) (Younos and Weigmann, 1988).

Chlordane was initially registered for use as an insecticide in the United States in 1948. From that time until the 1970s, it became one of the most heavily used household pesticides, and also gained widespread popularity as an agricultural and structural treatment. In 1955, tolerances of 0.3 parts per million (ppm) were established for chlordane on 46 fruit and vegetable crops, based on the results of toxicity studies presented at congressional hearings in 1950. In 1963, because of conflicting and inadequate data on toxicity, and inspired at least in part by the heightened public concern about pesticide use fostered by publication of Rachel Carson's *Silent Spring*, the Commissioner of Food and Drugs recommended a "zero" tolerance for chlordane on 54 agricultural commodities. In response to this recommendation, the Velsicol Chemical Corporation petitioned the National Research Council of the National Academy of Sciences to review the proposed change. This advisory panel studied more recent data on chlordane toxicity, and in 1965 concluded that "current technical chlordane is less toxic (than previous formulations) both acutely and chronically and is not a hazard as presently used in accordance with current practices" (Ingle, 1965). It was determined that the tolerance should remain at 0.3 ppm unless further evidence of potential hazard arose at a later date. As a result, an order reinstating the tolerance level at 0.3 ppm appeared in the Federal Register in April 1965 (Brooks, 1979a).

In 1974, the Administrator of the EPA proposed cancellation of all uses of chlordane due to positive results obtained in carcinogenicity tests, evidence of persistence in the environment, and evidence of bioaccumulation in food webs (U.S. EPA, 1976a; U.S. EPA, 1986a). In December of the following year, the EPA suspended production and use of chlordane and heptachlor for most food crops and all home and garden applications (Dick, 1982; Sittig, 1985). The use restrictions imposed by this action were estimated to potentially reduce the amounts of chlordane and heptachlor entering the environment by 90 percent and 85 percent, respectively, compared to 1974 levels (U.S. EPA, 1976a). The 1975 action permitted continued agricultural uses of chlordane and heptachlor for which there were no feasible alternatives and also allowed continued use of existing stocks of the compounds intended for restricted applications.

In March of 1978, 3 years of administrative litigation on the heptachlor/chlordane cancellation proceedings initiated by the EPA came to a close when an agreement on contested uses was reached. The settlement permitted limited use of chlordane by crop, location, time, and amount. Production and distribution restrictions were also imposed. A schedule was instituted for the 5-year phaseout of contested agricultural uses (*e.g.*, treatment of citrus, grapes, flax, strawberries, non-food/feed producing land, and nursery stock). Annual production limitations of no more than 7.25 million pounds of chlordane and heptachlor were also imposed for the still permitted termite control uses (IARC, 1979; Musselman, 1979; U.S. EPA, 1986a). By July of 1983, the only approved application of chlordane was for the control of underground termites (IPCS, 1984; G. LaRocca, U.S. EPA, pers. comm., 29 November 1988).

In 1987, the EPA and Velsicol Chemical Corporation entered into an agreement under which Velsicol voluntarily cancelled registration of many termiticide products and conditionally registered others. The agreement provided for a phased reduction in sales and use of most chlordane products until April 15, 1988, after which no sales or distribution of those products were permitted (U.S. EPA, 1987a; U.S. EPA, 1987b). While the EPA order authorized some limited chlordane termiticide applications, it is unlikely that Velsicol will promote or support any continued use of the compound in the United States (G. LaRocca, U.S. EPA, pers. comm., 29 November 1988).

Beyond these federal actions, individual states have also regulated use of chlordane compounds. Although the timing and content of these regulatory actions have varied from state to state, most have followed the lead of the EPA. After it was determined that chlordane and the cyclodienes caused cancer in laboratory animals, the State of New York issued an emergency directive halting all use of chlordane in March of 1985 [New York Department of Environmental Conservation (NYDEC), 1985]. This was the

first outright ban of chlordane in the nation, but it was followed shortly thereafter by similar actions in Massachusetts. In April of 1987, the interim regulations were made permanent (NYDEC, 1987). As was the case for EPA's restrictions on use of chlordane compounds, the driving force behind state actions was primarily one related to concern for human health, mostly associated with exposure to household residues after treatment for termite control. For example, the State of New York analyzed over 1,400 household air samples from 515 private homes in 1983. Detectable levels of chlordane were found in 1,039, or about 74 percent of the samples. This finding was cited in the NYDEC's announcement of the permanent ban (NYDEC, 1987).

The only U.S. producer of chlordane, the Velsicol Chemical Corporation of Rosemont, Illinois, is not presently manufacturing the compound, although current laws and regulations permit it to do so. Velsicol does, however, market remaining stocks of chlordane outside the United States Amounts sold and identity of receiving nations are proprietary information and therefore not available for this report (D. Jennings, Velsicol Chemical Corporation, pers. comm. 12 July 1989). Although other chemical producers in the United States and elsewhere have synthesized chlordane, the Velsicol Corporation believes it is presently the sole manufacturer of chlordane in the world (D. Jennings, Velsicol Chemical Corporation, pers. comm., 17 August 1989).

STANDARDS AND CRITERIA

In the United States, a number of standards for levels of chlordane in water, sediments, and biota have been established over the years, with varying degrees of acceptance and institutionalization. Recommendations for acceptable levels in drinking water and foods have been emphasized, with broader concerns about resource and ecosystem impacts more recently considered.

The predecessor to the EPA, the Federal Water Pollution Control Administration (FWPCA), established a standard for permissible levels of chlordane in public water supplies of 3.0 µg/l. The standard for farm use was also 3.0 µg/l. The FWPCA listed the 48-hour LC_{50} value for chlordane as 2.0 µg/l (in shrimp, one of the most sensitive groups of marine organisms) and based on an assumption used by FWPCA that 1/100 of the LC_{50} level represented a reasonable application level, the recommendation was made that environmental levels not be permitted to rise above 50 ng/l. It was noted that this criterion was so low that chlordane, among other pesticides, could not be applied in or near the marine habitat without danger of causing damage. (FWPCA, 1968).

In 1974, the U.S. National Academy of Sciences (NAS) convened a panel of experts to recommend acceptable environmental levels of contaminants, including chlordane. The NAS proposed that in order to protect wildlife predators, chlordane concentrations in freshwater fish should not exceed 0.10 ppm wet weight. For saltwater fish, the recommended limit was 0.05 ppm (NAS, 1972). However, the NAS guidelines were not adopted as formal criteria.

The National Shellfish Sanitation Program (NSSP), a multiagency cooperative effort coordinated by the U.S. FDA, proposed "alert" levels for excessive chlordane contamination of shellfish of 0.03 ppm wet weight (NAS, 1972). Like the NAS recommendations, the NSSP proposed limits were not accepted as regulatory standards.

The U.S. EPA 1980 revisions to water quality criteria for protection of aquatic and marine life list permissible concentrations of chlordane in water as the following: to protect freshwater aquatic life, 0.0043 µg/l as a 24-hour average, not to exceed 2.4 µg/l at any time; to protect saltwater aquatic life, 0.0040 µg/l as a 24-hour average, not to exceed 0.09 µg/l; to protect human health, preferably zero. Exposure to concentrations of 0.0046 µg/l, 0.00046 µg/l, and 0.000046 µg/l was estimated to result in an additional cancer risk of 1 in 100,000, 1 in 1,000,000, and 1 in 10,000,000, respectively, assuming consumption of 2 liters of water and 6.5 grams contaminated fish per day by a 70 kg adult (Dick, 1982; Sittig, 1985; U.S. EPA, 1988b).

The most familiar, and most widely cited administrative guideline for acceptable concentrations of chlordane is the FDA action level. This represents the limit at or above which the FDA will take action to remove contaminated products from the market. The action level for chlordane has been established at 0.3 ppm in edible portions of fish. In this case, residues of heptachlor, heptachlor epoxide, *cis*- and *trans*-chlordane, *cis*- and *trans*-nonachlor, oxychlordane, α-, γ-, and β-chlordene, and chlordene are summed into a total value for chlordane. Levels of individual components must be quantitated at 0.02 ppm or above and confirmed in order to be included in the total. If heptachlor and/or its epoxide are proportionally higher than

the amount of other chlordane compounds present, then action levels established specifically for those compounds are applied. In recognition of the complex composition of common chlordane formulations, the FDA specifies that when the analytical pattern of the residue matches that for technical chlordane, quantitation should be against a technical chlordane reference standard; when the pattern consists of identifiable individual components, appropriate individual reference standards should be used and the component values summed (U.S. FDA, 1985). Although often used as benchmarks against which measured residue concentrations are compared, FDA actions levels are administrative guidelines and are not generally coded into law.

In Australia, the National Health and Medical Research Council maximum residue limit for chlordane in public water supplies is 6.0 µg/l (ppb) (Ang, Meleady, and Wallace, 1989). In Brazil, the maximum permissible concentration of chlordane allowed in potable water is 3.0 µg/l, while that for heptachlor is 0.10 µg/l (Cáceres, Tundisi, and Castellan, 1980).

In an informal effort to provide guidance for interpretation of sediment monitoring results from NOAA's NS&T Program (discussed in greater detail elsewhere in this review), Long and Morgan (1990) gathered data from a variety of methods and approaches to evaluate potential for biological effects caused by sediment contamination. Three basic approaches were examined: equilibrium partitioning, spiked sediment bioassays, and miscellaneous methods for evaluation of synoptically collected field data. Chemical concentrations observed or predicted to be associated with adverse biological effects were determined and sorted to provide an estimate of sediment concentrations above which impacts could be expected. Based on the data they examined, Long and Morgan determined a lower 10 percentile chlordane sediment effects concentration at 0.5 ppb, dry weight, with a median effects concentration at 6.0 ppb.

PATHWAYS TO THE MARINE ENVIRONMENT AND ENVIRONMENTAL FATE

The behavior of chlordane-related compounds in the environment is of primary importance in determining their distribution on both large and small scales, as well as in determining to what extent specific resources may be impacted. The U.S. EPA (1988a) stated that available data were insufficient to fully assess the environmental fate of chlordane. However, some general characteristics were noted; among these, that chlordane is persistent and bioaccumulates, and that since the compound is nearly insoluble in water and should adsorb to soil surfaces, would not be expected to reach aquifers. Cardwell *et al.* (1977) stated that upon introduction into the environment, chlordane probably shares a fate similar to other chlorinated hydrocarbon compounds: some components vaporize and are carried away by thermal convection and other physical processes; while terrestrially, chlordane can be adsorbed by organisms or particulate matter or enter water bodies through dissolution or sorbing to particles.

In a report produced for the U.S. EPA, Callahan (1979) summarized knowledge of the environmental fate of chlordane in a table, reproduced below as Table 7.

Table 7. Summary of environmental fate of chlordane in aquatic media. Source: Callahan *et al.* (1979).

Environmental Process[a]	Summary Statement	Half-Life, $t_{1/2}$	Confidence of Data
Photolysis	Sensitized processes may be important	–	Low
Oxidation	No information available	–	-
Hydrolysis	Not an important process	>4 years	High
Volatilization	Probably an important process	–	Medium
Sorption	Probably an important process	–	Medium
Bioaccumulation	Is an important process	–	High
Biotransformation/ Biodegradation	Is very slow, but may be an important process for ultimate degradation	–	Medium

[a] There is insufficient information in the reviewed literature to permit assessment of a most probable fate.

U.S. EPA (1988b) stated that chlordane is hydrolyzed poorly, does not undergo significant biodegradation, and has a half-life in soil of 4 years. Residues were reported to persist for 14 years or longer. Lichtenstein and Polivka (1959) evaluated the residual effectiveness of both chlordane and heptachlor applications on experimental turf plots after 12 and 9 years, respectively. They found that for chlordane, between 12.4 and 17.8 percent of initial treatment concentrations remained. For heptachlor, although no residues were detected directly, a bioassay method indicated that between 3.4 and 5.9 percent of initial application remained. This resistance to degradation was echoed in a brochure prepared by the Velsicol Chemical Corporation (no publication date) which stated that a standard chlordane soil treatment remained 100 percent effective for 34 years, and that houses treated with chlordane may remain free of termites for more than 20 years.

Bennett *et al.* (1974) analyzed soil samples around the foundation of a home treated with chlordane 21 years before, and found that as much as 15 percent of the γ- isomer was still present in the soil. Concentrations in the upper layers of soil ranged between 4.35 and 15.0 ppm. They also determined that horizontal and vertical movement of chlordanes through the soil was limited, with concentrations of γ-chlordane dropping off markedly with depth in the soil and distance from the foundation. Based on these observations, it would be reasonable to suggest that chlordane also would not enter nearby waters in a dissolved state. However, entry via adsorption to particles transported by runoff or tributary waters would be a possibility.

Atmospheric Processes

Musselman (1979) noted the widespread usage (at that time) and high volatility of chlordane, and called chlordane vapor residues "ubiquitous" in the atmosphere. He stated that chlordane, its technical components, and degradation products enter open water via rainwater, storm sewer discharges, and by suspended sediments and cited the near-universal occurrence of those compounds in U.S. surface waters.

Cohen and Pinkerton (1966) collected and analyzed dust ,whose origins were known to have been a storm nearly two thousand miles distant from their sampling site, and found measurable concentrations of seven pesticides. Technical chlordane and DDT were the major pesticide components of the dust, with chlordane concentrations measuring 0.5 ppm, based on air-dried weight of dust. These findings were cited as evidence supporting a large-scale pesticide transport mechanism of soils containing the compounds mobilized by winds, carried at high altitudes over long distances, and deposited by sedimentation or in rainfall.

The possibility that volatilized and adsorbed residues of chlordane may be transported great distances to areas where the compound has been used infrequently, if at all, was supported by the work of Bidleman *et al.* (1981), who measured airborne concentrations of pesticides and PCBs circulating over the North Atlantic. They found chlordane (*cis-* + *trans*-chlordanes) at detectable concentrations in air samples from Barbados (3.9-22 x 10^{-12} g/m^3), Newfoundland (20-64 x 10^{-12} g/m^3), and on a trans-Atlantic crossing from the western coast of Africa to the northeastern coast of the United States (13-32 x 10^{-12} g/m^3).

As part of the Upper (Chesapeake) Bay Survey, Tzou (1975) evaluated several different phases of meteorological and hydrologic cycles as potential distribution and transport mechanisms for pesticides and PCBs to the Chesapeake Bay. Chlordane ($\alpha+\gamma$) was measured in atmospheric vapor, airborne dust, rainwater, storm sewer water, and groundwater in the Baltimore area, and in a major freshwater input (the Susquehanna River). Tzou combined these results with data for air circulation patterns, wind, precipitation, and river flow to estimate rates of chlordane transport in several media around the Chesapeake Bay. In the atmosphere, the rate of transport in a square meter section of the vertical air column was calculated to be 7 mg/m^2/month. The rate of deposition from the atmosphere via precipitation and particle sedimentation was estimated to be 21 mg/acre/month. For stormwater runoff, although a rate based on discharge volume was not calculated, measurements of chlordane concentration were made prior to a storm event and about six and a half hours after it began; chlordane concentration in the water increased significantly from 11 parts per trillion (ppt) to 63 ppt. Groundwater was found to contain relatively low levels of chlordane, 2 ppt. In the Susquehanna River, chlordane concentration in suspended sediment was found to be 0.030 ppm. Based on estimates of monthly sediment loadings from the river into the Chesapeake Bay, and assuming a constant chlordane concentration in those sediments as measured above, Tzou calculated a total chlordane input from the Susquehanna River to be 24 kg per year.

Waterborne Processes

Moser (1985) studied the storage and transport of sediments, pesticides, and PCBs in New Jersey fluvial systems, where chlordane had been found to be present at concentrations high enough to be of concern. She identified four depositional environments in the study area: non-tidal river channel, non-tidal flood plain, impoundments, and tidally influenced river. Moser found that chlordane concentrations tended to be highest in the low-energy non-tidal river channel, flood plain, and tidally influenced river environments. She also determined a relatively high degree of correlation among smaller fraction grain size, organic carbon content, and chlordane (α- and γ- isomers) concentration. Moser stated that based on physical process studies, pollutants like chlordane that show an association with particulate matter are available for resuspension and transport out of low-energy environments. Chlordane was characterized as an ubiquitous, nonpoint source, suburban pollutant with few recognizable inputs but most frequently occurring in areas of low-density residential land use.

Oloffs, Albright, and Szeto (1972) and Oloffs *et al.* (1973) examined the behavior of several chlorinated organic compounds, including α-chlordane and γ-chlordane, in natural waters incubated in the laboratory, both with and without underlying natural sediments. They found that both chlordane compounds accumulated at the water-air interface and escaped from natural waters into the atmosphere by evaporation if incubated without bottom sediments. However, in the presence of sediments, all detectable residues moved from the water and into the sediments after 6 and 12 weeks, respectively. This would suggest that in shallow, well-mixed water bodies, α- and γ- chlordanes could be expected to move into underlying sediments, while in deep waters with little movement, the fate of chlordane would be dependent upon depth, or degree of contact with bottom materials.

Oloffs *et al.* (1973) studied the effects of agitation and the presence of surface-active agents in water samples on the fate of γ-chlordane. They determined that while agitation had no significant effect, surface-active agents markedly inhibited the movement of the compound out of the water. For example, in samples without surfactants subjected to agitation, after 12 weeks, 4.6 and 13.8 percent of original concentration remained in water; similarly, in samples without surfactants and not subjected to agitation, 6.0 and <1 percent remained. However, in those samples with surfactant added, 82.9 and 79.1 percent (with agitation); and 75.1 and 71.4 percent (without agitation) of original γ-chlordane concentration remained. In both this study and the previous effort (Oloffs, Albright, and Szeto, 1972), the authors suggested that the presence of surfactants, such as industrial or domestic detergents, in sewage outfalls or near cities, may cause behavior of chlordane compounds there to differ substantially from that in other more removed natural waters.

Yamagishi *et al.* (1981) used measurements of chlordane (identified as *cis*-, *trans*-, and oxychlordane, and *cis*- and *trans*-nonachlor) in seawater, clams, and fish in and near Tokyo Bay to make inferences about routes of contamination in marine biota. They found that residual levels in fish and shellfish were lower at the entrance to Tokyo Bay than at inner portions of the bay. Measurements in freshwater fish upstream from the bay showed the common tendency of lower levels in upstream regions and higher levels in downstream regions. Yamagishi *et al.* stated that the river water contamination likely originated from sewage discharges into the river or from runoff following seasonal rainfalls fed into the rivers from small drainways. The apparent gradient of increasing concentrations in biota with proximity to inner bay regions was said to be a possible result of increased exchange between bay and ocean waters at the bay entrance and/or proximity to lumber factories and wooden residences where chlordane would have been used for protection from insects.

The State of Missouri and the U.S. Geological Survey (USGS), through the Missouri Water Resources Research Center (MWRRC), have sponsored studies focusing on the movement of chlordane in the environment. Because chlordane compounds have been found in relatively high concentrations in fish from lakes and rivers of the State, movement and fate of chlordane have been significant concerns. For example, under the auspices of MWRRC, Kapila, Puri, and Orazio (1988) investigated modes of chlordane transport from soil to the aquatic environment, while; Peyton, Anderson, and Gantzer (1988) focused on chlordane movement during rainfall. Kapila, Puri, and Orazio applied chlordane to test soils in controlled conditions simulating the natural environment and found that the rate of chlordane loss through volatilization from soil surface and leaching into the soil bed were low in all types of soils. Leachability was only marginally affected by the introduction of surfactants into the water applied to the soils, though in a saturated system, the presence of surfactant increased the water concentration of chlordane by a factor of five. The authors also found that unsaturated chlordane components such as heptachlor and chlordenes were readily transformed into oxychlordane and heptachlor epoxide, with the rate of transformation related to the moisture content of the soils.

Peyton, Anderson, and Gantzer determined from their rainfall-runoff tests that a significant mass of chlordane was mobilized in runoff and this mass increased with time to a steady state, regardless of soil type. Chlordane concentration was correlated with organic matter content in the soils. The vertical profile of chlordane concentrations was fairly constant throughout the soil profile, except at the surface. The concentration in the upper 0.5 cm was significantly lower than the rest of the profile, which was taken as an indication that runoff is an important transport mechanism for chlordane movement and that the uppermost soil layer is the major source. Analysis for individual chlordane components in leachate showed that the percent distribution of chlordane, heptachlor, and *cis*- and *trans*-chlordane was approximately the same as that in technical chlordane, except in leachate from sandy soil. In the latter, heptachlor, which is the most insoluble of the four components studied, did not desorb from soil into water as readily as the other compounds. Peyton, Anderson, and Gantzer commented that this suggested that heptachlor is not as mobile as other components in technical chlordane.

The investigations and reviews cited above indicate that chlordane follows pathways to the marine environment that are considered to be typical for the chlorinated organic pesticides. Demonstrated chlordane characteristics of low solubility in water, high volatility in the atmosphere, and high affinity for smaller fraction particles suggest that atmospheric transport of vapors and dust and hydraulic transport of sediment are important mechanisms by which chlordane moves from original application sites to estuarine and ocean environments.

There is evidence that chlordane, having entered nearshore or surface waters, can subsequently move down into the abyssal ocean. Knap, Binkley, and Deuser (1986) reported measurable fluxes of chlordane (sum of *cis*- and *trans*- isomers) to the deep north Atlantic Ocean. They used a sediment trap moored at a depth of 3,200 and 1,000 meters above the bottom of the Sargasso Sea and determined an average chlordane flux of 0.021 $\mu g/m^2$/year between 1978 and 1980. This was less than the flux determined for the related cyclodiene, dieldrin (0.038 $\mu g/m^2$/year), and substantially less than that for PCBs (1.590 $\mu g/m^2$/year).

TOXICITY AND TOXICOLOGY

General Toxicity

The toxic nature of chlordane has been known for over 40 years, yet the mechanisms involved remain unclear. A general discussion of metabolism of chlordane compounds is included in a review by Menzie (1978), but details of biochemical processes were not. Murphy (1986) described chlordane and the other cyclodiene pesticides as being neuropoisons with many signs and symptoms of poisoning resembling those for the more extensively studied DDT. Acute toxicity is believed to result from central nervous system stimulation. Biochemical studies have demonstrated alteration in ratios of amino acids and an increase in ammonia levels in brain tissues of animals poisoned with cyclodienes. Stickel *et al.* (1979) found that residues of several technical chlordane constituents were much higher in the brains of birds (cowbird, *Molothrus ater*; red-winged blackbird, *Agelaius phoeniceus*; grackle, *Quiscalus quiscula*; and starling, *Sturnus vulgaris*) that died following acute exposure to chlordane, compared to those that survived.

Recent research (Eldefrawi and Eldefrawi, 1987) has suggested that cyclodiene neurotoxicity is due at least in part to interference with ion transport. More specifically, this interference is thought to occur at the chloride channel proteins responsible for regulation of chloride across cell membranes. A major kind of chloride channel is the transmitter-operated chloride channel, which is a receptor activated by the appropriate transmitter to open its channel across the membrane. The primary inhibitory neurotransmitter receptor in the brain is the γ-aminobutyric acid (GABA) receptor. Both GABA receptors and chloride channel proteins are vital to proper functioning in the nervous system. Inhibition of GABA receptors produces convulsions and/or seizures in mammals, similar to the reaction induced by acute exposure to cyclodiene pesticides. Moreover, the pesticides have been shown to inhibit receptor function as reflected by GABA-induced flux of radioactively labelled chloride ions in mammalian brain membranes.

In studies with bobwhite quail (*Colinus virginianus*), Ludke (1976) found evidence for additivity of toxic effects when chlordane and endrin (another cyclodiene pesticide) were administered together. Birds treated with chlordane followed by endrin contained considerably more chlordane residues in their brains than did birds treated with chlordane alone. Based on his results, Ludke suggested that the vulnerability of an animal to a toxic pollutant may be increased if the individual already carries a body burden or is exposed to one or more closely related chemicals.

Chlordane has been implicated as a carcinogen in at least one mammalian species, the mouse. U.S. EPA (1988b) reported that the major target organ for carcinogenic effects of chlordane in mice is the liver, and that chlordane at dietary levels of 25 and 50 ppm over 18 months resulted in very high incidences of hepatic carcinoma in male and female mice. The NCI evaluated chlordane for carcinogenicity in 1977 and found that analytical-grade chlordane administered to mice at concentrations of 29.9 and 56.2 ppm (males) and 30.1 and 63.8 ppm (females) over 80 weeks resulted in significant occurrences of hepatocellular carcinoma. In the same study however, rats exposed to 203.5 and 407.0 ppm (males), and 120.8 and 241.5 ppm (females) did not show significant incidences of hepatocellular carcinoma (NCI, 1977).

Unlike many carcinogens, chlordane fails to exhibit binding to DNA and is apparently inactive in tests for genotoxicity. However, evidence exists that chlordane carcinogenicity occurs through promotion of growth in preexisting abnormal cells, possibly through inhibition of intercellular communication (Williams and Weisburger, 1986).

Chlordane is classified under U.S. EPA guidelines in Group B2: Probable Human Carcinogen. This category includes agents for which there exists inadequate direct evidence from human studies, but sufficient evidence from animal studies. The IARC (1979) placed chlordane under its Group 3 category, or not classifiable as to human carcinogenicity potential. The IARC found that evidence for carcinogenicity in animals was limited, and direct evidence for humans was inadequate.

One of the major uses of the results from toxicity tests is to provide a database for risk assessment for humans associated with a given exposure situation. Estimating health risk to human consumers of contaminated seafood is a process that has been recognized as having large uncertainties associated with it. Connor (1983) attributed these uncertainties to extrapolating from animal feeding studies at high doses to human health outcomes at low dose exposures, variation in exposure concentrations due to species differences, and differences in consumption patterns. However, such risk estimates were called the best assessments available. In 1987, the U.S. EPA estimated health risks from consumption of contaminated

fish based on data for seven contaminants contained in the large EPA database system called STORET. Among these contaminants for which a large number of data existed, was chlordane. Under a procedure detailed by Delos (1987), cancer risk estimates associated with consumption of fish were calculated from the STORET data for whole body and fillet concentrations of the contaminants. Table 8 shows calculated upper-bound estimates of mean cancer risks from consumption of fish tissue, assuming a daily per capita portion of 6.5 g.

Table 8. Upper-bound estimates of mean cancer risks from consumption of fish tissue, based on nationwide samples. Source: Delos (1987).

Compound	Number of Samples	Cancer Risk	
		Lower Estimate[1]	Upper Estimate[2]
Chlordane	5358	5.0×10^{-6}	2.0×10^{-5}
p,p'-DDE	4092	1.2×10^{-6}	9.9×10^{-6}
Dieldrin	5407	4.3×10^{-5}	2.0×10^{-4}
Heptachlor epoxide	3000	1.5×10^{-6}	6.0×10^{-6}
PCB 1254	1613	1.8×10^{-4}	4.2×10^{-4}
PCB 1260	2645	1.7×10^{-5}	8.9×10^{-5}
TCDD (dioxin)	90	6.7×10^{-6}	6.7×10^{-6}

1 Mean equated to median of detected concentrations multiplied by percentage detected.

2 Distribution constructed for all values, detected and undetected. All observations ranked and plotted as log-probability graph, treating undetecteds as low undefined values. Using central quartiles of detected values, the log mean (median) and log standard deviation (slope) of the combined distribution of detected and undetected values were estimated by linear regression. The arithmetic mean was then calculated from its theoretical relationship with log mean and variance. See reference for details of procedure.

The results portrayed in Table 8 show that for the compounds evaluated, PCBs contribute most of the cancer risk. Risk attributable to chlordane falls about midway between the high and low values for risk. Delos noted that while these risk levels for pesticides are substantially greater than risk levels calculated for consumption of peanut butter with 2 ng/g aflatoxin (upper-bound risk 1×10^{-7}) or a single serving of charcoal-broiled steak containing 6-50 ng/g benzo(a)pyrene (risk 4×10^{-9}-4×10^{-8}), it is also considerably less than the risk incurred in consuming a single beer.

A pesticide residue review and health advisory sponsored by the U.S. EPA (1988b) indicated that chlordane mutagenicity studies have been inconclusive, with negative results reported for mammalian and some bacterial assays, but positive results in other bacterial tests and in a study using maize plants. Houk and DeMarini (1987) evaluated a short-term assay based on induction of prophage lambda in *Escherichia coli* bacteria and found that it not only was a more sensitive assay for detection of carcinogenic chlorinated organic compounds, but also appeared to detect genotoxic activity more effectively than other tests. However, they also determined that of six pesticides testing positive for mutagenicity (lindane, nitrofen, chlordane, toxaphene, captan, and dichlorvos), chlordane was the least potent in terms of prophage induction.

In contrast, Vigfusson *et al.* (1983) exposed central mudminnows (*Umbra limi*) to four insecticides, including chlordane (identified as 43.2% 1,2,4,5,6,7,8,8-octachlor-2,3,3*a*,4,7,7*a*-hexahydro-4,7-methanoindane), and found that chlordane was the most potent inducer of sister-chromatid exchange among the tested compounds. The sister-chromatid exchange test was developed as a short-term test for potential genotoxic activity, and such an analysis in the mudminnow has been used as a test for the effects of waterborne chemicals. Four serial tenfold dilutions from the initial 5.4×10^{-10} M were required before a concentration was reached that showed no significant induction of sister-chromatid exchange over controls.

U.S. EPA (1988a) listed teratogenicity of chlordane as a subject area for which data were lacking. Ingle (1952) found that offspring of rats fed 150 to 300 ppm chlordane during and after gestation were normal,

although if kept with their lactating mothers, became excitable and developed tremors. Cranmer, Avery, and Barnett (1979) fed mice 8.0 ppm/day through pregnancy and determined that offspring were born with a defect in cell-mediated immune response.

In summary, chlordane appears to exert its acutely toxic effects through stimulation of the central nervous system. Some evidence exists that its toxicity may be additive with that for other cyclodiene pesticides. Chlordane has been shown to be carcinogenic in mice and is classified as a probable human carcinogen. Mutagenicity and teratogenicity studies have been inconclusive.

Toxicity to Aquatic Organisms

Several considerations make the interpretation of chlordane toxicity data difficult, especially in aquatic environments. As previously discussed, what is commonly referred to as "chlordane" is a complex technical mixture of isomers and related compounds. Relative toxicities of technical chlordane components may vary widely. While the basic formulation of chlordane has remained consistent since the early 1950s, toxicity studies occurring before or immediately after this time may have been conducted with older, inconsistent mixtures of the product and may not be comparable with more contemporary assessments.

Lack of reliable information on chlordane residues and effects may be further attributed to the previous lack of analytical standards and the lack of mass spectrometry for confirmations (Stickel *et al.*, 1979). Yet other complicating factors in environmental determinations of chlordane arise from the fact that constituent compounds are readily metabolized or oxidized by biochemical or abiotic environmental processes (Sovocool *et al.*, 1977).

The review on chlordane produced by the International Programme on Chemical Safety (U.N. World Health Organization [WHO], 1984) noted that aquatic toxicity data require interpretation, primarily because of the relative insolubility of the compound. Nominal concentrations administered in toxicity studies were likely to have exceeded actual exposure concentrations by a wide margin. Attempts to put chlordane into solution for toxicity tests may in fact introduce another uncertainty into the results, since it has been a common practice to use an organic solvent such as acetone to create test solutions. It is possible that the use of such solvents may either synergistically increase the nominal toxicity of chlordane compounds, or the solvents alone may be toxic to test organisms (NRCC, 1974). Studies conducted without controls to evaluate these effects should be considered with some skepticism.

Additive toxic effects of chlordane when administered with other chemical compounds have been discussed previously in this report (see page 20), and represent yet another possible complicating factor in toxicity assessments.

Finally, differences in exposure tolerance may complicate or preclude comparisons of interspecies toxicological results. Mishra and Srivastava (1984) cited influences of temperature, pH, oxygen content, and hardness of test water as extraneous factors that may influence toxicities observed among different species.

The studies summarized below include investigations in freshwater, estuarine, and marine organisms.

Effects of organic pollutants on methanogenesis, sulfate reduction, and carbon dioxide evolution in salt marsh sediments. Kiene and Capone (1984) exposed anaerobic salt marsh sediments collected on the north shore of Long Island to a variety of organic contaminants and then evaluated effects on methanogenesis, sulfate reduction, and carbon dioxide evolution. Among the compounds tested were chlordane and heptachlor (isomeric composition not specified).

It was found that methanogenesis in the salt marsh sediments was significantly inhibited ($p < 0.05$, single classification ANOVA) only at the relatively high administered concentration of 1,000 µg/g for both chlordane and heptachlor (88 and 52 percent reduction over controls, respectively). Concentrations of 10 and 100 µg/g for chlordane and 100 µg/g for heptachlor did not produce significant reductions in methane production over controls.

For hydrogen sulfide evolution, a significant inhibition in the sediments occurred only in one of two exposures at the highest chlordane concentration of 1000 µg/g. Sulfide production was not significantly affected at chlordane concentrations of 10 and 100 µg/g and one of two exposures at 1000 µg/g, as well as at heptachlor concentrations of 100 µg/g.

Effects on carbon dioxide evolution in the salt marsh sediments were minimal among the 14 compounds tested, with the notable exception of the 100µg/g chlordane concentration. Interestingly enough, this exposure level (but not those at 10 or 1,000 µg/g) stimulated carbon dioxide production to a significant degree over controls. Although heptachlor also appeared to stimulate CO_2 production, the increases were not significant.

The chlordane and heptachlor levels at which marsh processes were significantly affected in this study were high relative to those generally found in the natural environment. Kiele and Capone noted that the importance of contaminant effects on salt marsh processes stems from subtle changes that may be induced in predominant pathways of terminal carbon oxidation by pollutants. Although levels at which significant changes occurred were high, it was emphasized that these were acute exposures to individual compounds. Kiele and Capone felt that additional work was necessary to determine impacts of chronic exposures and synergistic influences of contaminant mixtures.

Effects of chlordane and heptachlor on the marine dinoflagellate, *Exuviella baltica.* Magnani *et al.* (1978) evaluated the effect of chlordane (identified as technical chlordane, 60 percent octachloro-4,7-methanotetrahydroindane and 40 percent related insecticidal compounds), and heptachlor (identified as technical heptachlor, 74% 1,4,5,6,7,8,8*a*-heptachloro-3a,4,7*a*-tetrahydro-4,7-methanoindene and 26 percent related insecticidal compounds) on a marine dinoflagellate (*Exuviella baltica*). They examined impacts of 50 µg/l (ppb) treatments of chlordane (dissolved in 50 µl of methanol) on growth, productivity, chlorophyll *a* content, and cell size distribution. They found that cell growth was significantly impaired in the chlordane treatment compared to control solutions and a heptachlor treatment. Figure 6 illustrates results for cell growth.

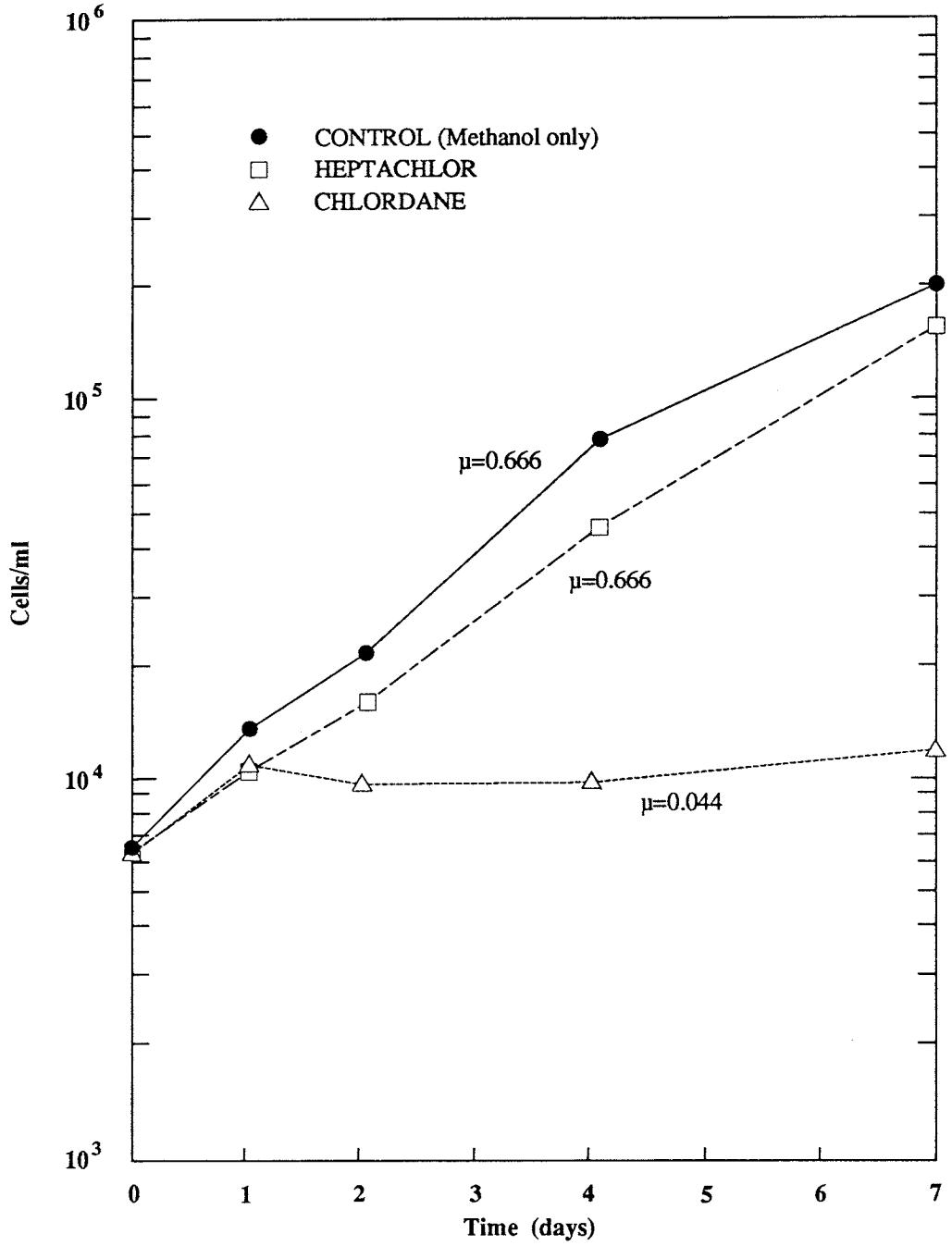

Figure 6. Cell growth of the dinoflagellate *E. baltica* exposed to 50 μg/l of chlordane or heptachlor. Growth rate, μ, expressed as divisions per day between days 1 and 7. Source: Magnani *et al.* (1978).

Magnani *et al.* also found that although chlorophyll *a* concentrations in test solutions were not reduced over controls, photosynthesis was inhibited, particularly with exposure to chlordane. Chlordane was more acutely toxic than heptachlor, causing the disintegration of many dinoflagellate cells.

Magnani *et al.* inferred possible impacts of chlordane exposure in the natural environment from their laboratory results (*i.e.*, effects on the availability of food for particle-feeding herbivores). However, the exposure concentrations used for the study, 50 µg/l, were substantially greater than any reported in natural waters, and were roughly equivalent to maximum solubility (0.056 ppm) of a 75:25 mixture of *cis-:trans-*chlordane in water at 25° C. (Sanborn *et al.*, 1976).

Comparison of the effects of chlordane and PCB on the growth, photosynthesis, and cell size of estuarine phytoplankton. Biggs *et al.* (1978) exposed cultures of mixed species of estuarine phytoplankton to concentrations of chlordane (identified as 60% octachloro-4,7-methanotetra-hydroindane and 40% related insecticidal compounds) of 10 ppm administered once, 5 ppm repeated daily, or 10 ppm repeated daily over 8- to 10-day test periods. Interestingly, although some short-term reductions in community cell growth rates and ^{14}C-uptake were noted following initial chlordane exposures (compared to methanol exposure controls), at the end of the test periods, particle concentration, total chlorophyll *a*, and ^{14}C uptake were equivalent to or greater than control values.

Although Biggs *et al.* noted that exposure to chlordane may initiate changes in cell division and photosynthesis in natural estuarine phytoplankton communities, the longer term effects are not clear. In interpreting their results, it is also important to consider that indicated concentrations represented chlordane added to the system and not measured exposure concentrations (*i.e.*, partitioning of chlordane among plankton, suspended matter, bottle walls, and water was not evaluated). Additionally, and as was the case in the experimental design of Magnani *et al.* above, exposure concentrations employed were high relative to those that have been reported in the aquatic environment.

U.S. Navy tests of alternative wood preservatives in Pearl Harbor, Hawaii. Between 1963 and 1966, the U.S. Navy treated and installed a number of experimental wood pilings along the Waipio Peninsula in Pearl Harbor, Hawaii. The purpose of the study was to evaluate several different treatments as to their long-term effectiveness in preventing the proliferation of and damage from marine boring organisms. Organisms responsible for damage that were recorded in the study were the isopod crustaceans, *Limnoria tripunctata*, and *Sphaeroma terebransi*, and the molluscs, the striate piddock, *Martesia striata*, and shipworms, *Teredo spp.*, and *Bankia spp.* Among the piling treatments tested were applications of chlordane, in 1.25, 2.5, and 5 percent additives to creosote; and in 5 percent solutions with xylene, with 1 and 2 percent tributyltin oxide. O'Neill (1983) described the program, provided details about treatments, and summarized results from piling inspections at specified time intervals.

Results from this test program are shown in Table 9 and stand as testimony to the long-term effectiveness of chlordane as a biocide. Percent loss in piling cross-sectional area for chlordane-treated members was estimated both visually and ultrasonically in 1982 (17 and 18 years after treatment). Damage due to boring was remarkably low relative to other treatments and considering the length of time the pilings had been exposed. Other formulations of tributyltin or copper compounds without chlordane resulted in considerable degrees of damage: five of five pilings treated with tributyltin oxide alone showed >50 percent damage; similarly, four of six treated with 14 percent copper naphthenate and 1 percent tributyltin also showed >50 percent damage. In contrast, none of the chlordane-treated pilings fell into that category, and 20 of 24 pilings showed <5 percent damage.

Table 9. 1982 results of 1964 and 1965 chlordane piling treatments at Pearl Harbor, HI. Vis = visual inspection, Ult = ultrasonic inspection. TBT = tributyltin oxide. Source: O'Neill (1983).

	Percent Loss of Piling Cross-Sectional Area											
	Piling 1		Piling 2		Piling 3		Piling 4		Piling 5		Piling 6	
Treatment	Vis	Ult	Vis	Ult	Vis	Ult	Vis	Ult	Vis	Ult	Vis	Ult
1.25% chlord in creosote[1]	2	10-25	2	10	4	–	15	20-25	3	–	10	0-25
2.5% chlord in creosote[1]	3	0	2	–	4	0-25	4	–	2	–	2	10-25
5.0% chlord in creosote[1]	2	0-25	3	10	2	–	1	0-25	3	0	0	0
5.0% chlord + 1% TBT[2]	3	–	2	–	4	–	–	6	5	10	2	–
5.0% chlord + 2% TBT[2]	2	–	3	–	2	–	2	–	4	–	2	–

[1]Treated in 1964.
[2]Treated in 1965.

While noting the excellent protective properties of chlordane, O'Neill pointed out that the EPA had banned its use in the marine environment. Given that restriction, he suggested research on alternative forms of compounds like chlordane, with the targeted result of an environmentally acceptable moiety or substitute based on toxicological mechanism.

Toxicities of eight organochlorine compounds in sediment and seawater to *Crangon septemspinosa.* McLeese and Metcalfe (1980) investigated the toxicities of eight organochlorine compounds, including technical chlordane, on the shrimp *Crangon septemspinosa.* Both water and sediments were evaluated, as well as partitioning of organics between sediments and water. Results for chlordane obtained by McLeese and Metcalfe are summarized in Tables 10 and 11 and Figure 7.

Table 10. *Crangon septemspinosa* 96-hour LC_{50}s and lethal thresholds for technical chlordane and other chlorinated organic compounds in seawater (20°C) and sediment (10°C). Source: McLeese and Metcalfe (1980).

	Seawater test		Sediment test	
Compound	96-h LC_{50} µg/l	Threshold µg/l	97-h LC_{50} µg/l	Threshold µg/l
Endosulfan	0.2	0.2	6.9	9.0
Endrin	0.6	0.5	47	41
DDT	0.4	0.2	31	20
Dieldrin	0.4	0.5	4.1	2.6
Technical chlordane	2.0	1.0	120	110
Aroclor 1242	13.0	6.5	>780[1]	-
Aroclor 1254	12.0	0.5	>3400[1]	-
HCB	>7.2[1]	-	>300[1]	-

[1]No mortalities at highest concentration tested.

Table 11. *Crangon septemspinosa* time-related lethality for technical chlordane and other chlorinated organic compounds in seawater. Source: McLeese and Metcalfe (1980).

	10°C		20°C	
Compound	Calc. avg. conc. µg/l	LT_{50} hours	Calc. avg. conc. µg/l	LT_{50} hours
Technical chlordane	12	30	12	35
	2.4	none dead @ 96 hrs	2.8	50
Endrin	2.3	23	2.4	20
	0.7	52	0.9	45
DDT	1.9	15	1.8	20
	1.1	15	0.9	16, 21
Aroclor 1254	20	75, 50	21	65
	14	30	15	40

Results from Tables 10 and 11 are best viewed within the context of the other compounds also examined by McLeese and Metcalfe. For example, Table 10 shows that chlordane toxicity was intermediate between four other pesticides exhibiting a higher toxicity to the shrimp in both water and sediment, while PCBs and HCB were relatively less toxic. Sediment concentrations in all compounds tested were less toxic than seawater concentrations by at least an order of magnitude. Table 11 suggests that toxicity differences at 10°C and 20°C were not significantly different, and shows that of the four compounds for which results were reported, technical chlordane was the least toxic in 96-hour exposures. Although exposure concentrations were relatively high for chlordane, resultant toxicity was low. In fact, at the lower exposure concentration of 2.4 µg/l at 10°C, no shrimp were dead after 96 hours.

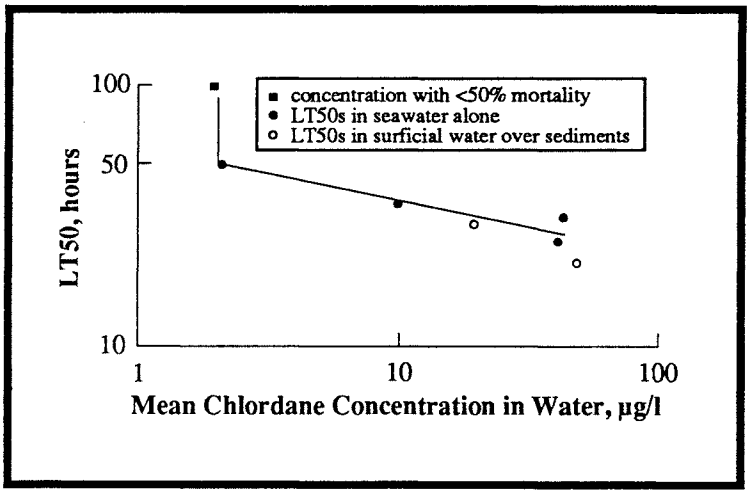

Figure 7. Lethality lines plotted for *Crangon septemspinosa* exposed to chlordane in seawater and surficial water over dosed sediments. Source: McLeese and Metcalfe (1980).

In some of the sediment tests, the concentration of technical chlordane in the overlying water was measured with time. Results portrayed in Figure 7 suggest little difference between LT_{50}s for *Crangon septemspinosa* at the average concentrations measured in water overlying sediment and similar concentration in water without sediment. McLeese and Metcalfe asserted that this was an indication that chlordane adsorbed to sediment contributed little toward observed toxicity.

Based on the results obtained in this study, McLeese and Metcalfe concluded that it was reasonable to consider chlordane in sediments as less toxic, relative to other pesticide compounds. However, they cautioned that physical parameters, such as sediment grain size, and flushing rates of water bodies, would affect toxicity of all compounds in the natural environment.

Toxicities of five organochlorine compounds in water and sediment to *Nereis virens.* McLeese, Burridge, and Van Dinter (1982) followed up the investigation with *Crangon septemspinosa* by McLeese and Metcalfe (1980) by examining the toxic effects of five organochlorines on the benthic polychaete worm, *Nereis virens.* For the latter study, the toxicities of endosulfan, endrin, DDT, dieldrin, and chlordane were evaluated. Static seawater and sediment tests were established in which five worms were exposed at varying concentrations.

Because no deaths were observed in 96-hour exposures to two of the compounds (endosulfan and chlordane), exposure periods were extended to 288 hours for both seawater and sediment tests. Results of these are shown in Table 12.

Table 12. *Nereis virens* 288-hour LC_{50}s for technical chlordane and other chlorinated organic compounds in seawater, sediment, and water overlying sediment. Source: McLeese, Burridge, and Van Dinter (1982).

	Seawater test	Sediment tests	
		Sediment	Overlying Water
	288 h LC_{50}	288 h LC_{50}	288 h LC_{50}
Compound	mg/l	mg/kg	mg/l
Endosulfan	0.10	0.34	0.10
Technical chlordane	0.22	≤5.8	0.19
Endrin	nm (0.11)[1]	2/5d (28)[2]	2/5d (0.11)
Dieldrin	nm (0.17)	nm (13)	nm (0.02)
DDT	nm (0.03)	nm (16.5)	nm (0.01)

[1]No mortalities at concentration tested (in parentheses).
[2]2/5d = two of five dead at specified concentration in parentheses at 288 hours.

Noting first the differences in units reported, if results from Table 12 are compared to results for shrimp obtained by McLeese and Metcalfe (1980) (Table 11), it is apparent that 288-hour LC_{50}s in water and sediment for *Nereis* were much greater than 96-hour LC_{50}s for *Crangon* shrimp. As observed by McLeese, Burridge, and Van Dinter, it would be expected that organisms of equal sensitivity would yield an LC_{50} at 288 hours less than or equal to that for 96 hours. Therefore, based on these results, it appeared that *Nereis virens* was much less sensitive than was *Crangon septemspinosa.*

Accumulation of biocide residues in tissues of *Lamellidens marginalis.* Agrawal (1986) cited the paucity of research on accumulation of pesticide residues in molluscs and investigated the uptake of two pesticides (chlordane and sevin) into gill, muscle, foot, and intestinal tissue of the freshwater bivalve, *Lamellidens marginalis.* Specimens were exposed to a sublethal concentration of chlordane (composition not specified) of 0.12 mg/l and accumulations were measured after 2, 4, 8, 15, and 30 days. Results for chlordane accumulations reported by Agrawal (1986) are summarized in Table 13.

Table 13. Accumulation of chlordane measured in tissues of *Lamellidens marginalis* exposed to 0.12 mg/l chlordane. Values in ppm wet weight. Mean ± standard deviation. Source: Agrawal (1986).

	Exposure period				
Tissue	2 days	4 days	8 days	15 days	30 days
Gill	4.97±0.0242	3.58±0.0314	2.40±0.0312	1.65±0.0312	1.01±0.0521
Foot	3.56±0.0321	2.18±0.0231	1.65±0.0401	0.90±0.0403	1.00±0.0331
Muscle	3.12±0.0263	2.01±0.0215	1.10±0.0257	0.21±0.0354	0.12±0.212
Intestine	2.21±0.0315	1.15±0.0295	1.25±0.0421	0.91±0.0322	0.42±0.0343

Agrawal speculated that the higher concentrations of chlordane measured in gill tissue probably resulted from the fact that route of exposure was via the gills. It was further asserted that the relatively steady declines in tissue burdens observed over the course of the 30-day exposure likely was due to metabolic degradation.

Effects of chlordane on the blood and tissue chemistry of a teleost fish, *Heteropneustes fossilis.* Noting that chlordane was used widely in India, especially for control of soil pests, and that relatively few investigations had focused on biological effects of the compound, Mishra and Srivastava (1984) performed a static bioassay to determine 24-, 48-, 72-, and 96-hour LC_0, LC_{50}, and LC_{100} values and then examined chlordane (identified as 1,2,4,5,6,7,8,8-octachloro-2,3,3*a*,4,7,7*a*-hexahydro-4,7,methano-indene) effects on the blood and tissue chemistry of a freshwater catfish species (*Heteropneustes fossilis*). More specifically, effects of exposure on carbohydrate metabolism and blood chloride concentration were studied. Behavioral changes were also described.

Mishra and Srivastava found that exposure to chlordane resulted in difficulty in breathing in the catfish with rapid pectoral and opercular action, erratic swimming, and gradual loss of equilibrium. Results of the static exposure to chlordane solutions are shown in Table 14.

Table 14. LC_0, LC_{50}, and LC_{100} values for freshwater catfish (*Heteropneustes fossilis*) exposed to static chlordane solution. 95 percent confidence limits in parentheses. Values in µg/l (ppb). Source: Mishra and Srivastava (1984).

Time	LC_0	LC_{50}	LC_{100}
24	2.28	3.50 (3.12-3.93)	5.25
48	0.25	1.25 (1.13-1.39)	5.00
72	0.20	0.29 (0.27-0.31)	4.50
96	0.15	0.275 (0.206-0.365)	3.50

In the tissue and blood chemistry experiment, the fish were exposed to 0.247 mg/l chlordane for 2, 6, 12, 48, and 96 hours. Control fish included a population kept in water and one exposed to the acetone solvent. Levels of glycogen were measured in muscle and liver tissue. Glucose and chloride were measured in blood in six individuals at each specified time interval. Table 15 illustrates summarized results from this portion of the investigation.

Table 15. Carbohydrate metabolite and blood chloride values in catfish exposed to 0.247 mg/l chlordane. Source: Mishra and Srivastava (1984).

	Control	Exposure period				
Parameter	(water)	2	6	12	48	96
Muscle glycogen (mg/100 mg wet wt.)	0.30±0.01	0.07±0.01	0.08±0.03	0.19±0.015	0.21±0.07	0.31±0.03
Liver glycogen (mg/100 mg wet wt.)	12.35±0.66	9.39±0.37	13.03±0.30	13.93±0.06	13.42±0.51	12.24±0.09
Blood glucose (mg/100 ml)	38.89±1.69	49.63±0.94	41.33±0.54	48.89±3.35	32.96±1.56	32.44±2.95
Blood chloride (meq/l)	94.85±1.84	77.11±0.86	90.15±1.41	81.82±2.35	159.09±2.02	169.69±2.95

Mishra and Srivastava reported that chlordane disturbed carbohydrate metabolism in the catfish. For example, Table 15 above shows that mobilization of muscle and liver glycogen between 2 and 12 hours occurred and the authors suggested that this was a result of increased secretion of catecholamines due to stress. The apparent recovery of liver glycogen levels at 12 hours indicated that either glycogenolysis was inhibited or gluconeogenesis was promoted. Mishra and Srivastava cited other studies that had shown that stressed fish secrete increased amounts of cortisol, which could induce synthesis of glycogen from sources other than carbohydrate precursors.

The fluctuation in blood chloride levels after chlordane exposure was attributed to altered osmoregulatory function in the fish, possibly due to differential changes in the water-electrolyte permeability and renal function.

Table 16 shows hematological variables measured in control and chlordane-exposed (0.247 mg/l, 96 hours) catfish. Erythrocyte, leucocyte, and thrombocyte counts were observed to decrease, while erythrocyte sedimentation rate and whole blood clotting time increased. These results were consistent with other fish studies referenced by Mishra and Srivastava.

Table 16. Hematological variables measured in blood of freshwater catfish. Source: Mishra and Srivastava (1984).

Parameter	Control (water)	Experimental (chlordane, 0.247 mg/l)
Erythrocytes (x 10^6/mm^3)	6.60±0.22	4.31±0.12
Leucocytes (x 10^3/mm^3)	41.17±0.60	36.33±1.38
Total differential cell count (per 10^3cells of all types)		
Small lymphocytes	70.00±2.52	64.17±2.13
Large lymphocytes	0.50±0.22	0.17±0.17
Monocytes	0.33±0.21	0.83±0.40
Basophils	0.17±0.17	0.33±0.21
Thrombocytes (x 10^3/mm^3)	22.50±0.92	19.67±0.71
Hematocrit (%)	48.28±0.69	40.63±0.31
Erythrocyte sedimentation rate (mm/hr)	1.33±0.11	2.00±0.10
Hemoglobin (g%)	12.51±0.12	11.33±0.87
Clotting time (sec)	113.33±4.77	127.50±4.16

In summary, the authors noted the biochemical and blood characteristic changes evoked by chlordane exposure, but were not able to determine whether the alterations reflected a general stress and compensatory reaction in the fish, or whether they represented a toxicological mechanism specific to chlordane.

Changes in selected biochemical parameters in the kidney and blood of tilapia exposed to heptachlor. Radhaiah, Girija and Rao (1987) studied biochemical changes induced in tissues and blood of the freshwater fish, *Tilapia mossambica*, exposed to heptachlor. Static bioassays were performed and a 48-hour LC_{50} value (0.15 mg/l) was determined for the fish. One fifth of the derived LC_{50} concentration was then employed as a sublethal exposure concentration for 5-, 10-, and 15-day periods. Tables 17 through 19 show levels of tissue parameters, enzymes, and blood components in fish exposed at this 0.03 mg/l heptachlor level.

Table 17. Levels of carbohydrates, proteins, amino acids, and lipids in the kidneys of fish exposed to 0.03 mg/l heptachlor. Each value mean of six observations ± standard deviation. Source: Radhaiah, Girija and Rao (1987).

Component	Control 15 days	Exposed Fish 5 days	Exposed Fish 10 days	Exposed Fish 15 days
Carbohydrates (mg/g wet wt.)	3.23±0.16	2.04±0.08	1.73±0.11	1.72±0.06
Proteins (mg/g wet wt.)	79.2±2.19	64.5±1.77	58.5±6.22	42.0±2.32
Amino acids (mg/g wet wt.)	1.50±0.04	1.83±0.08	1.95±0.09	2.97±0.25
Lipids (mg/g dry wt.)	162±26.5	178±29.5	182±14.2	205±39.8

The authors attributed the decreased carbohydrate content in heptachlor exposed fish to rapid utilization of carbohydrates by the tissue, possibly to overcome heptachlor-induced stress. The decreased protein levels and increased amino acid content were thought to possibly reflect protein breakdown and/or damage to liver tissue in response to pesticide exposure, and increase in lipids suggested lipogenesis under pesticidal intoxication.

Table 18. Activity levels of three enzymes in the kidneys of fish exposed to 0.03 mg/l heptachlor. Each value mean of six observations ± standard deviation. Source: Radhaiah, Girija, and Rao (1987).

Component	Control 15 days	Exposed Fish 5 days	Exposed Fish 10 days	Exposed Fish 15 days
Succinate dehydrogenase (µmol formazan formed/ mg/protein/h)	0.78±0.05	0.86±0.03	0.65±0.08	0.59±0.04
Lactate dehydrogenase (µmol formazan formed/ mg/protein/h)	1.85±0.14	1.58±0.07	1.92±0.28	2.19±0.34
Glutamate dehydrogenase (µmol formazan formed/ mg/protein/h)	0.23±0.03	0.29±0.03	0.31±0.02	0.35±0.02

Radhaiah, Girija, and Rao felt that the increased activity of lactate dehydrogenase at 10 and 15 days suggested that fish were favoring anaerobic respiration to meet energy demands. Glutamate dehydrogenase catalyzes key reactions to provide substrates for protein synthesis or carbohydrate metabolism in the kidney, and the observed increases were consistent with the above results for kidney tissue components.

Table 19. Levels of selected blood components in fish exposed to 0.03 mg/l heptachlor. Each value mean of six observations ± standard deviation. Source: Radhaiah, Girija and Rao (1987).

Component	Control 15 days	Exposed Fish 5 days	Exposed Fish 10 days	Exposed Fish 15 days
Urea (mg/100 ml)	718±65.2	815±27.4	835±21.5	967±15.9
Non-protein nitrogen (mg/100 ml	50.3±6.1	59.2±4.3	64.8±1.7	71.9±3.9
Creatine (mg/100 ml)	9.8±0.52	10.9±0.77	11.5±0.50	19.1±3.91

The increases observed in blood parameters were thought to have been due to the interruption of normal kidney function to clear such materials.

Determination of the maximum acceptable toxicant concentration and the safe concentration of certain aquatic pollutants. Verma, Tonk, and Dalela (1981) used larvae of common carp (*Cyprinus carpio*) to determine values for 96-hour LC_{50} and estimate safe concentrations for 15 compounds considered to be pollutants. Chlordane (specified only as commercial grade) was among the materials evaluated.

The 96-hour LC_{50} for chlordane in carp larvae was found to be 16 µg/l. This compared to a low value of 0.3 µg/l for thiotox to a high of 2,000 for sevin. Based on the LC_{50} information, and other data on effects of 60-day exposures on the larvae, Verma, Tonk, and Dalela calculated nominal safe concentrations. For chlordane, this level was found to be 0.154 µg/l. Range of values obtained for other compounds varied from a low of 0.00288 µg/l for thiotox to a high of 149.8 µg/l for a synthetic detergent.

Despite these findings, the authors cautioned that the values were of limited utility because doses and time of exposure considered safe for a certain age group and under certain environmental conditions could be potentially hazardous under other circumstances.

Biomonitoring of sublethal water contamination through a systems analysis approach. Verma and Tonk (1984) detailed inadequacies of environmental monitoring using exclusively physical and chemical approaches, and asserted the value of sublethal biological monitoring as a means of detecting detrimental impacts. As did Mishra and Srivastava (1984), Verma and Tonk used the freshwater Indian catfish, *Heteropneustes fossilis*, as the aquatic test organism for a series of tests related to effects of exposure to contaminants. Six different pesticides were evaluated, including chlordane (not defined, but listed as 20% emulsifiable concentrate).

Catfish were exposed for 30 days at 1/15th the 96-hour LC_{50}. Every 5 days, respiratory rate was determined. At the end of the test period, 13 blood parameters and adenosine triphosphatase and acetylcholinesterase activity were determined in liver, kidney, and gill tissue.

The chlordane exposure concentration, specified as 1/15th the 96-hour LC_{50}, was 0.42 mg/l. Effects of exposure on oxygen consumption rate of catfish at different time intervals are summarized in Table 20.

Table 20. Oxygen consumption rate of *Heteropneustes fossilis* exposed to 0.42 mg/l chlordane at various time intervals. Values represent mean of 3 measurements ± standard deviation. Source: Verma and Tonk (1984).

	Oxygen Consumption Rate mg/g•h O_2					
Exposure	**5 days**	**10 days**	**15 days**	**20 days**	**25 days**	**30 days**
Control	10.2±1.0	11.6±0.92	11.8±1.2	12.7±1.1	15.6±1.3	17.5±1.4
Chlordane	17.6±0.90[2]	9.3±0.75	6.5±0.70[1]	4.4±0.32[2]	1.7±0.18[3]	(dead)

[1] Significant at $p<0.05$.

[2] Significant at $p<0.01$.

[3] Significant at $p<0.001$.

As the table shows, significant changes in oxygen consumption over controls were evident in fish exposed to chlordane. Verma and Tonk attributed the initial alteration in oxygen consumption following exposure to chlordane to abnormal hyperactivity. They speculated that the subsequent depression in oxygen consumption could have been a result of changes in gill function, coagulation of gill mucus, asphyxiation, or inhibited enzymatic function.

Changes in hematological characteristics observed in fish exposed to 0.42 mg/l chlordane are summarized in Table 21. Significant ($p < 0.01$) differences between control and exposed values were observed for glucose, cholesterol, sodium, and chloride levels in blood. In all four cases, levels were elevated in blood of exposed fish.

Table 21. Hematological variables measured in blood of freshwater catfish, control and chlordane exposure (0.42 mg/l). Values represent mean of 3 ± standard deviation. Source: Verma and Tonk (1984).

Parameter	Chlordane	
	Control	0.42 mg/l
Clotting time (s)	125.0±5.0	110.0±4.0
Hemoglobin (g%)	14.4±0.7	14.9±0.8
Red blood cells (x 10^6/mm^3)	3.5±0.3	4.0±0.2
White blood cells (x 10^3/mm^3)	3.9±0.2	4.3±0.3
Erythrocyte sedimentation rate (mm/hr)	1.1±0.4	0.8±0.2
Packed cell volume (%)	38.5±1.1	40.5±1.0
Glucose (mg/dl)	88.7±1.4	102.3±1.1*
Protein (g/dl)	5.2±1.2	4.0±1.2
Cholesterol (mg/dl)	275.5±8.4	298.1±7.4*
Sodium (mg/dl)	295.0±4.5	354.0±6.5*
Potassium (mg/dl)	6.3±0.5	7.2±0.8
Magnesium (mg/dl)	1.3±0.3	2.4±0.2
Chloride (mg/dl)	278.0±4.6	312.0±4.6*

* Significant at $p<0.01$.

Table 22 illustrates summarized results for effects of chlordane exposure on selected enzyme activities. Significant inhibition occurred in nearly all tissues and enzymes examined. Although only results for control and chlordane solutions are shown in the table, Verma and Tonk presented measurements for five other pesticides (aldrin, metasystox, dichlorvos, sevin, and carbofuran). The fact that alterations in all ATPases were more pronounced and highly significant for chlordane (as well as aldrin) was taken as an indication that these enzymes were more sensitive to such organochlorine pesticides. In contrast, acetylcholinesterase activity appeared to be less affected by chlordane and aldrin than by the organophosphate compounds, metasystox and dichlorvos.

Table 22. Alteration in enzyme activity in freshwater catfish tissues following 30 days of exposure to 0.42 mg/l chlordane. Values represent mean of 3 ± standard deviation. Source: Verma and Tonk (1984).

Tissue	Enzyme	Control[1]	Chlordane[1] 0.42 mg/l	% Inhibition
Liver	Total ATPase	17.48±1.12	9.76±0.64	44.16[2]
	Mg^{2+} ATPase	10.08±0.95	5.60±0.30	44.44[2]
	$Na^{+}K^{+}$ ATPase	7.40±0.58	4.16±0.22	43.78[2]
	Acetylcholinesterase	32.47±1.80	22.41±1.85	30.98[3]
Kidney	Total ATPase	30.28±1.48	17.69±1.18	41.58[2]
	Mg^{2+} ATPase	14.20±1.05	7.15±0.45	49.64[2]
	$Na^{+}K^{+}$ ATPase	16.08±1.01	8.54±0.39	46.89[2]
	Acetylcholinesterase	12.19±0.77	10.82±1.00	11.24
Gills	Total ATPase	22.08±1.42	13.71±0.72	37.90[2]
	Mg^{2+} ATPase	13.57±0.95	8.44±0.28	37.80[2]
	$Na^{+}K^{+}$ ATPase	8.51±0.58	5.27±0.36	38.07[2]
	Acetylcholinesterase	18.24±1.32	14.09±1.06	22.75

[1] Units for ATPase, µmols inorganic phosphate liberated per mg tissue protein per hour; for acetylcholinesterase, µmols AChE hydrolyzed per mg tissue protein per hour.

[2] Significant at $p<0.01$.

[3] Significant at $p<0.05$.

The subtlety and complexity inherent in the actions of pesticides are made more apparent by studies such as those by Verma and Tonk. Chlordane affected oxygen consumption in *Heteropneustes fossilis* in different ways with time, significantly elevated certain blood parameters (but not others), and significantly inhibited certain enzyme activities (but not others). Although it appeared that generalizations about the action of chlordane can be made based on the results of Verma and Tonk (*e.g.*, 30-day exposure significantly inhibited the action of ATPases), it was also apparent that chronic exposures resulted in other physiological reactions of complex origin.

Chester River Study. The Chester River Study was a program of the Maryland Department of Natural Resources to provide environmental and resource management information relevant to the shellfish industry. Specifically, research was targeted on determination of sources and fate of such contaminants as chlorinated hydrocarbons and metals. The study was performed by the Oceanic Division of the Westinghouse Electric Corporation. As part of the Chester River Study, toxic effects of chlordane on the eastern oyster (*Crassostrea virginica*) and soft-shelled clam (*Mya arenaria*) were studied. This represents one of the more focused examinations of chlordane residues and impacts in the estuarine environment. Summary results are discussed in Clarke and Murdock (1972), while detailed discussions of analytical procedures and results are presented by Munson (1972). Chlordane was chosen from among the chlorinated hydrocarbons present in the Chester River because it had been detected in shellfish at levels which approached or exceeded reported "alert levels" of 30 parts per billion (ppb).

Four experiments of varying durations were performed during this study: 8- and 31-day experiments with oysters and 8- and 69-day experiments with soft-shelled clams. Chlordane (identification based on α- and γ-chlordane) was dissolved and administered in 2:1 acetone:water solutions, with acetone-water controls established to distinguish effects of chlordane. Reporting basis for tissue concentrations was not specified but was presumed to be wet weight because of references to wet weight regulatory guidelines.

In the 8-day experiments, oysters and clams were exposed to 1, 10, and 100 ppb concentrations of chlordane. Soft-shelled clams were found to concentrate the compound at levels varying from 400 to 1,000

times exposure concentrations. Oysters were found to accumulate chlordane at even greater rates, up to 10,000 times exposure concentrations. Although no oyster mortalities occurred in the 8-day experiment and the clam mortalities that occurred were not significantly correlated to elevated exposure, morphological abnormalities were observed at higher exposure concentrations. In the soft-shelled clams, these included elongation and swelling of siphons and blistering of siphon tissues. In oysters, decreased body weights and poorly developed gonads were observed in specimens exposed to higher levels of chlordane.

In the longer term experiments, the bivalves were exposed to much lower concentrations of 1, 10, and 100 ppt. Although the morphological abnormalities observed in the 8-day experiments were not seen, examination by scanning electron microscope revealed that the crystalline structure of new oyster shell growth was significantly affected by exposure to even the lowest concentrations of chlordane. The degree to which these changes were detrimental to the health of the organisms was not determined.

Chlordane uptake rates were studied over both short- and long-term exposures. Tables 23 and 24 show results of the short-term laboratory study of chlordane uptake in soft-shelled clams and oysters, respectively. In the long-term experiment, it was found that in oysters, the ratio of oyster tissue concentration to water concentration was not constant, and that at lower exposure concentrations, organisms accumulated greater amounts of chlordane relative to higher water concentrations. At 100 ppt exposure, the relative concentration factor in body tissues was found to be 9,000; at 10 ppt, it was 54,000; and at 1 ppt, it was 160,000. Figure 8 illustrates plots of concentration factors *vs.* chlordane concentration dosing levels for both oysters and clams.

In longer term exposures with clams, Munson found a considerable variability in results (Figure 9). This was attributed to factors such as differences in individual uptake rates, lack of precision in the analytical technique, and other extraneous events affecting laboratory conditions. Exposure at the highest test concentration of 0.1 ppb did show an uptake curve more typical of readily accumulated organic compounds.

Table 23. Short-term chlordane (α- and γ-chlordane) uptake by soft-shelled clams. Source: Munson (1972).

	Nominal Administered Chlordane Concentration			
Days of exposure	**0 ppb**	**1 ppb**	**10 ppb**	**100 ppb**
	Concentration measured in clams, ppm			
1	–	–	–	–
2	0.15	1.4	4.4	17
3	–	1.2	5.6	–
4	–	–	–	15
5	0.29	–	–	–
6	0.70	–	–	28
7	0.07	1.2	8.3	26
14 days after exposure cessation	0.05	0.45	0.73	9.0

Table 24. Short-term chlordane (α- and γ-chlordane) uptake by oysters. Source: Munson (1972).

	Nominal Administered Chlordane Concentration			
Days of exposure	**0 ppb**	**1 ppb**	**10 ppb**	**100 ppb**
	Concentration measured in oysters, ppm			
1	0.18	0.37	2.4	40
2	–	–	–	–
3	0.23	2.6	4.0	180
4	–	–	–	–
5	0.27	4.3	100	200
6	–	–	–	–
7	1.1	4.9	67	85

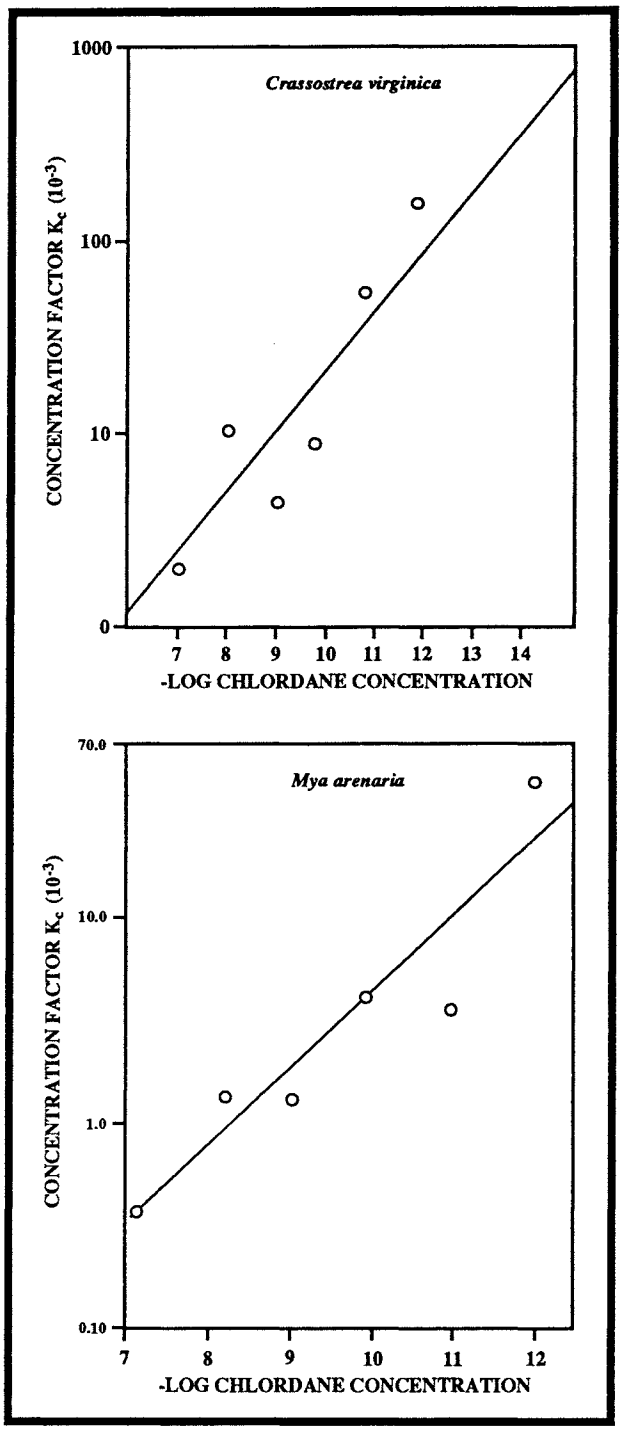

Figure 8. Log-log plot of chlordane concentration factors *vs.* chlordane concentration dosing levels for oysters and clams. Source: Munson (1972).

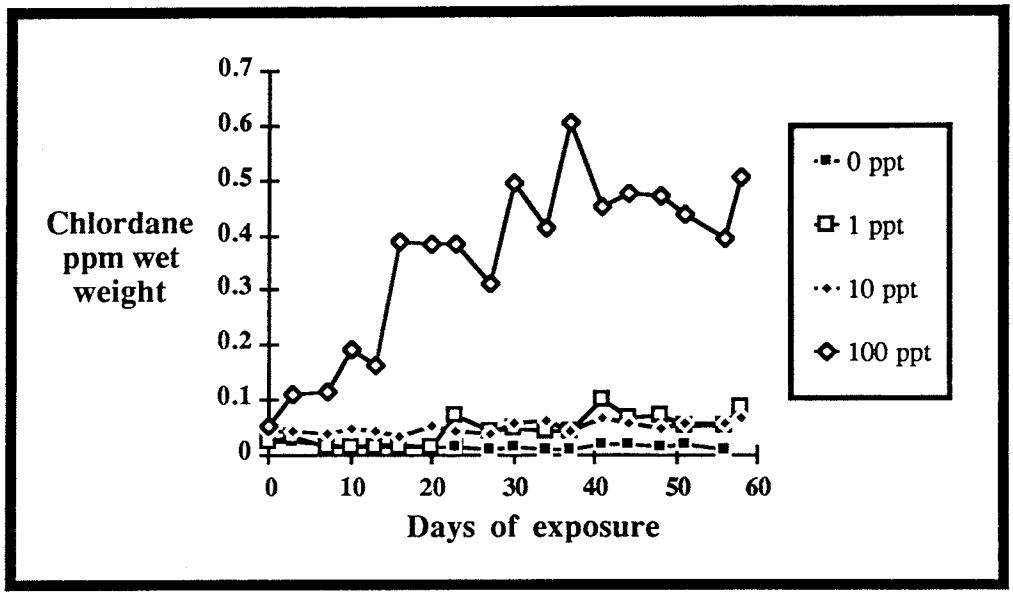

Figure 9. Long-term results for chlordane (α- and γ-chlordane) uptake by soft-shell clams. Nominal exposure concentrations in ppt; clam tissue concentrations in ppm. Source: Munson (1972).

Observations by Munson included the comment that even if body tissue accumulations of contaminants in shellfish stocks are considered safe by human (*e.g.*, FDA action level) standards, in most instances it has not been established that the levels are completely harmless to the organisms themselves. Furthermore, based upon observed chlordane uptake rates, it was noted that to maintain oyster body tissue concentrations below an advisory NSSP "alert" level of 30 ppb (a suggested reference concentration with no regulatory significance), chlordane concentrations in the environment inhabited by the oysters would have to be less than 0.2 ppt.

Other results found by Munson during the course of the Chester River Study will be discussed in subsequent sections focusing on regional findings.

Relative toxicity of ten chlorinated hydrocarbon insecticides to four species of fish. Henderson, Pickering, and Tarzwell (1959) conducted bioassays on four species of freshwater fish to evaluate the relative toxicities of ten chlorinated pesticides, including chlordane (identified as 1,2,4,5,6,7,8,8-octachloro-2,3,3*a*,4,7,7*a*-hexahydro-4,7-methanoindene). They determined 24-, 48-, and 96-hour LC_{50} concentrations in both hard and soft water and found that compared to aldrin, dieldrin, endrin, heptachlor, toxaphene, DDT, methoxychlor, lindane, and BHC, chlordane fell into the middle range of toxicity: roughly equivalent to DDT, methoxychlor, lindane, and heptachlor; less toxic than other cyclodienes such as aldrin, dieldrin, and endrin; and more toxic than BHC. Water hardness did not significantly affect toxicity of most pesticides tested, with chlordane appearing to be slightly more toxic in soft water. Chlordane results are summarized in Table 25.

Table 25. LC_{50} values for chlordane for four species of freshwater fish. Values in µg/l (ppb). Source: Henderson, Pickering, and Tarzwell (1959).

Fish species	Dilution Water[a]	24 hr. LC_{50}	48 hr. LC_{50}	96 hr. LC_{50}
Fathead minnows (*Pimephales promelas*)	Soft[b]	69	69	52
Fathead minnows (*Pimephales promelas*)	Hard[c]	98	69	69
Bluegills (*Lepomis macrochirus*)	Soft	36	32	22
Goldfish (*Carassius auratus*)	Soft	166	87	82
Guppies (*Lebistes reticulatus*)	Soft	560	190	190

[a] Reference standard chlordane, 100% active ingredients in acetone.
[b] 95:5 mix, distilled H_20:natural spring water, 25° C.
[c] Natural spring limestone water, 25° C.

Henderson, Pickering, and Tarzwell also evaluated the relative toxicities of different formulations of the same pesticides, and for chlordane found that the emulsifiable concentrate (75% chlordane, 25% inert ingredients) added directly to soft water at 25° C. was less toxic to fathead minnows than the technical product in acetone. The 96-hour LC_{50} value for the former was 127 µg/l (ppb); for the latter, 52 µg/l.

Acute toxicity of pesticides to fish, plankton, and worm. Mani and Konar (1986) evaluated two pesticides commonly used in India in terms of their acute toxicities to plankton, a worm, and fish. One of the two pesticides was Termex, a commercial formulation containing 20 percent chlordane. LC_5, LC_{50}, and LC_{95} values were calculated for three organisms exposed to Termex: the freshwater plankton species, *Diaptomus forbes;* the worm, *Branchiura sowerbyi*; and the fish, *Tilapia mossambica*. Results of the acute toxicity tests are shown below as Table 26.

Table 26. Acute toxicity values for plankton, worm, and fish exposed to Termex (20% chlordane). LC values in mg/l (ppm). Source: Mani and Konar (1986).

Test species	LC_5 ppm	LC_{50} ppm	LC_{95} ppm
Plankton (*Diaptomus forbesi*)	0.0006	0.00345	0.00665
Worm (*Branchiura sowerbyi*)	<0.0005	0.02200	0.04750
Fish (*Tilapia mossambica*)	<0.0050	0.02800	0.71500

Mani and Konar noted that plankton and worm specimens showed greater sensitivity to lower concentrations of Termex than did tilapia. Mortality was manifested in the former two organisms within 24 hours of initial exposure, and worms showed fragmentation even at the lowest exposure concentration. The authors noted that fish treated at the lowest concentration showed lethargic movement, frequent surfacing, and loss of balance.

Mani and Konar used their results as evidence that low concentrations of Termex (*i.e.*, 0.0006 ppm) in the natural environment could be hazardous, and asserted that it was essential to use such pesticides wisely to protect aquatic ecosystems.

Effects of chlordane on several estuarine organisms. Parrish *et al.* (1976) investigated toxicity and uptake of chlordane (identified as 1,2,4,5,6,7,8,8-octachloro-3*a*,4,7,7*a*-tetrahydro-4,7-methanoindane) in five species of estuarine organisms, ranging from oysters to fish. They determined 96-hour toxicity levels comparable to, but generally less than, those of Henderson, Pickering, and Tarzwell (1959) listed above. Parrish *et al.* stated that although chlordane appeared to be slightly more toxic to marine organisms than to freshwater organisms, direct comparisons were difficult due to different test conditions (*e.g.*, exposure duration and temperature). Table 27 summarizes these results in estuarine species, and also lists concentration factors (*i.e.*, whole body tissue concentration/concentration in water) determined in the 96-hour tests.

Table 27. EC_{50} values and concentration factors determined for chlordane, in five species of estuarine organisms. EC_{50} values in µg/l (ppb). Source: Parrish *et al.* (1976).

Test species	96 hr. EC_{50}, Nominal*	96 hr. EC_{50}, Measured*	Concentration Factor, Nominal	Concentration Factor, Measured
Eastern oyster (*Crassostrea virginica*)	11.6 (8.4-16.0)	6.2 (4.8-7.9)	1300-5000	3200-8300
Pink shrimp (*Penaeus duorarum*)	0.5 (0.4-0.8)	0.4 (0.3-0.6)	3000-6500	4000-6000
Grass shrimp (*Palaemonetes pugio*)	8.4 (6.9-10.3)	4.8 (4.0-6.0)	1000-2000	1900-2300
Sheepshead minnow (*Cyprinodon variegatus*)	39.8 (34.0-45.2)	24.5 (19.9-28.6)	8300-10000	12600-18700
Pinfish (*Lagodon rhomboides*)	10.4 (7.5-12.4)	6.4 (5.0-7.3)	2000-4800	3000-7500

* Effect expressed as % reduction in shell deposition for oysters, death for all others. 95% confidence limits in parentheses.

Parrish *et al.* found that chlordane was acutely toxic to all five estuarine species tested. Pink shrimp (*Penaeus duorarum*) was the most sensitive organism, with 96-hour EC_{50} values ranging from 0.3 to 0.6 ppb. Sheepshead minnows (*Cyprinodon variegatus*) were least susceptible to chlordane, differing by about a factor of 60 from pink shrimp (96-hour EC_{50} = 19.9 to 28.6 ppb). Eastern oysters (*Crassostrea virginica*), grass shrimp (*Palaemonetes pugio*), and pinfish (*Lagodon rhomboides*) showed 96-hour toxic effects (impaired shell deposition in oysters, mortality in shrimp and fish) at roughly equivalent concentrations, 4.0 to 8.0 ppb.

Parrish *et al.* also evaluated the effects of chlordane exposure on reproductive success and survival of sheepshead minnows. In 28-day tests, fertilization success and survival of embryos were not significantly affected by six water concentrations ranging from 0 to 100 µg/l chlordane (nominal concentration, 36 µg/l measured). However, survival of fry was significantly affected at nominal concentrations of 46 and 100 µg/l (17 and 36 µg/l measured), with no fry surviving longer than 10 days. At 21 µg/l nominal (7.1 µg/l measured), mortality was not significant but fry lost equilibrium and swam erratically.

Heptachlor toxicity to and uptake by several estuarine organisms. In a study related to the preceding effort, Schimmel, Patrick, and Forester (1976a) performed heptachlor 96-hour bioassays and uptake investigations using the same estuarine species as Parrish *et al.* (1976), with the addition of spot (*Leiostomus xanthurus*) (Schimmel, Patrick, and Forester identified grass shrimp as *Palaemonetes*

vulgaris; Parrish *et al.* called it *P. pugio*). Although analytical-grade heptachlor (99.8% heptachlor) was studied, the primary heptachlor used by Schimmel, Patrick, and Forester was the technical grade mixture which contained a sizable percentage of chlordane constituents: 65 percent heptachlor, 22 percent *trans*-chlordane, 2 percent *cis*-chlordane, and 2 percent nonachlor. Results for the 96-hour exposures and concentration factors are summarized in Table 28.

Direct comparison of Tables 27 and 28 clearly shows that the technical grade heptachlor used by Schimmel, Patrick, and Forester was consistently more toxic to the five common estuarine species than the chlordane used by Parrish *et al.* Measured concentration factors for heptachlor and chlordane did not show the same consistency.

Table 28. EC_{50} values and concentration factors determined for technical-grade heptachlor, in six species of estuarine organisms. EC_{50} values in µg/l (ppb). Source: Schimmel, Patrick, and Forester (1976a).

Test species	96-hr EC_{50}, Nominal*	96-hr EC_{50}, Measured*	Concentration Factor Ranges, Measured
Eastern oyster (*Crassostrea virginica*)	4.0	1.5	3900-8500
Pink shrimp (*Penaeus duorarum*)	0.21 (0.16-0.28)	0.11 (0.07-0.15)	200-300
Grass shrimp (*Palaemonetes vulgaris*)	2.08 (1.39-3.02)	1.06 (0.46-2.07)	500-700
Sheepshead minnow (*Cyprinodon variegatus*)	10.53 (7.39-13.71)	3.68 (2.74-4.67)	7400-21300
Pinfish (*Lagodon rhomboides*)	9.29 (6.98-12.59)	3.77 (2.02-8.80)	2800-7700
Spot (*Leiostomus xanthurus*)	2.18 (1.86-2.58)	0.85 (0.72-1.00)	3000-13800

* Effect expressed as % reduction in shell deposition for oysters, death for all others. 95% confidence limits in parentheses.

Heptachlor uptake, depuration, retention, and metabolism by spot (*Leiostomus xanthurus*). Schimmel, Patrick, and Forester (1976b) focused on the effects of technical-grade heptachlor (65% heptachlor, 22% *trans*-chlordane, 2% *cis*-chlordane, and 2% nonachlor) on the estuarine fish, spot (*Leiostomus xanthurus*). Specimens of juvenile spot were collected near the U.S. EPA laboratory at Gulf Breeze, Florida, and held for 10 days to acclimate prior to the investigation.

Fish were exposed to technical-grade heptachlor for 24 days with measured concentrations in water ranging from 0.14 µg/l to 2.55 µg/l. At the highest concentration, fish did not survive beyond 6 days. Levels of technical heptachlor constituents in both edible tissues and in whole fish were measured over the exposure period. The remaining test fish were held for 28 additional days with no further exposure to heptachlor to evaluate depuration.

Schimmel, Patrick, and Forester observed that accumulated concentrations of the measured heptachlor compounds were approximately three times greater in the head and viscera of spot as in the edible tissues. Table 29 summarizes accumulations of heptachlor (heptachlor, *trans*-chlordane, and *cis*-chlordane) after 24 days of exposure to various concentrations in water.

Table 29. Concentrations of technical heptachlor compounds in edible tissues and whole body samples of spot exposed for 24 days in flowthrough bioassay. Source: Schimmel, Patrick, and Forester (1976b).

Heptachlor Exposure in Water, (µg/l)		Hept, Edible	*t*-Chlord Edible	*c*-Chlord Edible	Hept Whole	*t*-Chlord Whole	*c*-Chlord Whole
Nominal	Measured	µg/g	µg/g	µg/g	µg/g	µg/g	µg/g
0.27	0.14	0.23	0.10	0.04	0.34	0.17	0.06
0.52	0.26	0.27	0.12	0.04	0.56	0.28	0.07
1.01	0.58	1.1	0.67	0.18	1.75	1.09	0.30
1.99	1.03	2.9	1.6	0.38	5.28	2.96	0.70

Figure 10 shows results of the bioconcentration/depuration study for *cis*- and *trans*-chlordane. For both compounds (as well as for heptachlor and heptachlor epoxide, not shown) and all exposure concentrations, accumulation in edible tissues was rapid. Maximum levels were reached at either day 10 or day 17, and in nearly all cases had begun to decline at day 24 when exposure was discontinued and depuration commenced. Figure 11 illustrates these results for *trans*-chlordane and heptachlor, but combines them with measured exposure concentration in water to give bioconcentration factors.

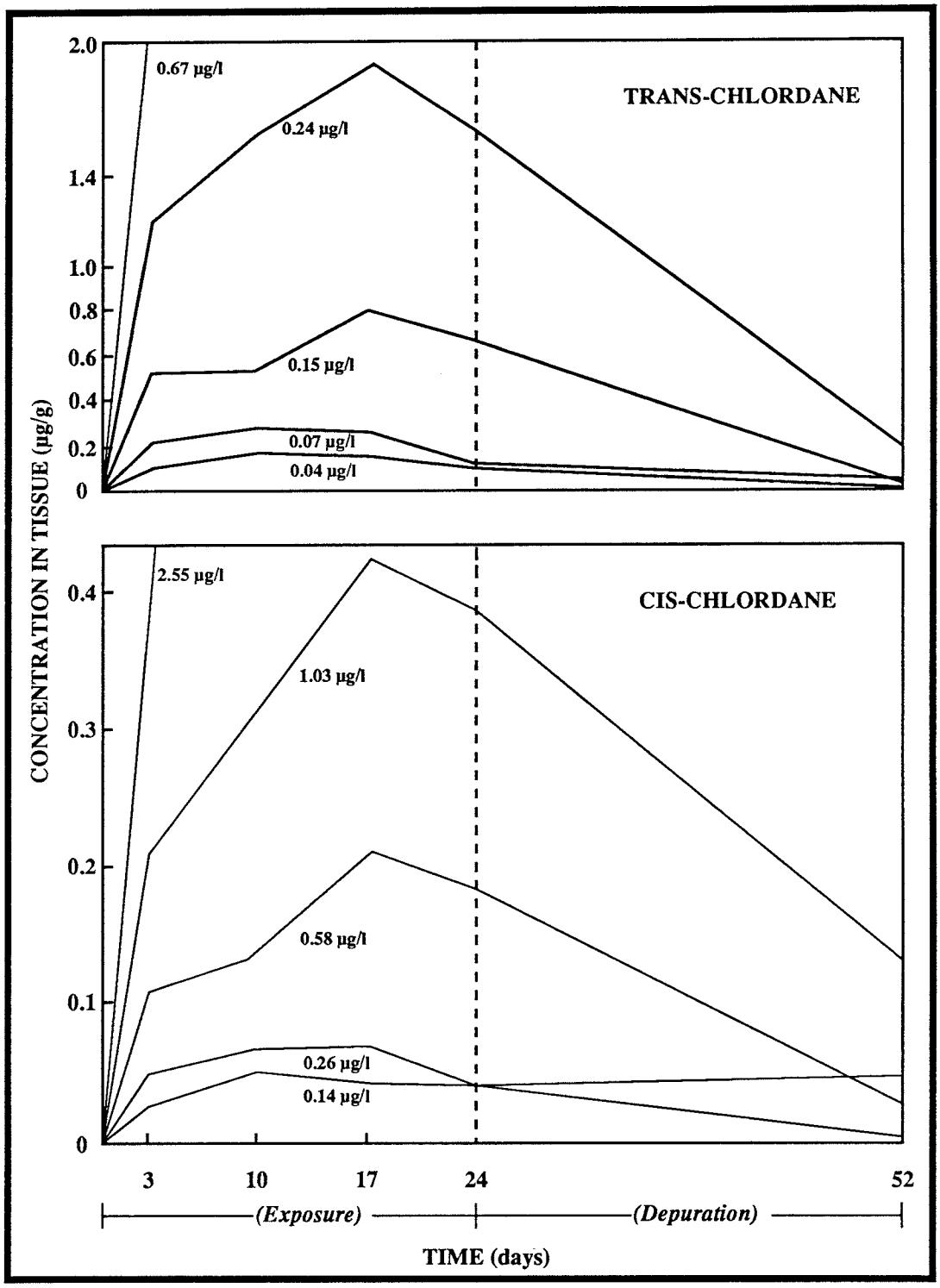

Figure 10. Bioconcentration of *trans*-chlordane and *cis*-chlordane in edible tissue of spot exposed in a 24-day bioassay, followed by a 28-day depuration'. Source: Schimmel, Patrick, and Forester (1976b).

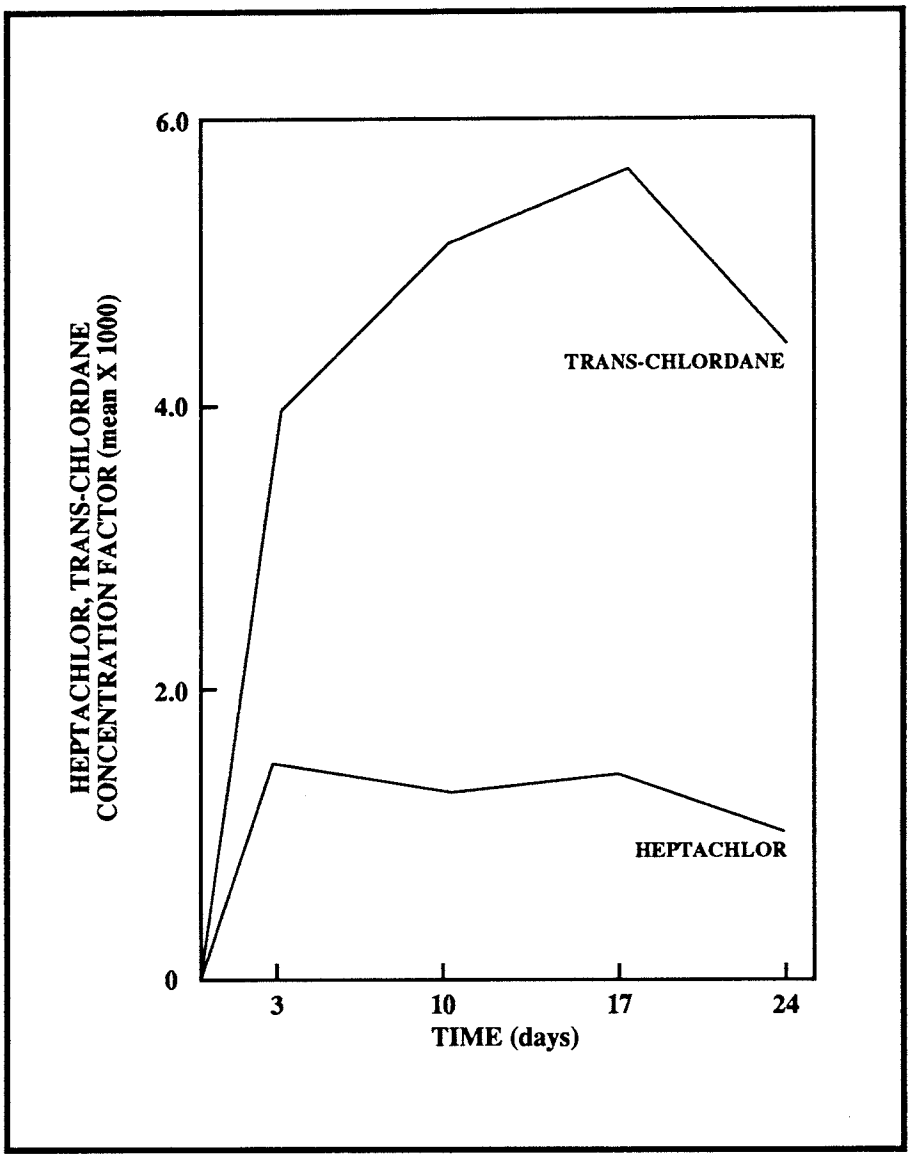

Figure 11. Average concentration factors of heptachlor and *trans*-chlordane in muscle tissue of spot continuously exposed to technical grade heptachlor for 24 days. Source: Schimmel, Patrick, and Forester (1976b).

Influence of lipid pool size on bioaccumulation of chlordane by suckers. Roberts, deFrietas, and Gidney (1977) investigated the relationship between liposity and body burden of chlordane compounds (identified as *cis*- and *trans*-chlordane) in the northern redhorse sucker (*Moxostoma macrolepidotum*) and white sucker (*Catostomus commersoni*). Using a radioactive isotope as a tracer, the assimilation efficiency from food and biological half-life of chlordane in individual fish was determined. 203 Hg-labeled methylmercury chloride was used to estimate the assimilation efficiency for chlordane, the amount of contaminated food actually ingested, and , consequently, the amount of chlordane ingested and its rate of clearing from tissues.

Uptake and retention of chlordane in tissues of the northern redhorse sucker was found to be directly related to lipid content of individual fish. For example, 62 days after a 5-day oral exposure period, total chlordane (sum of *cis*- and *trans*- isomers) normalized to ^{203}Hg concentration (in order to give a more representative value for chlordane actually ingested, rather than simply administered) varied considerably. However if the results obtained by Roberts, deFrietas, and Gidney are analyzed using the nonparametric

Spearmans rank correlation coefficient, lipid content and total chlordane are found to be highly significantly correlated ($r_S = 0.748$, $p = 0.0028$). Roberts, deFrietas, and Gidney estimated the half-lives in a fish with lipid content of 2 percent to be 60 days for *cis*-chlordane, and 33 days for *trans*-chlordane.

The authors also administered chlordane to white suckers by gastric intubation and found that up to 57 percent of the dose could be subsequently measured in tissues of the subject fish. Within 5 days of the dosing, less than 5 percent of the administered loading remained in the gut; 30 to 60 percent had moved into tissues. At this time, 23 to 40 percent of the body burden was partitioned to the lateral muscle tissue, while liver concentrations accounted for only 5 percent or less. Roberts, deFrietas, and Gidney noted that while chlordane compounds were shown here to be readily absorbed from the digestive tract, other chlorinated organics such as DDTs and PCBs had much higher assimilation coefficients.

Difference between fresh- and seawater acclimated tilapia in the accumulation of chlordane and pentachlorophenol. Noting that different physiological mechanisms govern the regulation of osmotic pressure in freshwater and saltwater organisms, Tachikawa *et al.* (1987) investigated the differences in chlordane and pentachlorophenol uptake between fresh- and seawater acclimated tilapia (*Tilapia nilotica*). Technical chlordane (with a listed content by constituent of 6.8%±0.3 α-chlordane, 8.3%±0.4 γ-chlordane, 3.7%±0.3 *trans*-nonachlor, 1.9%±0.1 *cis*-nonachlor, and undetected concentrations of oxychlordane) was used in the study and reported concentrations were the sum of concentrations of the five components. Three fish per group (*i.e.*, treated freshwater and treated seawater) were exposed to 50 ppb nominal concentrations in 50 liter aquaria for 24 hours. In a separate experiment, tilapia were fed food dosed with chlordane. Tissue residues from the exposure in water are shown in Figure 12, while those measured in the feeding study are summarized in Table 30.

Figure 12 shows that for tilapia, much higher concentrations of chlordane were found in liver tissue and fat than in other tissues analyzed. This may have reflected both the lipophilic nature of the chlordane compounds as well as sequestering of the contaminants in liver tissue for processing. Differences between freshwater- and seawater-acclimated fish were most evident in liver tissue where concentrations in freshwater fish exceeded those in seawater fish by about a factor of five.

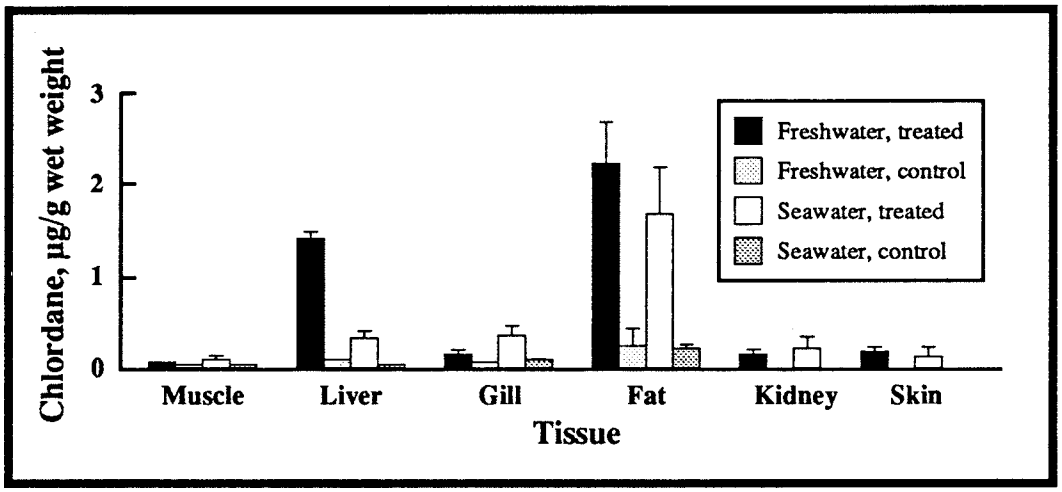

Figure 12. Chlordane measured in tissues of tilapia exposed to 50 ppb chlordane for 24 hours. Source: Tachikawa *et al.* (1987).

Table 30. Chlordane concentrations measured in tissues of tilapia 24 hours after being fed dosed food. Values in μg/g (ppm) wet weight. Source: Tachikawa *et al.* (1987).

Tissue or organ	0.833 mg[1] Freshwater[3] (n=4)	0.833 mg[1] Seawater[3] (n=3)	8.33 mg[2] Freshwater[4] (n=2)	8.33 mg[2] Seawater[3] (n=3)
Muscle	0.24±0.04	0.12±0.11	1.88-1.96	0.94±0.79
Liver	4.27±0.87	2.12±1.05	41.23-77.53	23.85±21.11
Gill	1.82±1.05	1.95±1.64	12.82-16.05	5.01±1.12
Fat	6.39±2.06	2.26±1.31	20.97-24.70	14.78±16.06
Digestive tract	2.01±0.27	3.66±2.48	97.01-272.40	153.50±30.40

1 Corresponds to 50 ppb exposure in 50 l of test water.
2 Corresponds to 500 ppb exposure in 50 l of test water.
3 Mean ± standard deviation.
4 Range of 2 values

The results for fish fed chlordane-dosed food were generally similar to those for fish exposed via water. Liver tissue and fat again contained the highest concentrations, while liver tissue reflected the most distinct differences between freshwater and seawater fish. In the case of digestive tract, contents in the tract were analyzed with tissue which may have accounted for the similarity in measured concentrations.

CHLORDANE RESIDUES IN THE ENVIRONMENT

Interpretation of environmental residues of chlordane is made difficult by a number of considerations. As has been noted previously, the complex mixture of compounds that comprises the commercial product technical chlordane, complicates both laboratory procedures and the evaluation of analytical results.

Metzger (1987) defined the most likely reasons for residue variation with different species, and his observations are:

1. Actual differences in residues in different kinds of fish due to different feeding habits.

2. Differences in residues due to a greater number of freshwater species (which contain higher residues than saltwater species) in a particular dataset.

3. Differences in residues due to a greater number of fish from a particular data subset being taken from a more highly contaminated (or less contaminated) location.

4. Differences in analytical method (recoveries), sampling techniques, data reporting (*e.g.*, data not corrected for recovery).

5. Differences in residues due to a greater number of high or low residue species in a specific data subset.

6. Different studies monitoring for all or only part of the residue of concern for a particular chemical.

Similarly, Garreis and Murphy (1986a) discussed the difficulties inherent in attempting to compare chlordane levels in fish from various areas experiencing chlordane contamination. They noted that such comparisons are complicated by lack of standardization in sample collection, sample preparation, and laboratory analysis as well as unavoidable differences in rates of chlordane bioaccumulation across species.

Many of the chlordane measurements made under the Chester River Study (Munson, 1972) and the Upper Bay Survey (Munson, 1975) were overseen by the same investigator, presumably using comparable methodologies. Nevertheless, Munson (1975) commented that chlordane concentrations from the two studies were *not* directly comparable: in the Chester River Study, pure standards of chlordane isomers were not available and the standard method employed technical chlordane as the reference standard. In the later Upper Bay Survey, pure standards of α- and γ-chlordane were available and incorporated into analytical methodologies. Munson (1975) noted that a reliable conversion factor between the two measurements would only be possible if the same gas chromatographic column packings were used under identical conditions, but he offered an approximate comparison between values reported as total (α+γ) chlordane and technical chlordane by multiplying total chlordane by 2.0.

All of these considerations are relevant when evaluating data results to be presented in this summary. Whenever possible, the actual chlordane compounds measured and identified in a particular dataset are specified. General analytical methods used are summarized in Appendix B. However, evaluation of methodologies employed to generate results is beyond the scope of the present effort and readers are advised to use caution in directly comparing results obtained by different researchers, or even the same investigator using slightly different methods.

Results are generally presented in the formats in which they were reported. That is, if data are given in wet weight, dry weight, or lipid weight, they are reproduced in those formats. In some isolated instances, results may be converted to other formats in order to facilitate a certain degree of comparability among studies. A "rule-of-thumb" conversion between wet weight and dry weight is that a dry weight concentration will exceed a correspondingly equivalent wet weight concentration by a factor of three to five, depending on the specific organism and tissue. Lipid weights are more variable and are highly dependent upon species and tissues examined.

National Studies

Estuarine mollusk studies, National Pesticide Monitoring Program (NPMP), 1965-1972; 1977. In order to assess the extent of estuarine contamination by organochlorine pesticides, the U.S. Bureau of Commercial Fisheries initiated a monitoring program for mollusks in 1965. In cooperation with local laboratories, approximately 180 monitoring stations in 15 coastal states were sampled monthly for any of ten species of bivalve mollusks. When the principal participating laboratory (the Gulf Breeze Laboratory, Gulf Breeze, Florida) became part of the newly formed U.S. EPA in 1971, the monitoring effort was assumed by the EPA as well. The program analyzed 15 persistent organochlorine compounds, among them, chlordane (not specifically identified as to isomers, but likely technical grade). Ten species of pelecypod mollusks were used as monitoring organisms: Pacific oyster, *Crassostrea gigas*; eastern oyster, *Crassostrea virginica*; Olympia oyster, *Ostrea lurida*; ribbed mussel, *Modiolus demissus*; northern horse mussel, *Modiolus modiolus*; California mussel, *Mytilus californianus*; blue mussel, *Mytilus edulis*; hard clam, *Mercenaria mercenaria*; soft-shell clam, *Mya arenaria*; and Asiatic clam, *Corbicula fluminea*, a freshwater species. Butler (1973) described the 1965-1972 results of this part of the NPMP, and Butler, Kennedy, and Schutzmann (1978) compared 1977 measurements against those obtained in 1972.

The results of these NPMP estuarine bivalve studies, as far as chlordane compounds are concerned, can be reported rather succinctly: Of 8,095 samples analyzed for chlorinated organic compounds between 1965 and 1972 and of 178 samples analyzed in 1977, no samples contained chlordane at levels exceeding the reported detection limit of 10 ppb wet weight. PCBs, DDT, and other pesticides were measured at significant levels over the course of the monitoring, but chlordane was not. This result will be discussed subsequently in the context of more contemporary measurements.

Estuarine fish studies, National Pesticide Monitoring Program, 1972-1976. As part of its NPMP, the U.S. EPA sponsored the collection and analysis of juvenile fish from estuarine regions of the United States, as well as Puerto Rico and the Virgin Islands, between the years of 1972 and 1976. Approximately 38,000 fish were collected semiannually from 144 estuaries, were pooled into 1,524 samples and screened for 20 pesticides as well as PCBs. Juvenile fish were selected for study because it was presumed that they were more likely to be representative of a given estuarine environment than wider ranging adult fish. Results

from this effort were summarized and reported in Butler and Schutzmann (1978), and more detailed supporting data were obtained from Butler and the U.S. EPA Gulf Breeze Laboratory. Among the compounds searched out were chlordane, technical chlordane, and γ-chlordane. Conversations with chemists and technicians at the Gulf Breeze Laboratory have confirmed that "chlordane" was likely to have been a mix of the α- and γ- isomers of chlordane, while "technical chlordane" was a mixture represented by a slightly different chromatogram not easily resolved into isomeric constituents (J. Moore, U.S. EPA Gulf Breeze Laboratory, pers. comm. 23 March 1989).

Although NPMP analyzed over 1,500 pooled samples of whole fish from around the country, chlordane compounds were detected in only 35 (2.3%). These positive measurements, above the stated detection limit of 10 μg/kg (ppb) wet weight, are shown in Table 31. The small sample size and the wide variety of species analyzed make meaningful comparisons difficult; however, fish from estuaries in Hawaii, around the Chesapeake Bay in Maryland, and in Texas were indicated as being most impacted by chlordane contamination in the given time period.

Table 31. Positive (above detection limit) measurements of chlordane compounds from NPMP analyses of juvenile estuarine fish, 1972-1976. Whole fish, except as noted. Values in μg/kg (ppb) wet weight. Source: Data supporting Butler and Schutzmann (1978).

Estuary	Species	Date	Chlordane	Technical Chlordane	Gamma-Chlordane
Mississippi Sound, AL	Striped mullet	6/75	57		
Nawiliwili Harbor, HI	Silverside	6/73	340		
Nawiliwili Harbor, HI	Blenny	6/73	250		
Lihue Bay, HI	Lizardfish	6/73	240		
Lihue Bay, HI	Blenny	6/73	240		
Capt. Cooks Landing,HI	Silverside	6/73	330		
Capt. Cooks Landing, HI	Lizardfish	6/73	340		
Potomac River, MD	Bay anchovy	10/74	21		
Potomac River, MD	Bay anchovy	10/74	16		
Potomac River, MD	Spot	10/74	35		
Potomac River, MD	Spot	10/74	60		
South River, MD	Bay anchovy	10/74	42		
South River, MD	Bay anchovy	10/74	21		
Patapsco River, MD	Alewife	5/73		350	
Patapsco River, MD	Alewife	5/73		710	
Patapsco River, MD	Atlantic menhaden	5/73		32	
Patapsco River, MD	White perch	5/73		210	
Patapsco River, MD	White perch	5/73		220	
Patapsco River, MD	Bay anchovy	10/74	34		
Patapsco River, MD	Bay anchovy	10/74	40		
Patapsco River, MD	Bay anchovy	10/74	29		
Patapsco River, MD	Atlantic menhaden	10/74	143		
Patapsco River, MD	White perch	10/74	264		
Patapsco River, MD	White perch	10/74	129		
Susquehanna Flats, MD	White perch	5/73		90	
Choptank River, MD	Bay anchovy	10/74	18		
Choptank River, MD	Bay anchovy	10/74	62		
Choptank River, MD	Bay anchovy	10/74	15		
Western Long Is. Sound, NY	Atlantic silverside	4/76			253
Western Long Is. Sound, NY	Atlantic silverside	4/76			181
Port Harlingen, TX	Gulf menhaden	1/76	41		
Port Harlingen, TX	Gulf menhaden	1/76	30		
Port Harlingen, TX	Gulf menhaden	1/76		344	
Arroyo City, TX	Spotted sea trout	2/76			
	Liver			822	
	Ovaries			188	
	Muscle			207	
Colorado River, TX	Striped mullet	2/76		104	

In general, the estuaries where elevated chlordane levels were measured in resident fish were not urban or industrial in nature, and in fact were regions receiving substantial amounts of runoff from agricultural lands. This contrasts with other studies and evaluations of previously collected data (*e.g.*, Carey, Wiersma, and Tai, 1976; see Table 4) which suggested that chlordane was a contaminant more likely to be associated with urban populations than agricultural areas. However, the dominant crops in the agricultural regions adjacent to the affected estuaries--corn, pineapples, and sorghum--represented important target crops for chlordane use in pest control. In the case of the Chesapeake Bay, where fish from a number of sampling locations were found to contain chlordane compounds, corn has been the primary regional agricultural crop (Pait *et al.*, 1989). According to one U.S. EPA estimate, chlordane application for protection of corn accounted for over 20 percent of the total U.S. use in 1974; if this was the case, the fact that fish in Chesapeake estuaries contained consistently measurable chlordane residues is not surprising. In Hawaii, chlordane was a major pesticide used for protection of pineapple crops as well as for structural treatments. In Texas, chlordane was applied as a sorghum seed treatment, but the relative extent that these contributed to residues in the estuarine environment is not clear.

EPA-NOAA Cooperative Estuarine Monitoring Program, 1976-1977. In 1976 and 1977, the U.S. EPA and NOAA sponsored the collection and analysis of contaminant burdens in fish from 11 U.S. estuaries: Long Island Sound, Chesapeake Bay, Savannah River estuary, Pensacola Bay, Mobile Bay, Galveston Bay, Laguna Madre, San Francisco Bay, Columbia River estuary, Coos Bay, and Puget Sound. Table 32 lists the common and species names of the 28 fish species sampled in this project. Results were summarized in a final report by Butler (unpublished).

Table 32. Common and species names of fish sampled for EPA-NOAA Cooperative Estuarine Monitoring Program, 1976-1977. Source: Butler (unpublished); Robins *et al.* (1980).

Common Name	Species Name
American eel	*Anguilla rostrata*
American shad	*Alosa sapidissima*
Atlantic croaker	*Micropogon undulatus*
Black drum	*Pogonias cromis*
Bluefish	*Pomatomus saltatrix*
Dover sole	*Microstomus pacificus*
English sole	*Parophrys vetulus*
Night smelt	*Spirinchus starksi*
Pacific cod	*Gadus macrocephalus*
Pacific hake	*Merluccius productus*
Pile perch	*Rhacochilus vacca*
Red drum	*Sciaenops ocellatus*
Redtail surfperch	*Amphistichus rhodoterus*
Shiner perch	*Cymatogaster aggregata*
Southern flounder	*Paralichthys lethostigma*
Southern kingfish	*Menticirrhus americanus*
Spanish mackerel	*Trachurus symmetricus*
Spiny dogfish	*Squalus acanthias*
Spot	*Leiostomus xanthurus*
Spotted seatrout	*Cynoscion nebulosus*
Starry flounder	*Platichthys stellatus*
Striped bass	*Morone saxatilis*
Striped mullet	*Mugil cephalus*
Striped surfperch	*Embiotoca lateralis*
Summer flounder	*Paralichthys dentatus*
Tautog	*Tautoga onitis*
Walleye pollock	*Theragra chalcogramma*
White sturgeon	*Acipenser transmontanus*

While a large number of data were reported for DDT compounds, dieldrin, and PCBs, 18 other organochlorine and organophosphate compounds, including chlordane and heptachlor, were not detected in any of the 561 measurements covering 514 individual fish. As in the previously detailed NPMP estuarine mollusk study (for which similar analytical methods were apparently employed), the detection limit for

pesticides was specified as 10 ppb, wet weight. The implications of this level of sensitivity will be discussed in the context of more contemporary measurements in subsequent sections.

EPA Bioaccumulation Study, 1986-1987. The Bioaccumulation Study of the U.S. EPA was an outgrowth of that Agency's National Dioxin Study, which investigated the extent to which dioxin (2,3,7,8-tetrachlorodibenzo-*p*-dioxin, or 2378-TCDD) was present in several environmental media. Concerns that other contaminants similar to dioxin could accumulate in aquatic food webs and present health threats to human consumers prompted the EPA to conduct a one-time aquatic monitoring study to support regulatory decisions mandated by the Clean Water Act. Contaminants analyzed under the Bioaccumulation Study were selected for their potential for human toxicity, and more specifically, for their carcinogenicity, bioaccumulation potential, and potential for human exposure through releases in the aquatic environment (U.S. EPA, 1986b).

The Bioaccumulation Study focused upon fish sampled at about 400 fresh- and saltwater sites in the United States. and Puerto Rico, with analyses primarily of whole body tissues. Among the approximately 60 highly bioaccumulative compounds targeted were six components of technical chlordane: *cis*-, *trans*-, and oxychlordane, *cis*- and *trans*-nonachlor, and heptachlor. Although most of the samples analyzed were freshwater fish, specimens from some estuaries and saltwater embayments were also sampled. Data for those coastal locations were extracted and provided for this report by the Monitoring and Data Support Division of the U.S. EPA's Office of Water Regulations and Standards.

Approximately 60 of the 400 total samples originated in coastal or estuarine waters. A large number of species were collected, including spot (*Leiostomus xanthurus*), red drum (*Sciaenops ocellatus*), spotted seatrout (*Cynoscion nebulosus*), starry flounder (*Platichthys stellatus*), southern flounder (*Paralichthys lethostigma*), Dolly Varden (*Salvelinus malma*), Atlantic salmon (*Salmo salar*), flathead sole (*Hippoglossoides elassodon*), Pacific cod (*Gadus macrocephalus*), surf smelt (*Hypomesus pretiosus*), bocaccio (*Sebastes paucispinis*), sea catfish (*Arius felis*), summer flounder (*Paralichthys dentatus*), spotted drum (*Equetus punctatus*), sheepshead (*Archosargus probatocephalus*), black drum (*Pogonias cromis*), weakfish (*Cynoscion regalis*), red snapper (*Lutjanus campechanus*), Atlantic croaker (*Micropogonias undulatus*), bluefish (*Pomatomus saltatrix*), winter flounder (*Pseudopleuronectes americanus*), white seaperch (*Phanerodon furcatus*), diamond turbot (*Hypsopsetta guttulata*), striped mullet (*Mugil cephalus*), gizzard shad (*Konosirus punctatus*), and stingray (unspecified, but probably round stingray, *Urolophus halleri*). In addition to the fish tissue results, a few measurements for soft-shell clam (*Mya arenaria*), Pacific oyster (*Crassostrea gigas*), and unspecified shellfish (*Crassostrea virginica*?) were made in Washington State and Louisiana waters. Two thirds of the marine and estuarine samples were whole body analyses of bottom-feeding fish. The other three sample categories for the subset were fillet tissue analyses of predators, whole body analyses of predators, and shellfish tissue measurements. Results by sample category are summarized in Table 33. Figure 13 shows marine and estuarine results in fish tissue plotted on a map of the United States.

Table 33. Summaries of summed concentrations of three chlordane compounds (Σ *cis*- and *trans*-chlordane, *cis*- and *trans*-nonachlor, oxychlordane, and heptachlor) in fish and shellfish collected in the coastal and estuarine environment of the United States, 1986-1987. Values in ng/g (ppb) wet weight. Source: Unpublished data provided by R. Yender, U.S. EPA.

Matrix	N	Σ chlordane Range	Min Species	Minimum Location	Max Species	Maximum Location
Bottom-feeding fish Whole body	40	6.91[a]-409	Red drum	So. Brunswick R., Georgia	Stingray	Colorado Lagoon, California
Predatory fish Fillet	8	6.94[a]-30.9	Spotted seatrout	North River, Georgia	Atlantic croaker	Houston Ship Channel, Texas
Predatory fish Whole body	8	7.50[a]-42.5	Sheepshead	North River, Georgia	Bluefish	Manhasset Bay, New York
Shellfish	6	7.50[a]-11.9	Softshell clam	Grays Harbor, Washington	Shellfish unidentified	Lake Pontchartrain, Louisiana

[a] All five chlordane compound concentrations below quantitation limits.; value shown reflects one- half specified limits.

Figure 13. Marine and estuarine results for summed concentrations of chlordane in fish sampled for the U.S. EPA Bioaccumulation Study, 1986-1987. Source: unpublished data compiled by R. Yender, U.S. EPA Office of Water Regulations and Standards.

In each of the four matrix categories, maximum concentrations occurred in waters near heavily populated urban centers. The highest level was found in whole body tissues of a stingray collected in Colorado Lagoon, a poorly flushed extension of Alamitos Bay near Long Beach, California. The wet weight concentration of 409 ppb exceeded the FDA action level of 300 ppb, but it should be noted that the former was a whole body measurement, as opposed to that in muscle tissues usually regarded as the edible portion of a fish. Of the 61 marine and estuarine samples analyzed, only one other contained summed chlordane compounds in excess of the FDA standard: a whole body sample of gizzard shad collected in the Susquehanna River in Maryland. The maximum value obtained for fillets (30.9 ppb in Atlantic croaker from the Houston Ship Channel) was an order of magnitude less than the FDA action level.

National Status and Trends Program Mussel Watch Project, 1986-. In 1986, the Ocean Assessments Division of NOAA resurrected and expanded the concept of a national program of bivalve monitoring in its NS&T Mussel Watch Project. It is an ongoing effort, with collections and analyses of both sediments and bivalves having been made on an annual basis since 1986 (sediments have been analyzed selectively since 1988). The NS&T Mussel Watch Project uses primarily the same sentinel organisms as the earlier efforts of Butler and the U.S. EPA. As in those efforts, because one species of bivalve does not occur ubiquitously around all the coasts of the nation, collection and analysis of several kinds of organisms are necessitated. Along the northeast Atlantic and Pacific coasts, including Alaska, *Mytilus edulis* are collected and analyzed; along the mid- and southeast Atlantic and Gulf of Mexico coasts, *Crassostrea virginica* oysters are used; along the Pacific coast, *Mytilus californianus* are collected in addition to *M.*

edulis; and at Hawaiian sites, *Ostrea sandvicensis* are targeted. When possible, sediment samples are collected near the sampling site for bivalves, and these have been analyzed for the same suite of chemical compounds as the biological samples. Sediments from most sites were analyzed in both 1986 and 1987; beginning in 1988 only sediments from new and previously unsampled sites were analyzed.

Three constituents of technical chlordane are measured in the NS&T Mussel Watch Project: α-chlordane, *trans*-nonachlor, and heptachlor. For the purposes of this review, the three compounds have been summed and are reported as Σ chlordane. Many concentrations of individual compounds were below quantitation or detection limits, and for summary and statistical purposes, one half the reported detection limit was substituted for those values below method detection limits (*i.e.*, "less than" values); the reported limit of quantitation was used for those concentrations not quantifiable (*i.e.*, "less than or equal to" values).

The tables below convey general characteristics of the analytical data generated in the first three years of the NS&T Mussel Watch Project. Figures 14 through 35 show more detailed results from specific sites in coastal and estuarine regions of the United States. In most cases, values portrayed are means of samples from three stations within a site. Figures 14 through 24 illustrate sediment results from 1986 and 1987, while Figures 25 through 35 show bivalve results.

Table 34. Summaries of summed concentrations of three chlordane compounds (Σ α-chlordane, *trans*-nonachlor, and heptachlor) in sediments collected in the coastal and estuarine environment of the United States, 1986-1987. Values in ppb dry weight. Source: NS&T Mussel Watch Project.

Year	N	Mean± Sdev	Min*	Min Location	Max	Max Location	Median
1986	393	1.51±3.06	0.06	(several)	22.5	Long Island Sound, NY	0.40
1987	366	2.07±6.35	0.05	(several)	76.1	Choctawhatchee Bay, FL	0.48

* These minima may reflect summation of converted concentrations below limits of detection. In many cases, value portrayed is sum of three below detected and/or unquantitated conversions.

Table 35. Species summaries of summed concentrations of three chlordane compounds (Σ α-chlordane, *trans*-nonachlor, and heptachlor) in mussels and oysters collected in the coastal and estuarine environment of the United States, 1986-1988. Values in ppb dry weight. Source: NS&T Mussel Watch Project.

Species	N	Mean± Sdev	Min*	Min Location	Max	Max Location	Median
1986							
M. edulis	141	42.2±51.0	0.78	Narragansett Bay, RI	312	New York Bight, NJ	24.7
C. virginica	216	20.6±24.9	0.78	(several)	168	Biloxi Bay, MS	12.0
M. californianus	72	12.7±13.6	0.97	San Simeon, CA	78.6	Anaheim Bay, CA	8.85
O. sandvicensis	6	25.8±25.6	2.00	Barbers Pt., HI	64.9	Honolulu Harbor, HI	19.7
1987							
M. edulis	151	41.2±45.4	0.81	(several)	231	Long Is. Sound, NY	23.5
C. virginica	211	25.6±47.6	1.47	Rookery Bay, FL	581	Choctawhatchee Bay, FL	14.4
M. californianus	72	22.9±32.3	0.81	(several)	217	Anaheim Bay, CA	11.1
O. sandvicensis	6	19.7±15.0	0.81	Barbers Pt., HI	37.0	Honolulu Harbor, HI	22.0
1988							
M. edulis	167	21.1±26.6	0.35	(3 locations)	212	Marina del Rey, CA	14.1
C. virginica	66	20.0±23.2	1.04	Laguna Madre, TX	162	Galveston Bay, TX	11.8
M. californianus	78	10.5±11.9	0.35	(2 locations)	68.1	Anaheim Bay, CA	7.28
O. sandvicensis	9	18.5±20.7	0.88	Kauai, HI	51.6	Honolulu Harbor, HI	2.13

* These minima may reflect summation of converted concentrations below limits of detection. In many cases, value portrayed is sum of three below detected and/or unquantitated conversions.

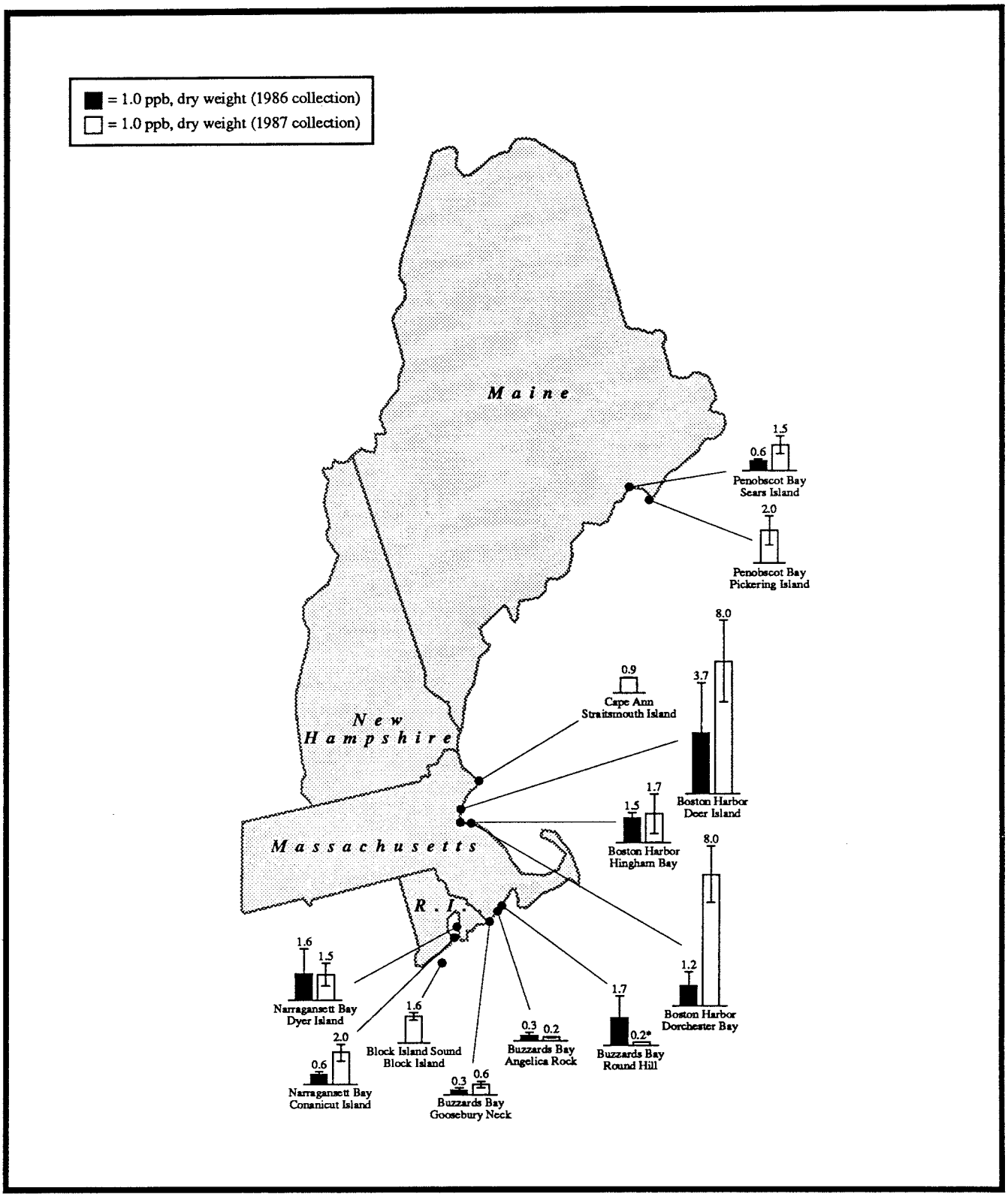

Figure 14. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments from Maine to Rhode Island, 1986 and 1987. Asterisk indicates all results below detection. Source: NS&T Mussel Watch Project.

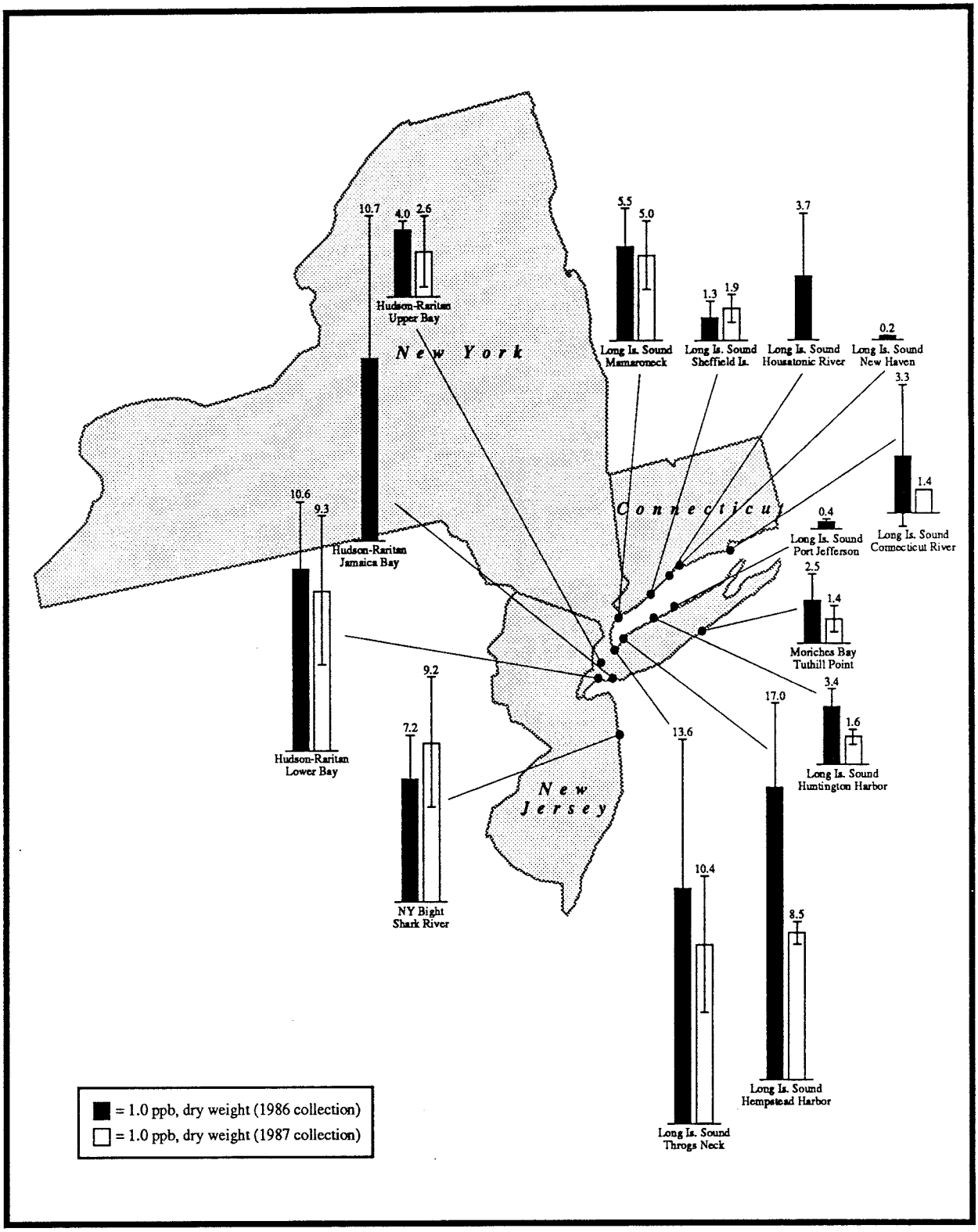

Figure 15. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments from Connecticut to New Jersey, 1986 and 1987. Source: NS&T Mussel Watch Project.

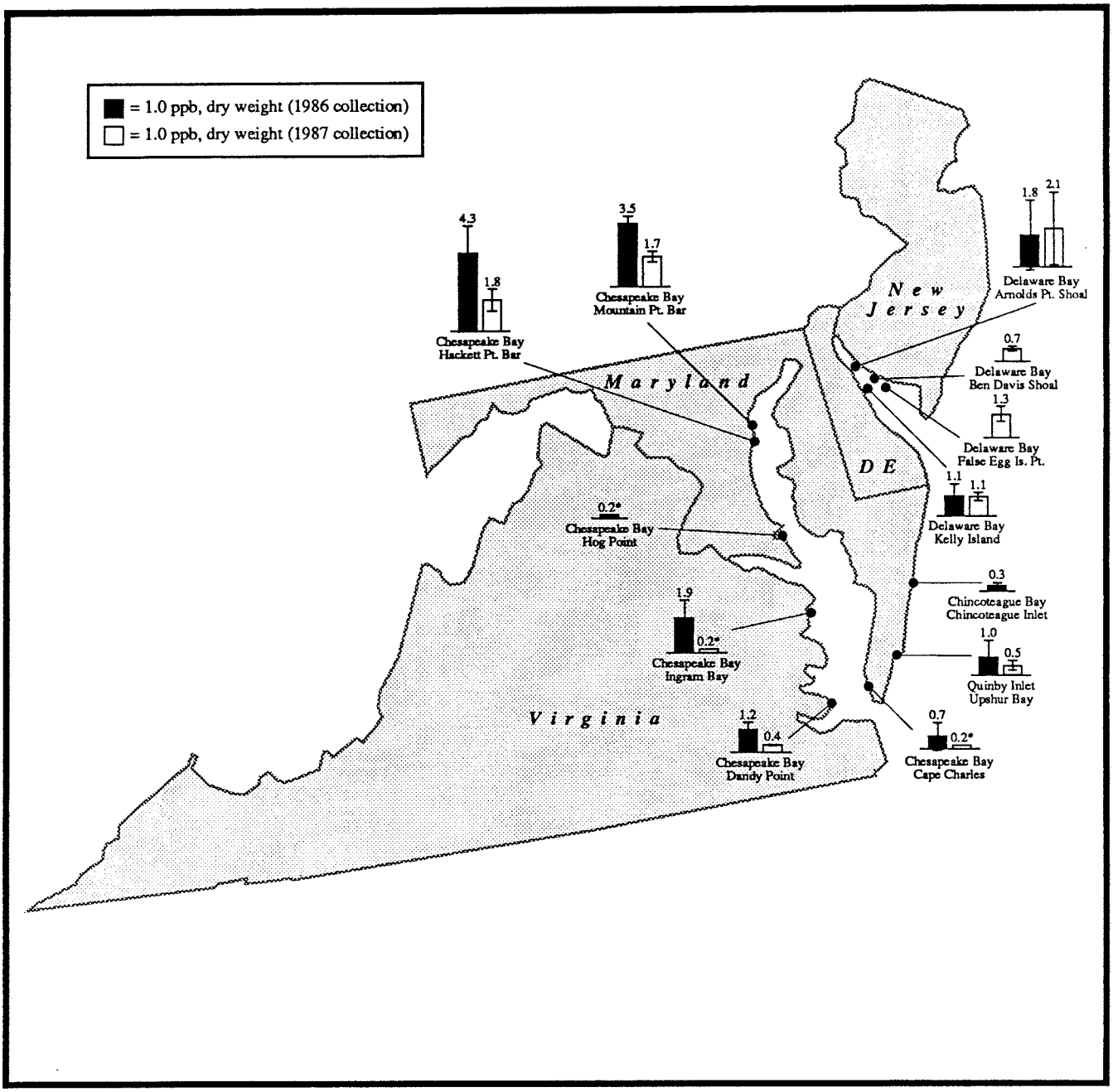

Figure 16. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments sampled along the Atlantic coast from New Jersey to Virginia, 1986 and 1987. Asterisk indicates all results below detection. Source: NS&T Mussel Watch Project.

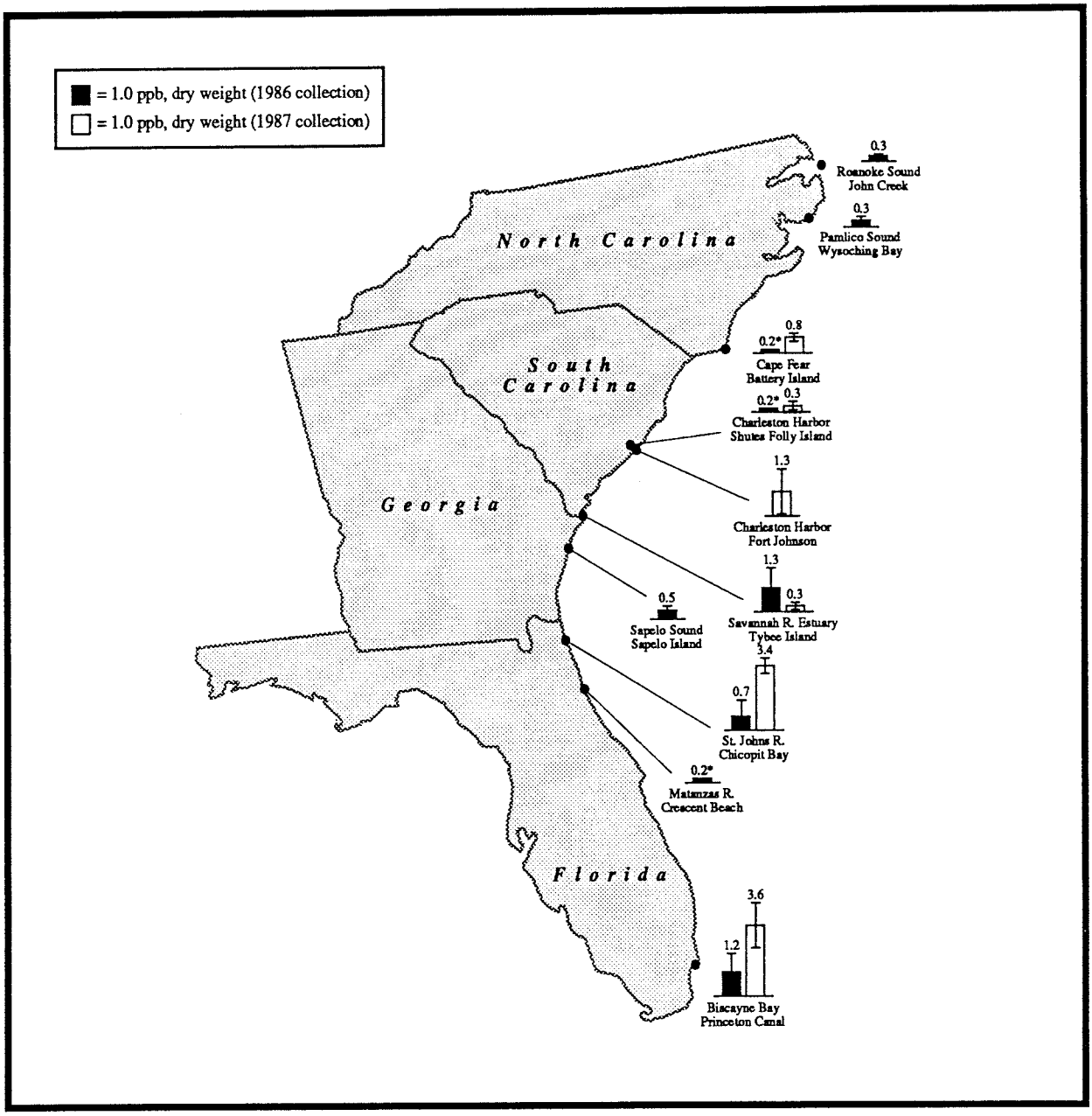

Figure 17. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments sampled along the Atlantic coast from North Carolina to Florida, 1986 and 1987. Asterisk indicates all results below detection. Source: NS&T Mussel Watch Project.

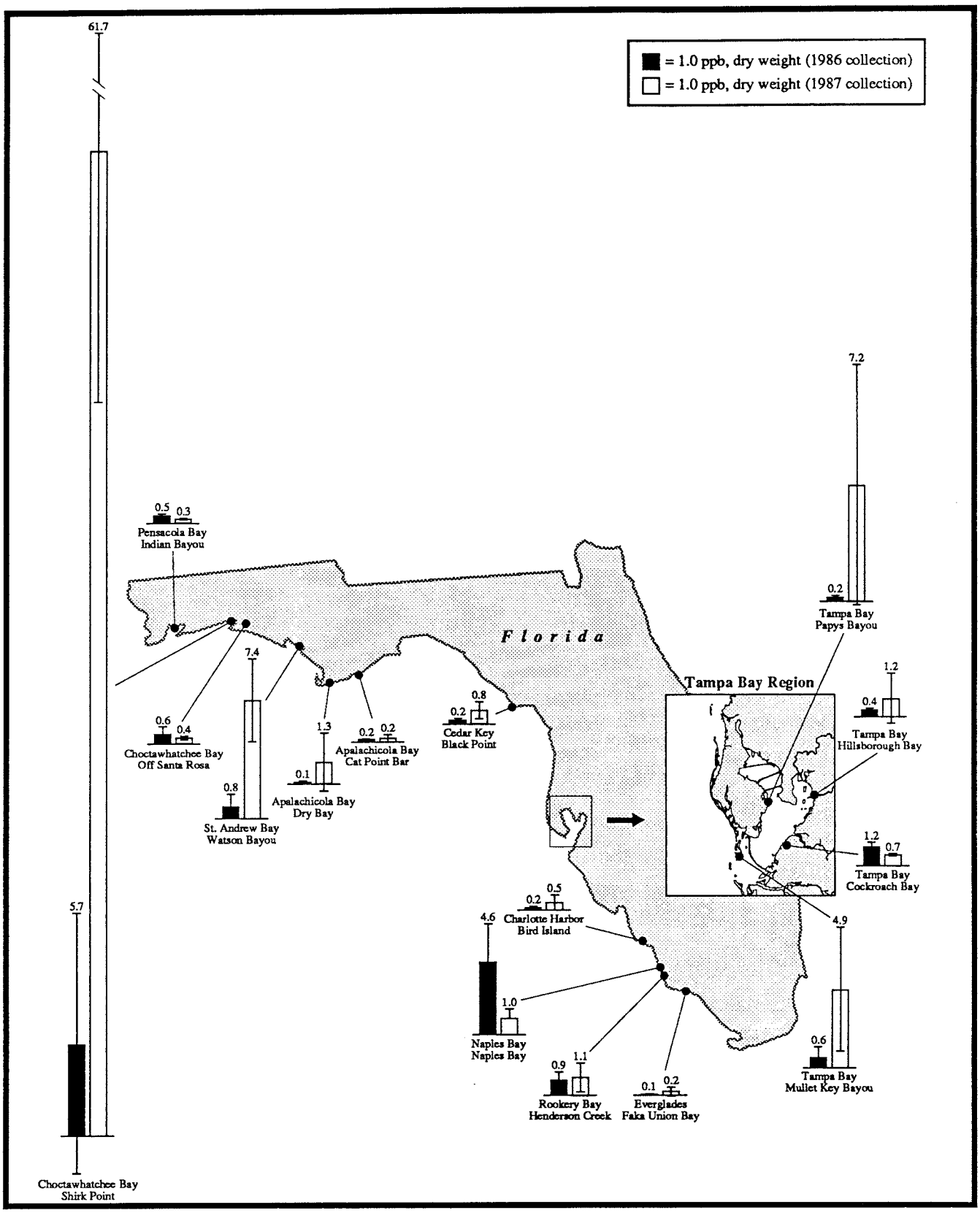

Figure 18. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments sampled along the Gulf of Mexico coast of Florida, 1986 and 1987. Source: NS&T Mussel Watch Project.

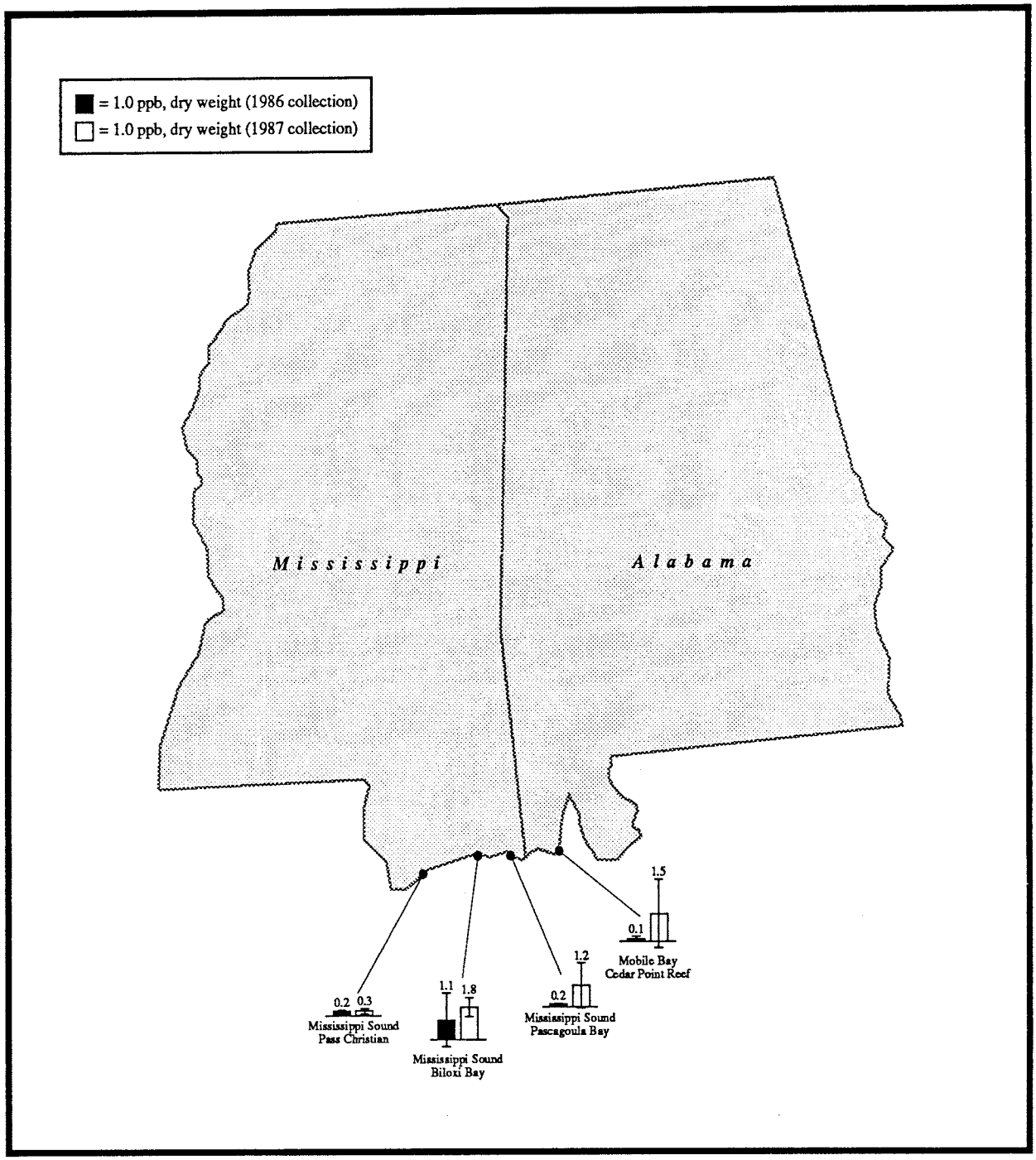

Figure 19. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments along the coasts of Mississippi and Alabama, 1986 and 1987. Source: NS&T Mussel Watch Project.

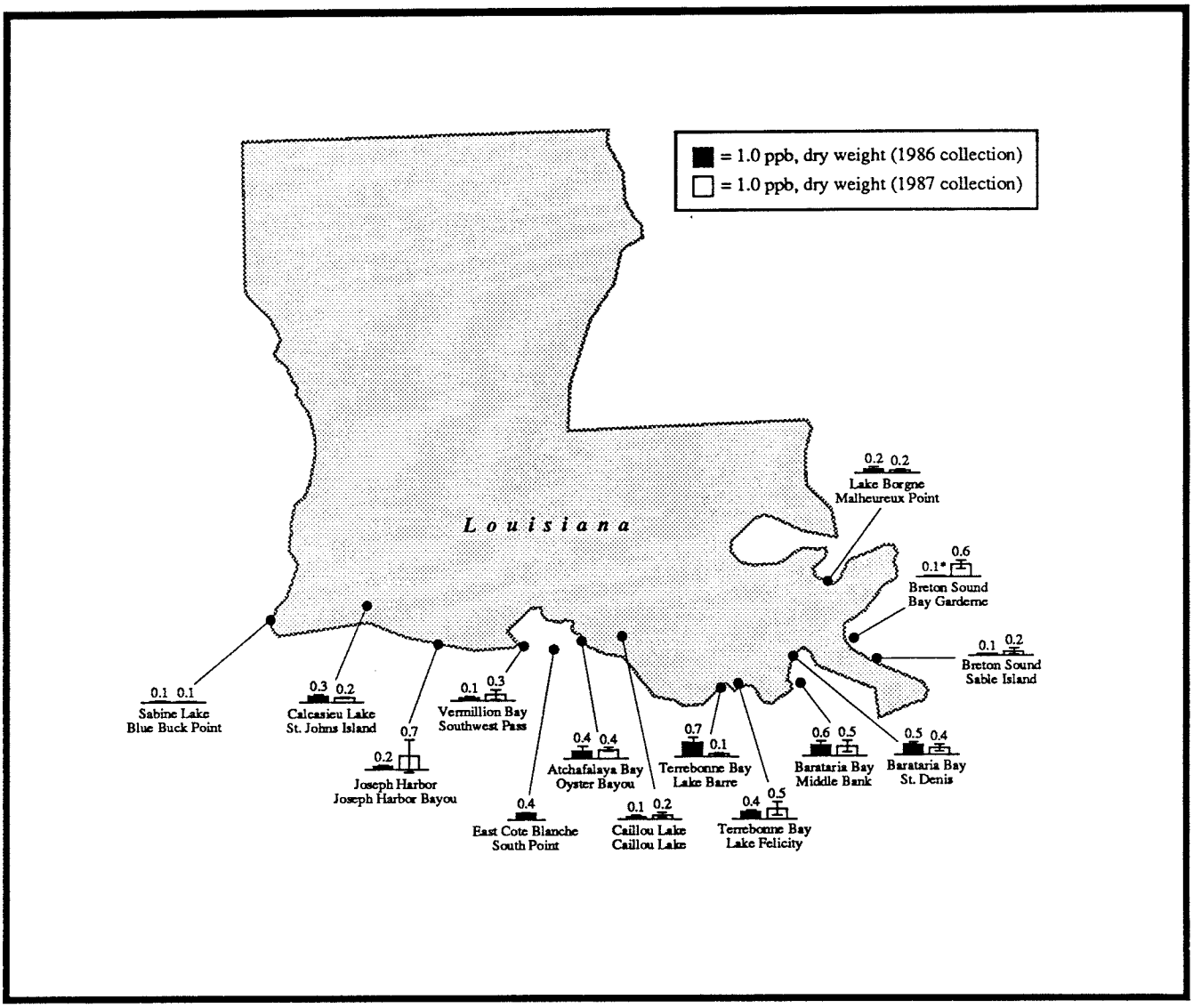

Figure 20. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments along the coast of Louisiana, 1986 and 1987. Asterisk indicates all results below detection. Source: NS&T Mussel Watch Project.

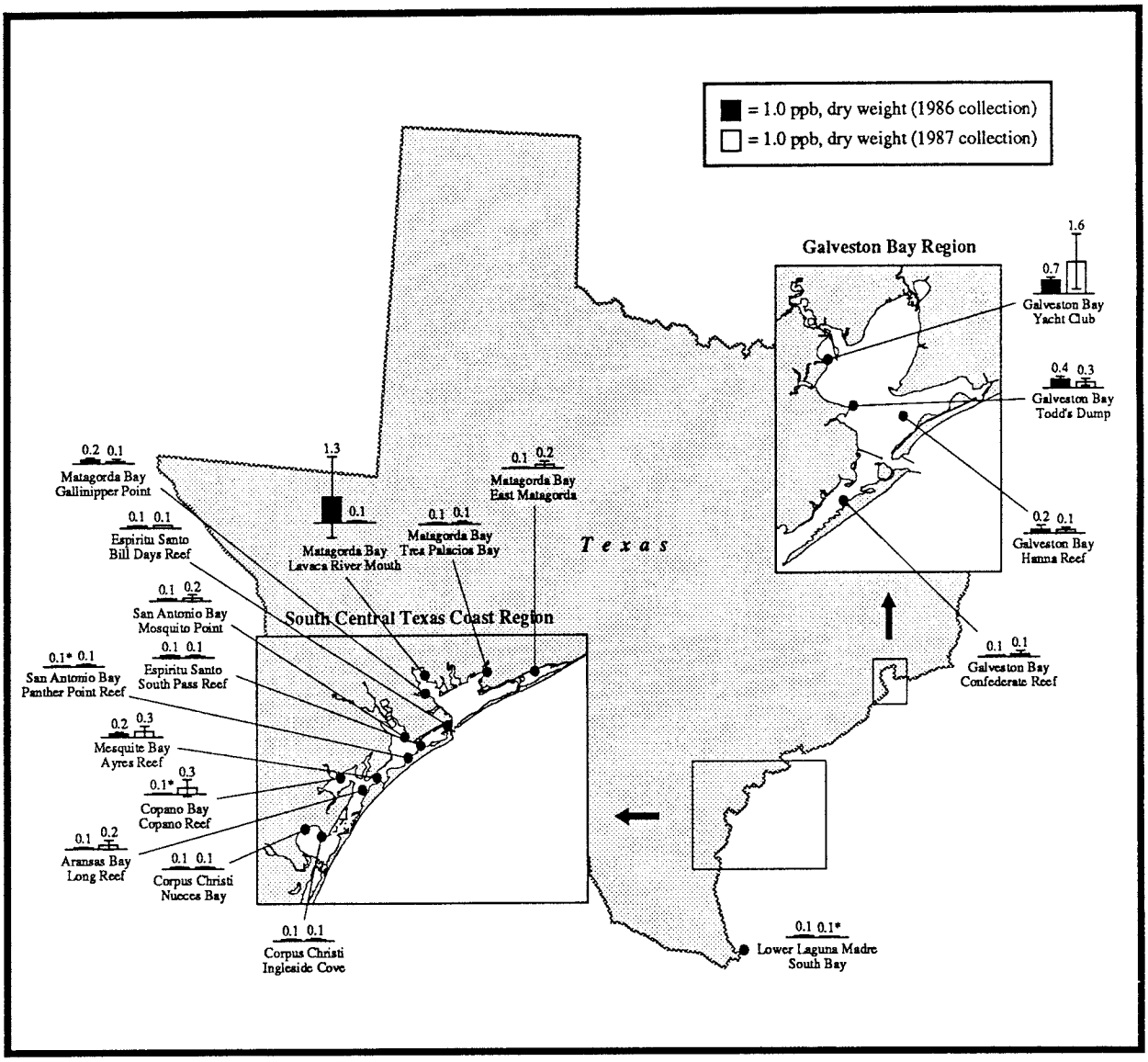

Figure 21. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments along the coast of Texas, 1986 and 1987. Asterisk indicates all results below detection. Source: NS&T Mussel Watch Project.

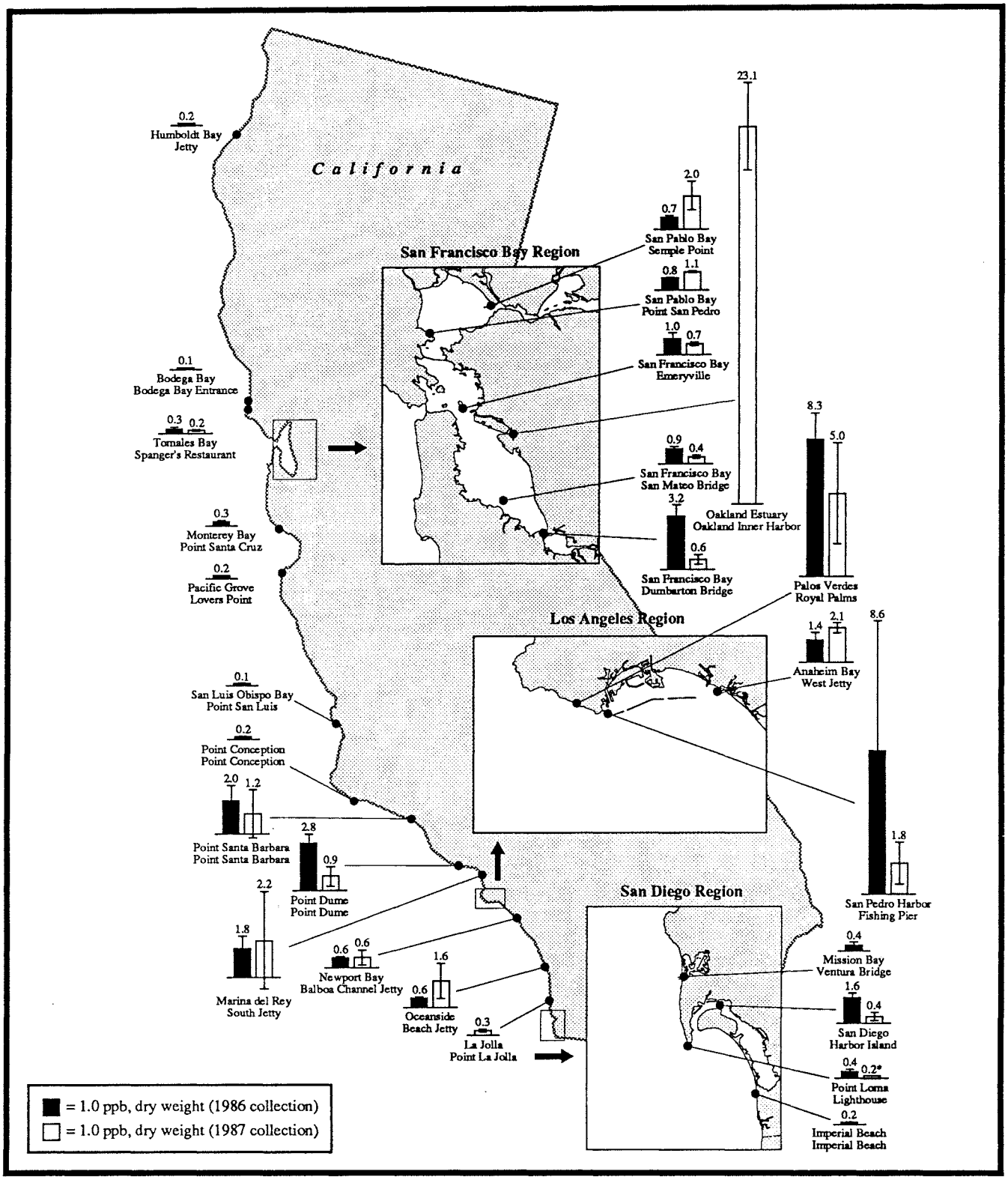

Figure 22. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments along the coast of California, 1986 and 1987. Asterisk indicates all results below detection. Source: NS&T Mussel Watch Project.

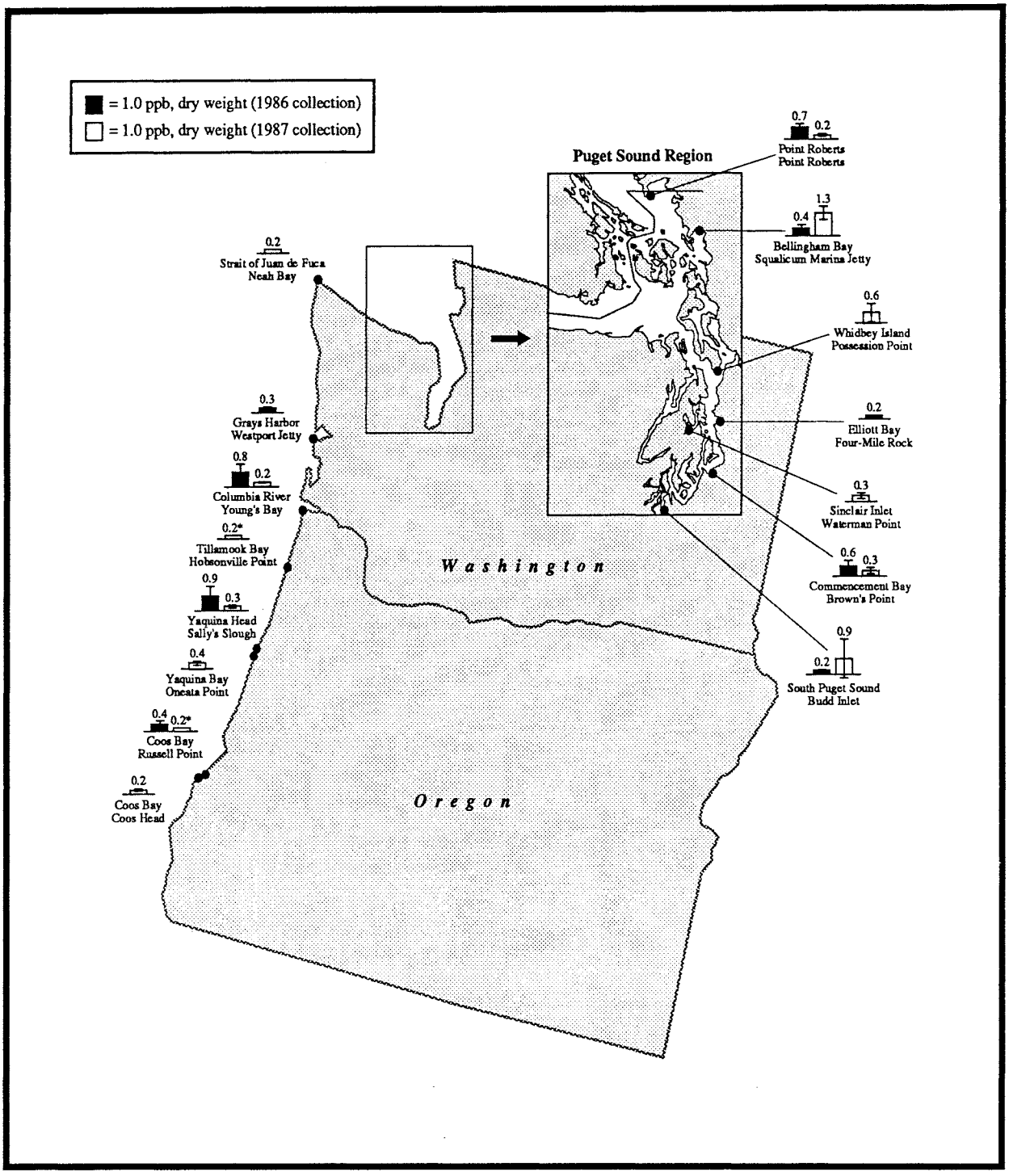

Figure 23. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments along the coasts of Oregon and Washington, 1986 and 1987. Asterisk indicates all results below detection. Source: NS&T Mussel Watch Project.

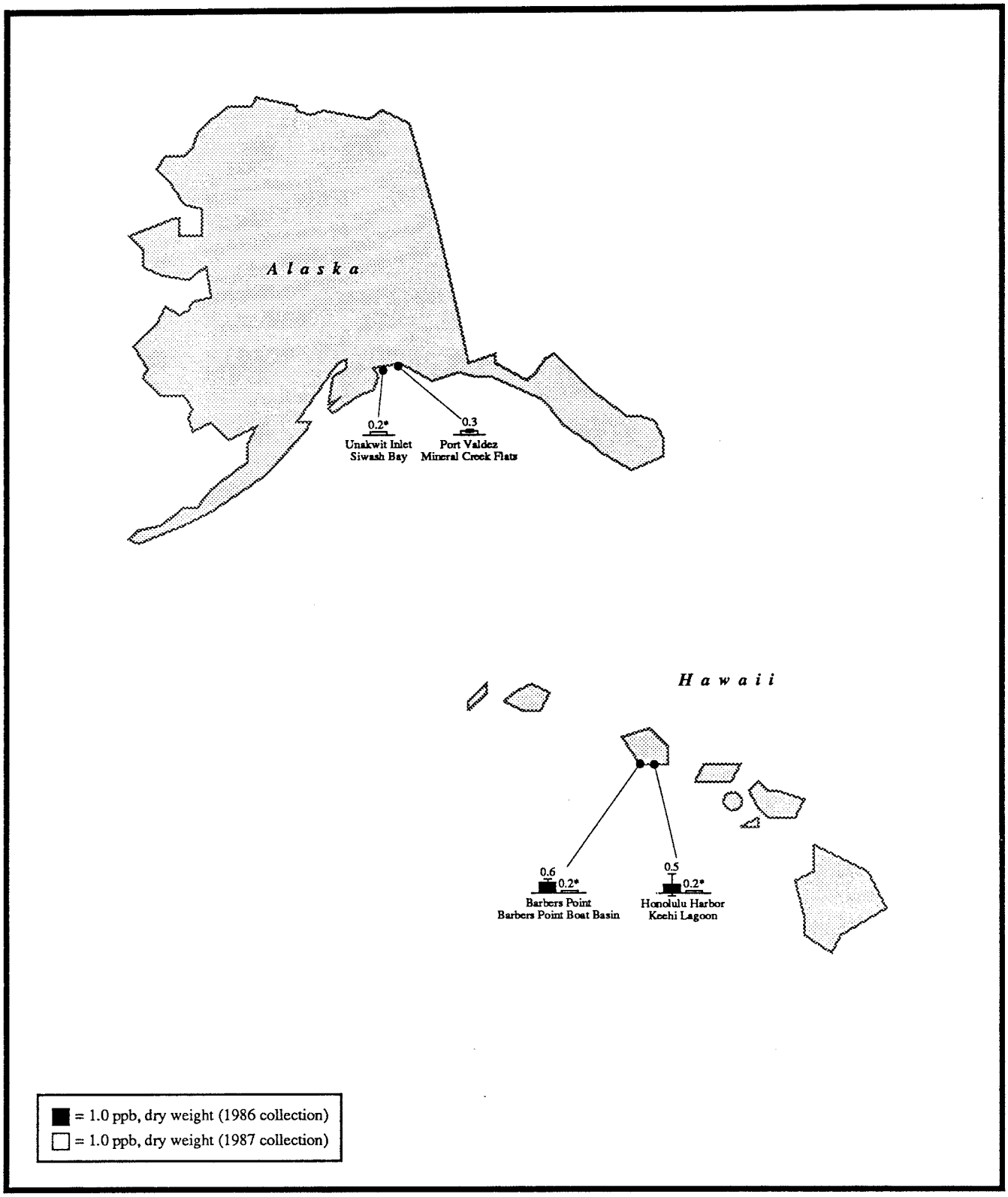

Figure 24. NS&T Mussel Watch Project results for summed concentrations of chlordane in sediments along the coasts of Alaska and Hawaii, 1986 and 1987. Asterisk indicates all results below detection. Source: NS&T Mussel Watch Project.

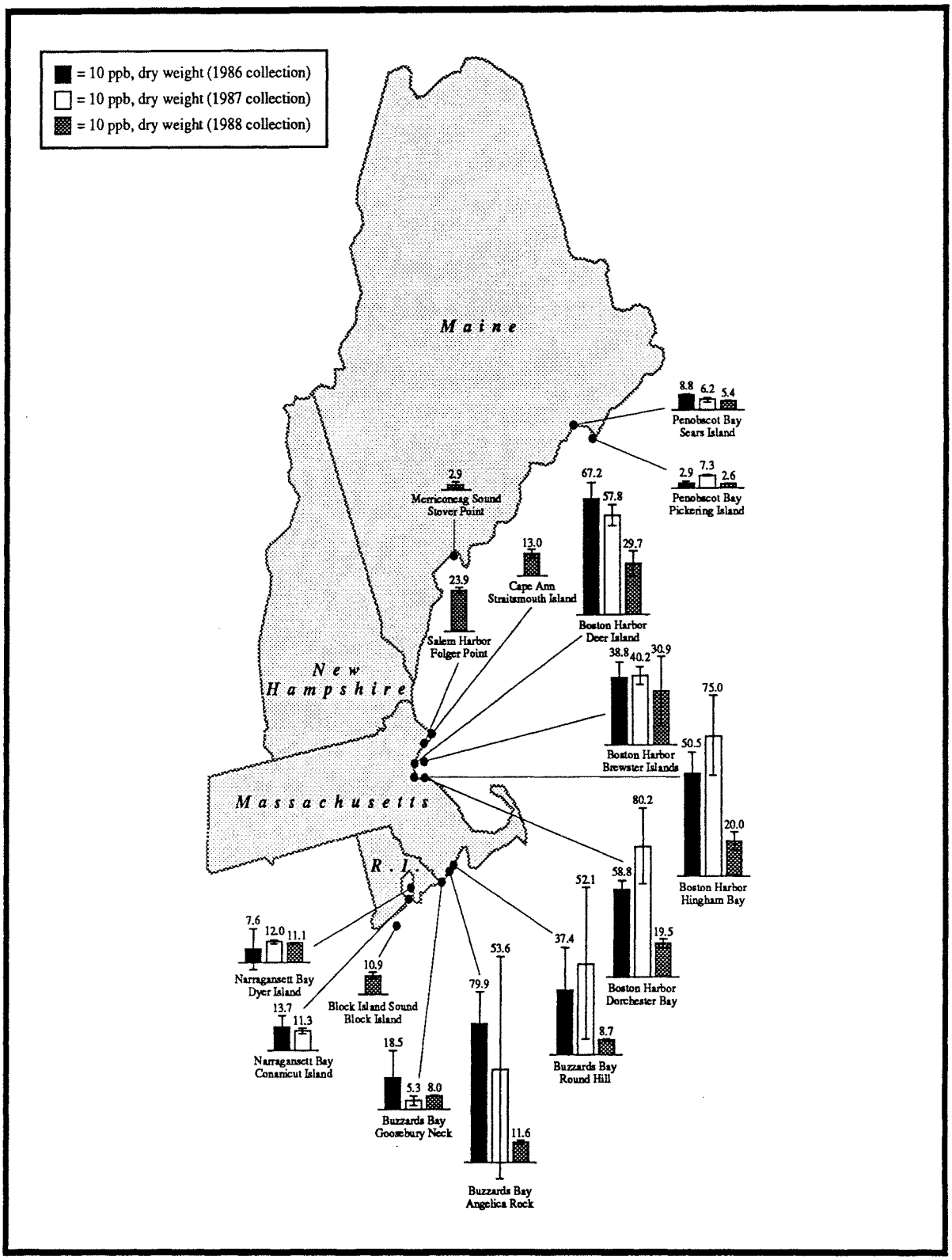

Figure 25. NS&T Mussel Watch Project results for summed concentrations of chlordane in *M. edulis* from Maine to Rhode Island, 1986-1988. Source: NS&T Mussel Watch Project.

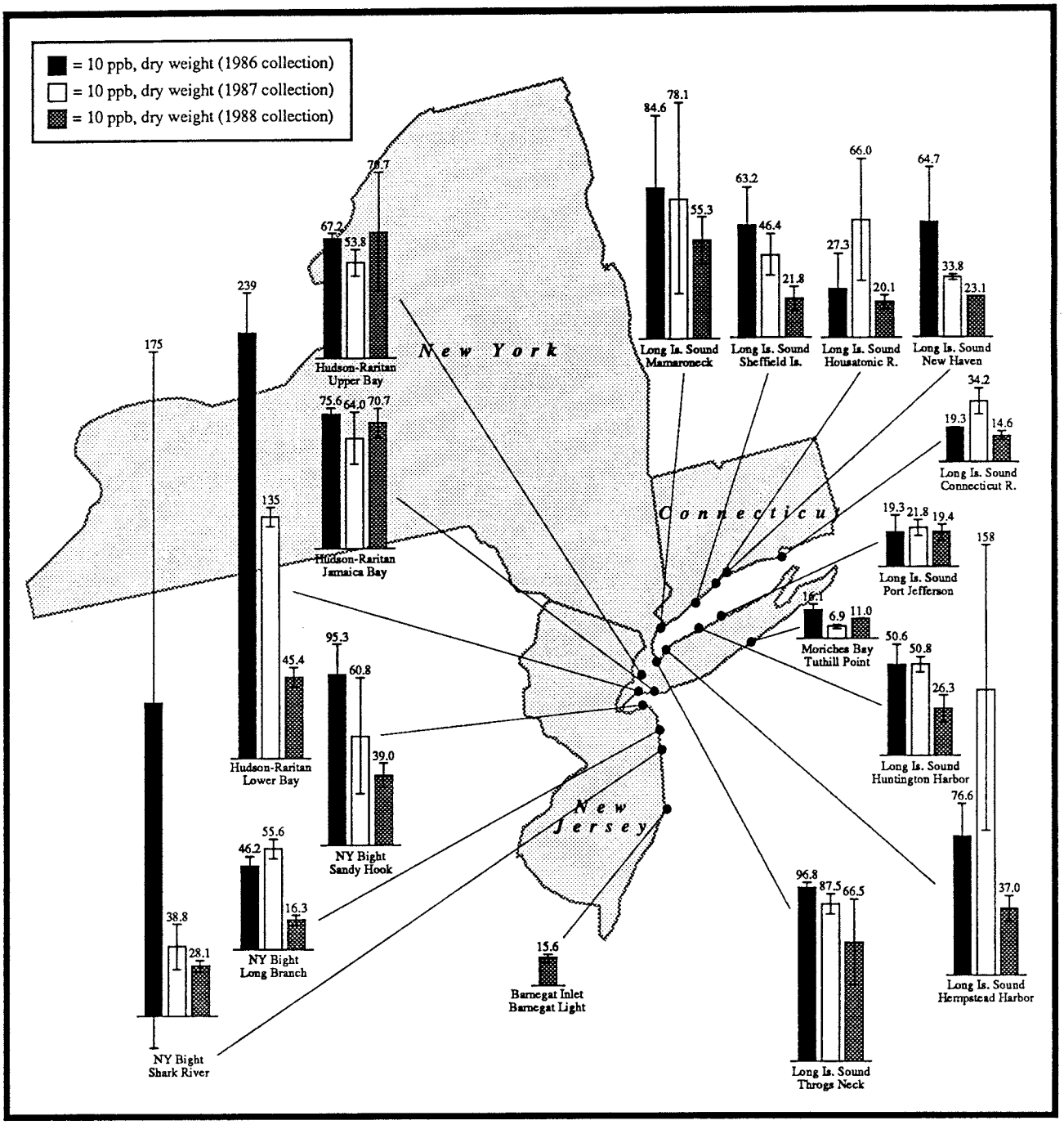

Figure 26. NS&T Mussel Watch Project results for summed concentrations of chlordane in *M. edulis* from Connecticut to New Jersey, 1986-1988. Source: NS&T Mussel Watch Project.

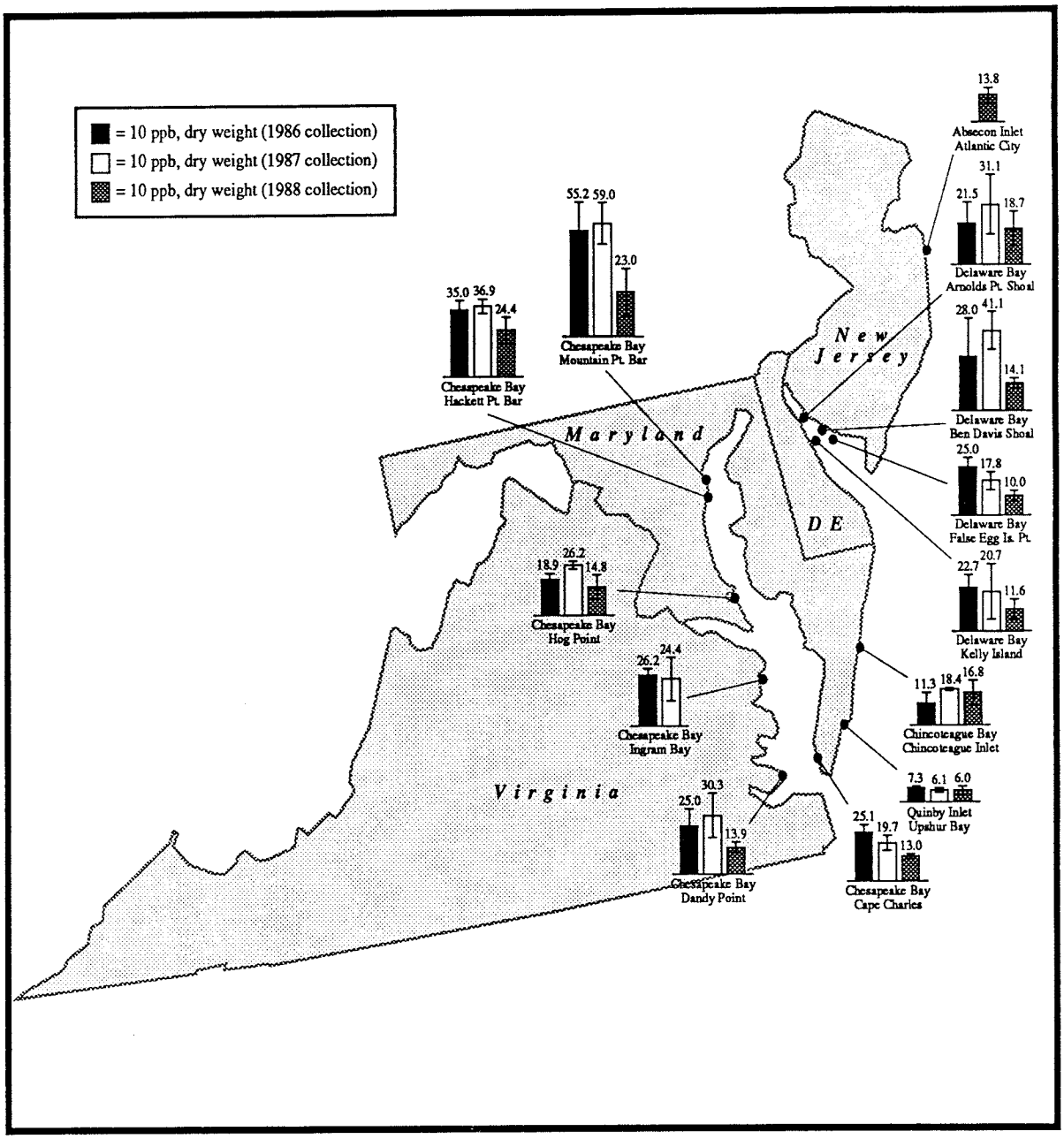

Figure 27. NS&T Mussel Watch Project results for summed concentrations of chlordane in *M. edulis* from New Jersey to Virginia, 1986-1988. Source: NS&T Mussel Watch Project.

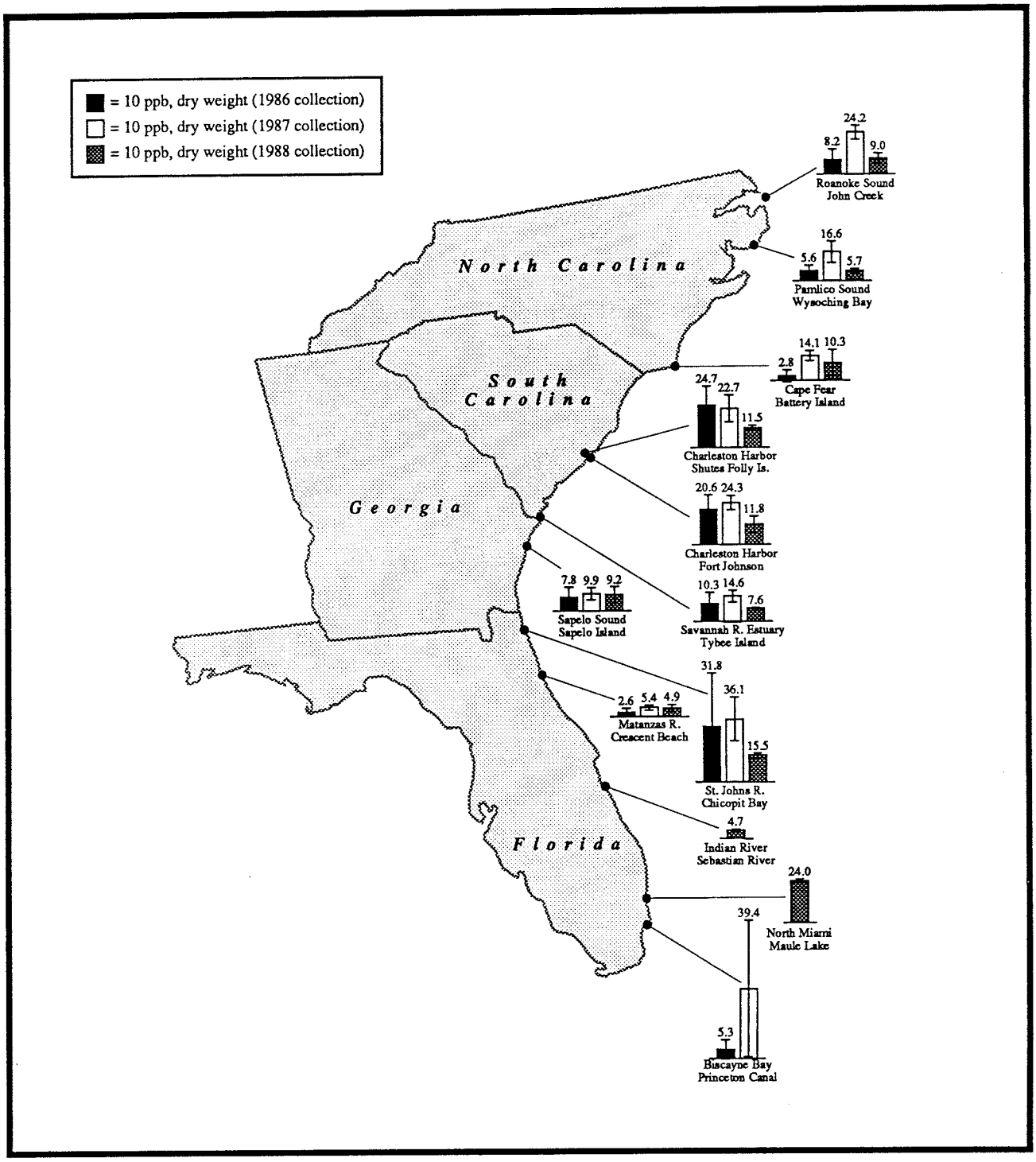

Figure 28. NS&T Mussel Watch Project results for summed concentrations of chlordane in *C. virginica* sampled on the Atlantic coast from North Carolina to Florida, 1986-1988. Source: NS&T Mussel Watch Project.

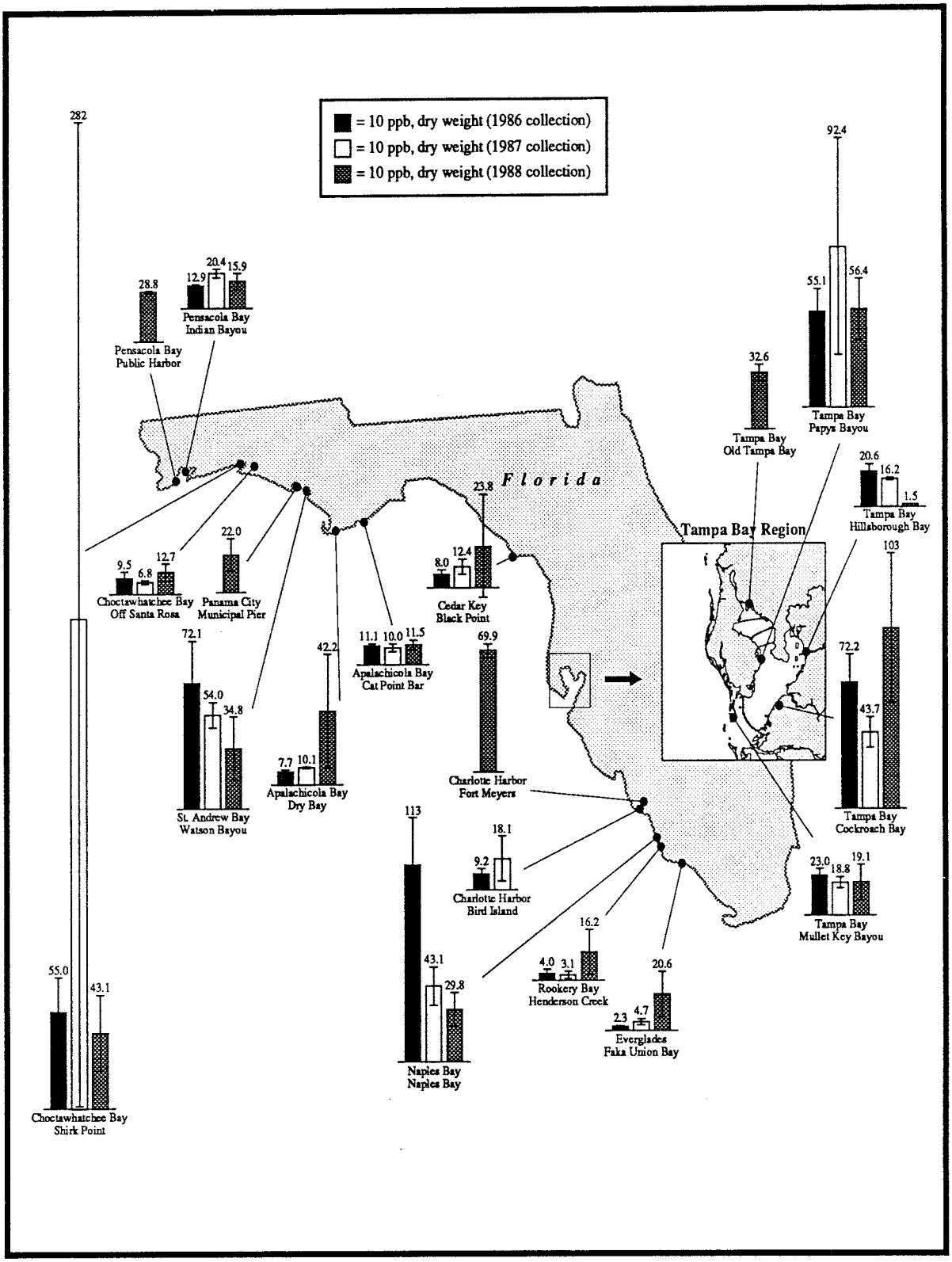

Figure 29. NS&T Mussel Watch Project results for summed concentrations of chlordane in *C. virginica* sampled on the Gulf of Mexico coast of Florida, 1986-1988. Source: NS&T Mussel Watch Project.

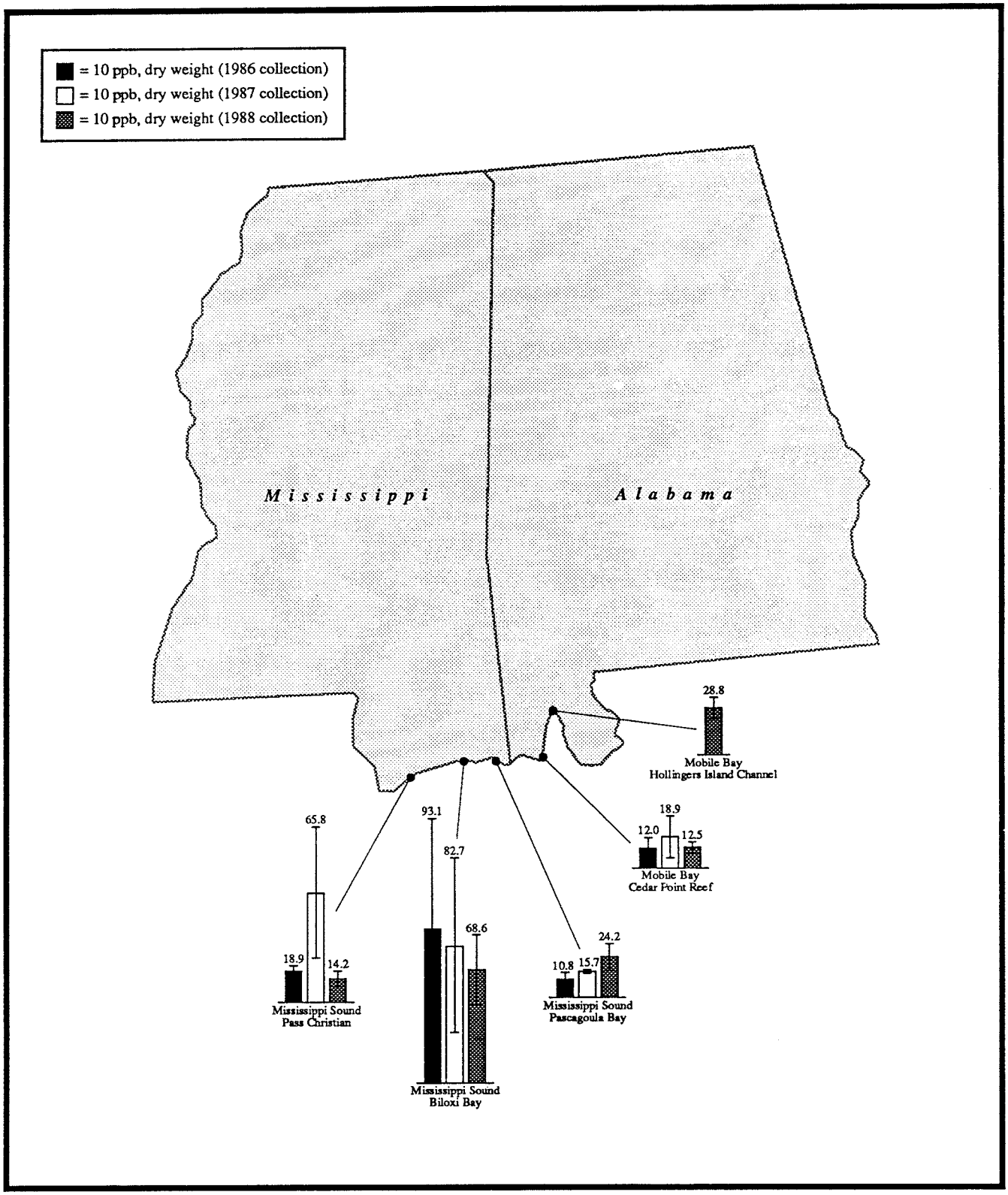

Figure 30. NS&T Mussel Watch Project results for summed concentrations of chlordane in *C. virginica* sampled on the Mississippi-Alabama coast, 1986-1988. Source: NS&T Mussel Watch Project.

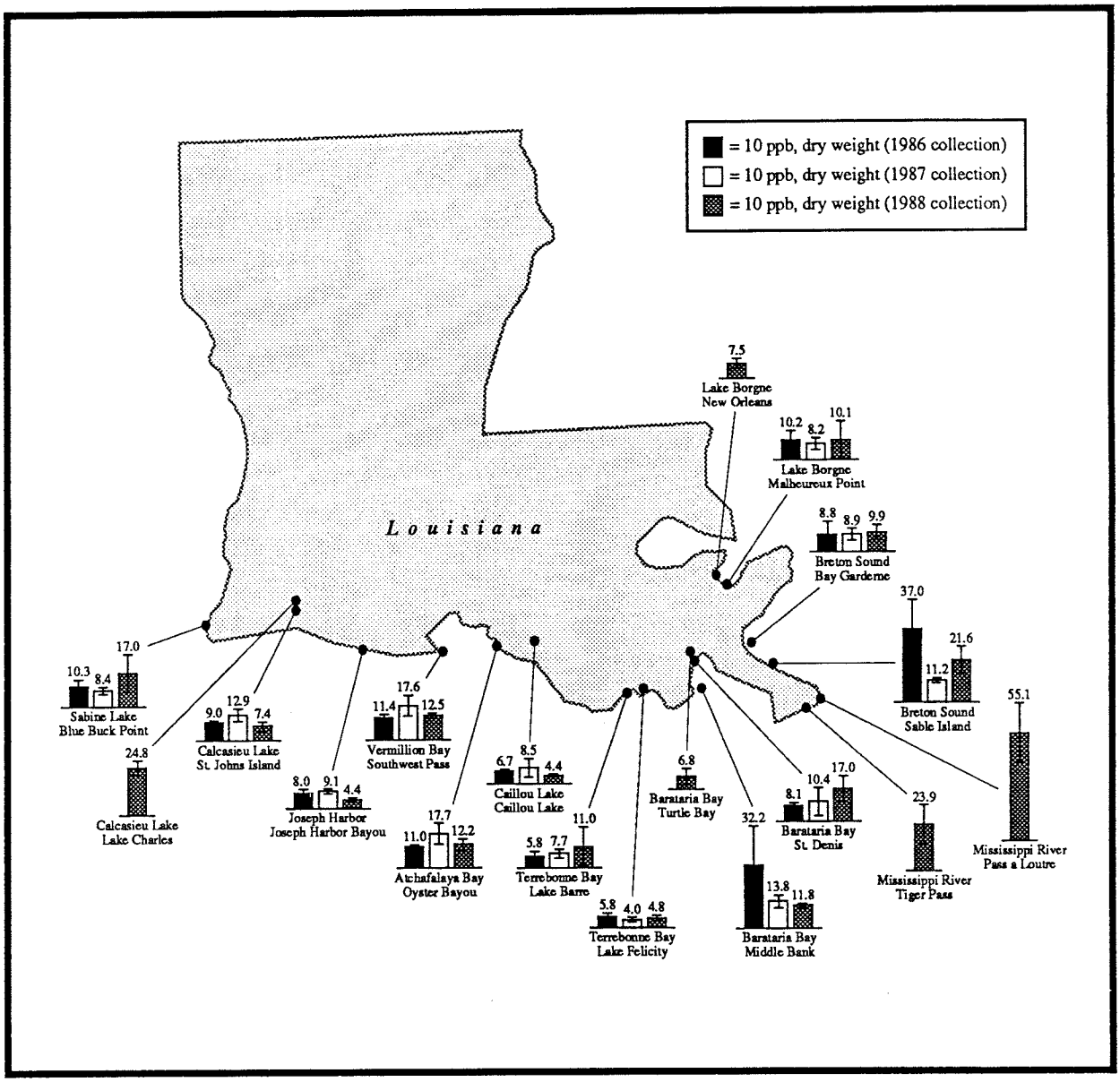

Figure 31. NS&T Mussel Watch Project results for summed concentrations of chlordane in *C. virginica* sampled on the Louisiana coast, 1986-1988. Source: NS&T Mussel Watch Project.

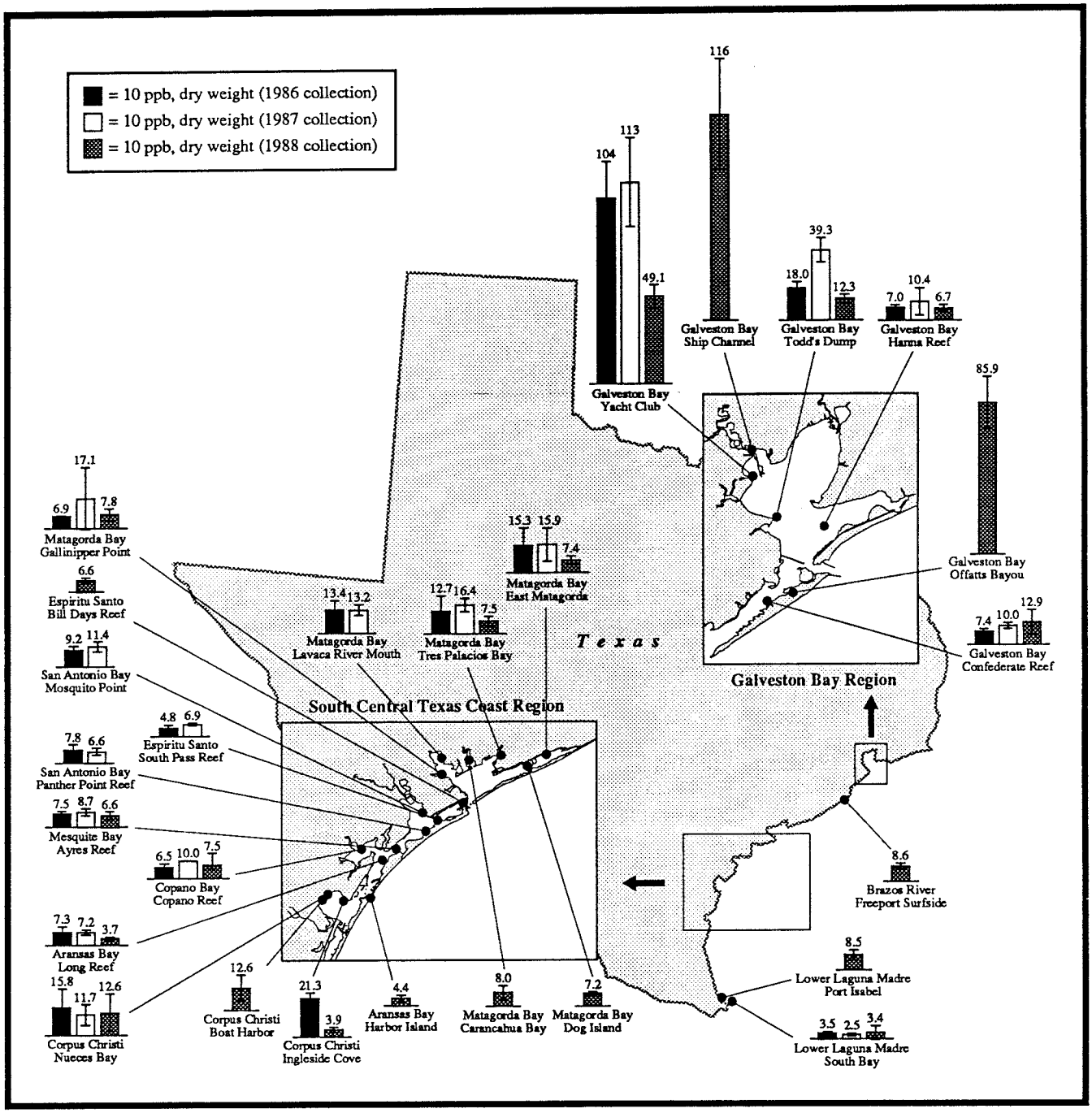

Figure 32. NS&T Mussel Watch Project results for summed concentrations of chlordane in *C. virginica* sampled on the Texas coast, 1986-1988. Source: NS&T Mussel Watch Project.

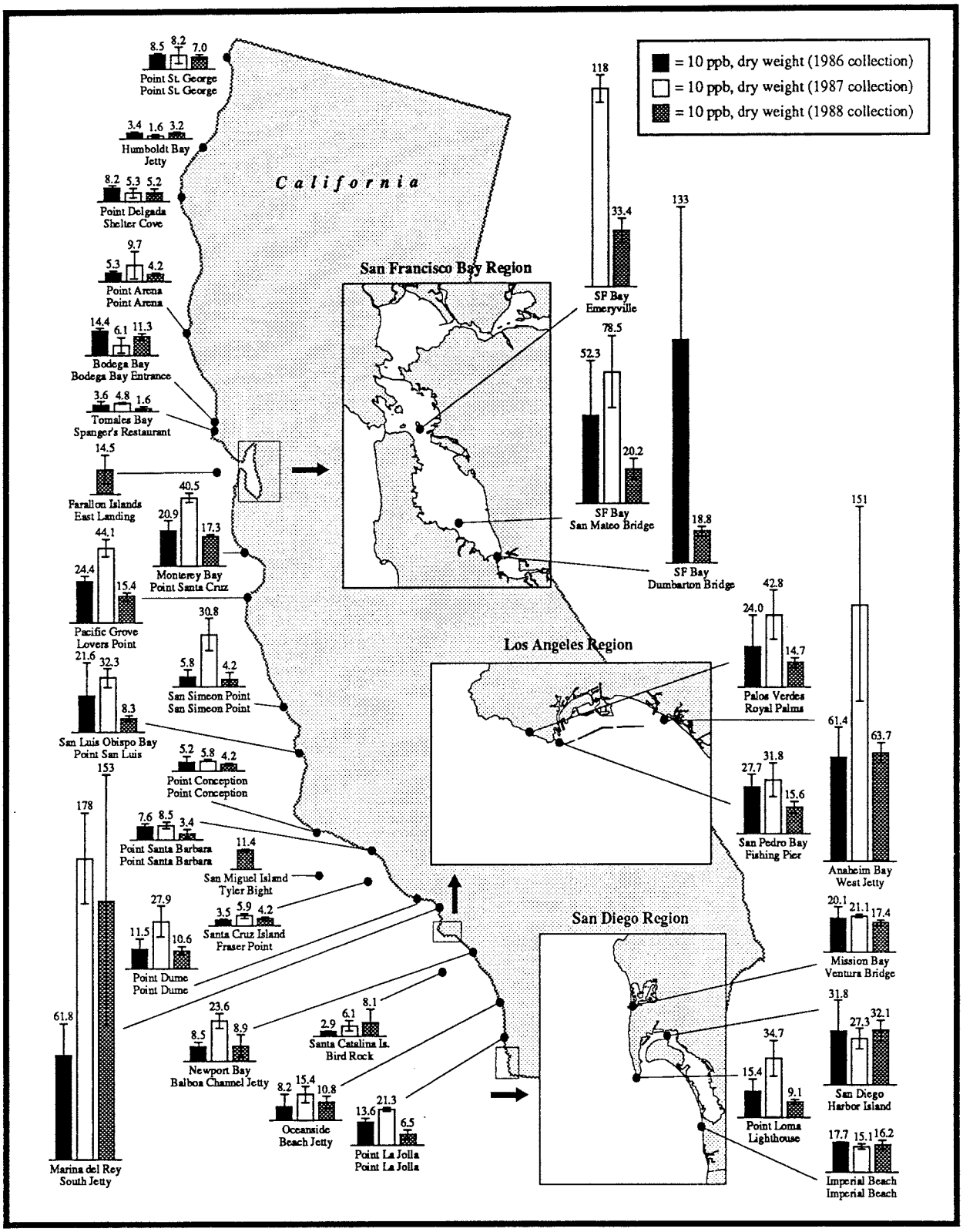

Figure 33. NS&T Mussel Watch Project results for summed concentrations of chlordane in *M. edulis* and *M. californianus* sampled on the California coast, 1986-1988. Source: NS&T Mussel Watch Project.

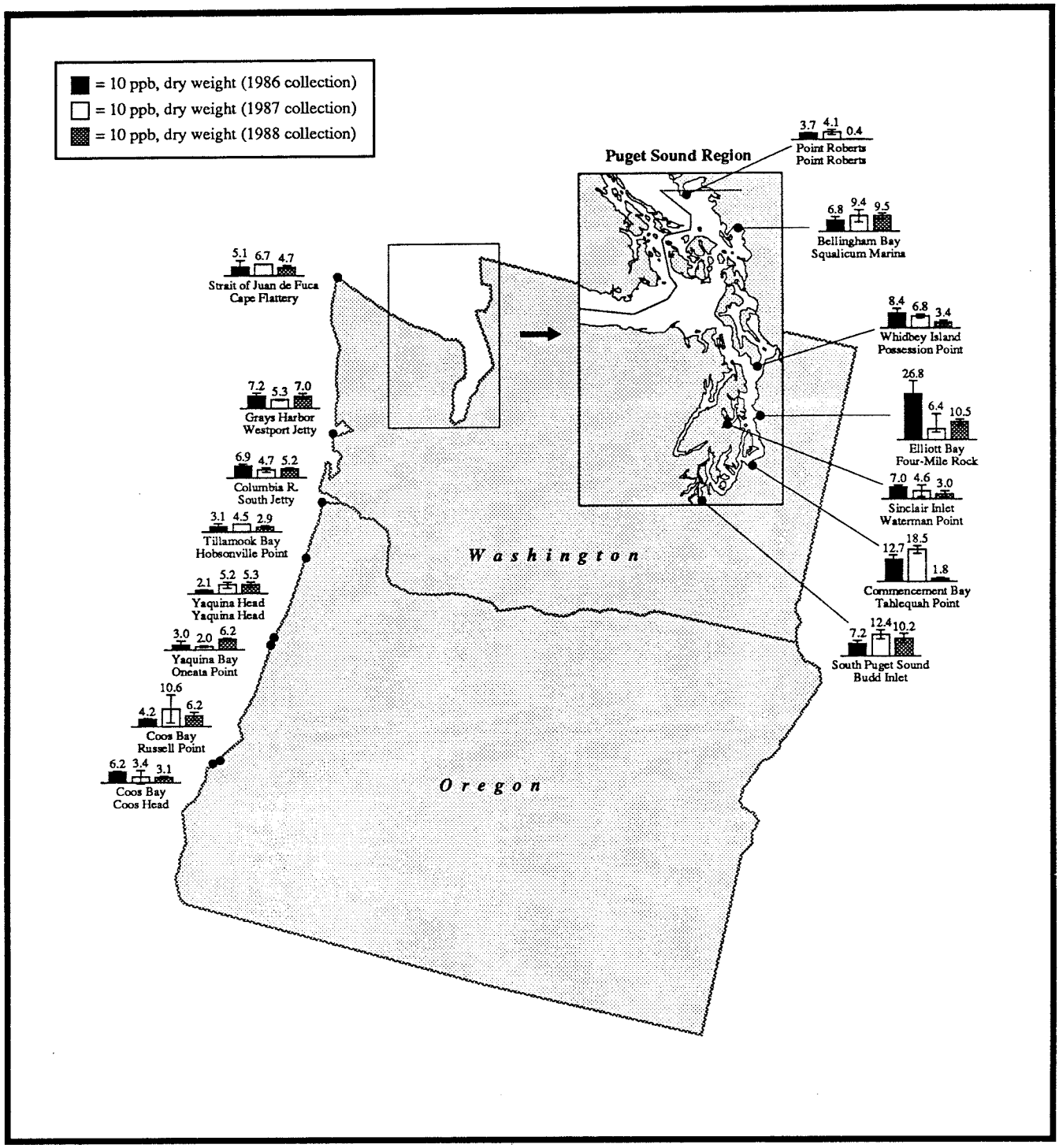

Figure 34. NS&T Mussel Watch Project results for summed concentrations of chlordane in *M. edulis* and *M. californianus* sampled along the Oregon and Washington coasts, 1986-1988. Source: NS&T Mussel Watch Project.

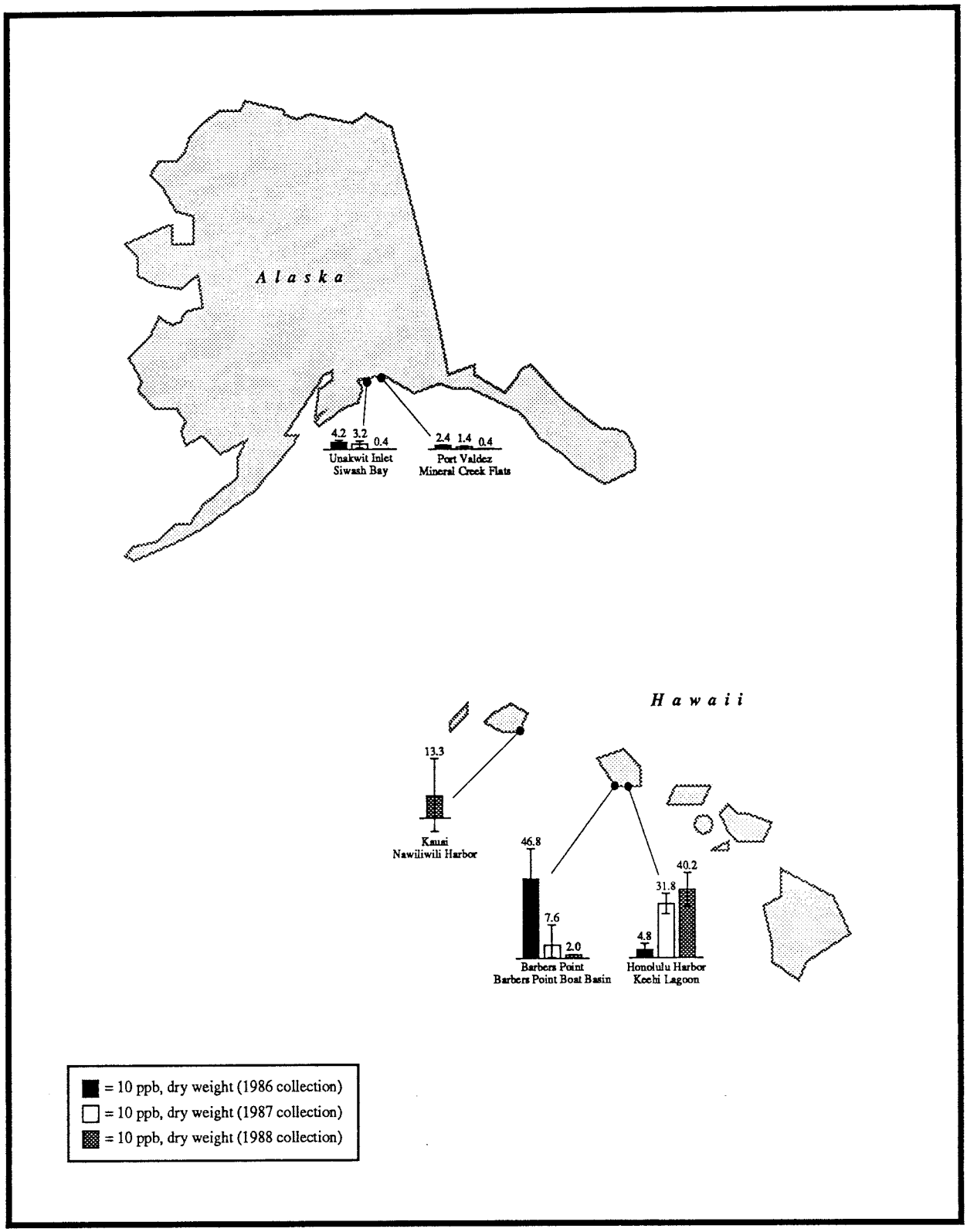

Figure 35 NS&T Mussel Watch Project results for summed concentrations of chlordane in *M. edulis*, *M. californianus* , and *O. sandvicensis* sampled along the coasts of Alaska and Hawaii, 1986-1988. Source: NS&T Mussel Watch Project.

Generally speaking, highest concentrations of summed chlordane compounds in both sediments and bivalves were found in heavily populated urban regions, as opposed to agricultural areas adjacent to the coast. This is consistent both with patterns of use discussed previously in this report and the history of agricultural use restrictions that have been in place since around 1975. That is, not only would most U.S. applications of chlordane in the past 15 years be expected to be for structural protection, the largest agricultural use for chlordane in the preceding period (about 20% of the total estimated use in 1974) was for protection of midwestern corn crops. Agricultural applications of chlordane, which might have resulted in significant and persistent residues in the marine environment had they been employed for protection of crops growing closer to coastal and estuarine regions, apparently were much less important relative to residential, commercial, and industrial applications. While NS&T Mussel Watch results show chlordane to be a nearly ubiquitous contaminant of the U.S. coastal and estuarine environment, it is clearly one linked to heavily populated regions of the coast.

An unusual finding of the Mussel Watch Project were the high concentrations of summed chlordane compounds measured in Choctawhatchee Bay, particularly in 1987. In both sediments and in bivalve tissue, the concentrations found at this location were the highest measured by the project. Given the general characterization of coastal environmental chlordane residues as correlating with urban population centers, these substantial concentrations found in Choctawhatchee Bay, which is not adjacent to a major city, are difficult to explain. Eglin Air Force Base borders most of the northern coastline of the bay, but there are no indications that this might have been the source of the elevated chlordane levels (It should be noted that other chlorinated pesticides like DDT were also found to be anomalously high in Choctawhatchee Bay.)

Figures 15 through 35 suggest that sites with elevated sediment concentrations of chlordane also have resident bivalves with relatively higher body burdens. This was tested using Spearmans rank correlation coefficient. Site mean data for 1986 and 1987 (the sampling cycles for which sediment and bivalve measurements were available for most sites) were combined, and each target species was evaluated separately. Results of the Spearmans rank procedure are shown as Table 36.

Table 36. Summary of Spearmans rank correlation analysis performed on sediment and bivalve results from 1986 and 1987 NS&T Mussel Watch samples, Source: NS&T Mussel Watch Project.

Species	N	r_s	Significance
Mytilus edulis	80	0.593	0.0001
Crassostrea virginica	132	0.521	0.0001
Mytilus californianus	25	0.263	0.1982
Ostrea sandvicensis	4	0.316	0.5839

The analyses show that sediment concentrations of summed chlordane compounds and bivalve concentrations were significantly correlated at rigorous levels of significance at those Mussel Watch sites where *M. edulis* and *C. virginica* were sampled. Sites along the Pacific where *M. californianus* and *O. sandvicensis* were sampled did not show this correlation.

In order to evaluate the percent contribution of the three constituent compounds (α–chlordane, *trans*-nonachlor, and heptachlor) to the summed chlordane values, results for 1986 through 1988 were grouped by species and averaged. Figure 36 shows the mean percent chlordane residue composition and the number of analyses for each species. For three of the four species (*M. edulis*, *M. californianus*, and *C. virginica*), α–chlordane represented the greatest proportion of the summed chlordane value, followed by *trans*-nonachlor, and then heptachlor. In the case of *O. sandvicensis*, the oyster species sampled at Hawaiian sites, *trans*-nonachlor ranked first proportionally, followed by α–chlordane and heptachlor. The percent composition of heptachlor was approximately two to three times higher in *O. sandvicensis* than in the other three species.

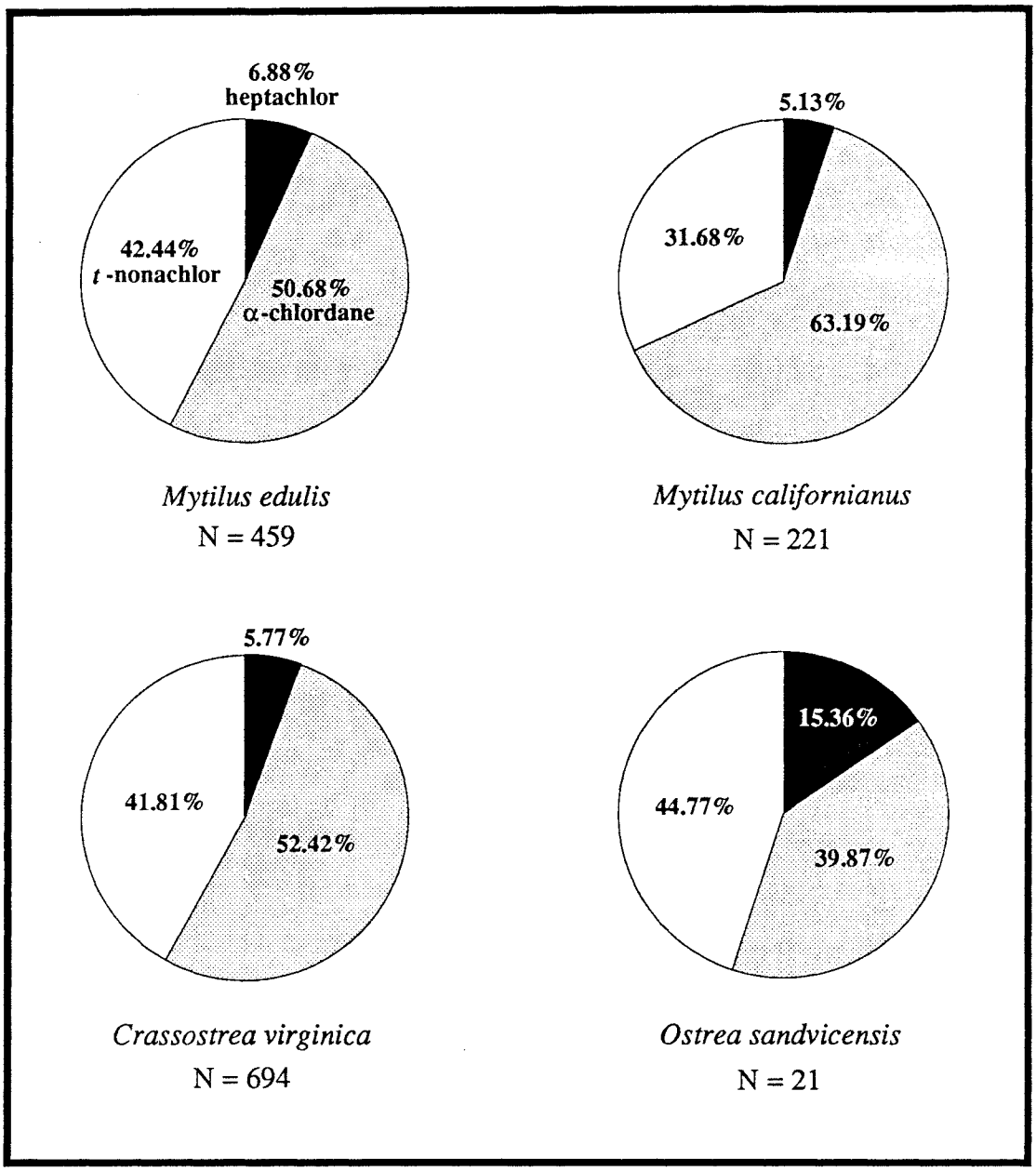

Figure 36. Mean percent chlordane residue composition by species of mussel, Source: NS&T Mussel Watch Project.

These differences may result from a number of considerations, including species uptake differences, variations in pesticide use patterns, or differences in environmental degradation processes between temperate and tropical waters.

As shown in Table 1, *trans*-nonachlor is a relatively minor constituent (approximately 7%) of technical chlordane. It proportionally greater prominence as a measured residue in bivalves suggests that other constituents are metabolized much more rapidly, and/or that environmental concentrations of *trans*-nonachlor increase as other chlordane compounds are metabolized or degraded (*i.e.*, *trans*-nonachlor is a breakdown product of other chlordane compounds). The observed differences between

rations of constituents in the technical formulation originally introduced into the environment and those found as residues offer a potential means of defining recent or illegal use of chlordane (R. Sloan, NYDEC, pers. comm., 24 May 1990). Clearly, however, additional and more refined information on metabolic pathways and rates would be helpful in making these kinds of assessments.

As previously noted, results from the NPMP estuarine bivalve analyses undertaken in the 1960s and 1970s indicated that chlordane was not present at levels above 10 ppb wet weight in any of nearly 8,300 samples of several species (Butler, 1973; Butler, Kennedy, and Schutzmann, 1978). Keeping in mind that NPMP results were reported as wet weight concentrations and NS&T Mussel Watch as dry weight, how do the results compare? Table 37 summarizes NS&T data for percent dry weight for each of the four bivalve species analyzed in the project.

Table 37. Summary of percent dry weight by species for four bivalve species collected in the NS&T Mussel Watch Project, 1986-1988. Source: NS&T Mussel Watch Project.

Species	N	Mean % dry weight	Standard deviation
Mytilus edulis	312	13.27	4.39
Crassostrea virginica	484	14.76	5.70
Mytilus californianus	207	14.97	4.23
Ostrea sandvicensis	21	15.30	3.57
All species	**1024**	**14.36**	**5.06**

Based on the mean percent dry weight value of 14.4 percent across all species sampled in NS&T Mussel Watch, an approximate dry weight equivalent for the 10 ppb wet weight NPMP detection limit can be calculated at 69.6 ppb. Although NPMP analyses would have measured technical chlordane as an undifferentiated quantity, NS&T Mussel Watch results include major constituents of the technical chlordane mix and the summed total could be considered to be roughly comparable. Using the dry weight detection limit of 69.6 ppb derived from the NPMP analyses, it was found that 103 of 1,396 NS&T Mussel Watch samples from 1986 to 1988 (7.38%) were measured at levels above NPMP detection limits. Therefore, while the NPMP findings of no measurable chlordane in estuarine bivalves were not impossible, it would have seemed likely that at least a few of the large numbers of bivalves sampled would have had some detectable residues. The period of sample collection for NPMP also would argue for measurable chlordane concentrations, as the compound was being used at that time as a pesticide in several kinds of applications. It is possible that unavailability of analytical standards or interference from other chlorinated compounds prevented identification or quantification of chlordane compounds. The comments of Cardwell *et al.* (1977), who postulated that less sensitive analytical methods or interference from DDT and its metabolites may have obscured residue measurements in another study where chlordane was not detected (Duke and Wilson, 1971), are equally applicable here as well.

National Status and Trends Program Benthic Surveillance Project, 1984-. An assessment and monitoring study complementary to the NS&T Mussel Watch Project was begun by NOAA in 1984. Known as the Benthic Surveillance Project, this effort has annually collected sediments and bottom-feeding fish from approximately 50 sites in coastal and estuarine regions around the country during the spring and summer. Tissues and sediments are subsequently analyzed by chemists of the National Marine Fisheries Service (NMFS) to provide annual datasets similar in nature to those of the NS&T Mussel Watch Project. In addition, results for the food items found in the stomachs of sampled fish have been generated. Bottom-feeding fish are expected to provide an integrative substrate for chemical contamination of a region, as they would be expected to range over a more extensive area than sedentary organisms such as bivalves. However, one individual bottom-feeding species is not indigenous to all coastal regions of the United States, dictating that several species be sampled in order to provide complete coverage around the country. In 1984 and 1985, the following species were sampled for the Benthic Surveillance Project: Longhorn sculpin, *Myoxocephalus octodecemspinosus*; winter flounder, *Pseudopleuronectes americanus*; windowpane flounder, *Scophthalmus aquosus*; Atlantic croaker, *Micropogon undulatus*; spot, *Leiostomus xanthurus*;

barred sand bass, *Paralabrax nebulifer*; white croaker, *Genyonemus lineatus*; diamond turbot, *Hypsopsetta guttulata*; hornyhead turbot, *Pleuronichthys verticalis*; starry flounder, *Platichthys stellatus*; English sole, *Parophrys vetulus*; flathead sole, *Hippoglossoides elassodon*; and four-horn sculpin, *Myoxocephalus quadricornis*. Because sediments are collected in association with the fish, the kinds of environments sampled may differ qualitatively from those included in the Mussel Watch Project (*i.e.*, potentially deeper water, farther from adjacent land masses). The Benthic Surveillance Project began in 1984, with collections made each year to date. The project is structured to allow a certain degree of flexibility in design elements so that additional chemicals, species, or more intensive sampling plans may be incorporated according to results obtained in previous cycles or from independent sources.

Complete sets of Benthic Surveillance results are available for the first two cycles of the project, 1984 and 1985. As was the case for the NS&T Mussel Watch Project, three constituents of technical chlordane were included in the suite of chemicals analyzed in the Benthic Surveillance Project: α-chlordane, *trans*-nonachlor, and heptachlor. Summary data for the two years are listed in Tables 38 and 39. For the purposes of summary and statistical analysis, those concentrations below detection were converted to one-half the listed limits; in some cases, where detection limits were not reported by the analytical facility (*i.e.*, "0"), one-half the lowest detection limit for that compound, substrate, and year was substituted. Site means for summed concentrations of the three compounds for the 2 years are plotted on maps of each coast and shown as Figures 37 through 39 (sediment) and Figures 40 through 42 (bottomfish liver tissue, by species).

Table 38. Summaries of summed concentrations of three chlordane compounds (Σ α-chlordane, *trans*-nonachlor, and heptachlor) in sediments collected in the coastal and estuarine environment of the United States, 1984-1985. Values in ppb dry weight. Source: NS&T Benthic Surveillance Project.

Year	N	Mean± Sdev	Min*	Min Location	Max	Max Location	Median
1984	152	1.27±2.76	0.03	E. Long Is. Sound	23.4	Salem Harbor, MA	0.242
1985	193	2.08±5.75	0.035	(several)	46.4	Boston Harbor, MA	0.20

* These minima may reflect summation of converted concentrations below limits of detection. In many cases, value portrayed is sum of three below detected and/or unquantitated conversions.

Table 39. Species summaries of summed concentrations of three chlordane compounds (Σ α-chlordane, *trans*-nonachlor, and heptachlor) in liver tissue of bottomfish collected in the coastal and estuarine environment of the United States, 1984-1985. Values in ppb dry weight. Source: NS&T Benthic Surveillance Project.

Species	N	Mean± Sdev	Min*	Min Location	Max	Max Location	Median
1984							
Atlantic croaker	27	23.4±19.5	0.26	San Antonio Bay, TX	58.5	Lower Chesapeake, VA	15.8
Barred sand bass	4	216±162	81.5	S. San Diego Bay, CA	405	Dana Point, CA	190
Diamond turbot	2	416±21.2	402	S. San Diego Bay, CA	432	S. San Diego Bay, CA	416
English sole	10	70.0±48.8	11.5	Nisqually Reach, WA	166	Commencement Bay, WA	71.2
Flathead sole	8	64.3±40.9	30.5	Lutak Inlet, AK	160	Elliott Bay, WA	51.5
Hornyhead turbot	14	111±70.3	31.5	Dana Point, CA	286	Seal Beach, CA	114
Spot	16	21.4±28.2	0.72	Apalachicola Bay, FL	114	St. Johns River, FL	12.6
Starry flounder	16	117±118	4.00	Coos Bay, OR	362	Hunters Point, CA	94.2
White croaker	11	226±253	4.00	Bodega Bay, CA	813	Seal Beach, CA	157
Windowpane fl.	3	38.1±11.8	25.8	Delaware Bay, DE	49.3	Delaware Bay, DE	39.2
Winter flounder	24	210±299	0.18	Salem Harbor, MA	1063	Boston Harbor, MA	70.9
1985							
Atlantic croaker	37	20.5±18.4	0.11	(2 stations)	95.3	Round Island, MS	16.2
Barred sand bass	7	126±96.3	0.11	Dana Point, CA	226	S. San Diego Bay, CA	163
English sole	17	53.8±54.4	7.50	Nisqually Reach, WA	152	Commencement Bay, WA	18.5
Four-horn sculpin	6	32.6±6.50	24.4	Endicott Fields, AK	42.0	Endicott Fields, AK	31.8
Hornyhead turbot	9	87.1±113	0.75	Outer San Diego Bay	367	Santa Monica Bay, CA	38.0
Longhorn sculpin	9	51.1±6.95	37.1	Machias Bay, ME	60.1	Machias Bay, ME	51.1
Spot	16	41.2±32.7	11.1	Sapelo Sound, GA	112	Upper Chesapeake, MD	14.6
Starry flounder	15	91.4±81.2	5.00	Coos Bay, OR	232	San Pablo Bay, CA	65.0
White croaker	18	267±330	0.65	San Pedro Bay, CA	994	Long Beach, CA	124
Windowpane fl.	3	43.5±11.6	33.2	Delaware Bay, DE	56.1	Delaware Bay, DE	41.2
Winter flounder	30	155±139	19.0	Buzzards Bay, MA	588	Boston Harbor, MA	111

* These minima may reflect summation of converted concentrations below limits of detection. In many cases, value portrayed is sum of three below detected and/or unquantitated conversions.

Although the areal coverage provided by the Benthic Surveillance Project differs from that of the Mussel Watch Project in several ways, ranging from the qualitative aspects of sites discussed previously to simple numbers of sites sampled around the country, the general correspondence of elevated sediment and biotic chlordane concentrations to more heavily populated urban centers remained. Urban embayments in Massachusetts (Salem and Boston) and in California (San Francisco, Los Angeles, and San Diego) yielded both sediments and fish tissues with relatively high concentrations of summed chlordane compounds in 1984 and 1985. The coastal sites sampled in the Gulf of Mexico were surprisingly free of contamination in both years in both matrices. Sites in southeastern Alaska and along the remote northern Alaskan coast showed levels of chlordane contamination in fish liver tissues (32 to 55 ppb dry weight) about equivalent to the highest level measured in the Gulf of Mexico, greater than tissue levels measured in many populous areas where chlordane use and subsequent environmental exposure would be expected to have been much greater.

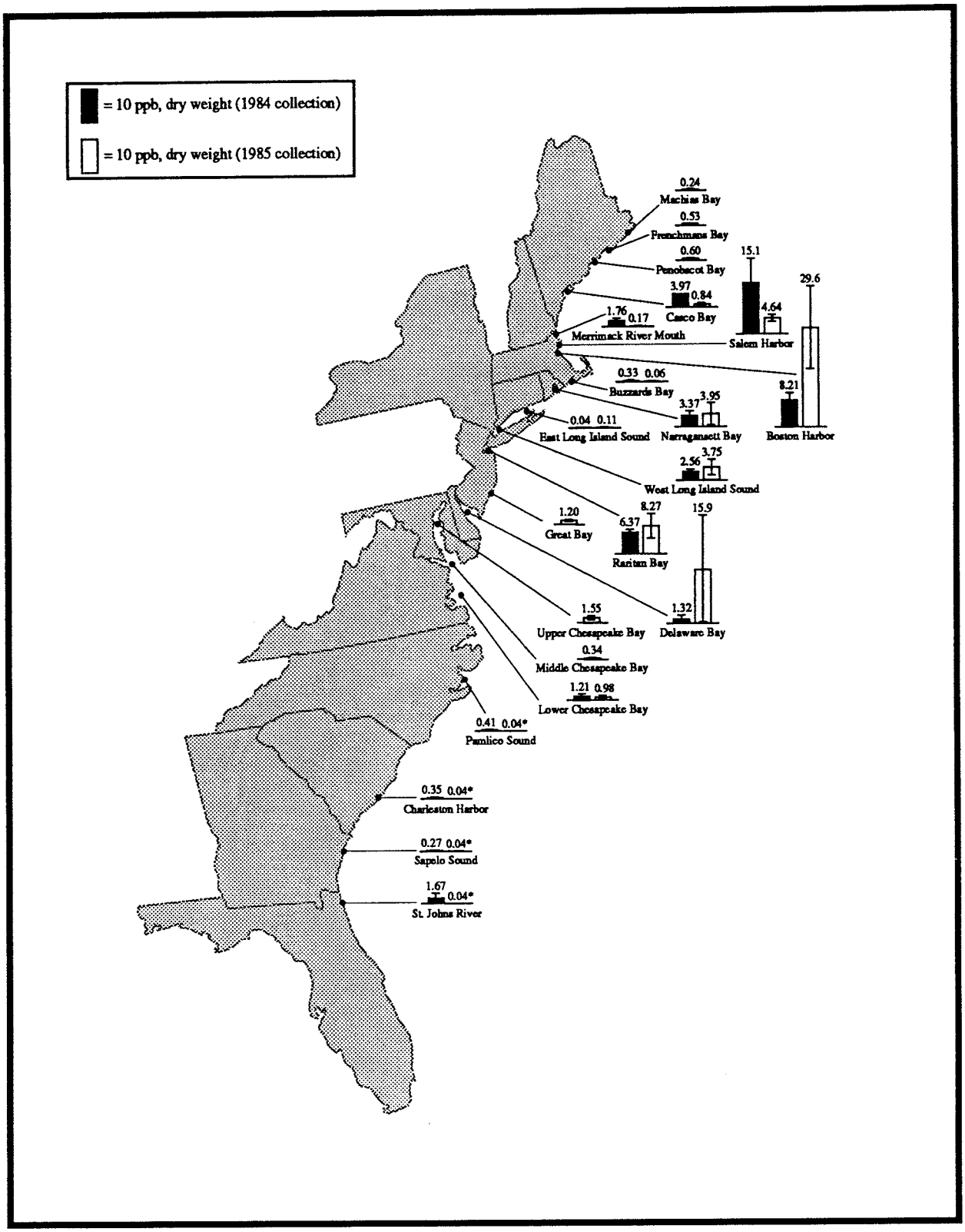

Figure 37. NS&T Benthic Surveillance Project results for summed concentrations of chlordane in Atlantic coast sediments, 1984 and 1985. Asterisk indicates all results below detection. Source: NS&T Benthic Surveillance Project.

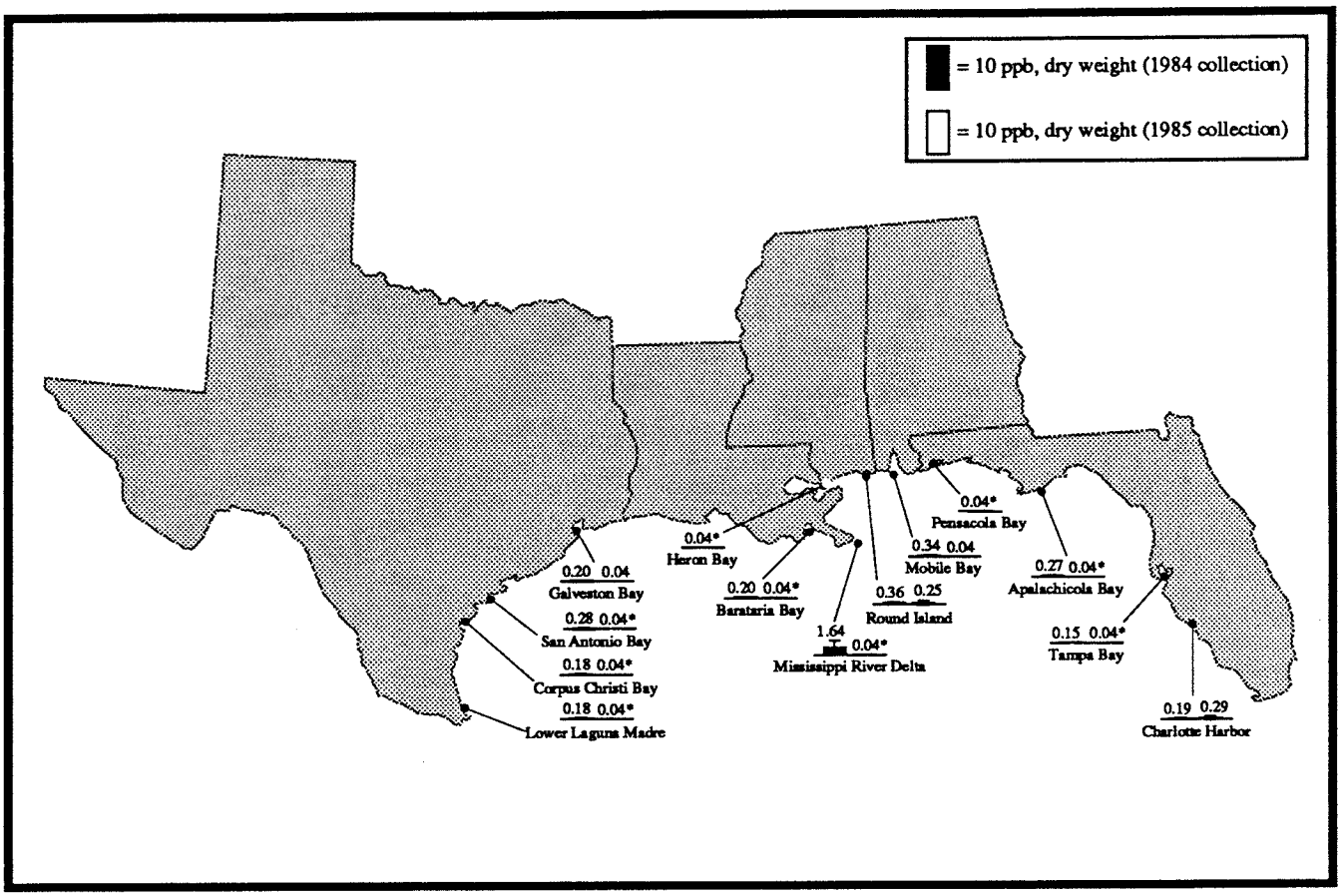

Figure 38. NS&T Benthic Surveillance Project results for summed concentrations of chlordane in Gulf of Mexico coast sediments, 1984 and 1985. Asterisk indicates all results below detection. . Source: NS&T Benthic Surveillance Project.

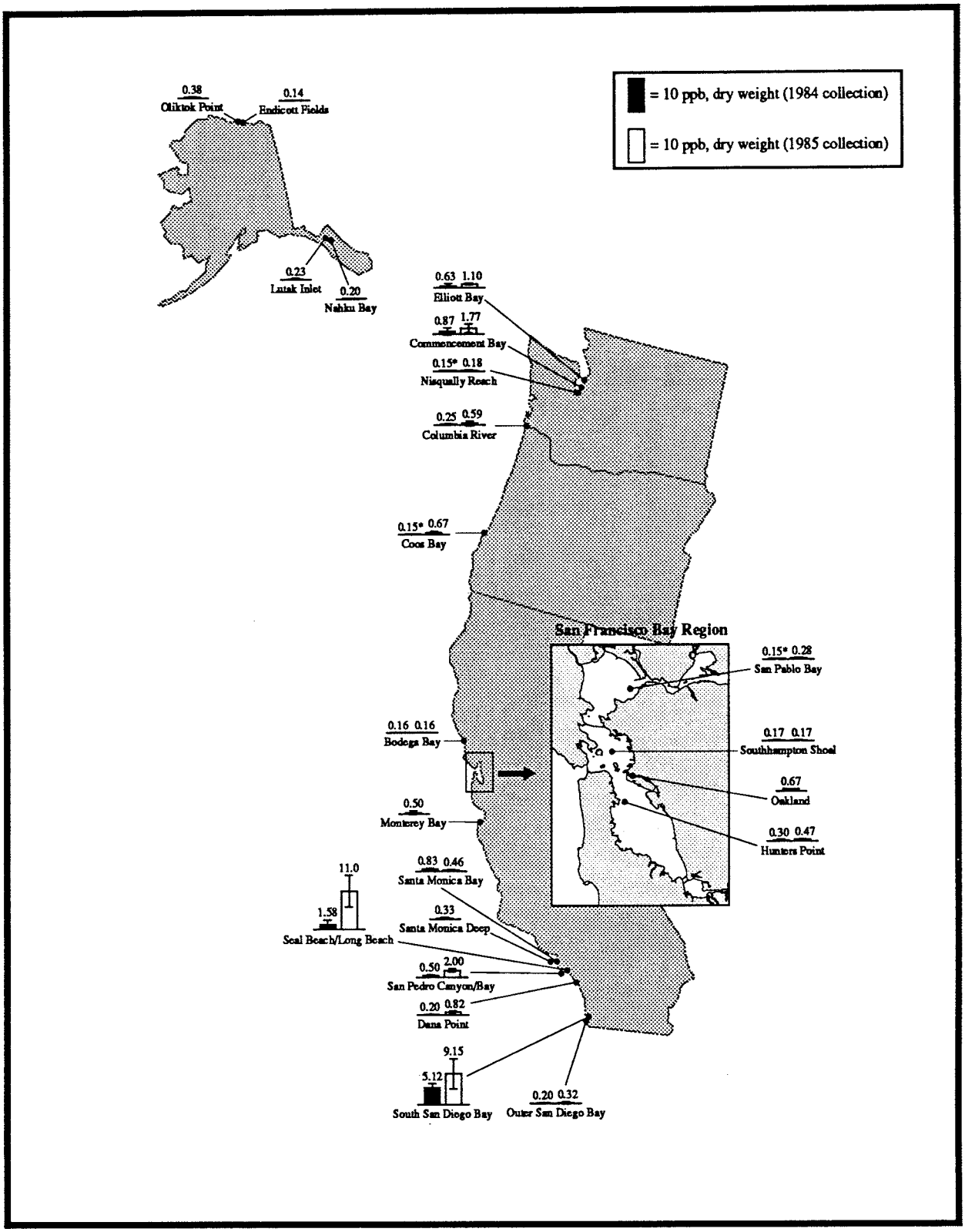

Figure 39. NS&T Benthic Surveillance Project results for summed concentrations of chlordane in Pacific coast sediments, 1984 and 1985. Asterisk indicates all results below detection. Source: NS&T Benthic Surveillance Project.

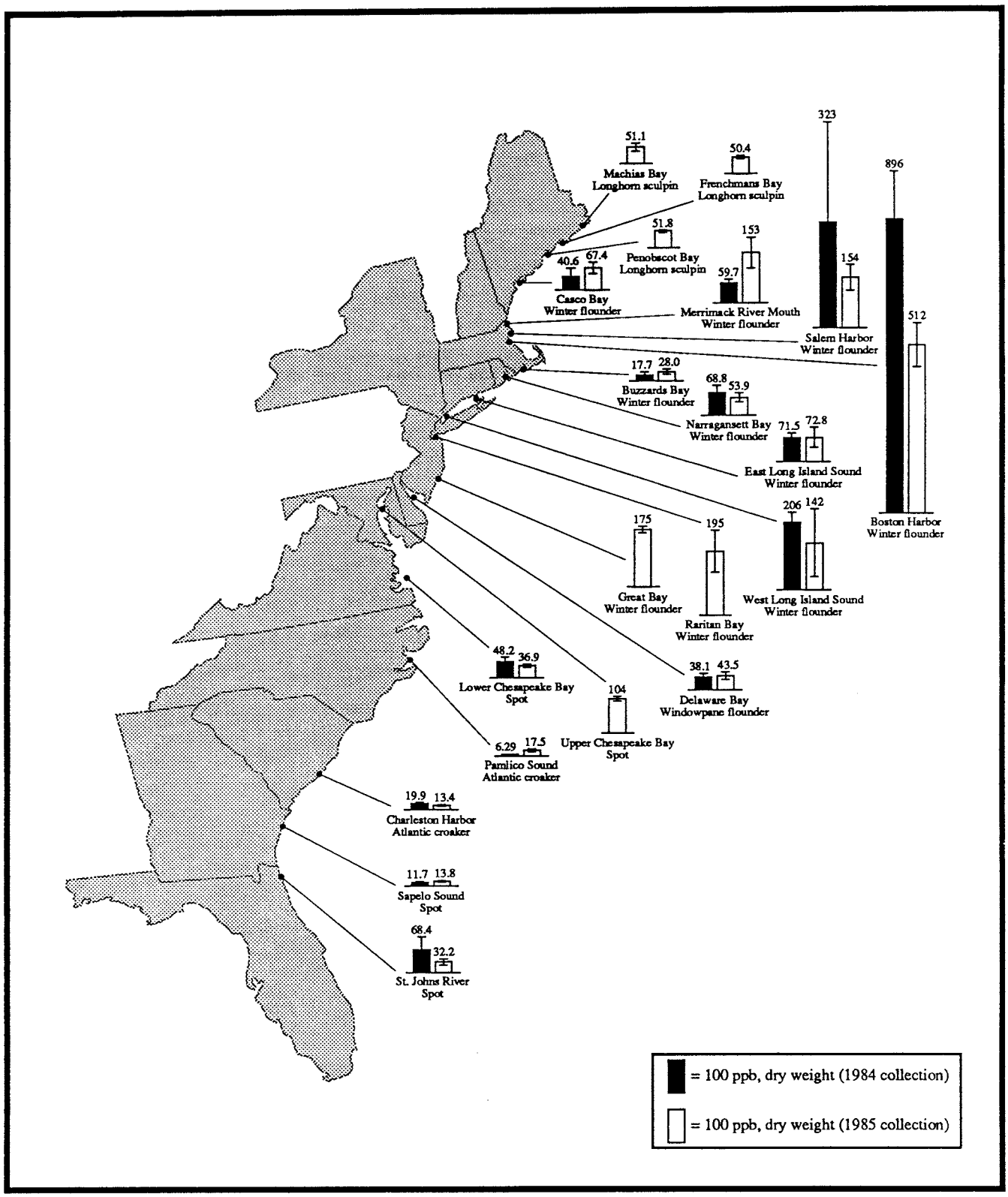

Figure 40. NS&T Benthic Surveillance Project results for summed concentrations of chlordane in liver tissue of Atlantic coast fish, 1984 and 1985. Source: NS&T Benthic Surveillance Project.

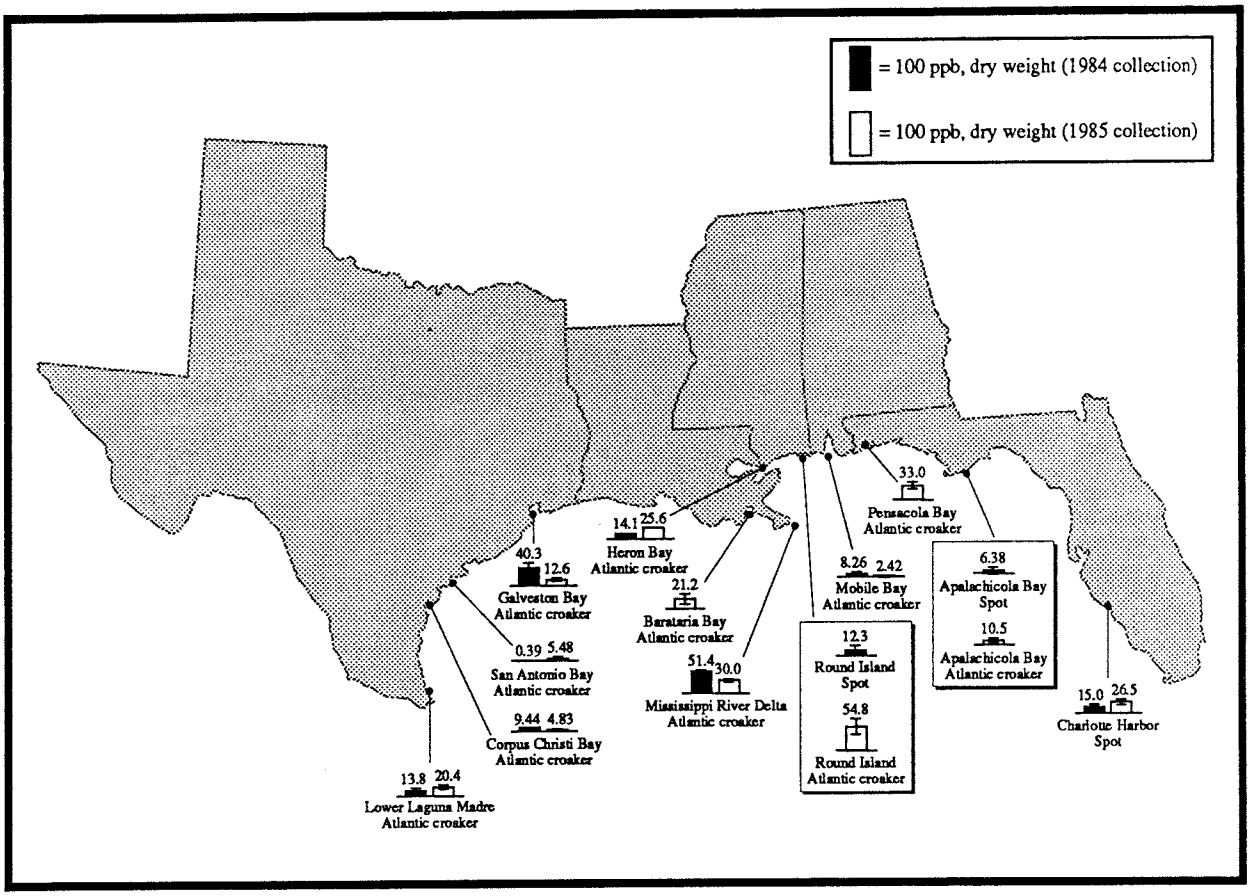

Figure 41. NS&T Benthic Surveillance Project results for summed concentrations of chlordane in liver tissue of Gulf of Mexico coast fish, 1984 and 1985. Source: NS&T Benthic Surveillance Project.

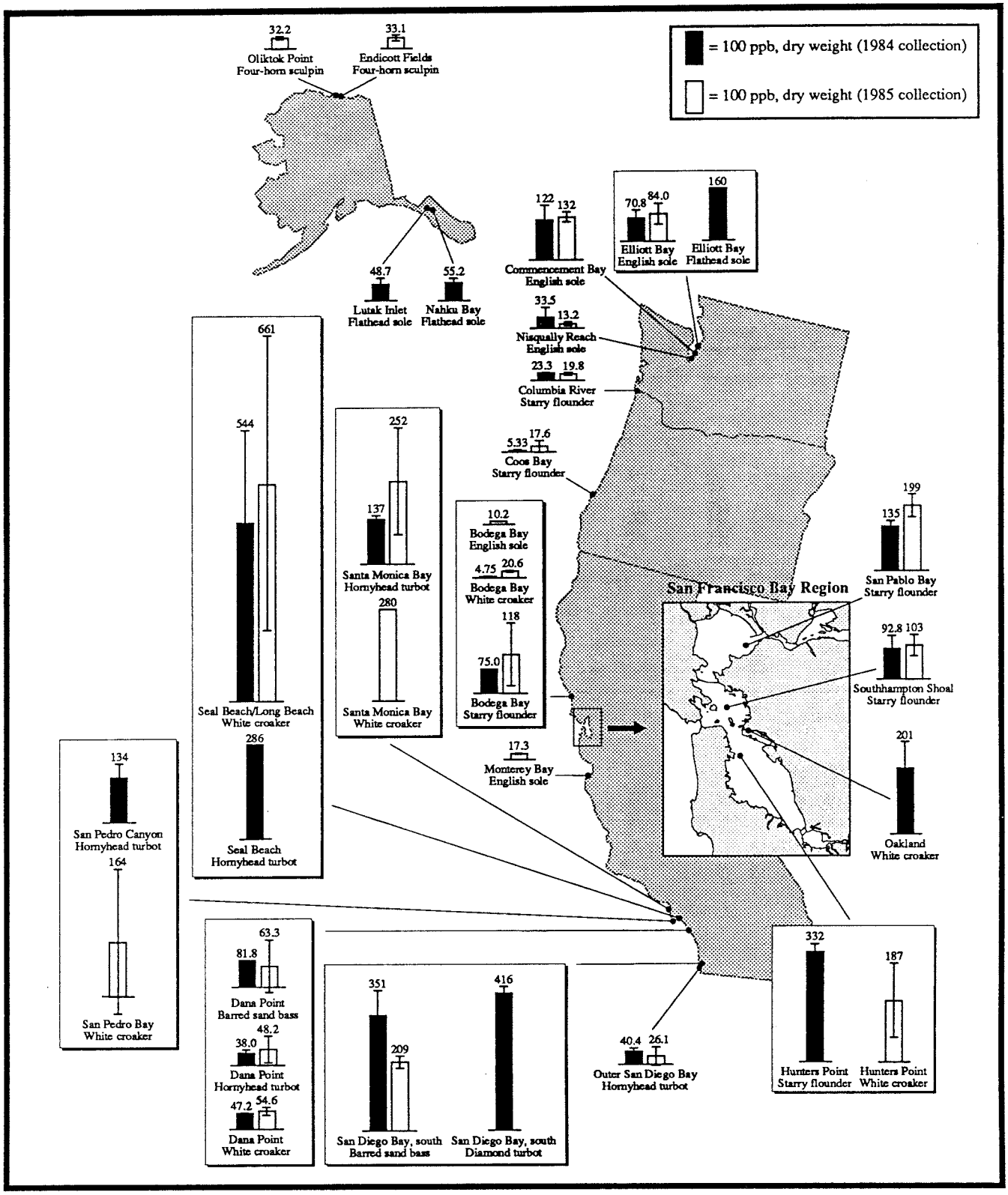

Figure 42. NS&T Benthic Surveillance Project results for summed concentrations of chlordane in liver tissue of Pacific coast fish, 1984 and 1985. Source: NS&T Benthic Surveillance Project.

To evaluate the relation between sediment concentrations of chlordane and liver tissue concentrations measured in fish collected from the same site, aggregated liver results for all species were averaged for sites, then compared to mean sediment concentrations for the same sites using Spearmans rank correlation coefficient. When data for both 1984 and 1985 were analyzed as a whole, it was found that sediment chlordane concentrations were correlated to liver concentrations at a highly significant level: $r_S = 0.601$, $p = 0.0001$, with 90 sediment-liver chlordane pairs.

Analysis of the percent composition of each of the three chlordane constituents was complicated by the relatively high proportion of concentrations below detection limits, primarily in sediments. For example, of a total of 343 sediment samples analyzed in 1984 and 1985, concentrations of all three chlordane compounds measured were not quantifiable in 236 (68.8%); heptachlor was measured at concentrations above detection limits at only 13 (3.79%) of all stations, and only 1 West Coast station in both years.

Composition of the summed chlordane value by constituent was calculated in each of the three substrates analyzed for the Benthic Surveillance Project, *i.e.*, sediment, fish stomach contents, and fish liver tissue. Data were restricted to those in which at least one of the three chlordane compounds was quantified at levels above limits of detection in order to reduce the influence of arbitrarily established values for below detected concentrations. Restricting the dataset in this fashion resulted in sample sizes of 107 (of 343 reported) for sediments, 74 (of 99) for stomach contents, and 271 (of 287) for bottomfish liver tissue. The decreasing percentages of samples in which all three chlordane compounds were not quantitated--68.8 percent in sediments to 25.25 percent for stomach contents to 5.57 percent for fish livers--is likely a reflection of the tendency of chlorinated hydrocarbon compounds such as chlordane to bioaccumulate.

Figure 43 illustrates aggregated results for percent composition of the three chlordane constituents in the three matrices examined. This graphic suggests a pattern in the distribution of chlordane residues in the matrices: It would appear that the relative contribution to the total measured chlordane value of α-chlordane and heptachlor decreases in moving from sediment to fish liver tissue, while the contribution of *trans*-nonachlor increases.

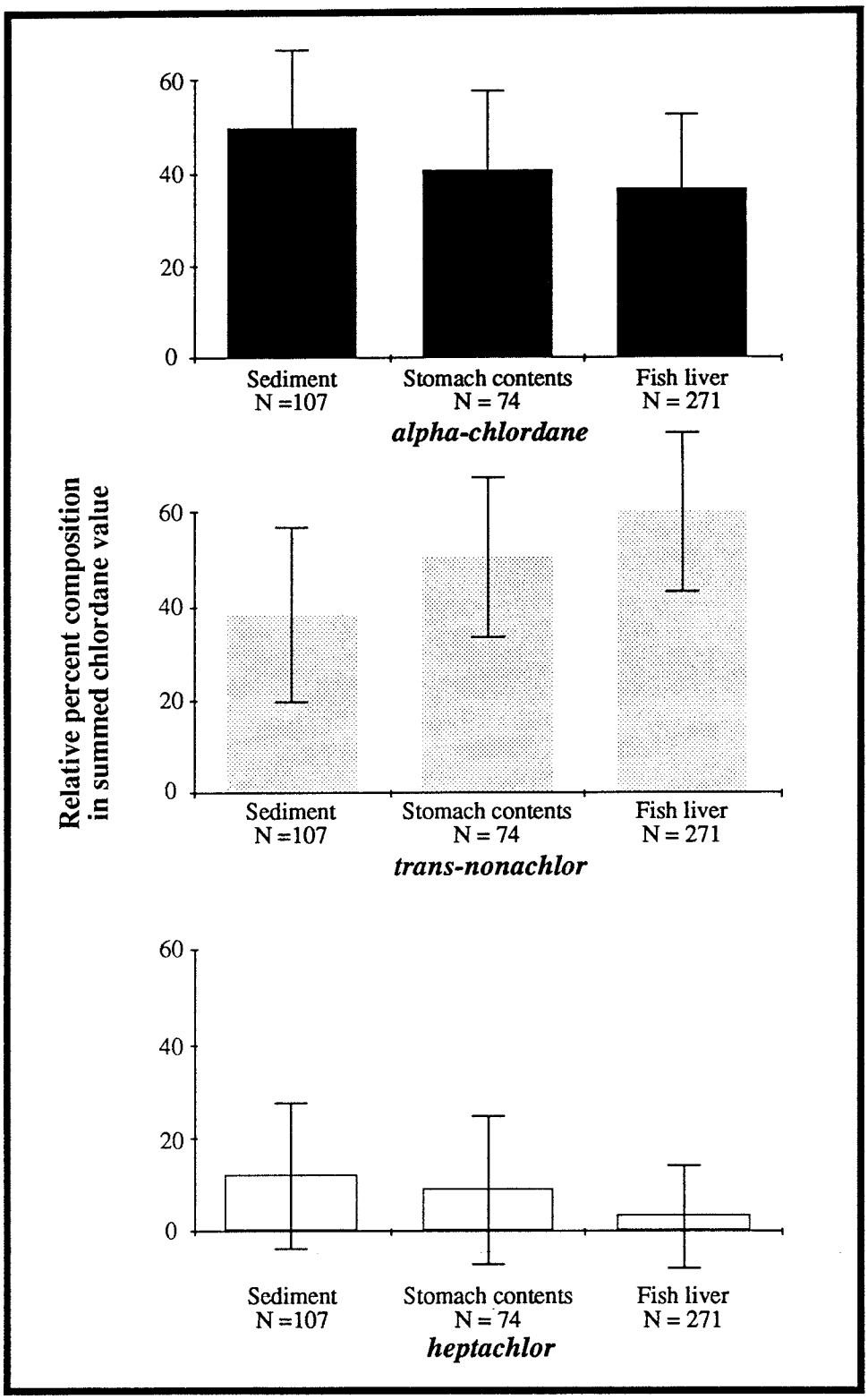

Figure 43. Percent composition of total measured chlordane value in the three matrices of sediment, fish stomach contents, and fish liver tissue. Source: NS&T Benthic Surveillance Project.

In the preceding discussion, liver data from 1984 and 1985 were combined to yield mean percent composition values, but no distinction among species was made. Figure 44 illustrates results in bottomfish liver tissue, differentiated into species groupings. Similar to aggregated species results discussed above, in all species, *trans*-nonachlor comprised the most prevalent constituent of the summed chlordane value, followed by α–chlordane and then by heptachlor. Concentrations of heptachlor in fish livers were generally an order of magnitude less than concentrations of *trans*-nonachlor. However, Figure 44 shows that actual percentages varied considerably among species suggesting possible differences in prey items and/or routes of exposure or differences in biochemical/metabolic processing of contaminants among species. Comparing these results with those for bivalve species analyzed for the NS&T Mussel Watch Project also suggests that distinct differences in chlordane residues exist between bivalve tissue and fish liver tissue: In three of four bivalve species, α–chlordane was the predominant chlordane constituent, in contrast to the fish liver tissue in which *trans*-nonachlor clearly was the most prevalent. The bivalve-fish differences in the distribution of residues could result from interspecies physiological differences, those resulting from the nature of tissues sampled (whole body *vs.* liver only), differences in routes of exposure (*e.g.*, chlordane adsorbed to particulate matter or chlordane in biotic food items), or differences resulting from the qualitatively different habitats of the organisms sampled.

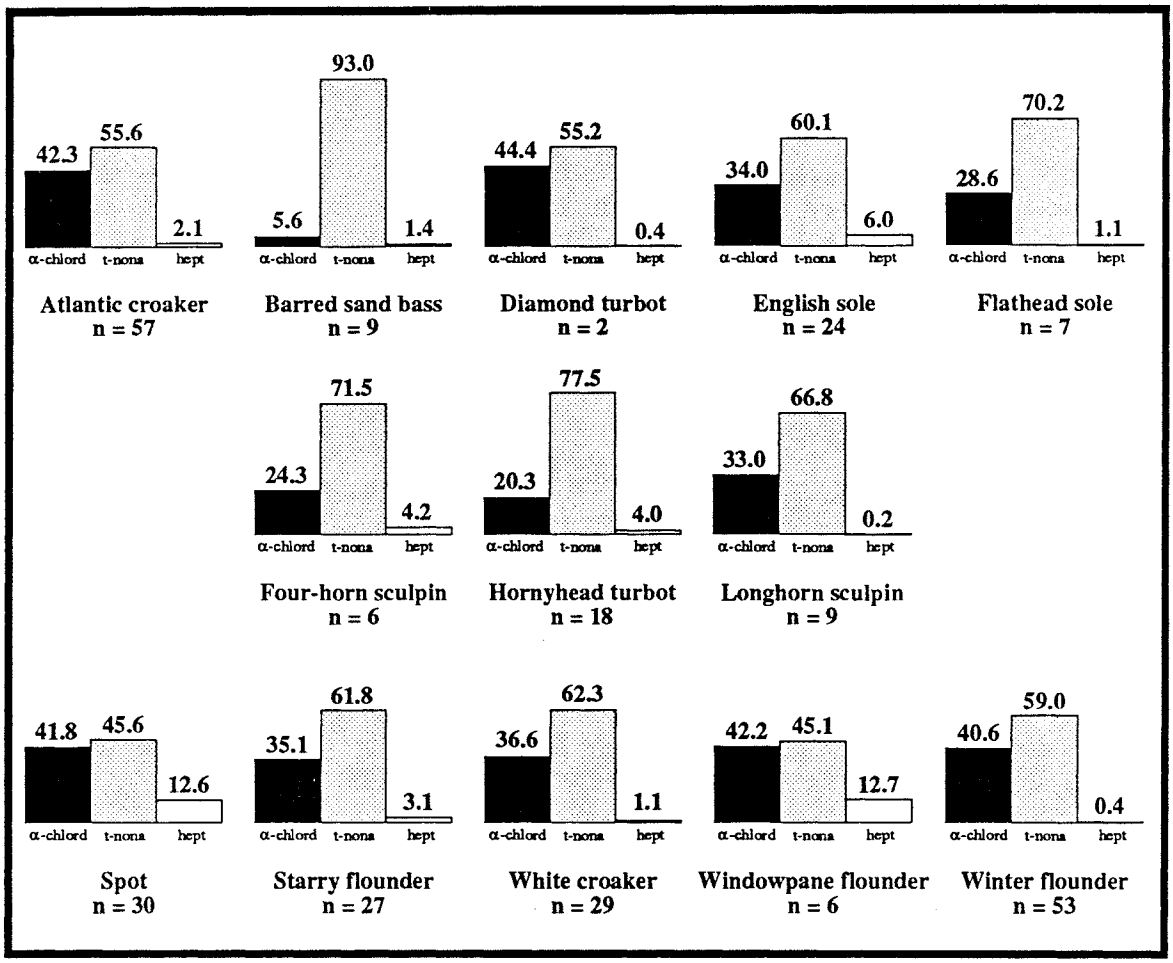

Figure 44. Percent composition of summed chlordane value, by constituent and fish species measured in fish liver tissue. Source: NS&T Benthic Surveillance Project.

There was fairly good correspondence between the Mussel Watch Project and the Benthic Surveillance Project in terms of those locations with highest environmental concentrations of chlordane compounds. For example, the Boston Harbor region of Massachusetts and the inner San Diego Harbor and Long Beach Harbor regions of southern California were flagged by Mussel Watch tissue results, Benthic Surveillance sediment results, and Benthic Surveillance liver tissue results as areas with consistently high values relative

to other sites along the nation's coast. In documenting the recent status and trends in chlordane contamination in the coastal environment of the United States, the two field sampling projects of the NS&T Program have shown chlordane residues to be more closely associated with urban regions than with agricultural areas. This is not unexpected, given the recent history of chlordane use for structural, as opposed to crop, protection.

One of the apparent major regional differences between results obtained in the two projects occurred between Connecticut and New Jersey, where the Mussel Watch Project measured some of the highest concentrations of summed chlordane compounds in mussels; yet, only moderate elevations in sediment and fish liver tissue concentrations were recorded by the Benthic Surveillance Project over the same span of coastline. This is possibly attributable to actual location of sites within an area or physiological characteristics of contaminant uptake in different species, but the more extensive coverage of the Mussel Watch Project, and the results of other researchers (to be discussed in the pages that follow) suggest that the Connecticut to New Jersey region was in fact more heavily contaminated by chlordane compounds than many other coastal areas.

The Choctawhatchee Bay site that yielded such high concentrations of chlordane compounds in both sediment and bivalve tissue analyzed by the Mussel Watch Project was not sampled under the Benthic Surveillance Project. The four Gulf Coast Florida sites that were sampled by Benthic Surveillance showed no unusual elevations in either sediment or fish liver tissue.

It is hoped that as more date are generated from the two major NS&T field projects, trends in chlordane contamination of the U.S. coastal sites sampled will become more apparent and amenable to interpretation. It will be interesting to track the concurrence (or lack of it) between environmental concentrations of chlordane compounds and their recent regulatory history.

Other Studies

Northeast Atlantic Coast: Maine to Virginia

NOAA/FDA/EPA survey of PCBs in Atlantic coast bluefish, 1984-1986. Between 1976 and 1982, the states of New York and New Jersey sponsored studies of environmental contamination by PCBs in the Hudson River and in other portions of both states. The findings of PCB levels in edible tissues exceeding FDA action levels (*i.e.*, >2.0 ppm, wet weight) in a relatively large number of those fish sampled became the source of concern in the states. In 1982, the New Jersey Department of Environmental Protection (NJDEP) announced fishing closures and health advisories for certain areas, based on the results of subsequent monitoring studies. These investigations will be discussed in more detail in this section.

The bluefish (*Pomatomus saltatrix*) was one of the species in which PCBs were found in relatively high concentrations. Because bluefish are migratory in nature, New Jersey and other east coast states expressed concerns that contamination observed in bluefish from industrial regions along the U.S. Atlantic coast could be indicative of a coast-wide problem. In 1984, the Committee on Commerce, Science, and Transportation in the U.S. Senate requested NOAA to coordinate a joint study with the FDA and the EPA to determine the extent of contamination in Atlantic coast bluefish stocks and evaluate levels of risk to human consumers. Summary results were presented in NOAA, FDA, and EPA (1986). Supporting data for the report are available through the National Technical Information Service and were obtained for the current review.

More than 4,200 individual bluefish were collected and analyzed for the project, although most were part of five-fish analytical composites. While the analytical emphasis focused on PCB contamination of the fish, some organochlorine pesticides were also measured. Among the latter were the chlordane compounds *cis*-chlordane, *trans*-chlordane, and *trans*-nonachlor. Approximately 3,500 fish, or 690 composites, in the total sample were analyzed for chlordane. Table 40 presents summary for grouped composite results; Figure 45 shows frequency distribution for the same grouped chlordane results.

Table 40. Summary of summed concentrations of three chlordane compounds (Σ *cis*-chlordane, *trans*-chlordane, and *trans*-nonachlor) in edible portions of bluefish collected along the U.S. east coast, 1984-1986. Values in ppm wet weight. Source: Data supporting NOAA/FDA/EPA (1986).

N	Mean	Sdev	Min*	Max	Median
690	0.09	0.057	0.01	0.36	0.08

* Detection limit reported as 0.01 ppm wet weight.

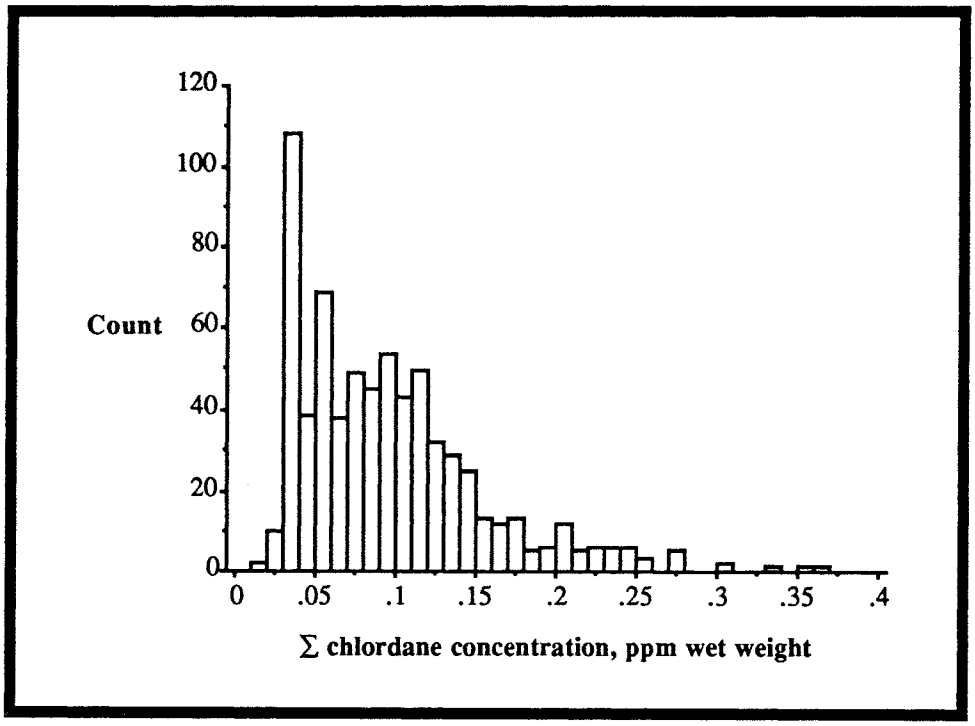

Figure 45. Frequency distribution for chlordane (Σ *cis*- and *trans*-chlordane, and *trans*-nonachlor) results generated in the Atlantic coast bluefish survey, 1984-1986. Source: Data supporting NOAA/FDA/EPA (1986).

As can be seen from both Table 40 and Figure 45, only 5 of the 690 composite samples (0.72%) contained summed chlordane concentrations above 0.3 ppm, the FDA action level. The large number of sample concentrations indicated at 0.03 ppm is probably indicative of the prevalence of samples at or below trace concentrations (listed as 0.01 ppm for each chlordane compound).

Ancillary measurements reported for bluefish tissue composites included lipid content. In order to determine the degree of statistical correlation between percent lipid and summed chlordane concentration measured in the edible tissue samples, the non-parametric (data were not normally distributed) Spearmans rank correlation coefficient was calculated. The resultant r_S value of 0.763 indicated that lipid content and chlordane concentration were significantly correlated in the 690 bluefish tissue composites, at $p = 0.0001$.

Massachusetts

Studies of the Marine Resources of Massachusetts Bays, 1966-1974. In Massachusetts, the Marine Fisheries Advisory Commission, formed in 1960, recommended a program of research focusing on the inshore waters of the Commonwealth. Beginning in 1963, the Division of Marine Fisheries of the Massachusetts Department of Natural Resources conducted an effort called the Estuarine Research Program, designed to provide information relevant to sound management of marine resources in those waters. Under this program, several embayments were sampled and characterized in the 1960s and 1970s. A wide range of

information was gathered, from historical and economic information on fisheries to resource inventories and pollution data. As part of the latter in several of the studies, the minor chlordane constituent heptachlor was targeted as an analyte in mud, soft shell clam (*Mya arenaria*), and winter flounder (*Pseudopleuronectes americanus*).

Although many embayments were sampled under the Estuarine Research Program, heptachlor was not analyzed at all of them. Individual studies which explicitly targeted the compound were those for the Merrimack River estuary (Jerome *et al.*, 1965); Quincy Bay (Jerome, Chesmore, and Anderson,1966); the North River estuary (Fiske, Watson, and Coates, 1966); the Parker River-Plum Island estuary (Jerome, Chesmore, and Anderson, 1968); Lynn-Saugus Harbor (Chesmore, Brown, and Anderson, 1972); Hingham Bay (Iwanowicz, Anderson, and Ketschke, 1973); and Plymouth, Kingston, and Duxbury Bay (Iwanowicz, Anderson, and Ketschke, 1974). Results from those studies are summarized in Table 41.

Table 41. Concentrations of heptachlor measured in water, mud, clam, and flounder tissues sampled in Massachusetts bays, 1964-1972. Note variations in units of measurement. Sources: as specified below.

Location	Year Sampled	Substrate	Heptachlor Range[1]	Source
Merrimack River estuary	1964	Clam	0.008-0.125, mw	Jerome, *et al.* (1965)
Quincy Bay	1964	Clam	bd-0.088, mw	Jerome, Chesmore, Anderson (1966)
North River estuary	1964	Clam	bd-2.255	Fiske, Watson, Coates (1966)
Parker River- Plum Island estuary	1965	Clam, flounder, white perch	All bd, mw	Jerome, Chesmore, Anderson (1968)
Lynn-Saugus Harbor	1968-69	Mud	bd-4.833, mw	Chesmore, Brown, Anderson (1972)
	1968-69	Clam	bd-0.431, mw	
	1968	Flounder: muscle	bd-0.517, mw	
	1968	Flounder: intestine	bd-1.638, mw	
Plymouth, Kingston,	1971	Mud	bd-0.016, md	Iwanowicz, Anderson, Ketschke (1974)
Duxbury Bay	1971	Clam	0.001-0.004, mw	
	1971	Flounder	0.001-0.026, mw	
Hingham Bay	1972	Water	<0.001-1.018, b	Iwanowicz, Anderson, Ketschke (1973)
	1972	Mud	<0.001-0.031, md	
	1972	Clam	<0.001-0.227, mw	

[1]Detection limits were not always specified; bd=below detection Measurement basis varied: b=parts per billion; mw=parts per million, wet weight; md=parts per million, dry weight.

Soft-shelled clams were the only substrate consistently sampled among the six locations where heptachlor was measured. Winter flounder tissues and mud were analyzed at three locations, and water and white perch (*Morone americanus*) tissue at one. Only one of the embayments, Lynn-Saugus Harbor, located approximately midway between Boston and Salem, showed concentrations of heptachlor that were notable. The maximum listed concentration in mud (4.8 ppm, specified as wet weight basis, although other sediment measurements in the Estuarine Research Program were consistently dry weight) was substantially higher--two orders of magnitude--than the two other locations where mud was sampled. In fact, although not shown here, the range of concentrations over a 6-month period at the station within Lynn-Saugus Harbor where the heptachlor maximum was measured also varied over two orders of magnitude. This would suggest that Lynn-Saugus Harbor was subject to relatively high sediment-associated burdens of heptachlor, but also that these high concentrations were infrequent, possibly resulting from pulses of heavy storm runoff, unidentified industrial inputs, or other unknown factors. The fact that both clams and winter flounder tissues in Lynn-Saugus Harbor also showed high concentrations (some levels in both organisms exceeding FDA action levels) supports the portrayal of the area as one that had experienced elevated inputs of heptachlor.

Bioaccumulation of metals, polychlorinated biphenyls, polyaromatic hydrocarbons and chlorinated pesticides in the musse', *Mytilus edulis*, transplanted to Salem Sound, Massachusetts, 1986. As part of the environmental assessment portion of the South Essex Sewage District (SESD) 301-h waiver application under the Clean Water Act, bioaccumulation of contaminants was evaluated in blue mussels (*Mytilus edulis*) transplanted to Salem Sound, Massachusetts. Analytical methodologies employed in the NOAA/ NS&T Program were used for the study, and many of the same analytes were targeted. As in the NOAA

program, the chlordane constituents α-chlordane, *trans*-nonachlor, and heptachlor were among the chlorinated pesticides measured. Results from the Salem Sound study were reported in Robinson and Ryan (1986), and are summarized below in Table 42.

Table 42. Concentrations of chlordane compounds measured in composites of 15 blue mussels transplanted into Salem Sound and held for 32 days, 1986. Values in ppb dry weight. Source: Robinson and Ryan (1986).

Sample Type	Sample #	Date	α-chlordane	*trans*-nonachlor	Heptachlor
Control, 0 days	738	7/10/86	8.8	8.4	bd
	691		8.8	7.9	0.3
	Mean		8.8	8.1	<0.3
Control, 32 days	416	8/11/86	6.4	6.4	bd
	435		4.7	5.4	bd
	853		4.5	5.5	bd
	Mean		5.2	5.8	bd
Upper Outfall, 32 days	300	8/11/86	6.2	6.1	bd
	349		6.1	5.9	bd
	700		5.5	5.1	bd
	Mean		5.9	5.7	bd
EPA Reference	177		15.0	14.0	1.0

Results found by Robinson and Ryan suggested that the original collection site for the mussels, at the University of Massachusetts Marine Station in Gloucester, Massachusetts, may have been more heavily contaminated with respect to pesticides like chlordane than the Salem Sound study site. Mussels transplanted and directly exposed to the SESD outfall plume contained uniformly lower concentrations of all three chlordane compounds analyzed. The decline in body burdens was also reflected in measured concentrations of PCBs, which were nearly twice as high in time-zero controls.

Histopathological and chemical assessment of winter flounder, lobster and soft-shelled clam indigenous to Quincy Bay, 1987. The Environmental Research laboratory of the U.S. EPA in Narragansett, Rhode Island, sponsored histopathological and chemical field studies in Quincy Bay, Massachusetts, to evaluate the extent of chemical contamination in that area. Organisms sampled for the study were the winter flounder (*Pseudopleuronectes americanus*), lobster (*Homarus americanus*), and soft-shelled clam (*Mya arenaria*). In addition, sediment cores and surficial sediment samples were collected and chemically analyzed. The pesticides targeted included α- and γ-chlordane. Results from this study of Quincy Bay were reported in Gardner and Pruell (1988). A summary of concentrations determined by Gardner and Pruell is shown below as Table 43.

Table 43. Summary of results for concentrations of chlordane compounds in sediments and biota collected in Quincy Bay, Massachusetts, 1987. Values in ng/g (ppb) dry weight. Source: Gardner and Pruell (1988).

Matrix	N	α-chlord Mean ±sdev	α-chlord Range	α-chlord Median	γ-chlord Mean ±sdev	γ-chlord Range	γ-chlord Median
Sediment scoop	18	2.8±2.28	0.11-6.48	1.905	1.76±1.45	bd[1]-4.63	1.375
Sediment core	16	3.91±4.99	bd-18	2.59	2.57±3.14	bd-11.7	1.62
Transplant oyster	6	13.15±4.90	5.28-18.6	14.2	15.15±4.93	7.42-21.1	16.25
Soft-shelled clam	2	8.38±2.72	6.45-10.3	8.38	10.78±2.72	8.86-12.7	10.78
Lobster muscle	16	0.14±0.34	bd-1.25	bd	0.95±0.73	bd-2.75	0.965
Lobster hepatopanc	8	61.7±45.3	14.9-148	48.6	128±85.42	31.3-256	97.5
Winter flounder musc	25	4.55±7.39	0-37.8	2.74	11.22±21.95	bd-110	4.81

[1]bd = below detection; based on extract detection limits reported by the authors, analytical detection limits for tissues and for sediment can be calculated to be in the ranges of 0.15-2.86 ng/g dry weight and 0.09-0.57 ng/g, respectively.

Comparing results across substrates, highest mean and median concentrations of both chlordane compounds were measured in hepatopancreas tissue of lobsters. The highest individual summed concentration determined in hepatopancreas was 404 ppb dry weight. Using the reported sample dry weight of 59.2 percent, the summed α- and γ-chlordane may be converted to 239 ppb wet weight, which approached, but did not exceed the FDA action level of 300 ppb. It would seem likely that if other chlordane compounds were targeted, the summed concentration would have exceeded the action level concentration.

Interestingly, lowest concentrations across substrates were found in lobster muscle tissue. In tissue samples, greater proportions of γ-chlordane were found over α-chlordane. In surficial sediments and sediment cores, the opposite appeared to be true. This result is portrayed graphically in Figure 46.

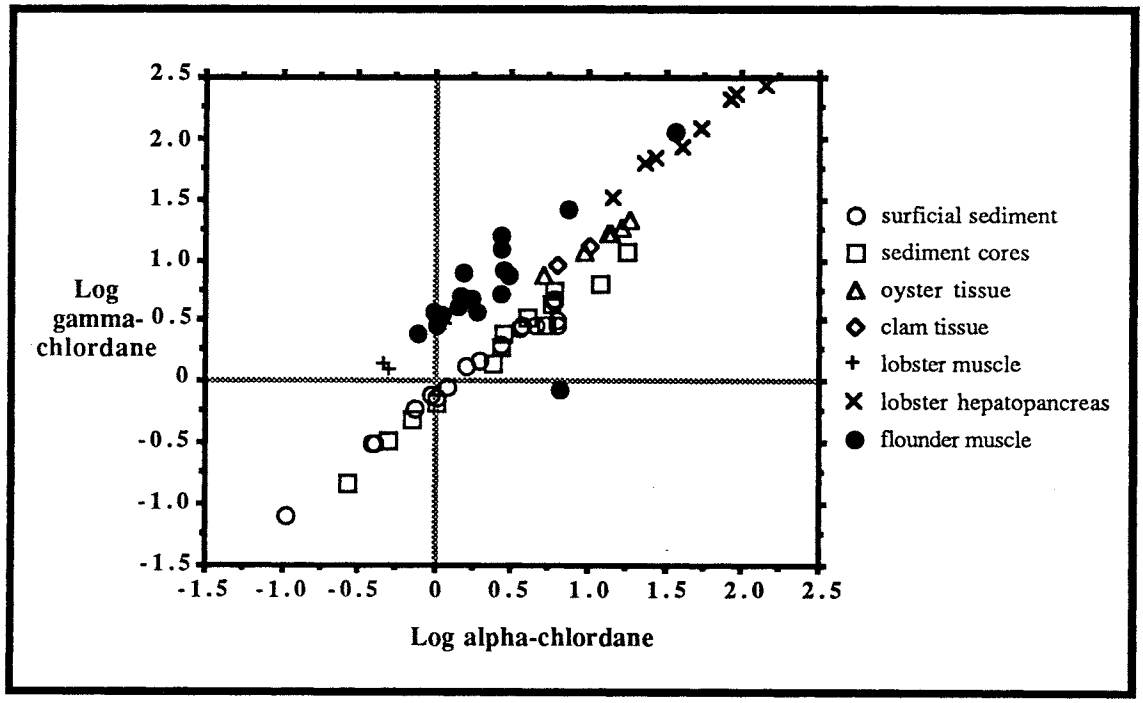

Figure 46. Relationship between α-chlordane and γ-chlordane in sediment and biotic substrates sampled in Quincy Bay, Massachusetts, in 1987. Log-log plot of concentrations expressed as ng/g dry weight. Source: Gardner and Pruell (1988).

These relationships at different substrate and trophic levels were notable in their consistency. That is, sediment (both surficial and core) ratios of α-chlordane to γ-chlordane defined one relationship, while lobster (both muscle and hepatopancreas tissues) and winter flounder muscle defined another. For both bivalves, a third relationship, intermediate between that for sediments and that for lobster/winter flounder, was apparent. These suggest that organisms of different trophic levels may metabolize chlordanes differently. If sediment concentrations of α- and γ-chlordane are taken to have been representative of chlordane mixtures in the environment, then the observed differences in occurrences of the two compounds may have reflected differential abilities to metabolize chlordane isomers.

Organochlorine residues in common terns and associated estuarine organisms from Massachusetts, 1971-1981. Between the years of 1971 and 1981, organochlorine residues were measured in eggs of common terns (*Sterna hirundo*); in body tissues of Atlantic silversides (*Menidia menidia*), mussels (*Mytilus edulis*); and miscellaneous other fish specimens collected along the Massachusetts coast. Goals of this investigation were to identify geographic patterns of contamination, relate them to use patterns and to evaluate potential declines in residues following use reductions. Included in the sample results were some measurements of α-chlordane and oxychlordane; however, because of analytical difficulties, chlordane isomers and oxychlordane were not identified and quantitated until 1976. Available results for chlordane in terns were reported in Nisbet and Reynolds (1984). Concentrations in silversides and mussels were apparently measured only infrequently and chlordane residue concentrations were not reported in the paper.

Chlordane results in Nisbet and Reynolds (1984) were limited to those for common tern eggs collected on Bird Island (in Buzzards Bay) and on Monomoy Island (off the southeastern shore of Cape Cod). These are summarized below in Table 44.

Table 44. Concentration of α- and oxychlordane measured in eggs of common tern collected in the vicinity of Cape Cod and Buzzards Bay, Massachusetts, 1976-1978. Values in ppm wet weight, mean ± standard deviation. Source: Nisbet and Reynolds (1984).

Location	Date	N	% Lipids	α-chlordane	oxy-chlordane
Bird Island	6/8/76	8	9.2±0.42	nm	0.026±0.005
	6/9/77	11 (pooled)	9.4	0.009	0.009
	6/11/78	11 (pooled)	9.2	0.017	0.034
	5/29/80	8 (pooled)	9.1	0.17[1]	0.009
	5/23/81	12 (pooled)	9.1	0.054[1]	0.009
Monomoy Island	6/10/76	8	9.3±0.61	nm	0.033±0.008
	6/8/77	6	7.7±0.71	0.019±0.004	0.011±0.001
	6/20/77	7	9.0±0.79	0.021±0.004	0.015±0.002
	6/11/78	12 (pooled)	10.0	0.01	0.019

[1]Includes γ-chlordane in γ-:α- ratio of approximately 1:2.

Sample size was small and no distinct temporal trends were apparent. Levels of chlordane were relatively low and Nisbet and Reynolds felt that these represented background contamination levels. Unfortunately, samples from sites closer to the Boston urban coast were not analyzed for chlordane isomers. However, the metabolite heptachlor epoxide was targeted in 1972 and the Snake Island site (just north of Boston) was the only location of four that contained a measurable concentration, 0.07 ppm wet weight.

Connecticut

Connecticut study of shellfish contamination in the Poquonock River, 1981. In May of 1981, a fire destroyed a hardware store in Groton, Connecticut. Water used to extinguish the fire also flushed pesticides from the structure and foundation of the store into the nearby Poquonock River. Initial measurements of chlordane (not specifically defined) in the basement of the building and in the runoff water were 3.78 and 0.17 ppm, respectively. Because of concern about the levels of chlordane reaching the Poquonock River and subsequently contaminating resident shellfish, the Groton Health Department and the Connecticut Department of Health Services closed the river to shellfishing. Details of the closure and subsequent actions were contained in a letter from the state Principal Sanitarian to the director of the Preventable Diseases Section (Shute, 1981).

The Bureau of Health Promotion and Disease Prevention within the Connecticut Department of Health Services analyzed bivalve samples collected between May and October of 1981 in order to evaluate the extent of contamination and to assess the need for continuing the moratorium on harvesting of shellfish. While concentrations of chlordane in oysters (unspecified species) were found to range from 0.1 to 0.3 ppm wet weight immediately after the fire, subsequent collections indicated a consistent decline in levels. For example, a given station yielded the maximum measured concentration, equal to the U.S. FDA action level of 0.3 ppm wet weight on May 28th. On June 12th, levels were measured at 0.2 ppm. On August 10, these had declined to 0.03 ppm. At another site, oyster concentrations decreased from 0.1 ppm on May 28 to 0.0013 in November of 1982.

Based on sampling results in oysters and clams at seven locations that showed a continuing drop in chlordane concentrations, the Preventable Diseases Section of the Connecticut Department of Health Services recommended the reopening of the Poquonock River to shellfishing in late October 1981.

New York

PCBs and organochlorine pesticides in New York Harbor waters, 1985-1986. The New York City Department of Environmental Protection has annually assessed water quality in the New York Harbor region for nearly 80 years, and as part of the 1986 effort, harbor waters were analyzed for 20 pesticides, related compounds, and PCBs. Results were reported in City of New York, Department of Environmental Protection (1987). Included in the survey were measurements of chlordane and heptachlor (compositions not specified). Samples were collected throughout the New York Harbor region, from the Hudson and East Rivers, south to Raritan Bay. Of the 50 samples analyzed, all chlordane measurements were below the detection limit of 0.027 µg/l (ppb). Two positive measurements for heptachlor (listed detection limits ranging from 0.000385 µg/l to 0.00068 µg/l) were determined at Rockaway Inlet (0.001 µg/l) and at Bergen Basin (0.0011 µg/l). The high proportion of undetected values for chlordane compounds is not surprising given their virtual insolubility in water. Additionally, as discussed previously in another section of this report, Oloffs, Albright, and Szeto (1972) and Oloffs *et al.*(1973) demonstrated that in shallow, well-mixed bodies of water, chlordane could be expected to relatively rapidly move from natural waters into the underlying sediments.

Geochemical assessment of sedimentation and contaminant distributions in the Hudson-Raritan estuary, 1974-1982. In an effort co-sponsored by NOAA and the U.S. Department of Energy, Olsen *et al.* (1984) summarized available research and data on the distribution of radionuclides and synthetic organic compounds in the Hudson-Raritan estuary. Among those compounds discussed was chlordane, reported as the sum of α- and γ- isomers, measured in sediment cores taken at various locations in the estuary.

Chlordane was found to be a major contaminant in sediments of inner New York Harbor, with concentrations ranging from 60 to 242 ng/g dry weight (ppb). However, concentrations dropped off markedly in upstream samples taken in the Hudson River and estuary, with levels less than 8 ng/g. Olsen *et al.* used these observations to assert that dominant sources of chlordane likely were urban runoff and sewage. They cited results from Bopp *et al.* (1982), in which sewage sludge from the Wards Island sewage treatment plant showed levels of pesticides greater than or equal to those found in the sediments as further support for this idea. Summarized results for chlordane from Bopp *et al.* and Olsen *et al.* are shown below in Table 45.

Table 45. Chlordane, PCB, and ^{137}Cs concentrations in sediments of the Hudson-Raritan estuary. Source: Bopp *et al.* (1982); Olsen *et al.* (1984).

Location	Sediment Depth (cm)	Σ Chlordane[a] (ng/g dry)	Σ PCB[b] (μg/g dry)	^{137}Cs (pCi/kg)
Inner Harbor	0-5	64	8.85	965
(mp 3.0)	5-10	60	11.9	1100
	10-18	164	10.6	720
	18-25	165	6.28	595
	25-30	160	6.19	690
	30-36	197	6.67	1010
	36-43	111	2.88	340
	43-50	81	1.80	37
Inner Harbor	0-10	165	2.24	260
(mp -1.5)	10-20	191	3.25	455
	30-40	202	4.40	420
	60-70	240	6.16	1090
	70-80	150	3.67	950
	80-90	200	4.79	1080
	120-130	121	2.75	290
	140-150	173	3.11	350
	150-160	178	3.64	365
	170-180	242	5.27	345
	190-200	81	1.07	200
	220-230	179	-	390
	240-250	203	3.34	500
Raritan Bay	0-7	8.2	0.20 (+ND)	70
(West, RB-1)	7-11	3.5	ND	ND
Raritan Bay (East, RB-15)	24-27	<4	ND	ND
Raritan Bay	0-1	13	0.56	140
(RB-27)	4-5	36	0.97	250
	8-9	13	0.95	230
	12-13	5	0.58	110
	16-17	3	0.22	33
Dredged Pit	0-2	21	0.76	350
	4-8 (sandy)	5	0.21	30
	24-28	35	1.09	470
Barnegat Shelf				
(BI-3)	0-5	14	0.70	310
(BI-7)	0-5	5	0.19	115
Hudson Canyon				
(590C)	0-5	<1	0.03	100

[a]Σ α– + γ–chlordanes.
[b]Σ Aroclor 1242 + Aroclor 1254.

Spearmans rank correlation procedure was used to evaluate the relation between chlordane and PCB concentrations, and chlordane and ^{137}Cs, in the sediments of the Hudson-Raritan estuary (For this analysis, "less than" values were converted to one half the listed quantity; "not detected" values were converted to zero). Although the two chlorinated compounds have qualitatively different ultimate sources, chlordanes and PCBs were found to be highly correlated (r_S=0.805, p=0.0001), further supporting the assertion that urban

runoff and/or sewage discharges were major sources of contaminants to the estuary system. Chlordanes and ^{137}Cs were also highly correlated (r_S=0.721, p=0.0001). Olsen *et al.* commented that the apparent covariation of organic and other contaminants with ^{137}Cs was an indication of the ability of fine particles to sequester, transport, and accumulate a wide variety of reactive pollutants, in spite of differences in chemistry and input mechanisms.

Olsen *et al.* summarized data collected by Bopp *et al.* (1982) that provided a history of chlordane contamination based on measurements of ^{137}Cs, $^{239,240}Pu$, and ^{60}Co in a sediment core collected at a New York inner harbor site. According to this analysis (shown in Figure 47), chlordane concentrations peaked around 1963, remained fairly constant between 1965 and 1970, then began a decline in the 1970s to approximately the same levels as measured in the 1950s.

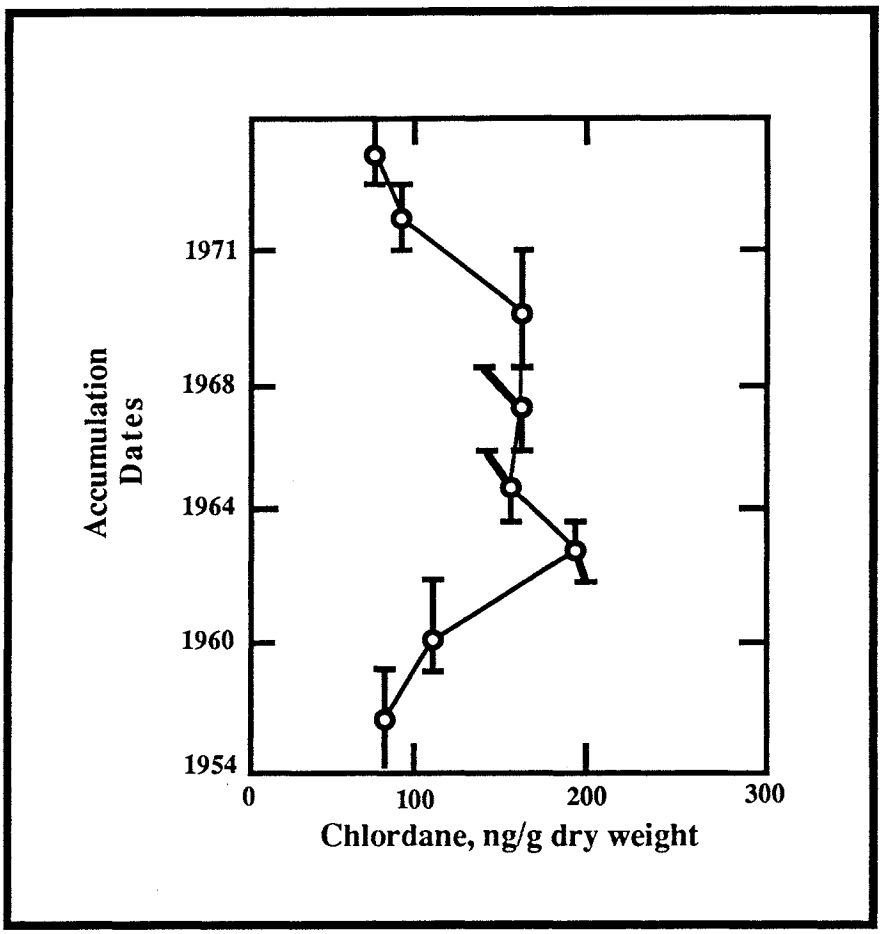

Figure 47. Chlordane contamination history for New York inner harbor site, based on radioisotope dating of sediment core. Source: Bopp *et al.* (1982); Olsen *et al.* (1984).

Chlorinated hydrocarbon residues in shellfish (Pelecypoda) from estuaries of Long Island, New York, 1968-1970. In 1968, the state of New York Department of Environmental Conservation began monitoring chlorinated hydrocarbon residues in shellfish taken in the waters off Long Island. Results are summarized in Foehrenbach, Mahmood, and Sullivan (1971). Shellfish species examined were the hard clam (*Mercenaria mercenaria*), the blue mussel (*Mytilus edulis*), eastern oyster (*Crassostrea virginica*), soft clam (*Mya arenaria*), ribbed mussel (*Brachidontes demissus plicatulus*), and the bay scallop (*Aequipecten irradians*), Among the compounds analyzed was the technical chlordane constituent, heptachlor. However, heptachlor was not detected (at a listed limit of 0.010 ppm wet weight) in any of the samples collected at any of the 10 sites around Long Island.

This does not necessarily indicate a lack of contamination by chlordane compounds, as heptachlor is only one of many components of technical chlordane. Recent (1986-1988) NS&T Mussel Watch results for Long Island sites show that concentrations of heptachlor have frequently been below detection, at *dry weight* detection limits which are more rigorous than those specified in Foehrenbach, Mahmood, and Sullivan. Moreover, NS&T results show that heptachlor is a comparatively minor biotic constituent relative to the other chlordane compounds α-chlordane and *trans*-nonachlor (see Figures 36 and 44 showing relative contribution of measured compounds to summed values for chlordane). While heptachlor may have been below quantitation levels in 1968-1970 shellfish even employing current methods, other chlordane compounds could have been and likely were present at higher levels.

NOAA/MESA analysis of residual chlorinated and aromatic hydrocarbons and related compounds in selected sources, sinks, and biota of the New York Bight, 1978-1980. As part of its Marine Ecosystems Analysis (MESA) Program, NOAA selected the New York Bight as an initial study area to evaluate the nature and resultant impacts of environmental problems. The New York Bight was chosen because of its multiple uses, its importance as an ecosystem, and its proximity to one of the most heavily populated and industrialized regions of the United States. A component of the New York Bight Project was a study of organic residue contamination of several substrates, including subsurface waters, the surface microlayer, bottom sediments, sewage sludge, sewage effluent, plankton, and tissues of invertebrates and fish. Included among the compounds measured were α-chlordane, *trans*-nonachlor, and heptachlor. Analyses were performed by the National Analytical Facility of the NMFS in Seattle. Results are summarized in MacLeod *et al.* (1981).

The wide variety of substrates analyzed by MacLeod *et al.* provides a broad perspective on ranges of concentrations encountered in the marine environment of the New York Bight region. Lowest concentrations were found in subsurface waters, with 29 of 30 measurements for chlordane constituents below detection limits. Highest concentrations were generally found in fish tissues, although the maximum among all substrates was that in hepatopancreas tissues of the American lobster (*Homarus americanus*). Figure 48 summarizes data of MacLeod *et al.*

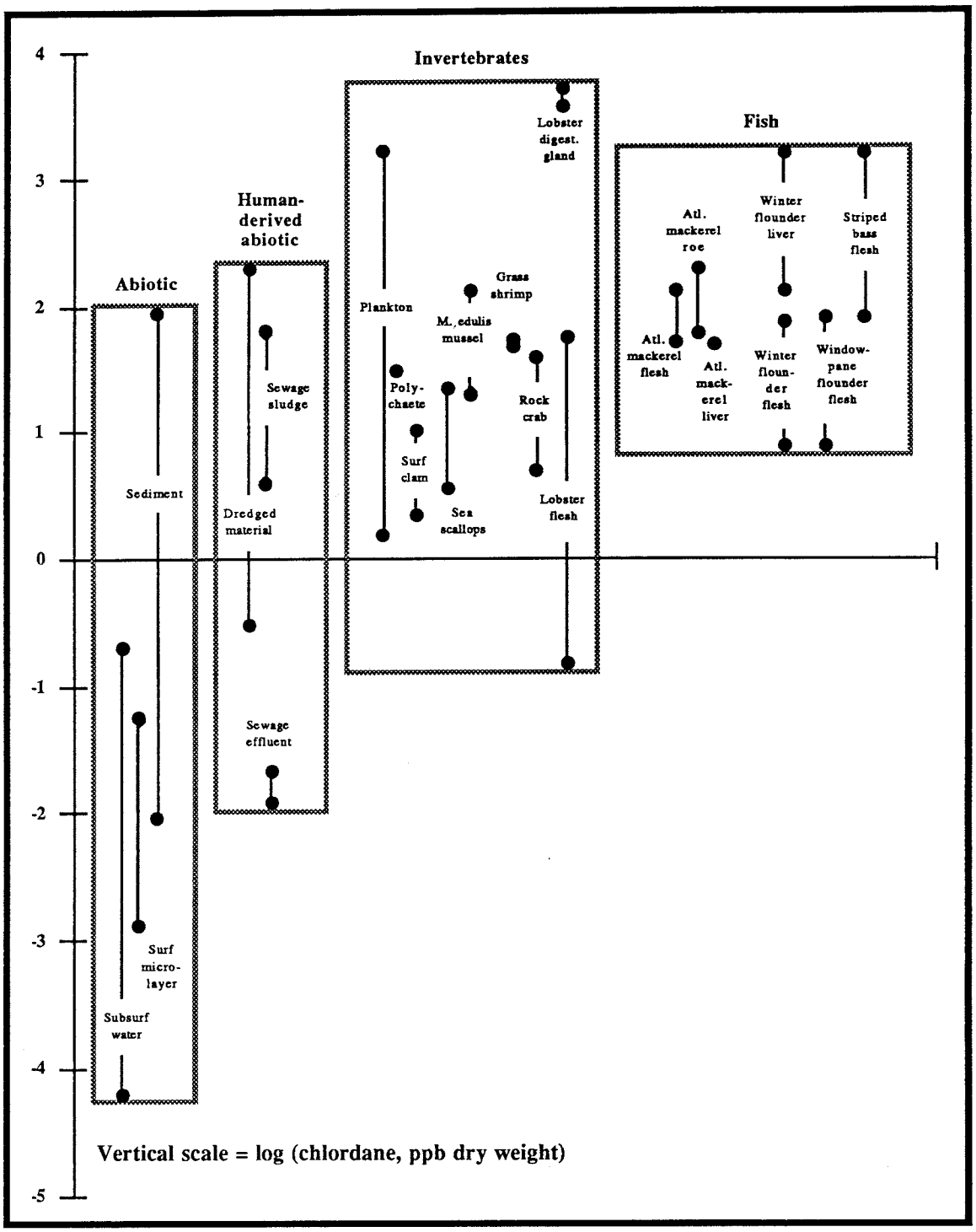

Figure 48. Chlordane concentration ranges by substrate sampled in the New York Bight region, 1977-1980. Source: MacLeod *et al.* (1981).

Contaminants in striped bass from the Hudson River and Marine District of New York State, 1982, 1985, and 1987. Chlorinated organic contamination of fish and wildlife resources in New York has been a cause of concern to state officials there for over two decades. As part of a continuing effort by the NYDEC to monitor and assess levels of PCBs, a major identified problem in the Hudson River, levels of chlordane were also measured in fillets of striped bass (*Morone saxitilis*) from the Marine District (marine and estuarine waters) of the State. Sloan and Horn (1986) contained summarized results for chlordane in muscle tissue of striped bass collected in the Hudson River near West Point, while Sloan *et al.* (1988) summarized results for striped bass collected in 1985 and 1987. Specific isomers of chlordane were not identified in the 1982 collection, although it was noted that heptachlor concentrations were <5 ppb for all samples. For 1985, measurements for chlordane (defined as the summation of several other chemicals, possibly including α-, γ-, and oxychlordane, *trans*-nonachlor, and compounds "C" and "E") and nonachlor (not specified as to isomeric composition) were given. In 1987, nonachlor alone was summarized for the striped bass tissue. Table 46 shows summarized results for 1982, 1985, and 1987.

Table 46. Chlordane and nonachlor concentrations measured in fillets of striped bass collected in marine waters of New York State in 1982, 1985, and 1987. Values in ppb wet weight. Source: Sloan and Horn (1986); Sloan *et al.* (1988).

Location	Month/ Season	Mean Chlordane	Chlordane Range	Chlordane N	Mean Nonachlor	Nonachlor Range	Nonachlor N
1982							
Hudson River West Point	May	160	50-500	26			
1985							
New York City Harbor area	Spring	70	<10-230	30	50	10-160	30
	Summer	120	20-830	30	90	20-750	30
	Fall	90	20-380	30	90	20-270	30
W. Long Island North shore	Spring	100	<10-550	30	80	10-550	30
	Summer	70	20-140	30	50	20-90	29
	Fall	60	20-180	30	60	20-180	30
W. Long Island South shore	Spring	140	<10-410	30	110	20-410	31
	Summer	80	10-180	21	60	10-130	21
	Fall	70	10-280	30	60	10-180	30
E. Long Island North shore	Spring	80	10-210	30	50	10-170	30
	Summer	40	10-100	19	30	10-60	19
	Fall	40	10-150	30	30	10-80	30
E. Long Island South shore	Spring	70	30-90	11	40	20-60	11
	Summer	50	10-230	22	30	10-140	22
	Fall	40	<10-120	30	30	10-80	30
1987							
New York City Harbor area	Spring				30	10-90	30
	Summer				30	<10-130	31
	Fall				30	10-60	27
W. Long Island North shore	Spring				30	10-160	61
	Summer				30	10-120	56
	Fall				20	10-90	45
W. Long Island South shore	Spring				20	10-40	46
	Summer				30	<10-180	72
	Fall				20	<10-160	68
E. Long Island North shore	Spring				20	10-330	55
	Summer				20	<10-100	65
	Fall				20	<10-130	54
E. Long Island South shore	Spring				20	10-70	51
	Summer				20	<10-80	65
	Fall				20	<10-90	64

Based on these data, it is clear that mean concentrations in striped bass fillets fell within FDA action level guidelines; however, some individual tissue samples were well above the 300 ppb limit. According to Sloan and Horn (1986), in 1982, 2 of 26 samples exceeded the FDA standard. By 1987, measured nonachlor concentrations appeared to indicate that chlordane levels in striped bass had declined substantially from 1982 and 1985 levels. Although chlordane was not measured in 1987, the upper range values for nonachlor concentrations had also declined: only one nonachlor maximum exceeded the FDA action level, compared to three in 1985

Toxic substances in New York fish and wildlife, 1977-1986. NYDEC (1978, 1979), Sloan (1981a, 1981b, and 1987) summarized results for a wide range of environmental studies undertaken by the NYDEC. While most of these involved freshwater species, some estuarine and marine samples were also taken, and some included analyses for chlordane compounds. Estuarine and marine fish species were not specifically analyzed for chlordane concentrations 1977-1979, although heptachlor was targeted and not found at levels above a detection limit of 10 ppb wet weight. Table 47 lists sites and species sampled between 1977 and 1979 for concentrations of heptachlor.

Table 47. Locations and species collected and analyzed for heptachlor in New York waters, 1977-1979. Values in ppb wet weight. Source: NYDEC (1978, 1979); Sloan (1981a, 1981b, and 1987).

Location	Species	Mean Heptachlor*	Heptachlor Range	Composite N	No. of Analyses
1977					
Hudson River					
Poughkeepsie	American shad	ND	-	2	1
Tappan Zee Bridge	American shad	ND	-	19	2
	Striped bass	ND	-	5	1
1978					
Long Island Sound					
New Rochelle	Winter flounder	ND	-	34	2
	Mummichog	ND	-	57	2
Huntington Harbor	Winter flounder	ND	-	33	2
	Mummichog	ND	-	93	2
Port Jefferson	Winter flounder	ND	-	56	2
	Mummichog	ND	-	71	2
Flushing Bay	Winter flounder	<10	-	13	1
	Mummichog	<10	-	12	1
	Bluefish	<10	-	1	1
	Blue crab claw	<10	-	5	1
Old Orchard Shoal	Winter flounder	<10	<10-<10	45	2
Atlantic Ocean					
Jamaica Bay	Winter flounder	ND	-	36	2
	Mummichog	ND	-	81	2
Southampton	Mummichog	ND	-	47	2
Moriches Bay	Winter flounder	ND	-	27	2
	Mummichog	ND	-	187	2
Shinnecock Bay	Winter flounder	ND	-	32	2
Great South Bay	Winter flounder	ND	-	36	2
	Mummichog	ND	-	64	2
Great Peconic Bay	Winter flounder	ND	-	37	2
	Mummichog	ND	-	141	2
Coney Island Creek	Mummichog	<10	-	31	1
Great Kills (Staten Is.)	Mummichog	<10	-	78	1

*Includes heptachlor epoxide.

Table 48 shows marine and estuarine results for chlordane reported in Sloan (1987).

Table 48. Chlordane concentrations measured in fish collected in estuarine and marine waters of New York State between 1980 and 1986. Values in ppb wet weight. Source: Sloan (1987).

Location	Species	Mean Chlordane	Chlordane Range	Composite N	No. of Analyses
1980					
Hudson River					
Poughkeepsie	American shad	20	20-30		
	Striped bass	130	-		
Tappan Zee Bridge	American shad	40	30-40		
	Striped bass	120	-		
City Island (NYC)	Winter flounder	30	-		
	Bluefish	30	-		
Verrazano Bridge	Winter flounder	50	-		
Arthur Kill, vicinity Con. Edison	Mummichog	110	80-120		
Atlantic Ocean, Rockaway Pt.	Winter flounder	20	-		
	Bluefish	80	-		
	Weakfish	50	-		
1981					
Flushing Bay	Mummichog	30	-	83	1
Little Bay	Bluefish	20	-	15	1
Huntington Bay	Winter flounder	10	10-10	25	2
	Mummichog	20	-	30	1
Port Jefferson Harbor	Winter flounder	10	10-10	25	2
	Mummichog	<10	-	30	1
Jamaica Bay	Winter flounder	10	10-20	20	2
	Mummichog	20	-	25	1
1982					
Flushing Bay	Winter flounder	40	30-50	8	2
	Striped bass	190	-	1	1
Great South Bay					
Captree State Park	Winter flounder	20	20-20	23	2
Great River	Mummichog	10	-	30	1
Moriches Bay					
Buoy 36	Winter flounder	20	10-20	23	2
Forge River	Mummichog	10	-	25	1
Shinnecock Bay	Winter flounder	30	20-30	25	2
Old Fort Pd.	Mummichog	10	-	25	1
Great Peconic Bay	Winter flounder	10	10-20	20	2
S. Jamesport Creek	Mummichog	10	-	25	1
1983					
Governors Island	American eel	210	110-420	3	2
East River	American eel	270	220-340	3	3
Queensboro Bridge					
Arthur Kill, near Con	American eel	700	560-830	2	2
Ed plant	Mummichog	110	-	15	1
Verrazano Narrows Bridge	American eel	280	200-610	15	3
Rockaway Point	Tautog	90	90-90	20	2
	American eel	100	-	1	1
1984					
Atlantic Ocean					
Smithtown Bay	Bluefish	100	10-150	8	8
Stonybrook	Bluefish	100	40-160	11	11
Jones Beach	Weakfish	60	-	1	1
Oak Beach	Weakfish	20	-	1	1
Robert Moses St. Pk.	Bluefish	30	10-100	17	17
Moriches Inlet	Weakfish	60	10-130	10	10
	Menhaden	70	20-120	5	5
Shinnecock Inlet	Bluefish	50	30-80	4	4
Montauk Point	Bluefish	110	30-210	26	26
Orient Point	Bluefish	80	20-200	5	5

Table 48. Continued

Location	Species	Mean Chlordane	Chlordane Range	Composite N	No. of Analyses
1986					
Hudson River Verrazano Bridge	American eel	250	60-510	20	20
Jamaica Bay Kings County	American eel	150	40-370	20	20
Barnums Island Channel, Oceanside Hempstead	American eel	130	40-290	20	20
South Oyster Bay Wantagh	American eel	120	30-510	20	20
Great South Bay Green Cr., Sayville	American eel	90	20-240	20	20

There clearly has been a large degree of variability in concentrations measured across different areas and species in New York waters. Unfortunately, the relatively small sample sizes within species and at given locations do not permit delineation of temporal or spatial trends. Highest concentrations of chlordane were measured in 1983, when the NYDEC sampled the urban waters around New York City collecting primarily American eel. At four of the five sites, the FDA action level of 300 ppb wet weight was exceeded in at least one composited sample. At the Arthur Kill site, located near a Consolidated Edison plant, both samples of American eel analyzed contained chlordane at concentrations well above the FDA limit (560 and 830 ppb). Samples of American eel appeared to contain relatively greater body burdens of chlordane than did other species, even in areas removed from urban harbors such as Great South Bay and South Oyster Bay.

NYDEC (1979) contains results for organochlorine and heavy metal analyses of mammals collected in 1978, and among these were concentrations of chlordane and heptachlor in liver and brain tissues of harbor seals (*Phoca vitulina richardsii*) collected in Suffolk County (eastern Long Island). These results are presented as Table 49.

Table 49. Concentration of chlordane and heptachlor in tissues of harbor seal (*Phoca vitulina richardsii*) from Suffolk County, New York, 1978. Values in ppb wet weight. Source: NYDEC (1979).

Species	Tissue	Mean Chlordn	Chlordn Range	Mean Heptaclr*	Heptaclr* Range	No. Analyzed
Harbor seal	Liver	<70	-	<20	-	1
	Brain	<60	-	<20	-	1
Harbor seal (juvenile)	Liver	<80	<70-<100	<20	<20-<20	2
	Brain	<60	<60-<60	<20	<20-<20	2

*Includes heptachlor epoxide.

Given the demonstrated propensity for organochlorines like chlordane compounds to accumulate in food webs, it is surprising that levels were not quantified in the seal tissues. Although concentrations would very likely have been easily measured in blubber tissue, had it been analyzed, the relatively high lipid content of liver and brain tissue also would have been expected to have been a ready matrix for higher levels.

New Jersey

Chlordane contamination in fish from New Jersey waterways, 1978. In April of 1978, a fire at a Burlington County, New Jersey, garden supply center resulted in an introduction of pesticide-contaminated water into nearby Pennsauken Creek and Strawbridge Lake and apparently caused a fish kill in those water bodies. Sampling efforts targeted on waters in and near this system revealed a degree of chlordane contamination in excess of that expected from the accident proper: 64 percent of the fish sampled in Pennsauken Creek (in the vicinity of Camden, New Jersey) had chlordane levels exceeding the FDA action level of 300 ppb. A study was therefore initiated to delineate the extent of chlordane contamination in New Jersey waters through analysis of sediment and edible fish tissues. Results from this study were summarized in Suchow, Lipsky, and Schulze (1980). Although the investigation targeted mostly freshwater tributaries flowing into the Delaware River, results are reported here because of possible downstream impacts on the Delaware Bay estuary, and because some target fish species were also examined in other studies taking place in the northeast Atlantic coast region.

Chlordane was specified as total chlordane, although no definition was provided. Suchow, Lipsky, and Schulze reported results for chlordane in fillets of nine fish species, two of which (American eel, *Anguilla rostrata* and white perch, *Morone americanus*) are common estuarine dwellers. The 56 sites sampled were grouped into "Camden area" and "outside Camden area" for comparison using the nonparametric Mann-Whitney test. Concentrations measured in all nine species are shown in Table 50.

Table 50. Total chlordane concentrations measured in muscle tissue of nine common fish species collected in New Jersey waters. Values in ng/g (ppb) wet weight. Source: Suchow, Lipsky, and Schulze (1980).

	N		Mean		Median		Range	
Species	"Camdn"	"Outsd"	"Camdn"	"Outsd"	"Camdn"	"Outsd"	"Camdn"	"Outsd"
American eel	20	13	2410	215	1720	211	ND-13900	ND-748
White perch	3	9	2960	61.2	376	33	154-8340	ND-178
Brown bullhead	19	11	460	46.4	280	17	ND-1720	ND-216
Carp	18	12	751	110	659	47	ND-1920	ND-556
Largemouth bass	7	10	160	66.1	65	33	ND-601	ND-213
Common sucker	4	12	448	52.7	325	29	ND-1261	ND-170
Pumpkinseed	8	5	334	37.4	73	40	ND-222	ND-61
Black crappie	8	5	164	83.2	81	15	ND-740	ND-288
Goldfish	10	3	712	224	489	264	45-1617	148-440

In comparing the Camden area with other areas sampled, it was found that residue levels for four of the nine species, including the two that range into estuarine waters, were significantly different for the two defined populations. American eel, brown bullhead, and carp were significantly different at p=0.01, while white perch were different at p=0.05. American eel populations showed a particularly high degree of contamination, with 80 percent of the Camden area sample and 30 percent of the outside Camden area sample exceeding the FDA action level of 300 ppb.

Some limited data for sediments were also available in Suchow, Lipsky, and Schulze. Analyses from 16 initial sediment samples collected from Pennsauken Creek showed a range of 64 to 3,649 ppb; other data from this investigation reported by Moser (1985) were from the Cooper River area, where the range of chlordane values was 35.3 to 198.2 ppb.

Storage and transport of sediments, pesticides, and PCBs in impounded fluvial systems in New Jersey. Moser (1985) characterized the depositional environments of an area in southern New Jersey that included the most highly chlordane-contaminated portions of the area examined by Suchow, Lipsky, and Schulze (1980). As part of her study, Moser measured the chlordane (α- and γ- isomers) concentrations of water and sediments at some of the same locations sampled by Suchow, Lipsky, and Schulze.

In the Cooper River, Moser found a sediment chlordane concentration range of not detected (<10 µg/kg) to 279 µg/kg (compared to 35 to 198 µg/kg in Suchow, Lipsky and Schulze); in Pennsauken Creek, Moser measured chlordane in the range of not detected to 973 µg/kg (compared to 64 to 3,649 µg/kg).

In eight Pennsauken Creek water samples, Moser found α-chlordane (γ-chlordane was measured less frequently) occurring in the range of not detected (<0.01 µg/l) to 0.36 µg/l. In sixteen Cooper River water samples, α-chlordane ranged from not detected to 0.08 µg/l; summed α- + γ-chlordane concentrations ranged from not detected to 0.13 µg/l.

Sediment cores were taken at impoundments associated with both the Cooper River and Pennsauken Creek systems. Although depositional rates and dating of strata presented a number of uncertainties, it is interesting to note the universal occurrence of chlordane residues at all depths down to 28 cm in the Evans Pond (Cooper River) core and 32.5 cm in the Middle Strawbridge Lake (Pennsauken Creek) core. Below these depths, no chlordane was measured despite the occurrence of other pesticide residues (DDT and BHC compounds).

New Jersey study of PCBs and chlordane in selected finfish, 1981-1982. Prompted by the discovery of widespread PCB contamination of fish and sediments in the upper Hudson River of New York state in the mid-1970s, the NJDEP initiated a comprehensive survey of the lower portions of the Hudson and other waters of the state. Results for the first 5 years (1975-1981) of the annual sampling effort were summarized in Belton, Ruppel, and Lockwood (1982), but the report contained results for PCBs only. In 1983, NJDEP released a followup report (Belton *et al.*, 1983) presenting the results of subsequent sampling in 1981 and 1982 which included limited data for chlordane in bluefish as well. The original intent of NJDEP was to include chlordane results in a separate report with other pesticide measurements; however, the levels of chlordane found in bluefish and other popular sport and commercial species mandated a more timely release of information, *i.e.*, through inclusion with the PCB results. Moreover, there was concern about possible synergistic effects between PCB and chlordane burdens in edible tissues of fish. As a result of the findings of this study, NJDEP issued emergency fishing closures or advisories for the taking of striped bass, American eel, bluefish, white perch, and white catfish, primarily from industrialized waters in the northeastern portion of New Jersey Hudson-Newark-Raritan area). Chlordane data presented in Belton *et al.* (1983) are shown in graphic and tabular form as Figure 49 and Table 51.

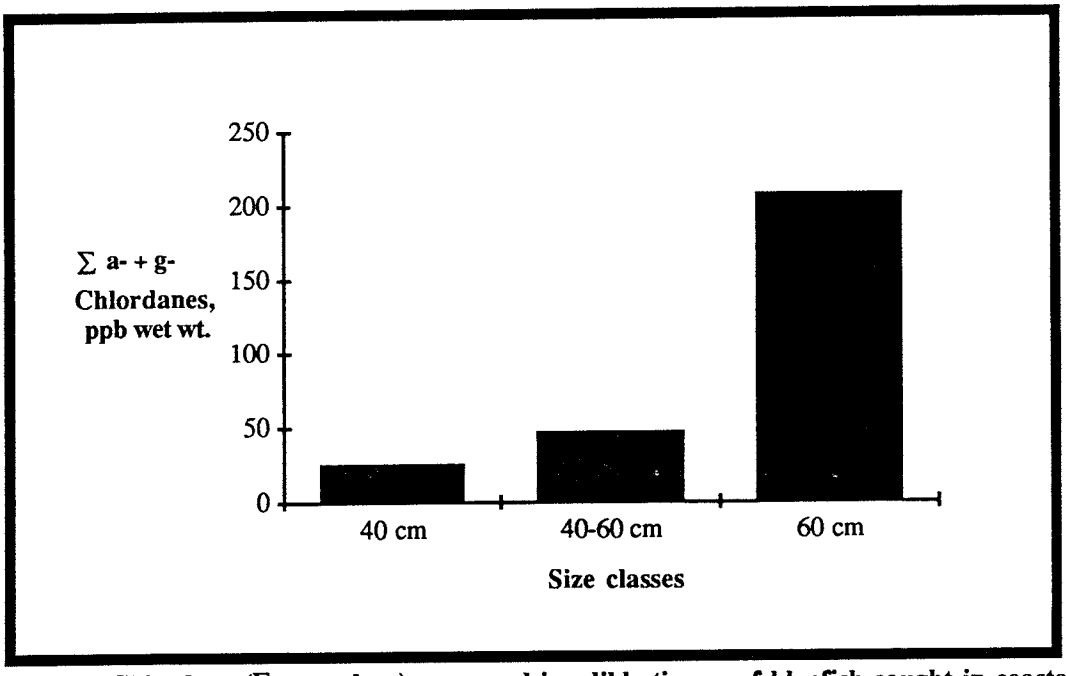

Figure 49. Chlordane (Σ α- and γ-) measured in edible tissues of bluefish caught in coastal and estuarine waters of New Jersey, 1982 by size class. Values in μg/kg (ppb) wet weight. Source: Belton *et al.* (1983).

Figure 49 clearly suggests a positive relationship between body burdens of chlordane occurring in edible tissues of bluefish and size of organism. Belton *et al.* noted that while the overall mean for chlordane in large fish (>60 cm) did not exceed the FDA action level of 300 ppb wet weight, 29 percent of the individuals were above the concentration.

Table 51 shows bluefish chlordane results for both 1981 and 1982 in different portions of New Jersey waters. The large decline in concentrations between 1981 and 1982 at the Hudson River and at Raritan/Sandy Hook locations is interesting, and not explainable with the summary information provided in Belton *et al.* Whether or not this represented a real decline in tissue burdens must be evaluated against size of fish sampled (as was shown by Figure 49) and other factors. Also of interest is the relatively high mean concentration and range of values found in bluefish sampled in coastal waters: the mean concentration of 126 ppb wet weight was more than three and four times that for the fish collected in the Hudson River and Raritan Bay, respectively. Belton *et al.* commented that the complexities of life history, migration, and seasonality make it difficult to assess where and how bluefish become contaminated, but it is of some interest to note that PCB contamination levels in the same samples of bluefish did not show the same degree of disparity among the three areas (mean PCB burdens were 1,550, 460, and 1,730 ppb for Hudson River, Raritan, and coastal fish, respectively).

Table 51. Results for chlordane (Σ α- and γ-) measured in edible tissues of bluefish caught in 1981 and 1982 in New Jersey waters. by general location. Values in μg/kg (ppb) wet weight. Source: Belton *et al.* (1983).

Sample Location	Year	Mean Chlordn*	Chlordn Range	No. of Fish	No. Analyses
Hudson River	1981	205.9	127.4-284.5	3	2
	1982	36.93	30.4-50.0	11	3
Raritan, Sandy Hook bays	1981	110.1	2.5-232.1	12	4
	1982	27.75	18.21-39.03	20	4
Coastal Waters†	1982	125.7	11.21-357.6	123	28

* Defined as the FDA mean, arithmetic mean giving equal weight to both single and composite samples.

† Coastal waters include sites from Sandy Hook south to Barnegat Inlet, from the surf line to 30 miles offshore.

New Jersey study of dioxin in aquatic animals and sediments, 1983-1984. In 1983, the NJDEP discovered extensive soil contamination of a terrestrial site (the former Diamond-Alkali site) located near Newark by the highly toxic compound dioxin (2,3,7,8-tetrachlorodibenzo-*p*-dioxin). Because the Passaic River flows past the impacted site, the Office of Science and Research within NJDEP collected and analyzed sediments and biota in the tidal river and found that both contained relatively elevated levels of dioxin. Based on these results, a more intensive investigation was initiated to determine the extent of contamination. Details and results of the studies were reported in Belton *et al.* (1985a).

Although the focus of the report was dioxin contamination in New Jersey coastal waters, some discussion of pesticide levels in biota was included as well. Results for levels of chlordane (α- and γ-chlordane) in tissues of American lobsters (*Homarus americanus*) caught in New Jersey waters were among the pesticide concentrations reported. Those chlordane results are summarized in Table 52.

Table 52. Results for chlordane (Σ α- and γ-) measured in tissues of American lobster sampled in New Jersey waters in 1982 and 1984. Values in µg/kg (ppb) wet weight. Source: Belton *et al.* (1985a).

Location	Date	Sample Size	Tissue Type	Chlordane ppb wet weight
Offshore - Mud Hole	11/16/82	5	Hepatopancreas	74.40
	11/16/82	5	Hepatopancreas	48.89
	11/16/82	5	Hepatopancreas	84.57
	11/16/82	5	Hepatopancreas	71.80
	11/16/82	5	Hepatopancreas	87.84
	11/16/82	10	Hepatopancreas	31.89
	11/29/82	15	Hepatopancreas	45.61
	11/29/82	15	Hepatopancreas	44.15
Raritan Bay - Chapel Hill Channel	7/20/84	5	Muscle and hepatopancreas	11.00
	7/20/84	5	Muscle and hepatopancreas	10.17

The offshore lobster samples were collected on the eastern slope of the "Mud Hole," approximately 20 miles east of Long Branch. Unfortunately, these were opportunistic samples for which only hepatopancreas tissue samples were available for chlordane analyses. However, hepatopancreas--"tomally"-- is an edible tissue of the lobster. Based on results for other contaminants analyzed and on additional results cited by Belton *et al.*, chlorinated organic compounds would be expected to be present at substantially higher concentrations in hepatopancreas tissue than in muscle tissue. The highest chlordane concentration measured in hepatopancreas was well below the FDA action level of 0.3 ppm wet weight.

New Jersey study of toxic hazards to urban recreational fishermen and crabbers, 1982-1983. As a follow-up to previous public health advisories, and in order to evaluate possible health risks to recreational consumers of fish and shellfish taken from the Hudson River-Upper Bay-Newark Bay system, the NJDEP sponsored a study (Belton *et al.*, 1985b) which measured chlorinated organic contaminants in resident seafood species and then estimated levels of risk based upon consumption patterns of fishermen. Concentrations of summed chlordane isomers (specified as α– and γ–chlordanes) were among those organochlorines measured, in various tissues of bluefish (*Pomatomus saltatrix*), striped bass (*Morone saxatilis*), blue crab (*Callinectes sapidus*), American eel (*Anguilla rostrata*), white perch (*Morone americana*), Atlantic menhaden (*Brevoortia tyrannus*), Atlantic tomcod (*Microgadus tomcod*), white catfish (*Ictalurus catus*), winter flounder (*Pseudopleuronectes americanus*), blueback herring (*Alosa aestivalis*), alewife (*Alosa pseudoharengus*), killifish (*Fundulus sp.*?), and American shad (*Alosa sapidissima*). Table 53 shows chlordane results reported in Belton *et al.* (1985b).

Table 53. Results for chlordane (Σ α– and γ–) measured in various tissues of fish sampled in upper New Jersey waters in 1982 and 1983. Values in µg/kg (ppb) wet weight. Source: Belton *et al.* (1985b).

Station	Species	Date	Sample Size	Tissue Type	Chlordane* ppb wet weight
Hudson River- Alpine	White perch	12/1/83	5	Muscle	31.44
		12/1/83	5	Muscle	45.27
	Atlantic menhaden	5/26/83	5	Whole	49.04
Hudson River- Piermont	Striped bass	10/18/83	1	Muscle	53.44
	White perch	12/4/83	5	Muscle	71.62
		12/4/83	5	Muscle	52.87
Hudson River- George Washington Bridge	Blue crab	9/28/82	5	Mix	31.25
		8/2/83	1	Mix	21.14
	White perch	11/29/83	5	Muscle	74.81
		11/29/83	5	Muscle	65.63
Hudson River- Bayonne to Weehawken	Bluefish	10/14/82	5	Muscle	108.7
	Striped bass	12/9/83	1	Muscle	ND
		4/13/83	5	Muscle	135.8
		10/19/83	1	Muscle	38.27
		12/9/83	1	Muscle	ND
		4/22/83	5	Muscle	29.32
		12/9/83	5	Muscle	38.27
		4/13/83	5	Muscle	24.00
		7/19/83	4	Liver	67.02
		12/9/83	5	Liver	167.7
	American eel	4/12/83	5	Muscle	133.6
		4/12/83	5	Muscle	128.6
	White perch	4/22/83	5	Muscle	77.01
		4/22/83	5	Muscle	56.77
		4/22/83	5	Muscle	32.07
		4/22/83	5	Muscle	76.54
		12/9/83	5	Muscle	40.02
		12/9/83	5	Muscle	33.16
		12/9/83	5	Liver	88.63
		12/9/83	5	Roe	53.30
		10/18/83	1	Muscle	62.20
		10/18/83	5	Muscle	63.55
	Atlantic tomcod	4/22/83	5	Whole	9.27
	White catfish	4/28/83	5	Muscle	99.43
		4/28/83	5	Muscle	108.1
	Winter flounder	4/12/83	4	Muscle	10.29
	Blueback herring	4/22/83	5	Whole	22.98
Hudson River- NY Bay, Caven Cove	Bluefish	6/13/83	1	Muscle	130.8
		5/26/83	1	Muscle	142.1
		6/13/83	1	Liver	535.7
	Striped bass	5/25/83	1	Muscle	30.61
		8/27/82	5	Muscle	29.03
		5/26/83	4	Muscle	45.87
		8/27/82	5	Muscle	27.44
Hudson River- NY Bay, Caven Cove	Striped bass	4/26/83	5	Liver	198.1
	American eel	8/17/82	5	Muscle	53.47
		8/17/82	5	Muscle	50.32
	Blue crab	7/6/83	1	Mix	20.49
		7/6/83	1	Mix	25.61
		8/17/82	3	Hepatopancreas	105.4
		7/6/83	5	Egg sponge	32.10
		5/26/83	5	Muscle	27.49
		7/6/83	5	Muscle	5.06
		5/26/83	5	Hepatopancreas	54.16

Table 53 continued

Station	Species	Date	Sample Size	Tissue Type	Chlordane* ppb wet weight
New York Bay-	Striped bass	10/19/83	1	Muscle	38.55
Kill Van Kull		10/19/83	1	Muscle	56.85
		10/19/83	1	Muscle	32.73
		10/19/83	1	Muscle	91.44
		10/19/83	1	Muscle	34.55
		10/19/83	5	Liver	28.70
		10/19/83	5	Liver	153.3
	American eel	8/26/83	1	Muscle	140.5
	Blue crab	8/27/83	1	Mix	58.58
		8/27/83	1	Mix	130.7
		8/27/83	1	Mix	16.24
		8/27/83	1	Mix	19.03
		8/27/83	1	Mix	49.03
Hackensack River-	Blue crab	8/20/82	5	Mix	27.66
Laurel Hill		8/20/82	5	Muscle	ND
		8/20/82	5	Hepatopancreas	118.3
		8/20/82	1	Mix	15.87
		8/20/82	5	Mix	23.84
Hackensack River-	Blue crab	8/18/82	5	Mix	41.33
Route 3 Bridge		8/18/82	5	Mix	46.61
Rahway River-	American eel	10/18/83	1	Muscle	180.3
Arthur Kill Confluence		7/19/83	4	Liver	217.3
		10/18/83	1	Liver	35.69
		10/18/83	1	Muscle	266.1
	Killifish	10/18/83	5	Whole	166.7
	American shad (juv)	10/18/83	5	Whole	178.0
Newark Bay-	Striped bass	6/14/83	5	Muscle	50.14
Bayonne		6/14/83	5	Muscle	55.10
Newark Bay-	Bluefish	8/24/83	3	Muscle	83.74
Shooter's Island	Striped bass	10/18/83	1	Muscle	242.9
		10/18/83	4	Muscle	79.93
		10/18/83	1	Muscle	95.66
	American eel	5/26/82	5	Muscle	46.29
		5/26/82	5	Muscle	12.61
		6/13/83	5	Muscle	141.8
		6/13/83	5	Muscle	142.6
		6/13/83	5	Muscle	110.1
		5/26/82	5	Muscle	52.68
	Alewife	10/18/83	5	Whole	100.7
Newark Bay-	Striped bass	6/14/83	5	Muscle	101.5
NJ Central RR Trestle		6/14/83	5	Muscle	78.85
		6/14/83	5	Muscle	96.51
		6/14/83	5	Muscle	76.72
		6/14/83	1	Muscle	52.00
	Blue crab	8/17/82	5	Mix	23.39
		8/17/82	5	Mix	23.47
		8/17/82	5	Muscle	ND
		8/17/82	5	Muscle	3.11
		8/17/82	5	Muscle	ND
		8/17/82	5	Mix	34.91
		8/17/82	5	Mix	45.90
		8/17/82	5	Muscle	ND
Newark Bay-		8/17/82	5	Hepatopancreas	119.7
NJ Central RR Trestle		8/17/82	5	Muscle	ND
Arthur Kill-	Bluefish	10/18/83	1	Muscle	44.82
Route 440 Bridge					

*defined as the FDA mean, arithmetic mean giving equal weight to both single and composite samples.
†Coastal waters include sites from Sandy Hook south to Barnegat Inlet, from the surf line to 30 miles offshore.

Maryland

Chester River Study, 1972. In addition to the previously discussed laboratory studies conducted under the auspices of the Chester River Study, a series of samples was collected from the river for analysis of chlorinated hydrocarbon content. Among the compounds targeted were α- and γ-chlordane. The samples included water, bottom sediments, suspended sediments, and biota. Results from analysis of these collections were reported in Munson (1972).

Attempts to quantify chlordane concentrations in water samples taken in the Chester River were unsuccessful. When no traces of chlordane were found in 1-gallon samples, 5-gallon samples were analyzed with the same results. According to Munson, the analytical detection limit for chlordane was approximately 5 ppt, and accordingly, the chlordane concentrations of river water and suspended particulate material were presumed to be less than this value.

Results for chlordane concentrations measured in Chester River sediments are summarized in Table 54. Munson encountered difficulties with analysis of sediment samples, and the set of results was not as comprehensive as it might have been. Nevertheless, analysis of chlordane concentration in sediments *vs.* sediment grain size appeared to show an inverse relationship between the two parameters. However, the data were presented in graphical form only, and because underlying numerical values did not accompany the plot, it was not possible to apply statistical tests to evaluate the degree of correlation. The graphic representation of these results is reproduced here as Figure 50.

Table 54. Summary data for chlordane concentrations in sediments collected in the Chester River, 1971-1972. Values in ppb. Source: Munson (1972).

	November			April			June		
	Mean	Sdev	Range	Mean	Sdev	Range	Mean	Sdev	Range
Chlordane, ppb	1.7	3.9	nd-10	4.5	2.4	nd-9.0	9.3	4.9	nd-21

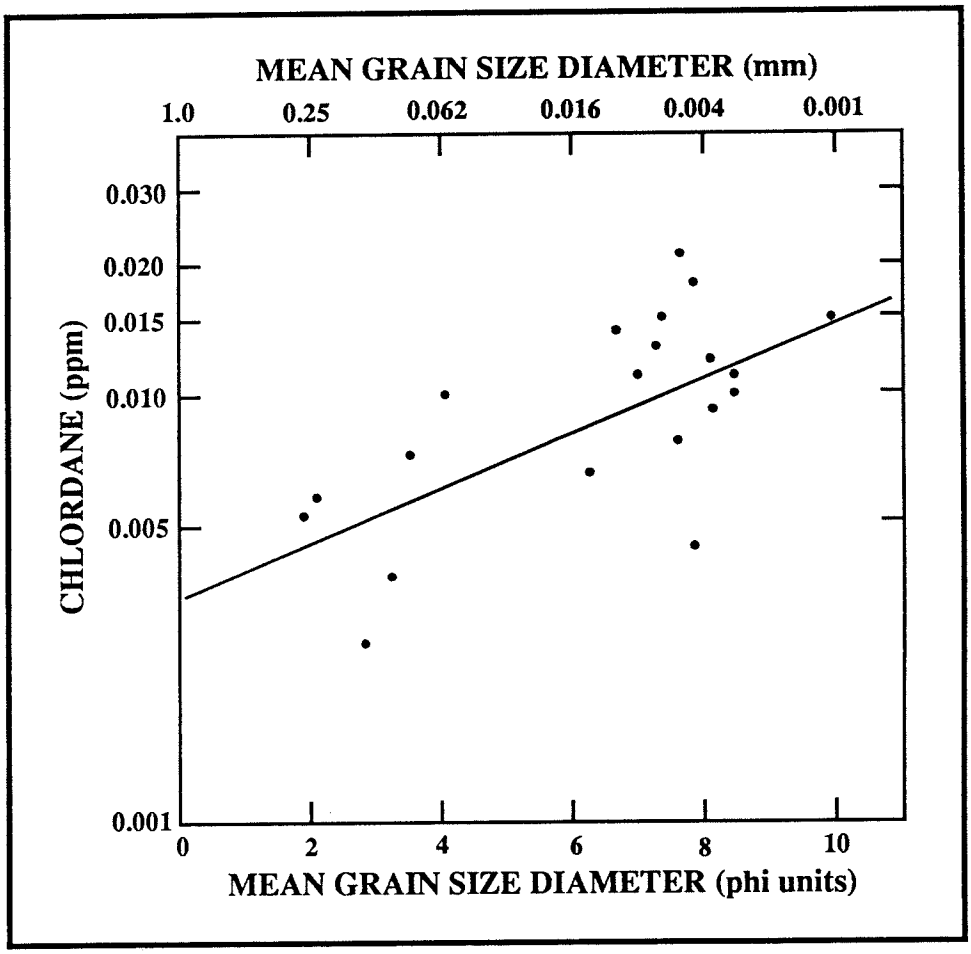

Figure 50. Log chlordane concentration vs. mean grain size diameter of sediments collected from the Chester River in April 1972. Source: Munson (1972).

Chlordane concentrations and grain size measurements in a sediment core taken near the mouth of the Chester River were also presented by Munson. Sediment concentrations of chlordane measured in the core were specified, although grain size data were presented in graphic format only. Both are illustrated in Figure 51. Munson used these core data as supporting evidence for the existence of an inverse relationship between chlordane concentrations and sediment grain size. That is, the occurrence of smaller fraction sediment grain size appeared to correspond to elevated chlordane concentration in the sediment. However, whether actual temporal variability in environmental levels of chlordane that also might account for some peaks in the sediment core record was not addressed by Munson. It is, therefore, difficult to speculate whether the chlordane concentrations in the core result from levels of chlordane present in the environment, or from sediment grain size considerations. Although the nature of the data presentation does not allow a very sophisticated analysis, it appears that normalization of the chlordane data to sediment grain size would remove a large degree of the variability in concentration and give a normalized core profile with minimal trend in chlordane concentration. Given the time of sampling (1972, prior to substantive use restrictions imposed by the EPA), this could in fact be an accurate reflection of chlordane concentrations in the environment.

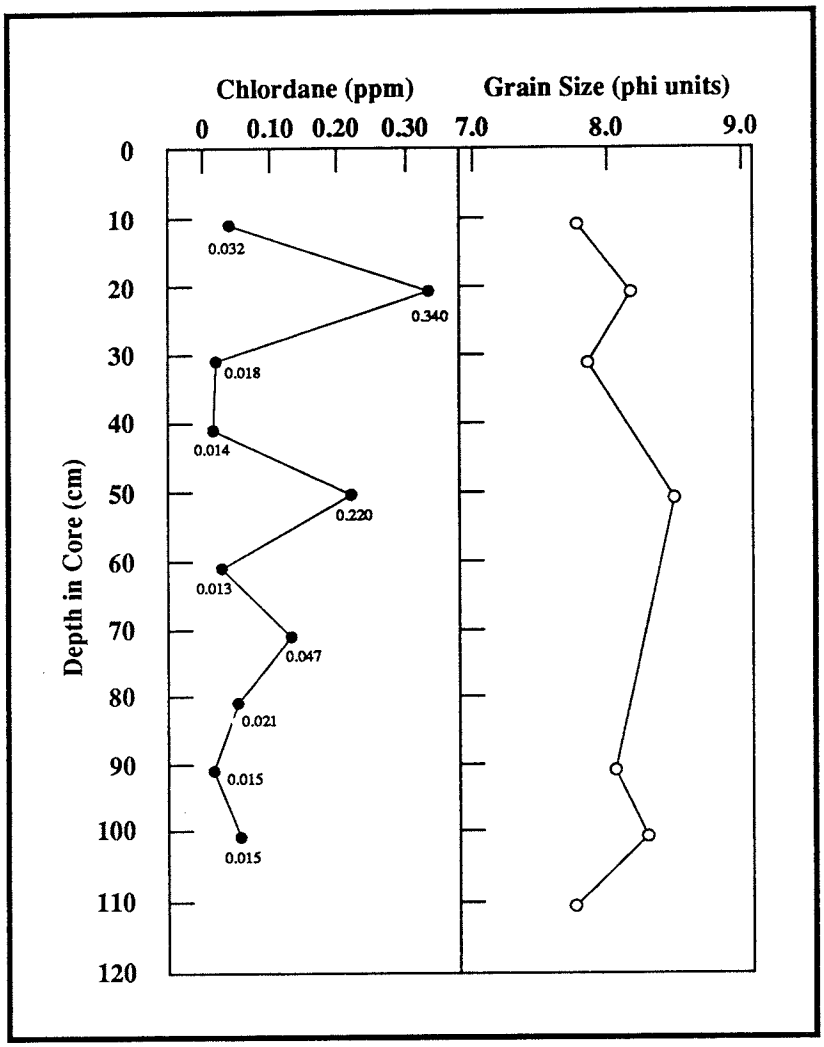

Figure 51. Chlordane concentration and mean grain size diameter measured in a sediment core collected in the Chester River, 1972. Source: Munson (1972).

In addition to sediments, biological samples were also collected and analyzed for the Chester River Study. These included oysters (*Crassostrea virginica*), soft-shelled clams (*Mya arenaria*), blue crabs (*Callinectes sapidus*), and unidentified fish. Summarized data were presented in Munson (1972) and are reproduced here as Table 55. Again, because the data are highly summarized, it is difficult to assess their significance. In fact, Munson noted that these averaged values masked seasonal differences, intraspecific differences, and differences related to location in the river environment.

Table 55. Summary data for chlordane concentrations in biota collected in the Chester River, 1971-1972. Values in ppb. Source: Munson (1972).

	Oysters		Soft Clams		Fish		Crabs	
	Mean	Range	Mean	Range	Mean	Range	Mean	Range
Chlordane, ppb	36	9-160	14	nd-38	74	34-180	14	3-24

None of the measurements for chlordane in biota exceeded FDA action levels, despite the fact that whole soft body tissues and not simply edible tissues were analyzed. Fish and oysters apparently accumulated chlordane to a greater degree than did soft-shelled clams and blue crab. Munson attributed this to higher lipid content and the feeding habits of the former.

Upper Bay Survey, 1975. The concept of this study was adapted from that of the Chester River Study, discussed previously. In the Chester River effort, sediments from the upper portion of the Chesapeake Bay were indicated as potential sources of toxicity to organisms in the Chester River. In 1973, the scope of the original study was expanded to cover a larger area and included additional tests and was called the Upper Bay Survey. Objectives of the Upper Bay Survey were to characterize concentrations and transport mechanisms for chlorinated hydrocarbons, examine impacts of chlorinated hydrocarbon exposure on commercially important fish and shellfish species, perform bacteriological studies with particulate material, develop models for predicting contaminant distribution in the bay, and facilitate access to results via computer retrieval. Results were reported in Munson (1975).

In addition to the chlordane distribution and transport mechanisms investigated by Tzou (1975) and discussed in a previous chapter of this review, Munson (1975) measured levels of chlordane (identified as the sum of α- and γ- isomers) in water, on suspended sediment, and in several species of shellfish sampled in the Upper Bay. Figure 52 illustrates the general geographic coverage and principal sampling sites for the Upper Bay Survey. Figure 53 shows average chlordane concentrations measured at selected sites (not all sites designated in Figure 53 were identified on the reference maps) in several media: on suspended sediment, in water on suspended sediment; on plankton; and in water on plankton. Figure 54 shows distribution of results by month sampled.

These figures illustrate the high degree of variety in chlordane concentrations found by Munson with respect to location within the upper portion of the Chesapeake Bay and with respect to sampling period within the year. In Figure 53, the occurrence of relatively higher concentrations of chlordane found in all media at sites 7a and 7b (the Patapsco River site near Baltimore) affirmed the characterization of the compound as an urban contaminant. These concentrations, and results for stations 8a, 8b, and 8c, suggested the urban area around Baltimore as a source for higher levels of chlordane that in turn fed into the Chesapeake Bay via the Patapsco River. Munson speculated that chlorinated hydrocarbons, like the chlordanes, may be trapped out of the overall system by the high depositional rate in Baltimore Harbor.

Figure 54, illustrating temporal variation in chlordane concentrations measured between December 1973 and September 1974, shows less consistent patterns of relative concentration across media. The apparently elevated levels measured in water on zooplankton in December through March were reflected to some extent in zooplankton, but not on suspended sediment or in water on suspended sediment. The terminology used here by Munson was not fully explained, but presumably "water on suspended sediment" and "water on zooplankton" refer to interstitial water and/or water adhering to outer surfaces of sediment and zooplankton.

Munson also measured α- and γ-chlordanes in four species of bivalves and in blue crab. Results of these analyses are shown in Table 56. It is not clear whether analyses were performed on individual organisms or on composited samples. Concentrations across the different species ranged from not detected in eastern oyster at two sites, to 0.15 ppm wet weight in blue crab collected near the mouth of the Magothy River. All measurements were well below the FDA action level of 0.3 ppm.

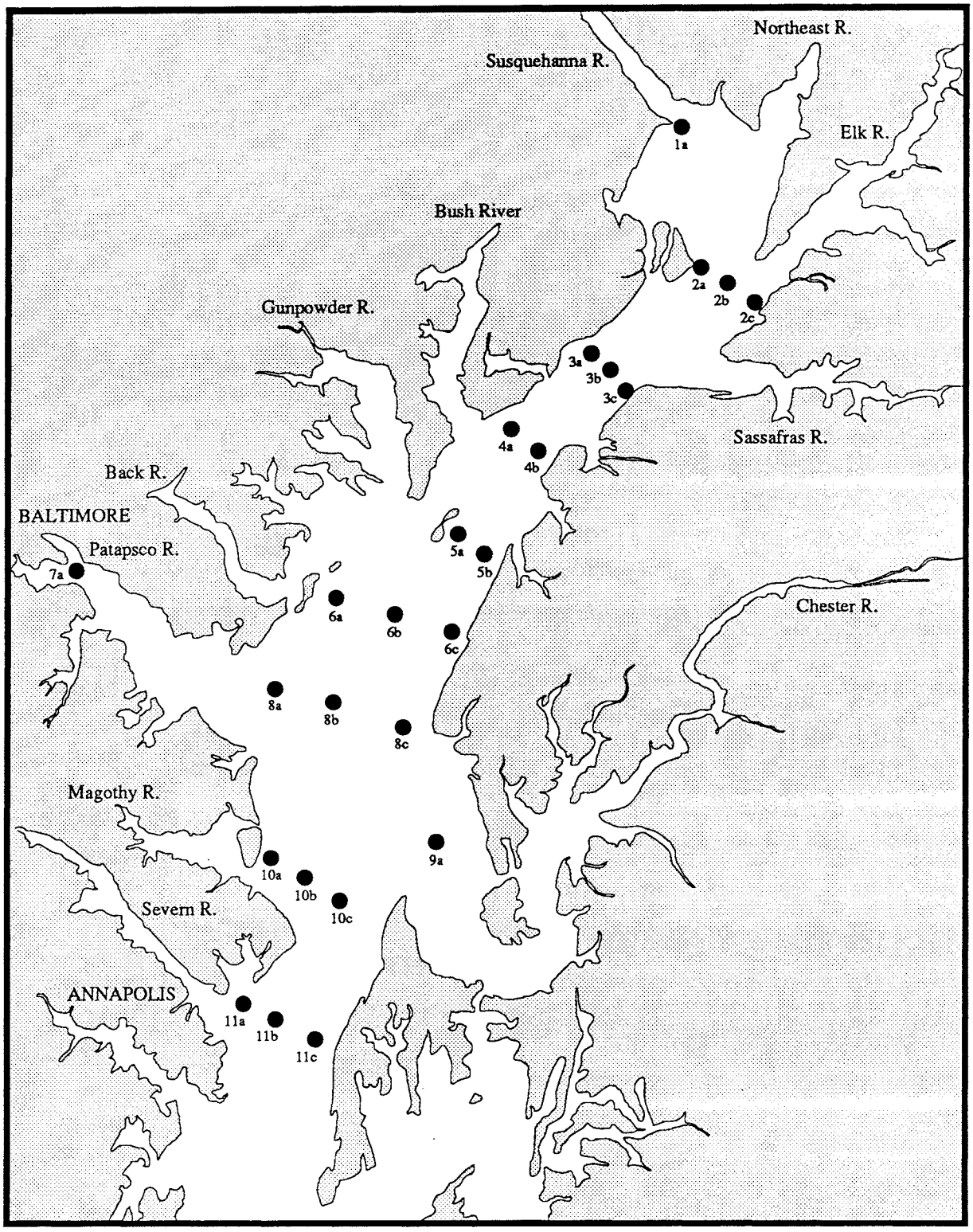

Figure 52. Principal site locations for the Upper Bay Survey, 1973-1974. Source: Munson (1975).

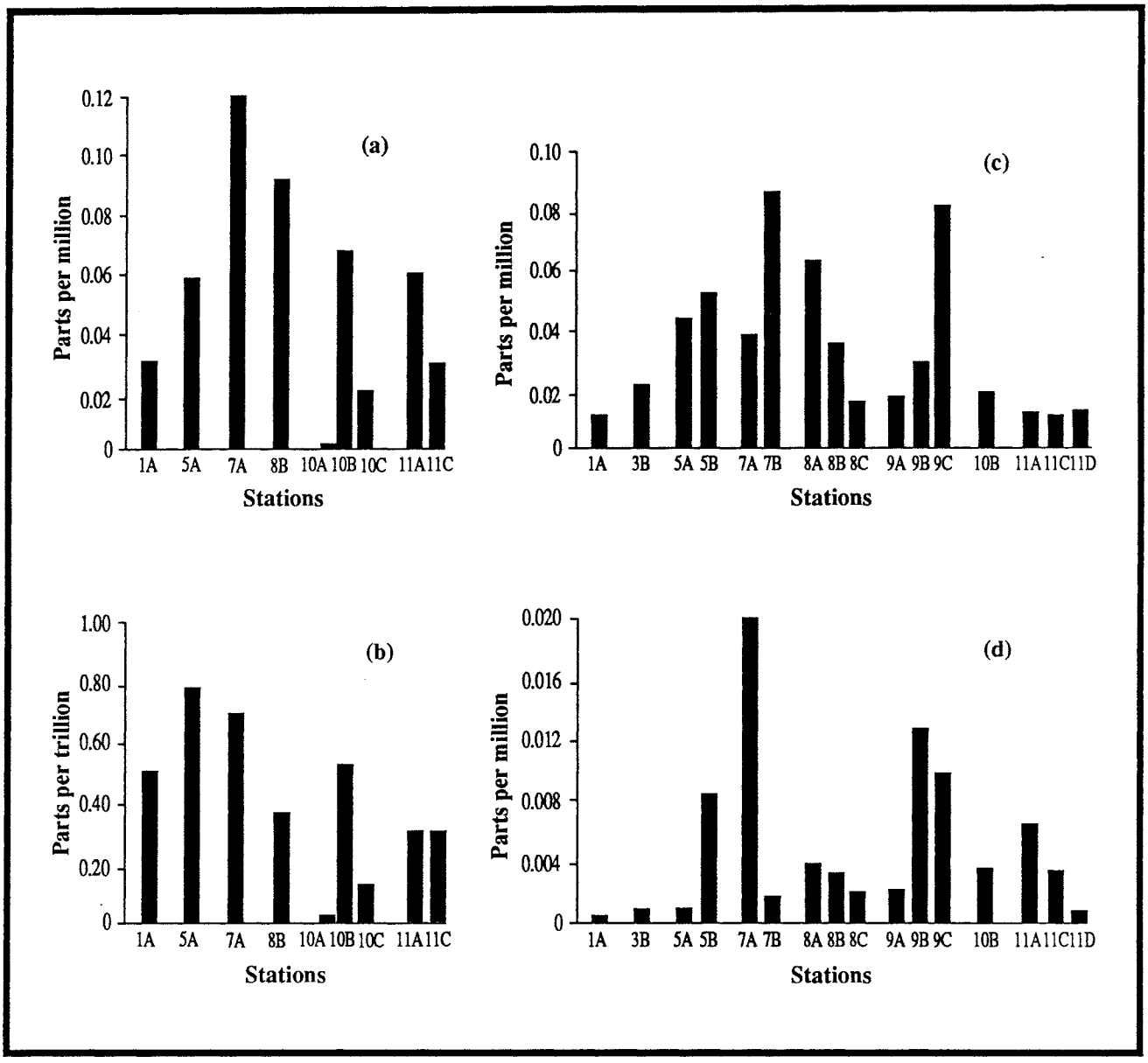

Figure 53. Average chlordane concentrations by station in the upper portion of the Chesapeake Bay. (a) on suspended sediment; (b) in water on suspended sediment; (c) in zooplankton; (d) in water on zooplankton. Refer to Figure 51 for site locations; not all site locations were shown in reference figures. Source: Munson (1975).

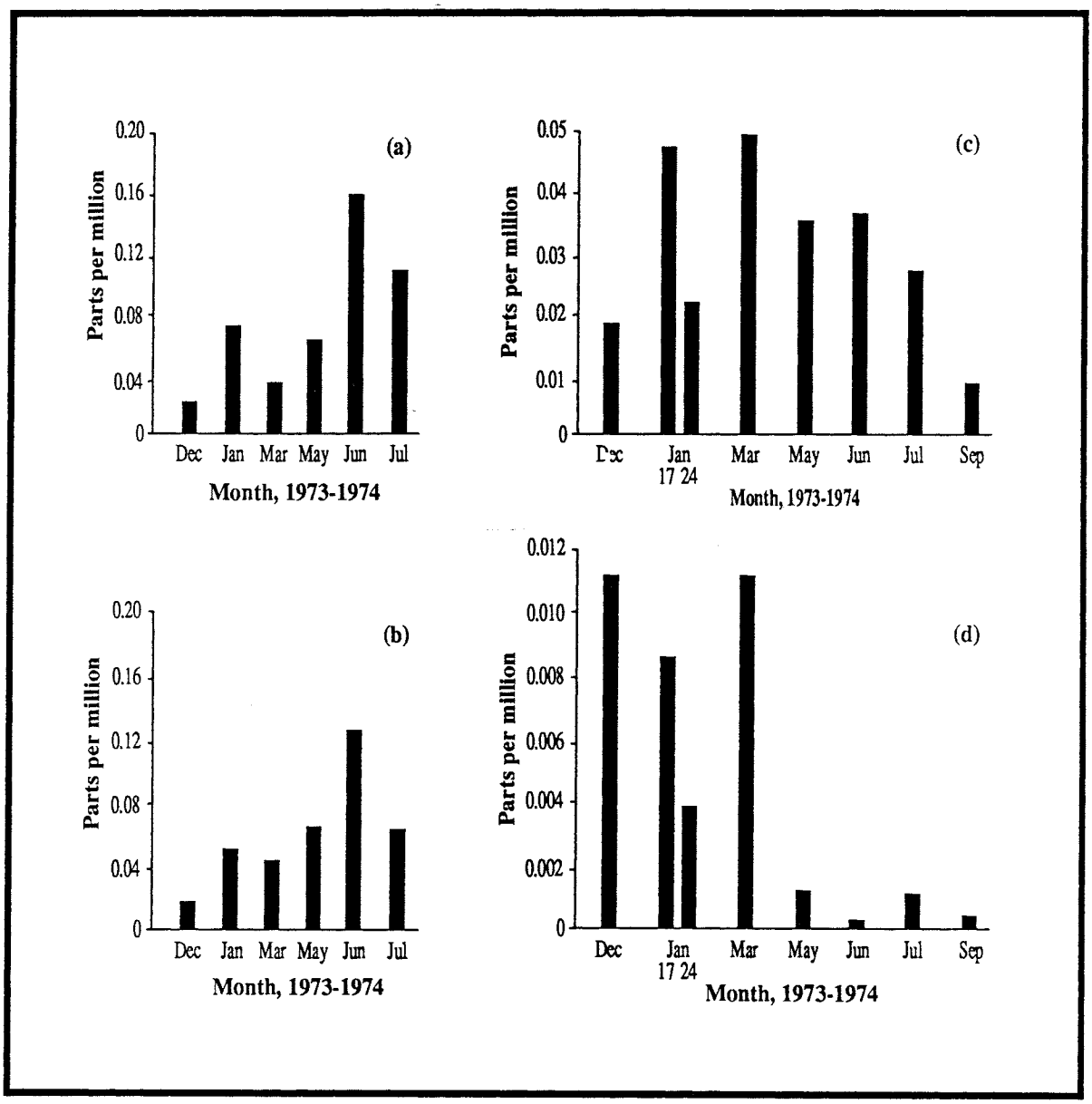

Figure 54. Average chlordane concentrations by month, 1973-1974, in the upper portion of the Chesapeake Bay. (a) on suspended sediment; (b) in water on suspended sediment; (c) in zooplankton; (d) in water on zooplankton. Source: Munson (1975).

Table 56. Results for chlordane (Σ α- and γ-) measured in various tissues of shellfish sampled in upper Chesapeake Bay waters in 1974. Values in mg/kg (ppm) wet weight. Source: Munson (1975).

Organism	Station	Date	Chlordane, ppm wet weight
Rangia cuneata (Atlantic rangia)	4A	9/20/74	0.0092
	5C	5/3/74	0.0040
	5C	8/2/74	0.0100
	6A	3/22/74	0.016
	6A	5/3/74	0.054
	6A	8/2/74	0.064
	6A	9/20/74	0.0066
	6B	3/22/74	0.025
	6C	3/22/74	0.016
	6C	5/3/74	0.0002
	6C	8/2/74	0.0059
	6C	9/20/74	0.0042
	8A	3/22/74	0.017
	8A	8/2/74	0.0017
	8A	8/26/74	0.033
	8A	9/20/74	0.012
	8B	3/20/74	0.011
	10A	5/3/74	0.0094
	10A	9/20/74	0.051
Crassostrea virginica (eastern oyster)	8B	1/25/74	0
	10A	1/25/74	0.0098
	10A	3/22/74	0
	10A	8/2/74	0.0099
	11A	3/20/74	0.017
Brachiodontes recurvus (mussel)	8A	3/22/74	0.016
Macoma sp.	10A	8/2/74	0.0099
Bivalves, all			0.016
Callinectes sapidus (blue crab)	10A	3/2/74	0.15

A summary table of information from Munson (1975) follows as Table 57. In evaluating these results, differences in units of measurement (*e.g.*, ppm or ppt; wet weight or dry weight) should be taken into account. For those matrices measured in both studies, values obtained by Munson (1975) for the Upper Bay Survey were roughly equivalent to those in Munson (1972), the Chester River Study.

Table 57. Summary of results for chlordane (Σ α- and γ-) measured in various media from the upper Chesapeake Bay in 1973 and 1974. Note difference in units among media. Source: Munson (1975).

Sample Type	No. of Samples	Chlordane, ppm wet weight	Std. dev.
Bivalves (ppm, wet weight)[1]	26	0.016	0.017
Plankton (ppm, wet weight)	70	0.041	0.032
Suspended sediment (ppm, dry weight)[2]	66	0.061	0.086
Bottom sediment (ppm, dry weight)	54	0.0052	0.014
Plankton (ppt, in water)[3]	69	0.0038	0.0083
Suspended sediment (ppt, in water)	68	0.53	0.88

[1]µg chlordane per gram wet weight of material extracted.
[2]µg chlordane per gram dry weight of material extracted.
[3]ng chlordane per liter of water filtered to collect material extracted.

Heavy metal, polychlorinated biphenyl and pesticide levels in shellfish and finfish from Maryland waters, 1976-1980. The Maryland Department of Health and Mental Hygiene sponsored a program which monitored the edible shellfish and finfish of the Chesapeake Bay and its tributaries for a number of contaminants, including chlordane (constituents not specified) and heptachlor. The program was designed to provide both information on the suitability of the resources for human consumption as well as longer term information on the ecological health of the Chesapeake Bay system. Four shellfish species were sampled (eastern oyster, *Crassostrea virginica*; soft-shell clam, *Mya arenaria*; hard-shell clam, *Mercenaria mercenaria*; and blue crab, *Callinectes sapidus*) as were a number of estuarine and marine finfish species (striped bass, *Morone saxitilis*; bluefish, *Pomatomus saltatrix*; white perch, *Morone americana*; American shad, *Alosa sapidissima*; blueback herring, *Alosa aestivalis*; spot, *Leiostomus xanthurus*; spotted seatrout, *Cynoscion nebulosus*; and menhaden, *Brevoortia tyrannus*). Freshwater species were also analyzed, among these was yellow perch (*Perca flaviscens*). Results for the period 1976 through 1980 are summarized in a technical report, Eisenberg and Topping (1981) and two journal articles, Eisenberg and Topping (1984) and (1985).

Heptachlor results for 1976 and 1977 were not reported. During the period 1978 through 1980, all heptachlor measurements in shellfish and finfish were below limits of detection (*i.e.*, <1.0 ppb wet weight).

Chlordane was among the small number (4 in shellfish and 6 in finfish) of the 18 targeted chlorinated compounds that demonstrated any widespread occurrence in the Chesapeake bay and its tributaries. Quantifiable concentrations of chlordane were found in 95 percent of shellfish and 97 percent of finfish analyzed between 1976 and 1980. However, none of the shellfish sampled exceeded the FDA action level of 300 ppb wet weight; nine of the finfish (3.1%) exceeded the level.

Eisenberg and Topping cautioned against overinterpretation of pooled summary data they provided in their report, citing environmental variables as important causes of residue variability in biotic samples. Nevertheless, the reported chlordane means and medians by year for oysters,

shown in Table 58, suggest a possible decline with time over the 5-year period covered by the program. Examination of site specific data, however, such as that shown in Figure 55, fails to support this trend delineation. The Chesapeake Bay/Severn River site data shown in Figure 55 were selected because the greatest number of samples between 1976 and 1980 were collected and analyzed from this specific site.

Table 58. Summary data for chlordane concentrations in biota of the Chesapeake Bay and its Maryland tributaries, 1976-1980. Edible tissues of organisms, values in ppb wet weight. Source: Eisenberg and Topping (1981, 1984, 1985).

Location	1976	1977	1978	1979	1980
Oyster					
Mean	18	20	13	10	10
Median	20	10	10	10	11
Range	nd - 40	nd - 50	nd - 40	nd - 100	nd - 70
Clams					
Mean	16	20	32	18	20
Median	20	20	30	10	20
Range	nd - 30	5 - 40	10 - 70	nd - 80	2 - 50
Finfish					
Mean	100	70	80	80	120
Median	90	70	60	50	60
Range	nd - 370	10 - 260	nd - 340	nd - 700	4 - 310

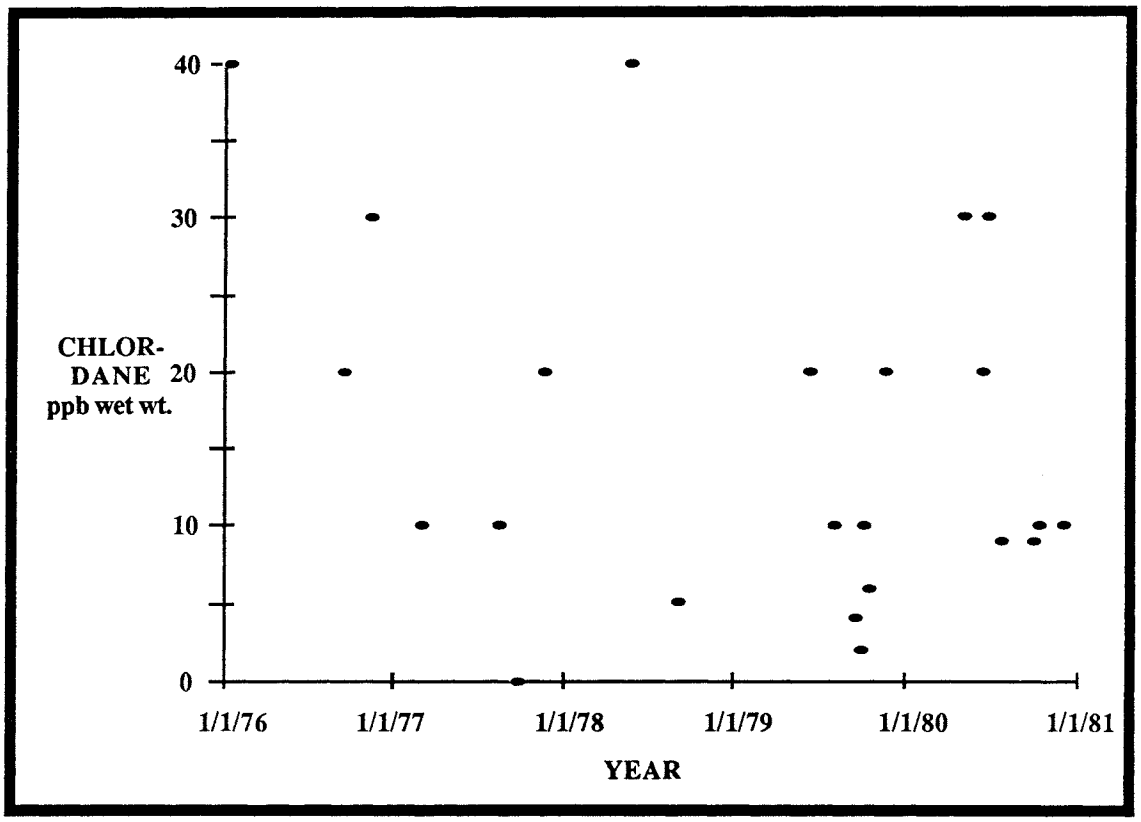

Figure 55. Concentrations of chlordane in eastern oysters sampled at a site in the Chesapeake Bay at the Severn River, 1980. Source: Eisenberg and Topping (1981, 1984, 1985).

Although long-term oyster data (*i.e.*, over the 5-year period) do not demonstrate significant temporal trends, data for 1980 alone appear to reflect a consistent phenomenon among many sites sampled. During that year, elevated concentrations of chlordane were observed in oysters collected in late spring and early summer, with lower levels observed from mid-summer through the end of the year. As summer is the spawning season for oysters, this consistent decline in concentrations of chlordane may reflect a post-spawning decrease in tissue lipid content associated with those reproductive activities (D. Murphy, Maryland Department of the Environment, pers. comm., 8 June 1990). Figure 56 shows 1980 results for chlordane in oysters sampled at eight different locations around the Chesapeake Bay.

It is possible that these body burden patterns in 1980 reflect precipitation patterns experienced in the Chesapeake Bay region during that year. Climatological records (NOAA, National Climatic Center, 1981) indicate that the region experienced near normal to very dry conditions in the first part of 1980, with some areas such as Chestertown receiving only 32 percent of its normal precipitation totals in the month of February. This was followed by an unusually wet March and April and a normal to dry summer. Precipitation records for two locations, one on the western side of the Chesapeake Bay and the other on the eastern side, are shown in Figure 57. The higher concentrations observed in Chesapeake Bay oysters in 1980 may have resulted from the greater amounts of rainfall in March and April and resultant runoff containing particulate material that may have accumulated to a greater degree because of the preceding dry February. However, September and October were also wetter than normal months in the Chesapeake Bay region and samples collected afterward do not clearly reflect this fact.

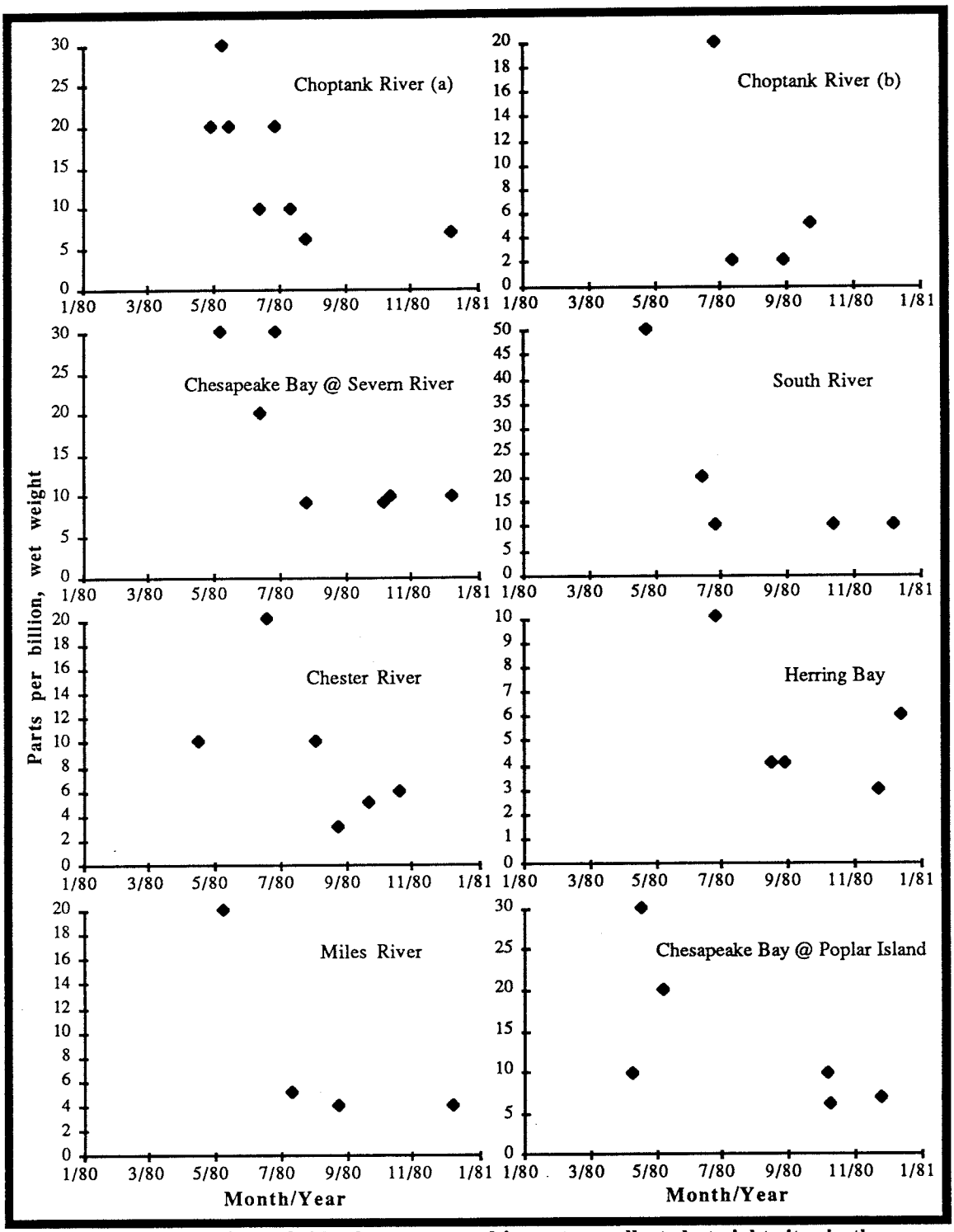

Figure 56. Concentrations of chlordane measured in oysters collected at eight sites in the Chesapeake Bay and its Maryland tributaries, 1980. Values in ppb wet weight. Note scale differences among plots. Source: Eisenberg and Topping (1981, 1984, 1985).

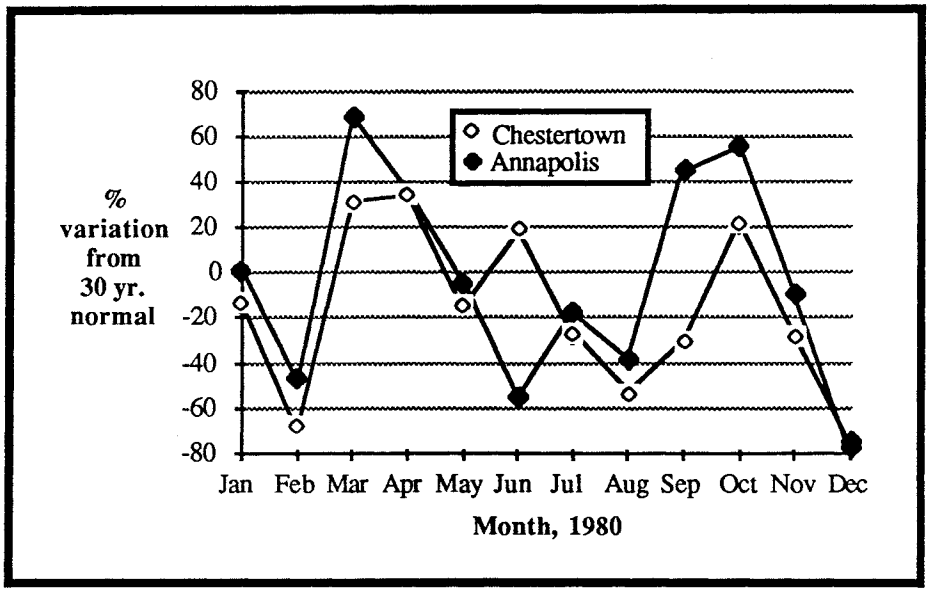

Figure 57. Precipitation records for 1980 from two locations on the Chesapeake Bay. Values in percent deviation from 30-year normal. Compare to Figure 56, concentrations of chlordane measured in Chesapeake Bay oysters in 1980. Source: NOAA, National Climatic Center (1981).

Eisenberg and Topping also analyzed samples of fish muscle tissue and gonadal tissue to compare relative levels of bioaccumulation. Samples of fish were collected from the Patapsco and York river estuaries in 1976 and from Chesapeake Bay near Rock Hall in 1979. These results are shown in Table 59.

Table 59. Results for chlordane measured in tissues of finfish sampled in Chesapeake Bay waters in 1976 and 1979. Values in mg/kg (ppm) wet weight. Source: Eisenberg and Topping (1981, 1985).

Species	Location	Tissue	Date	Chlordane, ppm wet weight
White perch	Patapsco River	Flesh	Spring, 1976	0.30
		Roe	Spring, 1976	0.24
Yellow perch	Patapsco River	Flesh	Spring, 1976	0.05
		Roe	Spring, 1976	1.15
Yellow perch	Patapsco River	Flesh	Spring, 1976	0.09
		Roe	Spring, 1976	0.92
Striped bass	Patapsco River	Flesh	Spring, 1976	--
		Roe	Spring, 1976	1.28
Striped bass	Chesapeake/Rock Hall	Flesh	April, 1979	0.08
		Gonad	April, 1979	0.73
Striped bass	Chesapeake/Rock Hall	Flesh	April, 1979	0.12
		Gonad	April, 1979	0.80
Striped bass	Chesapeake/Rock Hall	Flesh	April, 1979	0.08
		Gonad	April, 1979	0.58
Striped bass	Chesapeake/Rock Hall	Flesh	April, 1979	0.08
		Gonad	April, 1979	0.98
Striped bass	Chesapeake/Rock Hall	Flesh	April, 1979	0.12
		Gonad	April, 1979	0.89
Striped bass	Chesapeake/Rock Hall	Flesh	April, 1979	0.10
		Gonad	April, 1979	1.80
Striped bass	Chesapeake/Rock Hall	Flesh	April, 1979	0.10
		Gonad	April, 1979	0.61

Table 59. Continued

Species	Location	Tissue	Date	Chlordane, ppm wet weight
Striped bass	Chesapeake/Rock Hall	Flesh	April, 1979	0.12
		Gonad	April, 1979	0.36
Striped bass	Chesapeake/Rock Hall	Flesh	April, 1979	0.10
		Gonad	April, 1979	0.50
Striped bass	Chesapeake/Rock Hall	Flesh	April, 1979	0.20
		Gonad	April, 1979	1.90
American shad	Chesapeake/Rock Hall	Flesh	April, 1979	0.04
		Gonad	April, 1979	0.01
American shad	Chesapeake/Rock Hall	Flesh	April, 1979	0.04
		Gonad	April, 1979	0.01
American shad	Chesapeake/Rock Hall	Flesh	April, 1979	0.04
		Gonad	April, 1979	0.01
American shad	Chesapeake/Rock Hall	Flesh	April, 1979	0.04
		Gonad	April, 1979	0.01
American shad	Chesapeake/Rock Hall	Flesh	April, 1979	0.03
		Gonad	April, 1979	0.01
American shad	Chesapeake/Rock Hall	Flesh	April, 1979	0.03
		Gonad	April, 1979	0.03

In yellow perch and striped bass, distinct differences were apparent between concentrations of chlordane measured in flesh and gonads. For these two species, gonadal concentrations exceeded those in flesh by about an order of magnitude. In contrast, white perch and American shad flesh and gonadal tissues showed very little difference, with some indications that muscle tissue concentrations were higher than gonadal levels. Although Eisenberg and Topping cautioned that a much larger number of samples would have to be analyzed to statistically validate conclusions, they commented that the body of results appeared to suggest a physiologically different mechanism of uptake in different species.

Maryland intensive chlordane study, 1983-1985. Because several other states, primarily in the midwest and in the northeast, had issued health advisories due to excessive chlordane levels in fish in the 1980s, and because fish sampled under the Maryland Basic Water Monitoring Program (see Murphy, 1988a) had shown occasional chlordane concentrations exceeding the FDA action level of 0.3 ppm, the Office of Environmental Programs of the Maryland Department of Health and Mental Hygiene initiated a directed sampling program to investigate chlordane concentrations in estuarine fish (whole body analyses) from Maryland waters in 1983. Initial sampling efforts and studies in other states had indicated that chlordane contamination was a problem associated with urban waters. For this reason, the 1983 Maryland chlordane study targeted estuaries around the Baltimore urban area. These included estuaries of the Bush, Severn Gunpowder, Magothy, Back, Chester, Patuxent, Patapsco, Choptank, and Susquehanna rivers. A follow-up survey of the areas indicated as being most contaminated (Back and Patapsco rivers) was conducted in 1985. Fish species sampled included spot (*Leiostomus xanthurus*), white perch (*Morone americanus*), American eel (*Anguilla rostrata*), channel catfish (*Ictulurus punctatus*), and brown bullhead (*Ictulurus nebulosus*). The results of both sampling efforts are contained in Garreis and Murphy (1986a).

Reported chlordane concentrations were summations of the concentrations of *cis-* and *trans-*chlordane. Results from the 1983 survey are summarized in Table 60. Although such fish as channel catfish and brown bullhead are not considered to be marine species *per se*, results are included here because the fish apparently were collected in the brackish waters of the river estuaries.

Table 60. Concentrations of chlordane (Σ *cis*- and *trans*-chlordane) measured in muscle tissue from fish collected in ten Chesapeake Bay estuaries, 1983. Source: Garreis and Murphy (1986a).

Estuary	Species	No. of samples	Mean (ppm wet wt.)	Median (ppm wet wt.)
Chesapeake Bay/ Susquehanna Flats	Channel catfish	28	0.1462	0.1155
	White perch	13	0.0891	0.0610
Bush River	Brown bullhead	9	0.1686	0.1580
	Channel catfish	42	0.1540	0.0770
	White perch	54	0.1200	0.0880
Gunpowder River	White perch	28	0.0992	0.0930
	Spot	30	0.0500	0.0285
Back River	Channel catfish	14	0.6714	0.5840
	Brown bullhead	15	0.3119	0.2430
	White perch	19	0.2746	0.2220
Patapsco River	Spot	14	0.3204	0.2455
Chesapeake Bay/ Seven Foot Knoll	Spot	7	0.1691	0.1270
Magothy River	White perch	14	0.2146	0.1635
	Spot	7	0.0620	0.0530
Severn River	White perch	25	0.0842	0.0840
Patuxent River	Spot	15	0.0425	0.0330
	White perch	15	0.0318	0.0300
Chester River	Brown bullhead	14	0.1374	0.0560
	White perch	15	0.0895	0.0830
	Spot	9	0.0401	0.0280
Choptank River	Spot	27	0.0364	0.0300
	White perch	28	0.0222	0.0180

As Table 60 shows, mean concentrations of chlordane compounds measured in the three fish species sampled in the Back River estuary approached or exceeded the FDA action level of 0.3 ppm wet weight. Spot collected in the Patapsco River also exceeded the FDA level and white perch in the Magothy River approached it. Fish from the Bush River contained moderate levels in muscle tissue, while those from the Patuxent and Choptank Rivers were relatively uncontaminated.

In 1985, the Back River and Patapsco River estuaries, the areas the 1983 survey indicated as containing fish relatively contaminated with chlordane compounds, were revisited and sampled in order to further quantify the extent of chlordane burdens in fish tissue. Results for this study are show in Table 61.

Table 61. Concentrations of chlordane (Σ *cis*- and *trans*-chlordane) measured in muscle tissue from fish collected in the Back and Patapsco river estuaries, 1985. Source: Garreis and Murphy (1986a).

Estuary	Species	No. of samples	Range (ppm wet wt.)	Mean (ppm wet wt.)	Std. Deviation	Median (ppm wet wt.)
Back River	White perch	63	0.0050-0.6990	0.1656	0.1473	0.1210
	Channel catfish	40	0.0820-1.7910	0.5079	0.4758	0.2270
	Brown bullhead	88	0.0050-0.6140	0.1527	0.1097	0.1325
	American eel	6	0.1840-0.4820	0.3068	0.1160	0.2825
Patapsco River	White perch	29	0.0040-0.6140	0.1084	0.1060	0.0830
	Channel catfish	4	0.1870-0.8580	0.4558	0.2522	0.3890
	Brown bullhead	7	0.0200-0.2050	0.1137	0.0663	0.1100
	American eel	19	0.0800-0.6680	0.3483	0.1856	0.3400

In the 1985 sampling cycle, channel catfish and American eel from both estuaries contained mean levels of chlordane in excess of the FDA action level of 0.3 ppm wet weight.

Based on the results of the study, the Maryland Department of Health and Mental Hygiene recommended the issuance of an advisory to recreational fishermen concerning possible health risks from consumption of certain fish species. Specifically, the public was warned against substantial consumption of channel catfish and American eels from the Back River estuary and Baltimore Harbor (Patapsco River) areas. Women of childbearing age, infants, and children were advised not to eat these fish.

Baltimore Inner Harbor/Chesapeake Bay crab survey, 1983. The blue crab, *Callinectes sapidus*, has represented a highly valued segment of the Chesapeake Bay fishery in Maryland. Because of that fact, and because the diet of the crab ranges over a variety of plant and animal materials, some concern about possible contamination of the blue crab resource was expressed in the State of Maryland. As a result, in 1983, the Maryland Department of Health and Mental Hygiene sponsored a survey of contaminants in blue crab sampled in Baltimore Harbor and at nine other locations around the Chesapeake Bay. Body meat, claw meat, and hepatopancreas tissue were homogenized to produce samples for analysis. Results were reported in Garreis and Murphy (1986b).

There were 14 results reported for blue crab taken from Baltimore Harbor and 97 for the other locations around the Chesapeake Bay and its tributaries. For Baltimore Harbor blue crab, 1 chlordane concentration of 14 (7.1%) was below the detection limit of 0.001 ppm. Values ranged from below detection to 0.625 ppm. One composite sample from the Toms Cove site exceeded the FDA action level of 0.3 ppm. Mean concentration among the Baltimore Harbor crab was 0.065±0.163 ppm, reflecting the high degree of variability in concentrations encountered (for statistical purposes, those values not detected were converted to one half the detection limit, or 0.005 ppm). Median concentration was 0.017 ppm. In contrast, 79 of 97 chlordane measurements around the rest of the Chesapeake Bay (81.4%) were below detection. The range was from below detection to 0.055 ppm. The mean concentration, highly influenced by the large number of converted below detected values, was 0.004±0.011 ppm. The median concentration for the bay sites was below detection, or a converted value of 0.005 ppm.

The nonparametric two-sample rank test, the Mann-Whitney test (Zar, 1984), was used to evaluate whether the results from Baltimore Harbor sites were significantly different from those at the Chesapeake Bay sites. At p=0.0001, the Mann-Whitney test indicated that the two groups of results were different. Although the crab from Baltimore Harbor appeared to pose a minimal health risk to human consumers based on the FDA action level for chlordane, the concentrations of chlordane in the crab population reflected the higher environmental exposure in the area that resulted in the 1986 health advisory cautioning against consumption of fish.

Trace contaminants in Chesapeake Bay bluefish, 1984-85. The bluefish is an important commercial and recreational fishery in Atlantic coast waters and in Maryland, 470,000 pounds were taken in State waters in 1980. Of that total, 180,000 pounds, or more than 38 percent, were caught in the main stem of the Chesapeake Bay (Murphy, 1988b). In 1982, the State of New Jersey closed some waters to bluefish fishing and issued a health advisory cautioning against consumption due to levels of PCBs exceeding the FDA action limit. Public concerns were raised about the geographic extent of bluefish contamination, and as a result, the Maryland Department of the Environment undertook the analysis of 24 trace contaminants, including chlordane, in fillets of 71 bluefish (*Pomatomus saltatrix*) sampled at five locations in the Chesapeake Bay between September of 1984 and October of 1985. Murphy (1988b) summarized results from that study; supporting data were obtained from the State of Maryland for this review.

There were 70 analyses for chlordane in muscle tissue of bluefish made. Chlordane concentrations measured ranged from below detection (<0.01 ppm) to 0.25 ppm. Although the upper range of values approached the FDA action limit of 0.30 ppm, none of the 70 samples exceeded it. The two highest concentrations of 0.25 and 0.23 ppm were observed in the two largest fish taken in the study, 100 and 96 cm in length, respectively. Mean concentration for all samples was 0.051±0.047 ppm. The median value was 0.030 ppm.

Spearmans rank correlation analysis was performed to evaluate the relationship between chlordane concentrations and weight of specimens, as well as between chlordane and lipid content of muscle tissue. In the case of chlordane *vs.* weight, a significant correlation (r_S=0.416) existed at the p=0.0005 level. For chlordane *vs.* lipid content, a significant correlation also existed (r_S=0.478) at the p=0.0001 level.

Murphy (1988b) also used Spearmans rank procedure to show a correlation between lipid content and chlordane concentration, but separated the sample into a large size fraction and a small size fraction for analysis. A practical observation resulting from the determination of a significant correlation between lipid and chlordane was the suggestion that consumers could reduce contaminant intake by trimming away the fatty portions of the fish and broiling, rather than frying, the fillet. The correlation between size of fish and chlordane content of the tissue discussed above also suggested that it would be prudent to avoid larger Chesapeake Bay specimens.

Trace contaminants in Chesapeake Bay striped bass, 1986. The striped bass, *Morone saxitilis*, has been an important traditional fishery in the Chesapeake Bay, both recreationally and commercially. In 1983, prior to the imposition of a ban on fishing for the species, about 26 percent of the East Coast catch (approximately 446,000 pounds) originated in Maryland. In 1986, the State of Maryland sponsored a study of organochlorine compounds and metals in muscle tissue of striped bass from estuarine portions of two rivers feeding into the Chesapeake Bay. Results from this study are summarized in Murphy (1988c) and supporting data were obtained and evaluated for the present review.

There were 35 specimens of striped bass collected from the Potomac and Choptank rivers. The mean concentration of chlordane in muscle tissue of the fish was 0.174±0.186 ppm, with a median value of 0.133. Chlordane was detected in all fish analyzed (detection limit = 0.01 ppm) with a range of concentrations from 0.019 to 1.144 ppm. Three individual muscle samples (8.6%) exceeded the FDA action level for chlordane of 0.30 ppm.

Liver chlordane analyses were also performed on the ten specimens collected from the Choptank River. To determine if a significant correlation existed between liver and muscle concentrations of chlordane among the Choptank River specimens, the nonparametric statistic, Spearmans rank correlation analysis, was applied. This indicated that the correlation between liver and muscle concentrations of chlordane was not significant (r_S = 0.624, p = 0.0614).

Both length of fish and lipid content of muscle tissue samples were evaluated against muscle chlordane concentrations to determine degree of correlation. Again using Spearmans rank correlation coefficient procedure, r_S for the length vs. chlordane comparison was 0.176; r_S for lipid content vs. chlordane was 0.313. With a sample size of 35 fish, neither showed a significant degree of correlation (p = 0.3045 and 0.0684, respectively). This may be contrasted to chlordane results in other fish species (see, for example, discussion of NOAA/FDA/EPA Atlantic coast bluefish survey) where strong correlations were observed between chlordane concentrations and ancillary measurements such as lipid content.

Contaminant levels in oysters and clams from the Chesapeake Bay, 1981-1985. As part of its regular monitoring program for shellfish growing waters, the Maryland Department of the Environment collects and analyzes oyster and soft clam samples for levels of metals, PCBs, and organochlorine pesticides. Among the pesticides targeted was chlordane (Σ*cis*- and *trans*-isomers). Results for the period 1981 through 1985 were reported in Murphy (1990), and are summarized below in Tables 62 and 63.

Between 1981 and 1985, nearly 1,000 oyster (*Crassostrea virginica*) samples were collected from the middle and lower Maryland portions of the Chesapeake Bay and tributaries. Of these, about 800 were analyzed for chlordane concentrations. Chlordane was detected in 58 percent of the tissues samples, with a reported detection limit of 0.01 ppm wet weight. All measured concentrations were below 0.1 ppm, and correspondingly, well below the FDA action level of 0.3 ppm.

Table 62. Concentrations of chlordane measured in oyster samples from seven Chesapeake Bay watersheds, 1981-1985, by area. Source: Murphy (1989).

Watershed	No. of samples	No. > detection	Range (ppm wet weight)	Mean ± std. dev.	Median
Chester R.	155	103	<0.01-0.086	0.018±0.014	0.016
Choptank R.	189	108	<0.01-0.080	0.015±0.013	0.013
Lower Potomac R.	152	95	<0.01-0.059	0.015±0.010	0.014
Nanticoke R.	56	22	<0.01-0.040	0.011±0.009	<0.01
Patuxent R.	79	39	<0.01-0.038	0.013±0.010	<0.01
Pocomoke R.	90	36	<0.01-0.044	0.010±0.008	<0.01
West Chesapeake	87	69	<0.01-0.066	0.022±0.014	0.020

Table 63. Concentrations of chlordane measured in oyster samples from seven Chesapeake Bay watersheds, 1981-1985. by year. Source: Murphy (1989).

Year	No. of samples	No. > detection	Range (ppm wet weight)	Mean ± std. dev.	Median
1981	212	21	<0.01-0.043	0.008±0.007	<0.01
1982	162	70	<0.01-0.080	0.019±0.014	0.016
1983	156	63	<0.01-0.059	0.017±0.013	0.015
1984	139	89	<0.01-0.086	0.022±0.013	0.020
1985	139	65	<0.01-0.040	0.014±0.008	0.014

Interestingly, chlordane compounds were far more prevalent in oyster samples than were the usually widely distributed residues of DDT. Chlordane was the most commonly detected of the organochlorine pesticides targeted in the Maryland program. Between 1981 and 1985, DDT and its principal metabolite DDE were detected in no oyster samples, while DDD compounds were found in only one tissue sample. No clear temporal trends were shown by the oyster results.

In addition to oysters, soft shell clams (*Mya arenaria*) from five Chesapeake Bay watersheds were also sampled under the same program. Chlordane was detected in 77 percent of a total of 77 clams; concentrations ranged from <0.01 to 0.082 ppm wet weight. Mean and median values, as well as standard deviation were 0.02 ppm. No discernible trends were apparent in these results for clams.

Relationship between body contaminants and bone development in East Coast striped bass, 1978. In order to assess the relationship, if any, between skeletal deformities in striped bass (*Morone saxatilis*) and contaminant body burden, Mehrle *et al.* (1982) analyzed whole fish samples from four locations along the northeast Atlantic coast for concentrations of a number of inorganic and organic contaminants. Chlordane (identified as 1,2,4,5,6,7,8,8-octachlor-2,3,3*a*,4,7,7*a*-hexahydro-4,7-methanoindane) was among the compounds targeted by Mehrle *et al.* Approximately 50 fish were collected three times during 1978 from locations along the Hudson River near Indian Point, New York; The Potomac River near Blossom Point, Maryland; from the Nanticoke River near Vienna, Maryland; and from the Edenton National Fish Hatchery, Edenton, North Carolina. Although chlordane residues were detected, Mehrle *et al.* noted that all concentrations were 0.06 µg/g wet weight or less and were not considered to be at significant levels relative to other compounds like PCBs.

Southeast Atlantic Coast: North Carolina to Florida

Clinical investigation of mass mortality of bottlenose dolphins along the U.S. central and south Atlantic coast, 1987-1988. In 1987 and 1988, highly unusual numbers of bottlenose dolphins (*Tursiops truncatus*) died along the central and southern portions of the U.S. Atlantic coast. Between June 1987 and May 1988, over 740 animals were known to have died. Because of the unprecedented and highly publicized nature of the event, an investigation was launched in an attempt to determine the cause of the mortality. Funding and administrative support were provided by the NMFS, The U.S. Marine Mammal Commission, the Office of Naval Research of the U.S. Navy, Natural Sciences and Engineering Research Council of Canada, and the University of Guelph (Ontario, Canada). Ultimately, the researchers attributed the mortality to complications arising from a biological neurotoxin produced by the dinoflagellate, *Ptychodiscus brevis*. Results of the study were reported in Geraci (1989).

The forensic nature of the investigation necessarily involved a wide range of disciplines. One such aspect was the analysis of contaminant burdens in individual animals and among the compounds analyzed were the chlordane compounds *cis-* and *trans*-nonachlor, *cis-*, *trans-*, and oxychlordane, and heptachlor. However, in the data summaries presented in Geraci (1989), *trans*-nonachlor, which was the most prevalent of the chlordane compounds, was used as the representative compound for the chlordane family. Most of the analytical results presented were for *trans*-nonachlor concentrations in tissues of *Tursiops truncatus*, but some concentrations for pilot whales (*Globicephala melaena*), harbor porpoises (*Phocoena phocoena*), and humpback whales (*Megaptera novaeangliae*) were also summarized for comparative purposes. Among bottlenose dolphins, both stranded and captive animals were sampled, also for comparison. Results were reported for lipid weight in blubber tissue, and both lipid and wet weight for liver tissue. Table 64 shows summarized chlordane results from Geraci (1989).

Table 64. Concentrations of *trans*-nonachlor measured in marine mammals sampled between New Jersey and Florida, 1987. Values in ppm, basis as noted. Source: Geraci (1989).

Specimen	Tissue	N	Reporting basis	Mean ± std. dev.	Range
Tursiops truncatus	Blubber	56	Lipid wt.	14.6±12.0	1-58
Immature female	Blubber	18	Lipid wt.	15.3±12.1	1-51
Mature female	Blubber	9	Lipid wt.	7.4±8.4	1-28
Immature male	Blubber	22	Lipid wt.	16.8±13.3	1-58
Mature male	Blubber	6	Lipid wt.	20.7±5.3	13-28
Captive	Blubber	3	Lipid wt.	18.4±10.9	5-32
Globicephala melaena	Blubber	11	Lipid wt.	6.6±3.8	4-18
Phocoena phocoena	Blubber	8	Lipid wt.	7.8±2.6	5-12
Megaptera novaeangliae	Blubber	8	Lipid wt.	1.5±2.0	0.2-7

Table 64. Continued

Specimen	Tissue	N	Reporting basis	Mean ± std. dev.	Range
Tursiops truncatus	Liver	53	Lipid wt.	8.1±9.5	0-52
Immature female	Liver	21	Lipid wt.	7.6±6.4	0-25
Mature female	Liver	11	Lipid wt.	2.5±4.4	0-13
Immature male	Liver	17	Lipid wt.	12.4±13.2	0-52
Mature male	Liver	4	Lipid wt.	7.4±5.6	0-15
Captive	Liver	3	Lipid wt.	11.4±4.1	7-17
Globicephala melaena	Liver	11	Lipid wt.	1.6±4.4	0-15
Phocoena phocoena	Liver	9	Lipid wt.	3.8±3.2	0-8.7
Tursiops truncatus	Liver	53	Wet wt.	0.8±1.2	0-5.5
Immature female	Liver	21	Wet wt.	1.2±1.7	0-5.5
Mature female	Liver	11	Wet wt.	0.1±0.1	0-0.3
Immature male	Liver	17	Wet wt.	0.8±0.8	0-2.7
Mature male	Liver	4	Wet wt.	0.6±0.8	0-2.0
Captive	Liver	3	Wet wt.	0.3±0.05	0-0.4
Globicephala melaena	Liver	11	Wet wt.	0.03±0.06	0-0.2
Phocoena phocoena	Liver	9	Wet wt.	0.14±0.13	0-0.4

Geraci (1989) stated that blubber and liver tissue burdens were both evaluated in order to assess the capacity of the liver to process contaminants. With respect to the chlordanes, as represented by *trans*-nonachlor, it was found that only two individual animals contained higher levels in liver than in blubber. According to Geraci, this suggested that *trans*-nonachlor can be processed by the liver as it is delivered. The fact that it was undetected in many liver samples was taken as an indication that the compound is rapidly metabolized or excreted.

Brain samples from 18 stranded bottlenose dolphins were also analyzed for their organochlorine contaminant burdens. Wet weight concentrations of *trans*-nonachlor ranged from 0-0.3 ppm and were not significantly correlated with levels in other tissues.

Geraci commented that the levels of contaminants in the necropsied dolphins' blubber were among the highest recorded for cetaceans, although he cautioned that direct comparisons to levels found in other studies of bottlenose dolphins could not be made due to methodological differences. However, to ensure that the high levels determined were not artifacts of the method employed, the samples of other species listed in Table 64 were analyzed and compared to results in the literature. These values were found to be comparable, reinforcing the ranking of the concentrations in *Tursiops truncatus* as among the highest known.

South Carolina

South Carolina coastal contaminants study, 1985-. The Marine Resources Research Institute of the South Carolina Wildlife and Marine Resources Department has coordinated a modest program of sampling for trace contaminants in the coastal environment of that state since 1985. Sediments and oyster (*Crassostrea virginica*) have been analyzed for a range of contaminants, including α- and γ-chlordane, and occasionally other chlordane constituents like α-chlordene. Chlordane results from the program were obtained from the principal investigator in the South Carolina Wildlife and Marine Resources Department, Dr. Thomas D. Mathews.

Chlordane compounds have been detected only infrequently along the South Carolina coast. Positive results reported by Mathews (unpublished) are summarized below as Table 65.

Table 65. Chlordane compounds measured in the South Carolina coastal environment, 1985-1988. Values in ppb wet weight. Source: Mathews (unpublished data).

Location	Medium	Date	Constituent	Concentration (ppb wet wt.)
N. Santee River	Sediment	5/85	α-chlordane	16.3
Broad River	Sediment	6/86	α-chlordane	1.66
N. Santee River	Sediment	11/86	γ-chlordane	0.26
S. Santee River	Sediment	11/86	γ-chlordane	0.29
N. Santee River	Oyster	11/86	γ-chlordane	0.50
S. Santee River	Oyster	11/86	γ-chlordane	1.8
Cooper River	Oyster	11/86	γ-chlordane	0.82
Cooper River	Sediment	5/87	γ-chlordane	0.83
Cooper River	Sediment	5/87	α-chlordane	0.71
Charleston Harbor	Sediment	5/87	γ-chlordane	0.92
Charleston Harbor	Sediment	5/87	α-chlordane	1.1
N. Santee River	Oyster	5/87	γ-chlordane	1.1
N. Santee River	Oyster	5/87	α-chlordane	0.41
S. Santee River	Oyster	5/87	γ-chlordane	0.55
Charleston Harbor	Oyster	1988	(not specified)	6.36

Due to methodological difficulties associated with consistent drying of samples, Mathews reported results on a wet weight basis. More recently, he has found that most sediment samples contain between 40 to 60 percent water (Mathews, *pers. comm.*, 18 October 1989).

In September of 1989, Hurricane Hugo caused widespread damage along the South Carolina coast. Mathews commented that it is possible that particulate material to which chlordane compounds were adsorbed may have been mobilized in the flooding that resulted from heavy rains associated with the hurricane, through the washing away of material around termiticide-treated foundations. If sufficient quantities were to reach estuaries, this may be reflected in future samples collected and analyzed under the program.

Florida

Florida Deepwater Ports Maintenance Dredging Study, 1983. In recognition of the information shortcomings associated with estuarine systems in the State of Florida, the Department of Environmental Regulation initiated a program in 1981 to evaluate needs and to provide more reliable assessments of environmental conditions. As the program name implies, the Deepwater Ports Maintenance Dredging Study had as one of its primary foci, potential environmental problems associated with maintenance dredging, but stated goals also included collection of data for characterization of estuaries and development of investigative approaches and tools.

Twelve estuaries--Jacksonville, Canaveral, Fort Pierce, Palm Beach, Port Everglades, Miami, Manatee, St. Petersburg, Tampa, St. Joe, Panama City, and Pensacola--were targeted for sampling. At this writing, samples have been collected and analyzed at all port sites (S. Schropp, Florida Department of Environmental Regulation, *pers. comm.*, 3 November 1989); data have been interpreted and published for one site, the Port of Miami (Biscayne Bay)/Miami River (Ryan *et al.*, 1985). Lower portions of the Miami River were included in the Port of Miami study because the river was identified as a major contributor of sediment loading to the urban bay region. Chlordane (isomeric composition not specified) was measured in sediments collected at 14 locations within the Port of Miami-Miami River region. Figure 58 illustrates the location of sites sampled in the Miami area by the study, and Table 66 presents summarized data for analyses of chlordane (concentrations of chlordane at nine sites in the Miami River could not be quantitated due to interference from high levels of PCBs).

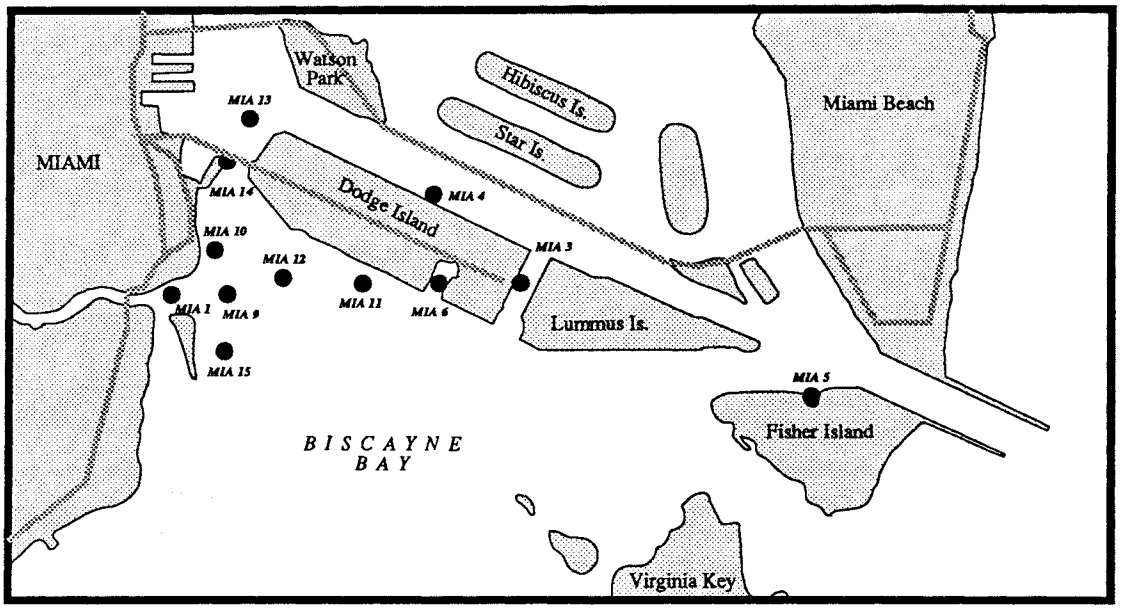

Figure 58. Map showing relative station locations at the Port of Miami, the Intracoastal Waterway, and the mouth of the Miami River, sampled in 1983. Source: Ryan *et al.* (1985).

Table 66. Concentrations of chlordane measured in sediments collected near Miami, 1983. Refer to Figure 58 for site locations. Source: Ryan *et al.* (1985).

Site	General Location	N	Reporting basis	Mean ± std. dev.	Range
MIA 1	Miami River mouth	3	ppm dry wt.	0.16±0.12	0.08-0.29
MIA 2	Miami River channel	3	ppm dry wt.	0.13±0.06	0.09-0.20
MIA 3	Port of Miami berth	3	ppm dry wt.	<0.006	-
MIA 4	Port of Miami berth	3	ppm dry wt.	<0.006	-
MIA 5	Port of Miami berth	3	ppm dry wt.	<0.006	-
MIA 6	Port of Miami berth	3	ppm dry wt.	<0.006	-
MIA 8	Miami River channel	3	ppm dry wt.	<0.006	-
MIA 9	Intracoastal Waterway	3	ppm dry wt.	<0.006	-
MIA 10	Intracoastal Waterway	3	ppm dry wt.	<0.006	-
MIA 11	Port of Miami channel	3	ppm dry wt.	<0.006	-
MIA 12	Port of Miami channel	3	ppm dry wt.	<0.006	-
MIA 13	Port of Miami channel	3	ppm dry wt.	<0.006	-
MIA 14	Intracoastal Waterway	3	ppm dry wt.	<0.006	-
MIA 15	Intracoastal Waterway	3	ppm dry wt.	<0.006	-

Biscayne Bay receives limited natural sedimentary input and is defined as a confined limestone basin, and as a result, materials from the adjacent urban land areas entering the bay could potentially influence the degree of contamination there. However, if the above results can be taken as representative of the Miami region, chlordane was apparently not a contaminant of concern at the time the sediments were collected. Both sites where chlordane was measured in quantifiable amounts were located in the Miami River and this was consistent with findings for other chlorinated hydrocarbons: they were detected only infrequently, with most occurring at river stations.

Gulf of Mexico Coast: Florida to Texas

Chlorinated pesticide levels in the eastern oyster in the south Atlantic and Gulf of Mexico, 1964-66. Bugg, Higgins, and Robertson (1967) collected and analyzed eastern oysters (*Crassostrea virginica*) from six southeast Atlantic and Gulf coast states for levels of chlorinated pesticides, including chlordane (identified as 1,2,4,5,6,7,8,8-octachloro-3*a*,4,7,7*a*-tetrahydro-4,7-methanoindane) in soft body tissues. Of 132 tissue samples collected between February 1964 and August 1966, chlordane was not detected in 112 (85%), was detected but not quantitated (< 10 ppb, wet weight) in 19 (14%), and was positively measured at a level of 10 ppb in one sample from Louisiana (1%).

The results of Bugg, Higgins, and Robertson support those of Casper (1967) discussed below. Both studies found levels of chlordane at or below detection limits in all oyster samples. It should be noted that in both of these surveys, oysters were obtained either from established shellfish growing beds or from commercial oyster dealers. It is possible that shellfish from other non-commercial areas would have shown higher tissue concentrations.

Florida

Water quality of the Charlotte Harbor estuarine system, 1982-1984. In 1982, the USGS began a 7-year multidisciplinary study of the Charlotte Harbor estuarine system. A number of physical, chemical, and biological parameters were assessed as part of the project, and chlordane (not specifically identified) and heptachlor were measured in both water and sediment samples collected at five transects located in Charlotte Harbor and in estuaries of the tributary Myakka, Peace, and Caloosahatchee rivers. Results for the first 2 years of the Charlotte Harbor study were reported in Stoker (1986).

Water samples collected and analyzed showed no concentrations of chlordane or heptachlor above analytical detection limits. In fact, no pesticide was measured above detection limits in any of the water samples. In sediments, chlordane was among the few targeted organic compounds detected: Besides three measurements of DDT-related compounds, a single measurement of chlordane at 1.0 µg/kg at the mouth of the Peace River, in upper Charlotte Harbor, was the only quantitated value obtained.

These results suggested that the Charlotte Harbor system was relatively uncontaminated by chlordane compounds at the time of sampling in 1982.

Hydrocarbons in Tampa Bay. A review by Van Vleet (1985) examined and summarized research on hydrocarbons in the Tampa Bay region. Among the studies discussed was one sponsored by the U.S. Army Corps of Engineers (USACOE) and performed by the USGS. Results were originally reported in U.S. Army Engineer District Jacksonville (1974). Relatively low levels of chlorinated organic compounds were found in sediments from the main shipping channel of Tampa Bay. Chlordane (not specifically identified) and heptachlor were among those not detected (detection limits not specified). However, in East Bay, the industrialized upper northeastern section of Hillsborough Bay, chlordane was measured in concentrations ranging from 5 to 13 ppb. Van Vleet noted that East Bay acts as a settling basin for most of the pesticides entering from municipal and industrial sources, and that any accumulation would not be rapidly flushed out into Hillsborough Bay and Tampa Bay.

Water and biota were also analyzed in the USACOE study. Two of the five stations at which water samples were collected and analyzed contained measurable amounts of chlordane, at 0.1 ppb. The organisms which were sampled included starfish (1 of 7 sites), stone crabs (4 of 7 sites), blue crabs (1 of 7 sites), tunicates (5 of 7 sites), crown conchs (3 of 7 sites), and oysters (3 of 7 sites) (species names were not specified). Ranges of values were starfish, 0 ppm; stone crabs, 0-0.0001 ppm; blue crabs, 0.031 ppm; tunicates, 0-0.001 ppm; crown conchs, 0-0.065 ppm; and oysters, 0.003-0.006 ppm. Presuming that these concentrations were wet weight basis, all were well below the FDA action level of 0.3 ppm

Van Vleet referred to work underway (at that time) by the Marine Research Laboratory of the Florida Department of Natural Resources which investigated chlorinated compounds in shellfish of Tampa Bay. Oysters were found to contain approximately 0.040 ppm chlordane, an order of magnitude greater than concentrations found in the USACOE study, but still an order of magnitude less than the FDA level.

Alabama

Organic pollutant levels in bivalves of Mobile Bay, 1982-1984. Like many other coastal urban regions, Mobile Bay has experienced substantial population and industrial growth in the past two decades. In order to provide reliable baseline information on current conditions that might be used for resource management decisions in the bay, Marion, Barker, and Settine (1987) identified and quantified organic contaminants in the eastern oyster (*Crassostrea virginica*) and the Atlantic rangia (*Rangia cuneata*). Oysters, whose distribution was limited to areas of moderately high salinity, were collected at seven sites in the lower portion of Mobile Bay in 1982 and 1983. Clams were sampled at ten sites in the upper portion of the bay in 1983 and 1984.

Chlordane (not specified) was targeted in initial screening samples; however, it was not found at concentrations above detection limits, and was not among the compounds scanned in most samples (K. Marion, pers. comm., October 1989). However, heptachlor was routinely searched for in the study.

In oysters, heptachlor was detected in 9 percent of the samples and at 29 percent of the sites. Concentrations ranged from below detection to 53 ppb (reporting basis not specified, presumed to be wet weight). Mean level was 32 ppb. In clams, heptachlor was not measured at levels exceeding detection limits. These results would suggest that heptachlor contamination of bivalves during the period sampled was not severe; even the maximum concentration detected in oyster tissue (53 ppb) was well below the FDA action level for heptachlor (300 ppb). The authors commented that tissue level of organic pollutants was below what they considered to be high or critical. However, they cautioned that continued pressures of growth and development could result in more severe degradation of the Mobile Bay environment.

Texas

Galveston Bay pesticide study, 1964. In the summer of 1964, following an outbreak of equine encephalitis, a broad program of pesticide-spraying was undertaken to control mosquito populations in the Houston area. Large quantities of pesticides such as malathion, DDT, and BHC were used in this program. Because of concern about possible pesticide contamination of shellfish beds in Galveston Bay, the U.S. Public Health Service, in cooperation with state agencies, sampled water and oysters in the bay in September and October of 1964. Casper (1967) reported the results of this effort. Laboratory analyses were made for 11 pesticide compounds, including chlordane (identified as 1,2,4,5,6,7,8,8-octachloro-3*a*,4,7,7*a*-tetrahydro-4,7-methanoindane).

Casper found pesticide levels in water and oysters to be universally low, despite the intense program of mosquito abatement. In fact, he concluded that levels of DDT and DDE in oysters were higher prior to the beginning of the control effort than after. For chlordane, Casper measured no detectable concentrations in oysters collected at ten Galveston Bay sites. In water analyses, concentrations of chlordane were not detected in seven of nine site samples, and not quantitated at levels <0.001 ppm at the remaining two sites.

Texas Statewide Monitoring Network, 1973-present. In order to evaluate the extent of environmental contamination by pesticides and PCBs in the State of Texas, the Texas Department of Water Resources has maintained the Statewide Monitoring Network since the early 1970s. Sites under this program were chosen for their proximity to urban or agricultural areas, or for their location in bays and estuaries receiving water and sediment from major river systems. Although the majority of sites were located in freshwater bodies, estuarine areas have also been sampled. Water, sediment, and fish tissue were target matrices. Four technical chlordane constituents (*cis*- and *trans*-chlordane, *cis*- and *trans*-nonachlor) have been among those pesticide compounds analyzed for the Statewide Monitoring Program. Results obtained through 1977 were summarized in Dick (1982); more recent measurements between 1980 and 1988 were obtained for this report directly from the Texas Department of Water Resources (unpublished). Dick (1982) also included results from measurements made by the USGS from a number of freshwater bodies in the State of Texas as well as some results from other state and federal efforts.

Freshwater data, generated primarily by the USGS, reflected a trend of elevated chlordane (and most other pesticides) near major urban centers like Houston, Dallas, and San Antonio. For example, while chlordane was detected above analytical detection limits of 0.1 µg/l in 9 percent of

563 water samples collected statewide between 1973 and 1977, positive determinations for chlordane were made in over 90 percent of samples taken at urban locations. At estuarine sites, which were sampled only by the Texas Department of Water Resources with higher detection limits (5.0 µg/l), none of the 64 samples collected between 1973 and 1977 contained chlordane at detectable concentrations.

In freshwater sediments, 38 percent of the 344 USGS measurements taken were at or above detection limits. In samples collected by the Texas Department of Water Resources, chlordane was detected in 4.3 percent of a total of 749 measurements. Maximum concentration in the latter set of samples was 610 µg/kg, dry weight. Only 3.1 percent of 196 concentrations in estuarine samples occurred at detectable levels, with the highest concentration of 130 µg/kg found in Arroyo Colorado. Detection limits for chlordane in freshwater sediments were lower than in estuarine sediments (10 µg/kg, *vs.* 20 µg/kg, respectively).

For chlordane in estuarine biota, Dick summarized results from early (1965-1969) collections by the Texas Parks and Wildlife Department, and subsequent measurements by the Texas Department of Water Resources. Although analytical difficulties limited results from the Texas Parks and Wildlife Department program, it was noted that chlordane was detected in only two of the six embayments sampled: San Antonio Bay and Aransas Bay. Maximum concentration in fish tissue was 122 µg/kg wet weight in San Antonio Bay, and in oyster tissue, 340 µg/kg in Aransas Bay. Maximum concentrations measured by the Texas Department of Water Resources were 220 and 750 µg/kg at Corpus Christi Inner Harbor.

Data obtained from the Texas Department of Water Resources (unpublished) for the period between 1980 and 1988 were consistent with earlier measurements in that chlordane compounds were targeted but only rarely detected.

Temporal accumulation of organochlorine pesticides in shorebirds wintering on the south Texas coast, 1979-1980. The lower Rio Grande Valley in Texas has been one of the most intensely farmed regions in the United States. Crops have been grown year-round, and the land has been subjected to repeated applications of pesticides and other agricultural chemicals. Runoff from croplands eventually reaches the Laguna Madre, an estuarine region along the south Texas coast that serves as a breeding, wintering and nursery ground for fish and wildlife species. In 1979 and 1980, three species of shorebirds (long-billed dowitcher, *Limnodromus scolopaceus*; western sandpiper, *Calidris mauri*; and American avocet, *Recurvirostra americana*) were collected and body tissues (less skin, feet, beaks, wingtips, and gastrointestinal tracts) were analyzed for a suite of organochlorine pesticides. Chlordane (not specifically identified) was among the compounds targeted. Results were reported in White, Mitchell, and Kaiser (1983).

While DDE (the principal metabolite of DDT), dieldrin, and toxaphene were regularly detected in the shorebirds collected, chlordane was not. Of the 165 birds analyzed for the study, chlordane was detected in only two carcasses, and at concentrations less than 0.2 ppm wet weight. In contrast, DDE was found in 157 of the 165 samples, at concentrations ranging as high as 68 ppm.

Reproductive success of black skimmers in Texas relative to environmental pollution, 1978-1981. Because waterbird censuses had indicated that breeding populations of the black skimmer (*Rhynchops niger*) were possibly on the decline, their reproductive status and impacts of organochlorine contaminants on reproduction along the southern coast of Texas were studied between 1978 and 1981 (White, Mitchell, and Swineford, 1984) . Sites sampled ranged from Corpus Christi to Laguna Vista, in the Laguna Madre region. Pesticides and PCBs were measured in 205 black skimmer eggs from three locations, although compounds targeted were not listed completely. Analyses of egg contents showed that 99.5 percent of the 205 eggs contained DDE and 77 percent contained PCBs. Although exact frequency of occurrence was not specified, the authors noted that chlordane isomers were detected less frequently, and that residues were usually less than 0.5 ppm wet weight basis. As a result, the authors did not consider these to be "biologically significant." This was contrasted to the situation for DDE residues, which ranged as high as 51 ppm.

Organochlorine contaminants and reproductive success of black skimmers in south Texas, 1984. Custer and Mitchell (1987) studied reproductive success and organochlorine contamination

in eggs of black skimmers (*Rhynchops niger*) in the Laguna Madre region of lower Texas in 1984. Among the 12 compounds targeted, were four chlordane constituents: *cis*- and oxychlordane, and *cis*- and *trans*-nonachlor. Similar to results found previously by White, Mitchell, and Kaiser (1983) in three shorebird species, chlordane compounds were detected only infrequently (with reported detection limits at 0.1 ppm wet weight) relative to other compounds such as DDE, or PCBs. Specifically, DDE was found in 98.5 percent of 53 eggs analyzed (maximum concentration 28.4 ppm wet weight), and PCBs in 72.6 percent (9.1 ppm maximum), while oxychlordane was found in only 5.2 percent (0.21 ppm maximum), and *trans*-nonachlor in 2.5 percent (0.09 ppm maximum). The other two chlordane compounds were not detected in any of the eggs sampled.

Organochlorine contaminants in white-faced ibis eggs in southern Texas, 1985. As the results found by White, Mitchell, and Kaiser (1983) indicated, problems had been identified with elevated concentrations of chlorinated pesticides in the DDT family accumulating in fish and fish-eating birds from the Laguna Madre area, and in 1985, contaminant concentrations were investigated in eggs of white-faced ibis (*Plegadis chihi*) nesting along agricultural drainage canals in southern Texas. This species had been identified in other studies as being sensitive to organochlorine concentrations, particularly those related to DDT. Results were reported in Custer and Mitchell (1989).

The study was conducted near two dredged material islands in the lower Laguna Madre. In addition to concentrations of 12 organochlorine compounds, including 4 constituents of technical chlordane (*cis*-chlordane, oxychlordane, and *cis*- and *trans*-nonachlor), shell thickness was also measured and other reproductive impacts were assessed.

While residues of DDT-related compounds were detected in 75 percent of the eggs analyzed, no detectable concentrations of any other organochlorine compounds--including the four chlordane compounds--were found. The lower limit of detection was 0.1 ppm wet weight basis.

Results from the three studies in the Laguna Madre region of Texas suggest that while DDT compounds may continue to pose a threat to birds indigenous to or migrating through the area, chlordane has found only at very low levels, if at all.

Pacific: California , Oregon, Washington, and Hawaii

Chlorinated hydrocarbons in livers of fish from the northeastern Pacific Ocean, 1969-1970. Residues of pesticides in seafood species from California prompted the U.S. Bureau of Commercial Fisheries, subsequently the NMFS, to conduct a preliminary survey of pesticides in coastal fish sampled in the northeastern Pacific Ocean. The purpose of the survey was to establish whether fish were accumulating chlorinated pesticides, and if so, where problem areas existed. Among the chlorinated hydrocarbons searched out was chlordane (not identified as to isomeric composition, but probably technical chlordane). The study analyzed liver tissues of several species of fish (listed in Table 67) collected in the Pacific Ocean between Mexico and Alaska. Results were discussed in Duke and Wilson (1971).

Table 67. Fish species collected and analyzed by Duke and Wilson (1971)

Common name	Genus/species	No. individuals	No. analyses
Bonito	*Sarda chiliensis*	16	2
English sole	*Parophrys vetulus*	14	1
Hake	*Merluccius productus*	168	4
Jack mackerel	*Trachurus symmetricus*	29	3
Lingcod	*Ophiodon elongatus*	1	1
Lizardfish	*Synodus lucioceps*	13	1
Ocean white fish	*Caulolatilus princeps*	32	4
Pacific mackerel	*Pneumatophorus diego*	19	2
Blue rockfish	*Sebastes mystinus*	5	1
Bocaccio	*Sebastes paucispinis*	9	1
Olive rockfish	*Sebastes serranoides*	5	1
Rosy rockfish	*Sebastes rosaceus*	23	2

Table 67. Continued

Common name	Genus/species	No. individuals	No. analyses
Starry rockfish	*Sebastes constellatus*	18	2
Treefish	*Sebastes serriceps*	17	2
Vermillion rockfish	*Sebastes miniatus*	10	1
Sablefish	*Anoplopoma fimbria*	20	2
Sand bass	*Paralabrax nebulifer*	20	2
Sardine	*Sardinops caerulea*	32	1
California scorpionfish	*Scorpaena guttata*	10	1
Spiny dogfish	*Squalus acanthias*	18	3
White croaker	*Genyonemus lineatus*	16	1
Yellowfin tuna	*Thunnus albacares*	11	1 + replicate
Albacore tuna	*Thunnus alalunga*	10	1
Chinook salmon	*Oncorhynchus tshawytscha*	20	2
Silver salmon	*Oncorhynchus kisutch*	40	4
Sockeye salmon	*Oncorhynchus nerka*	177	18
Chum salmon	*Oncorhynchus keta*	40	4
Pink salmon	*Oncorhynchus gorbuscha*	10	1
Rainbow trout	*Oncorhynchus mykiss*	10	1

Of the approximately 73 samples of composited fish liver tissues analyzed by Duke and Wilson, no detectable residues of chlordane were reported. In referring to this fact, Cardwell *et al.* (1977) noted that while it might have implied that the stability and susceptibility of chlordane to biomagnification were less than those of DDT and its metabolites (for which significant residues were measured by Duke and Wilson), the result may in fact have reflected the lack of sensitivity in analytical methodologies for chlordane or interference from other coeluting compounds.

DDT and PCBs in blubber of harbor seals, 1971. A coastwide survey of organochlorine contaminants in harbor seal (*Phoca vitulina richardii*) blubber was reported by Anas (1974). Seals were sampled at sites on San Miguel Island, California (five animals); the Columbia River estuary, Oregon (three animals); Puget Sound, Washington (two animals); and Pribilof Islands, Alaska (three animals). Blubber samples were analyzed for 16 organochlorine compounds, including chlordane (not defined) and heptachlor. While DDT and PCB compounds were identified, measured, and confirmed by thin-layer chromatography, most other pesticides, including chlordane and heptachlor, were not measured above detection limits (not specified for chlordane compounds, but listed at 0.02 ppm for DDT compounds).

California

Chlorinated hydrocarbons in San Francisco Bay tributary sediments, 1972. In a study sponsored by the USGS, Law and Goerlitz (1974) evaluated sediment concentrations of several chlorinated hydrocarbons, including chlordane (specific chlordane compounds analyzed not defined), in 26 streams tributary to San Francisco Bay. Field sampling took place in February and March of 1972.

Chlordane was detected in 36 of 39 stream sediment samples collected around San Francisco Bay. Highest concentrations of 660, 670, and 800 µg/kg (ppb) dry weight were found in Belmont Creek, San Francisquito Creek, and San Rafael Creek, respectively. Sediment concentrations were below detection limits of 0.1 µg/kg in three locations: Los Gatos Creek, Green Valley Creek, and at one site in the Napa River. Mean chlordane concentration for the 36 sites where chlordane was detected was 130±194 µg/kg. Median concentration for those sites was 57 µg/kg.

Although emphasizing that the study was exploratory in design and scope, Law and Goerlitz noted that some generalizations could be made. One surprising finding was the widespread occurrence of chlordane compounds and the magnitude of residues detected relative to those of DDTs and PCBs. Not only was chlordane as ubiquitous as the other compounds, but concentrations often exceeded those of DDT and PCB at the same location: sediment concentrations

of chlordane exceeded those of PCB compounds at 29 of 39 stations and exceeded those of measured DDT compounds at 27 of 39.

California Mussel Watch Program, 1977-88. The California Mussel Watch (CMW) Program, administered by the California State Water Resources Control Board, is one of the longest continuous monitoring programs in the United States. CMW began analyzing bivalve tissues, primarily those of California, or coastal, mussels (*Mytilus californianus*), and bay, or blue, mussels (*Mytilus edulis*) in 1977. Collections have continued on an annual basis since then at a number of coastal and harbor sites along the entire California coast. Results from the CMW Program have been made available in several forms: annual reports of results are issued by CMW and discuss in some detail data of note for particular years; Phillips (1988) summarized data from the first 10 years of the program in a single report; and digitized results for all years are also available. For this discussion, the computer database was obtained from the CMW Program and Phillips (1988) was used as an additional reference.

Analytes in the CMW Program have included 13 trace elements and 45 organic compounds, although not all contaminants have been measured in all samples. Among the organic compounds analyzed by CMW are eight constituents of technical chlordane: α–chlordene, γ–chlordene, *cis*-chlordane, *trans*-chlordane, oxychlordane, *cis*-nonachlor, *trans*-nonachlor, and heptachlor. The 1980 annual sampling cycle, which included collections made in 1979 and 1980, was the earliest in which analyses of chlordane compounds were made.

The CMW Program analyzes bivalves transplanted to given locations as well as resident bivalves. For this discussion, results for resident mussels are emphasized (although subsequent discussions of interspecies differences include results for transplanted specimens). A total of 143 CMW samples of resident mussel tissue collected between 1979 and 1988 were analyzed for one or more chlordane compounds. Of this total, 89 were analyzed for all eight chlordane compounds targeted by the CMW Program, while the other 54 were analyzed for seven or fewer. Table 68 contains summary results for resident *M. edulis* and *M. californianus* for which all eight selected compounds were measured. Values of those compounds which were analyzed but not detected were converted to one half the listed detection limit for summary and statistical purposes.

Table 68. Summary of summed concentrations of eight chlordane compounds (Σ α-chlordene, γ-chlordene, *cis*-chlordane, *trans*-chlordane, oxychlordane, *cis*-nonachlor, *trans*-nonachlor, and heptachlor) in mussels collected along the California coast, 1979-1988. Values in ppb dry weight. Source: CMW Program.

Species	N	Mean± Sdev	Min[a]	Site/Date, Min	Max	Site/Date, Max	Median
M. edulis	28	149±144	4.00	Humboldt Channel 2/20/85	506	Colorado Lagoon 2/19/85	100.7
M. californianus	61	16.7±15.7	3.75	Trinidad Head 9/16/82	73.8	Royal Palms 2/19/85	11.45

[a] These minima reflect summation of converted concentrations below limits of detection. In both cases, value portrayed is sum of eight below detected conversions.

Table 69 lists summary results for summed concentrations of the five technical chlordane components measured most frequently and consistently: *cis*-chlordane, *trans*-chlordane, oxychlordane, *trans*-nonachlor, and heptachlor. Limitation of the data in this fashion reduced sample size from the dataset as a whole by only one sample, from 143 to 142, but enabled a broader, more consistent basis for comparison of results.

Table 69. Summary of summed concentrations of five most frequently measured chlordane compounds (Σ *cis*-chlordane, *trans*-chlordane, oxychlordane, *trans*-nonachlor, and heptachlor) in mussels collected along the California coast, 1979-1988. Values in ppb dry weight. Source: CMW Program.

Species	N	Mean± Sdev	Min[a]	Site/Date, Min	Max	Site/Date, Max	Median
M. edulis	50	172±237	2.50	Humboldt Channel 2/20/85	1500	Colorado Lagoon 1/11/82	85.4
M. californianus	92	18.7±18.3	2.25	Trinidad Head 9/16/82	93.8	Oceanside 3/2/81	12.0

[a] These minima reflect summation of converted concentrations below limits of detection. In both cases, value portrayed is sum of five below detected conversions.

The data for the five summed chlordane compounds were segregated by species, collection site, and collection date in order to permit evaluation of temporal trends. Figures 58 and 59 illustrate results for the two species for sites where the number of sufficient data existed to enable temporal analysis. Figure 59 represents results for *M. californianus*. Seven sites spanning the length of the California coastline were sampled relatively regularly (*i.e.*, more than five times) between 1979 and 1988. These data suggest the trend of tissue concentration declines with time, with the most recent measurements (with the exception of that from the Crescent City sewage treatment plant site) near or below detection limits. However, this is somewhat inconsistent with the reported use totals for chlordane in California (portrayed in Figure 5) which indicated that applications attributable to structural protection accelerated greatly between 1985 and 1987. This increased use of chlordane just prior to virtual elimination of permitted applications in 1988 was not reflected in CMW results shown in Figures 59 and 60. It is possible that subsequent measurements will show the increase, assuming that the use totals were representative of areas in the state influencing the coastal and estuarine environment.

In the case of *M. edulis* (Figure 60) only two locations were sampled more than twice between 1979 and 1982. Because data are not numerous, it is difficult to evaluate temporal trends for the species. However, the available results do not disagree with the declines in chlordane compound concentrations with time observed in *M. californianus*--but again, failed to show the increase in chlordane use in the 4-year period between 1984 and 1987

In order to examine spatial distribution of results, the data for summed concentrations of the five chlordane compounds were sorted by species and annual sampling cycle. No sampling cycle provided comprehensive coverage of the California coastline for either species; however, data for the sampling cycle with the greatest number of results for *M. californianus* and *M. edulis* were plotted and are presented below as Figures 61 and 62, respectively. In the case of *M. edulis* samples, Figure 62, data are shown for the Alamitos Bay-Anaheim Bay region, in which the largest number of analyses for the 1982 cycle were available.

Figure 61, summarizing results for *M. californianus* collected in the 1983 sampling cycle, suggests a general association of elevated concentrations of chlordane compounds increased with human population and proximity to urban/industrial centers. This is similar to trends observed for other chlorinated hydrocarbons (NOAA, 1987).

M. edulis data portrayed in Figure 62 represent much higher levels of chlordane compounds, and probably reflect degree of circulation at the sites. All five locations are situated in the greater Los Angeles-Long Beach area, but the most elevated concentrations were observed at sites with the most restricted circulation. The site experiencing the greatest degree of exchange with open waters, the Anaheim Bay entrance site, also yielded the lowest chlordane concentration in its resident mussels (though still well above the maximum value obtained for *M. californianus*).

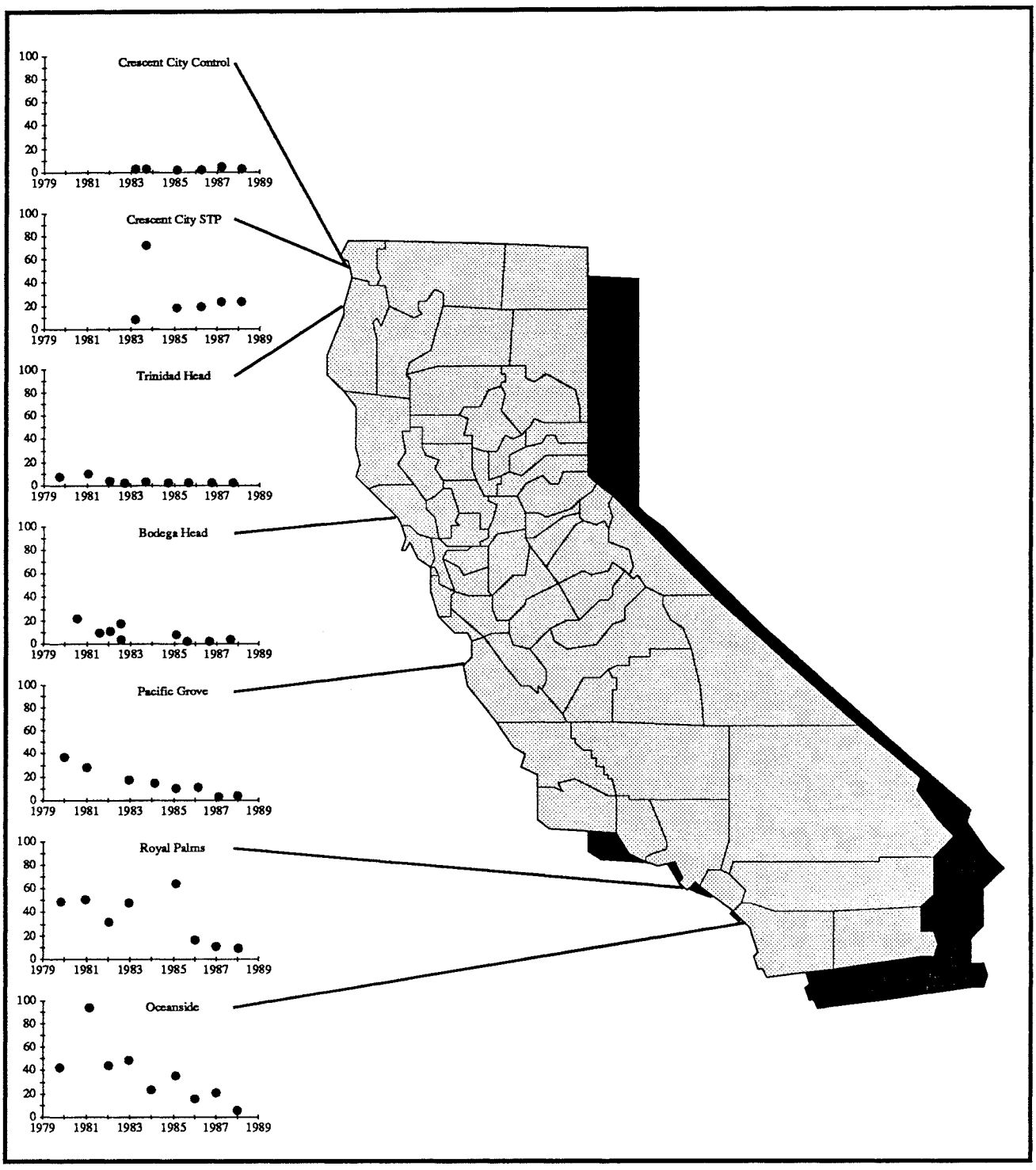

Figure 59. Temporal trends of summed chlordane compounds in tissues of *M. californianus* collected along the California coast, 1979-1988. Values in ppb dry weight. Source: CMW Program.

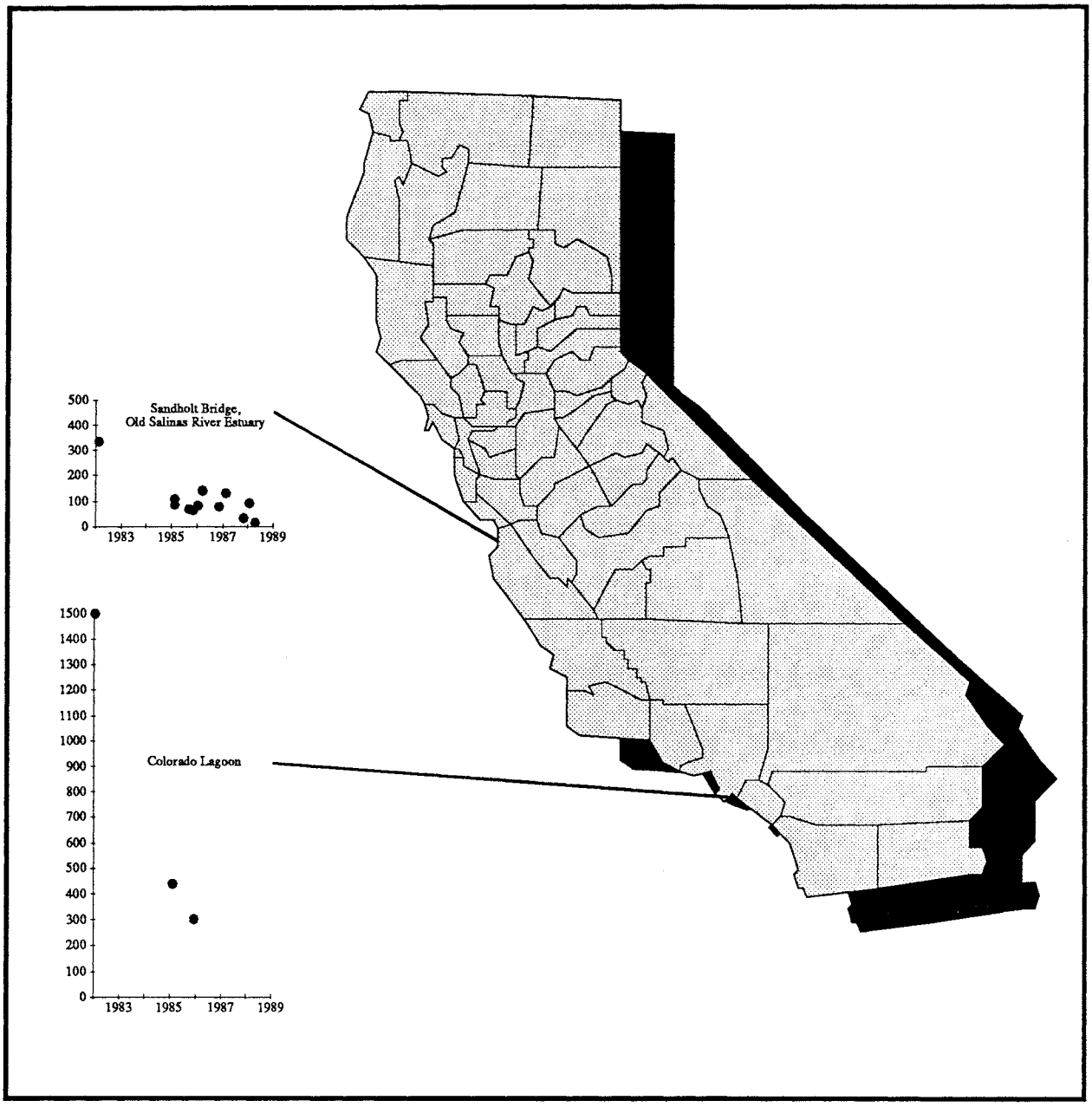

Figure 60. Temporal trends of summed chlordane compounds in tissues of *M. edulis* collected along the California coast, 1979-1988. Values in ppb dry weight. Source: CMW Program.

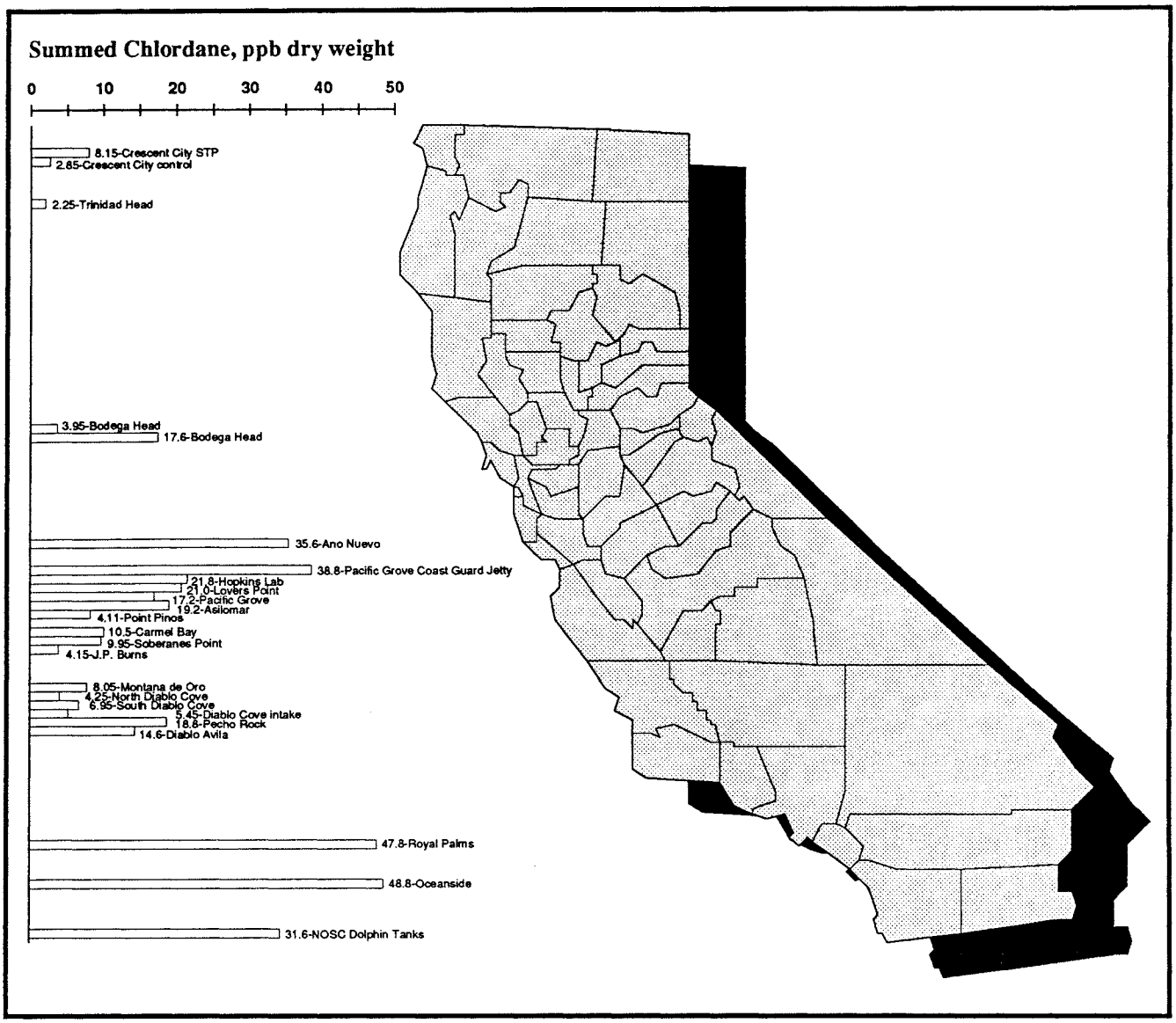

Figure 61. Levels of summed chlordane compounds measured in tissues of *M. californianus* collected along the California coast during the 1983 sampling cycle. Values in ppb dry weight. Source: CMW Program.

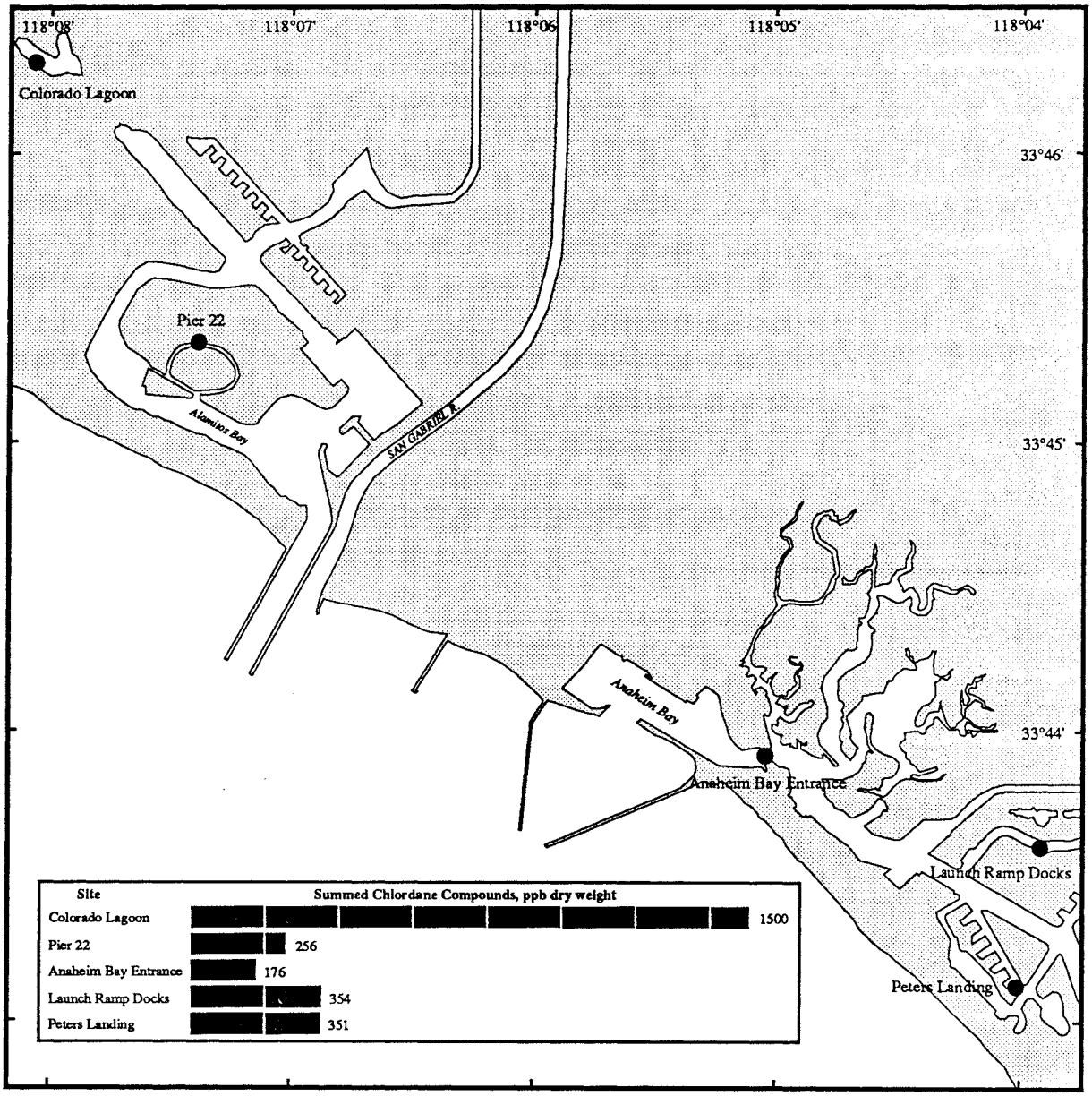

Figure 62. Location of sites and concentrations of summed chlordane compounds in tissues of *M. edulis* collected in the Alamitos-Anaheim region of southern California in the 1982 sampling cycle. Values in ppb dry weight. Source: CMW Program.

Tables 68 and 69 and Figures 59 through 62 clearly indicate differences in levels of chlordane compounds measured in the two species of mussels by the CMW Program. In general, maximum concentrations found in *M. edulis* exceeded those in *M. californianus* by about an order of magnitude. However, it is not clear whether this reflects a difference in uptake between the species, or differences in levels of exposure (*M. edulis* prefer protected habitats such as those found in harbors and embayments where exposure to contaminants could be expected to be greater; while *M. californianus* are typically found in exposed, rocky coastal areas usually more removed from direct inputs). Locations where both species naturally occur are relatively rare, and direct comparisons of chlordane uptake were not explicitly undertaken. Results for *M. edulis* and *M. californianus* transplanted to the same sites at the same time are also not available. However, three occurrences exist among the data in which transplanted *M. californianus* and

resident *M. edulis* were collected at the same site and time. The results for the sum of the five chlordane compounds most frequently analyzed, as specified previously, are shown in Table 70.

Table 70. Analytical results for Σ chlordane compounds in tissues of transplanted *M. californianus* and resident *M. edulis* located at the same site and collected on the same date. Values in ppb dry weight. Source: CMW Program.

Site	Date	*M. californianus* concentration	Transplant duration, mos.	*M. edulis* concentration
Sandholt Bridge Old Salinas R. estuary	2/19/85	78.1	4.4	86.0
Sandholt Bridge Old Salinas R. estuary	1/16/86	62.0	4.9	84.7
Shelter Island San Diego Bay	5/16/80	71.5	5.2	51.8

Although this sample is small, the differences between the chlordane concentrations in *M. edulis* were not found to be significantly different (using the Mann-Whitney test) from those in *M. californianus*. If, as these results suggest, interspecies rates of uptake can be considered to be similar, then the wide disparity in maximum concentrations observed in the dataset as a whole could be attributed to such other factors as increased level of environmental exposure.

Table 71 summarizes results for summed concentrations of *cis*-chlordane, *trans*-nonachlor, and heptachlor only, in order to facilitate comparison to results from NOAA's NS&T Mussel Watch Project, which are also included in the table.

Table 71. Comparison of summed concentrations of chlordane compounds (Σ *cis*-chlordane, *trans*-nonachlor, and heptachlor) measured by the California Mussel Watch Program in resident mussels collected along the California coast 1979-1988,and by the NOAA NS&T Mussel Watch Project in bivalves along all the U.S. coasts 1986-1988. Values in ppb dry weight. Source: CMW Program and NOAA NS&T Mussel Watch Project.

Species	N	Mean± Sdev	Min*	Site/Date, Min	Max	Site/Date, Max	Median
CALIFORNIA MUSSEL WATCH RESULTS, 1979-1988							
M. edulis	50	114±156	1.50	Humboldt Channel 2/20/85	995	Colorado Lagoon 1/11/82	54.5
M. californianus	92	12.4±11.9	1.25	Trinidad Head 9/16/82	57.2	Oceanside 3/2/81	9.58
CALIFORNIA MUSSEL WATCH RESULTS, 1986-1988 ONLY							
M. edulis	14	66.8±67.8	8.0	Elkhorn Slough, CA	274	Lauritzen Canal, CA	48.0
M. californianus	19	6.42±5.67	1.50	(several)	16.9	Newport, CA	3.80
NOAA/NS&T-1986							
M. edulis	141	42.2±51.0	0.78	Narragansett Bay, RI	312	New York Bight, NJ	24.7
C. virginica	216	20.6±24.9	0.78	(several)	168	Biloxi Bay, MS	12.0
M. californianus	72	12.7±13.6	0.97	San Simeon, CA	78.6	Anaheim Bay, CA	8.85
O. sandvicensis	6	25.8±25.6	2.00	Barbers Pt., HI	64.9	Honolulu Harbor, HI	19.7
NOAA/NS&T-1987							
M. edulis	151	41.2±45.4	0.81	(several)	231	Long Is. Sound, NY	23.5
C. virginica	211	25.6±47.6	1.47	Rookery Bay, FL	581	Choctawhatchee Bay, FL	14.4
M. californianus	72	22.9±32.3	0.81	(several)	217	Anaheim Bay, CA	11.1
O. sandvicensis	6	19.7±15.0	0.81	Barbers Pt., HI	37.0	Honolulu Harbor, HI	22.0
NOAA/NS&T-1988							
M. edulis	167	21.1±26.6	0.35	(3 locations)	212	Marina del Rey, CA	14.1
C. virginica	66	20.0±23.2	1.04	Laguna Madre, TX	162	Galveston Bay, TX	11.8
M. californianus	78	10.5±11.9	0.35	(2 locations)	68.1	Anaheim Bay, CA	7.28
O. sandvicensis	9	18.5±20.7	0.88	Kauai, HI	51.6	Honolulu Harbor, HI	2.13

* These minima reflect summation of converted concentrations below limits of detection. In both cases, value portrayed is sum of three below detected conversions.

The first data grouping in Table 71 includes all available CMW results for resident mussels between 1979 and 1988. The second restricts the data to the period between 1986 and 1988 to match the temporal coverage of NS&T results. Comparison of the full set of CMW results and the 1986 through 1988 data suggests that more elevated concentrations in both *M. edulis* and *M. californianus* occurred prior to 1986, since excluding those early results lowers the mean, median, and maximum values. Comparing the 1986 through 1988 CMW data to 1986 through 1988 NS&T data shows that maximum concentrations of chlordane compounds encountered in California mussels (*M. edulis*) were about equivalent to maximum concentrations measured by the NS&T Program nationwide. This indicates that the portrayal in NS&T results of certain California locations (specifically, southern California and the Los Angeles region) as areas with relatively elevated chlordane concentrations in biota were accurate. Concentrations found in *M. californianus* by the CMW Program were less than those from the NS&T Mussel Watch Project.

Organic contaminants in coastal areas of Los Angeles and the Southern California Bight, 1985. In a report sponsored by the Los Angeles Region of the California Regional Water Quality Control Board, Risebrough (1987) analyzed samples of sediment and biota collected in the Southern California Bight near Los Angeles for organochlorine compounds, including five known constituents of technical chlordane (*cis*-, *trans*-nonachlor, α–, γ–, and oxychlordane). Samples were collected near urban/industrial centers, as well as in remote island areas. Organisms and tissues examined included whole tissue samples of mussels (*Mytilus californianus*) and benthic invertebrates; muscle and liver of Pacific whiting (*Merluccius*

productus), long-spined thornyhead (*Sebastolobus altivelis*), short-spined thornyhead (*Sebastolobus alascanus*), California rattail (*Nezumia stelgidolepis*), kelp bass (*Paralabrax clathratus*), black surf perch (*Embiotoca jacksoni*), and white croaker (*Genyonemus lineatus*); and blood and egg of bald eagle (*Haliaetus leucocephalus*).

In most cases, data reported by Risebrough for the five chlordane compounds were summed to give a single value for chlordane. Values below detection limits were converted to one half the listed detection limit for statistical purposes; those concentrations detected but not quantitated (*i.e.*, reported as ≤ values) were converted to the value of the quantitation limit. Only those samples for which all five chlordane compounds were measured were used for statistical analysis; if any of the five compounds were not measured, then the sample was excluded from analysis. Table 72 summarizes dry weight results (as reported) for analyses made by Risebrough. Table 73 summarizes the same information, converted to wet weight concentration in order to facilitate comparison to such values as FDA action levels.

Table 72. Summary of summed concentrations of chlordane compounds (Σ *cis*-, *trans*-nonachlor, α-, γ-, and oxychlordane) in sediment, mussels, and fish collected in the Southern California Bight in 1985. Values in ppb dry weight. Source: Risebrough (1987).

Species/Matrix	Tissue	N	Mean± Sdev	Range	Median
Sediment[a]		31	5.36±23.2	0.028-130	0.36
Mussel	Soft body	11	28.12±68.57	1.85-233.8	5.95
Kelp bass	Muscle	41	7.89±6.72	0.52-26.9	4.70
Kelp bass	Liver	57	82.28±91.34	3.20-405	48.3
Black perch	Muscle	10	1.94±2.40	0.42-8.48	1.12
Black perch	Liver	11	10.45±9.51	3.68-37.8	7.08
White croaker	Muscle	13	67.26±70.06	6.20-259.7	48.4
White croaker	Liver	11	176.9±123.8	24.6-375.6	130
L-spined thornyhd	Muscle	24	13.1±12.61	0.55-55.0	9.03
L-spined thornyhd	Liver	15	53.27±14.50	27.2-73.5	59.9
S-spined thornyhd	Muscle	6	7.49±4.51	2.33-14.7	6.61
S-spined thornyhd	Liver	6	64.77±24.0	39.6-93.3	62.5
Cal. rattail	Muscle	12	4.43±3.61	1.7-15.0	3.26
Cal. rattail	Liver	13	63.32±34.98	3.1-143	66.4
Pacific whiting	Muscle	1	1.23	1.23	
Pacific whiting	Liver	1	158.6	158.6	

[a] Relatively high limits of detection and quantitation for sediment analyses necessitate the use of caution in interpretation of these results. For example, the maximum value listed here is actually the sum of five concentrations below limits of detection ranging from 50 to 60 ppb dry weight; conversion protocols, however, yield a value of 130 ppb.

Table 73. Summary of summed concentrations of chlordane compounds (Σ *cis*-, *trans*-nonachlor, α-, γ-, and oxychlordane) in mussels and fish collected in the Southern California Bight in 1985. Values in ppb wet weight. Source: Risebrough (1987).

Species	Tissue	N	Mean± Sdev	Range	Median
Mussel	Soft body	11	10.6±30.7	0.33-103	1.05
Kelp bass	Muscle	41	1.78±1.50	0.12-5.78	1.25
Kelp bass	Liver	57	29.9±31.8	0.83-134	15.65
Black perch	Muscle	10	0.43±0.56	0.09-1.98	0.25
Black perch	Liver	11	4.44±2.83	1.48-11.9	4.03
White croaker	Muscle	13	15.6±16.0	1.15-58.17	11.7
White croaker	Liver	11	79.5±76.8	11.2-240	32.8
L-spined thornyhd	Muscle	24	3.28±2.94	0.11-12.2	2.58
L-spined thornyhd	Liver	15	33.7±9.08	16.3-47.3	35.4
S-spined thornyhd	Muscle	6	3.38±2.11	0.72-6.17	3.46
S-spined thornyhd	Liver	6	13.8±4.47	9.50-19.6	12.0
California rattail	Muscle	12	1.07±0.86	0.32-2.85	0.83
California rattail	Liver	13	46.4±23.3	2.76-88.7	43.8
Pacific whiting	Muscle	1	0.48	0.48	
Pacific whiting	Liver	1	79.1	79.1	

For the six species of fish, a total of 86 muscle-liver pairs were available from the data of Risebrough. Log-transformed values for dry weight concentrations and the regression line for the undifferentiated mixed-species data, are plotted in Figure 63.

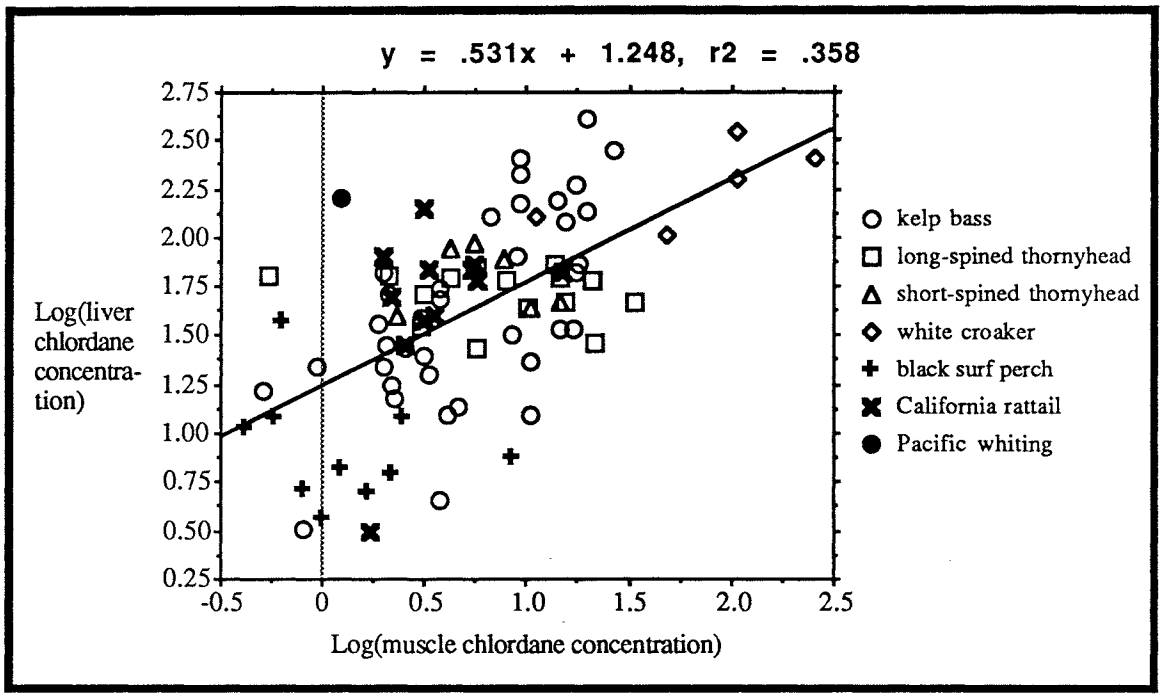

Figure 63. Log transformed values for summed chlordane compounds in liver tissue *vs.* muscle tissue for fish sampled in the Southern California Bight in 1985. Regression line calculated for undifferentiated mixed-species data. Source: Risebrough (1987).

Tables 72, 73, and Figure 63 illustrate the variability inherent in the data, yet also suggest species differences in the relationship between muscle and liver concentrations of chlordane compounds. For example, black surf perch appeared to accumulate chlordane in both tissues to a relatively lesser degree than other species, while white croaker apparently accumulated greater concentrations. Table 73 shows that although some concentrations in mussel tissue and liver tissue of white croaker approached FDA action levels of 0.3 ppm wet weight (=300 ppb), none of the muscle tissues analyzed exceeded that level.

Toxic chemicals and liver lesions in white croaker from the vicinity of Los Angeles, 1984. Because the marine waters near Los Angeles, California, are known to receive substantial amounts of municipal and industrial wastes, resident fish present an opportunity to study the uptake and effects of various kinds of chemical contaminants. Malins *et al.* (1987) collected and studied white croaker (*Genyonemus lineatus*) from the vicinity of Los Angeles and from a nonurban reference area to the southeast, Dana Point. Sediments from collection areas were also analyzed. The same analytical methodologies employed in the NOAA, NS&T Program were used for this study, and the two sets of results can be considered comparable. Figure 64 illustrates approximate site locations in the Los Angeles area, while Table 74 shows results reported in Malins *et al.* (1987).

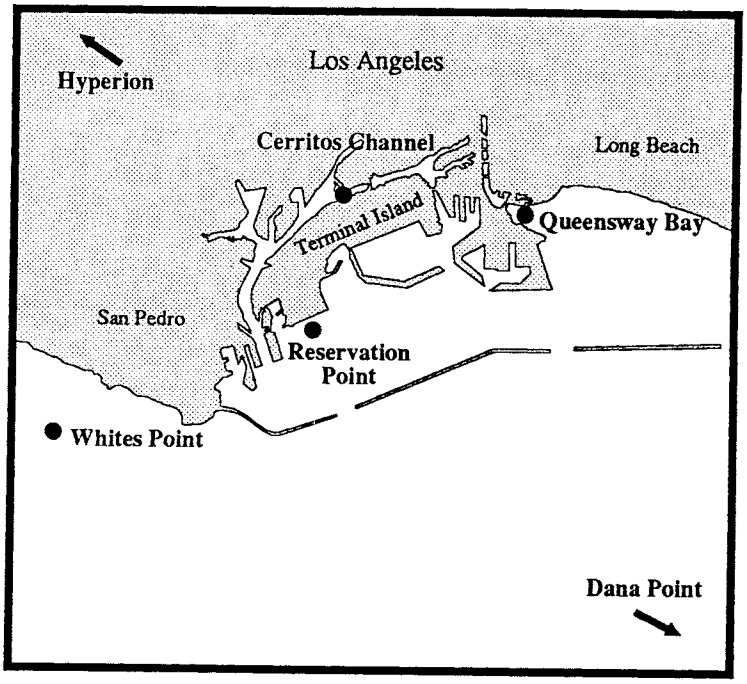

Figure 64. General locations of sites sampled by Malins *et al.* (1987) in the Los Angeles/Long Beach region in 1984. Source: Malins *et al.* (1987).

Table 74. Concentrations of chlordane compounds measured in sediments and tissues of white croaker collected near Los Angeles, California, 1984. Source: Malins *et al.* (1987).

Location	Matrix	α-chlordane ppb dry wt.	*trans*-nonachlor ppb dry wt.
Queensway Bay	Sediment	22	18
	Stomach contents	160	130
	Liver	1000	1300
Cerritos Channel	Sediment	7	6
	Stomach contents	<1	<1
	Liver	460	620
Reservation Point	Sediment	<1	1
	Stomach contents	5	5
	Liver	63	99
Whites Point	Sediment	7	4
	Stomach contents	10	10
	Liver	140	170
Hyperion	Sediment	2	1
	Stomach contents	20	18
	Liver	86	110
Dana Point	Sediment	1	<1
	Stomach contents	75	4
	Liver	19	42

Not surprisingly, the highest concentrations on chlordane compounds reported by Malins *et al.* were at sites located in enclosed portions of Los Angeles and Long Beach harbors. The Queensway Bay site in Long Beach Harbor showed maximum levels of α-chlordane and *trans*-nonachlor in all three substrates analyzed (40 ppb dry weight summed concentration in sediment, 290 ppb in white croaker stomach contents, and 2,300 ppb in white croaker liver). Although Cerritos Channel in inner Los Angeles Harbor yielded the next highest white croaker liver concentration (1,080 ppb), concentrations in sediment and stomach contents were surprisingly low. The sediment concentration of chlordane compounds was about equivalent to those found at the open ocean sites in Santa Monica Bay and off the Palos Verdes Peninsula, while levels for both α-chlordane and *trans*-nonachlor in stomach contents at Cerritos Channel were below detection--the only site of all six to be so distinguished.

It is possible that this distribution of concentrations at the sites resulted from two influences: runoff and dredging activities. Because the Queensway Bay site was located near the mouth of the Los Angeles River, which carries a considerable amount of urban runoff from the region, chlordane loadings and subsequent exposure for resident biota could be expected to be relatively high. The values measured in sediment, croaker stomach contents, and croaker liver appeared to reflect this loading. The Cerritos Channel site, while also receiving industrial runoff from adjacent areas, is regularly dredged for navigation. The low sediment and stomach contents concentrations may have resulted from dredging activities physically removing contaminants from the channel.

San Francisco Ocean Outfall Monitoring Program, 1982-1983. As part of an extensive study to collect information on ecological impacts associated with construction and subsequent operation of the City of San Francisco's ocean outfall in the Gulf of Farallones, the San Francisco Bureau of Water Pollution Control collected and analyzed sediments from four sites (TO01-04) along the open Pacific coast west of the city. Samples of mussels (*Mytilus californianus*) were collected and analyzed from three locations between Point Lobos and Point San Pedro, and one on Southeast Farallon Island. English sole (*Parophrys vetulus*) were collected from two sites (TO01 and TO04) and whole body tissues were analyzed. Among the synthetic organics targeted in the study were the chlordane compounds α- and γ-chlordane, *cis*- and *trans*-nonachlor, and oxychlordane. Results were reported in Bureau of Water Pollution Control and CH^2M Hill (1984). Figure 65 shows locations of the sites sampled and of the ocean outfall. Sediment chlordane concentrations measured around the prospective outfall site are summarized in Table 75. Concentrations in sediment were reported in pg/g (ppt), dry weight basis.

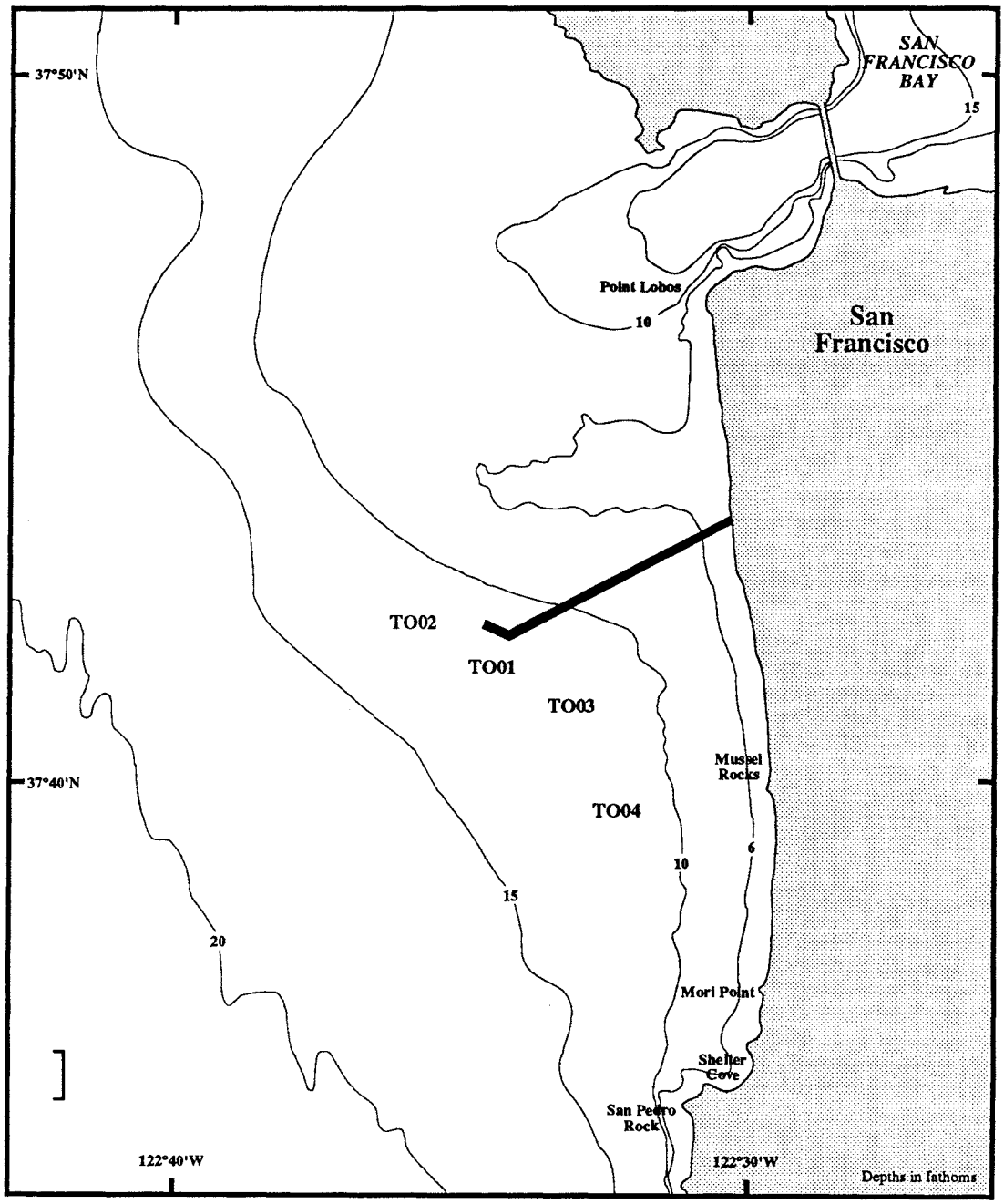

Figure 65. Sediment, mussel, and fish sampling stations for the San Francisco Ocean Monitoring Program, 1982 and 1983. Source: City of San Francisco Bureau of Water Pollution Control and CH^2M Hill (1984).

Table 75. Concentrations of chlordane compounds measured in sediments collected near the site for the San Francisco ocean outfall, 1982. Values in picograms/g (ppt), dry weight. Source: City of San Francisco Bureau of Water Pollution Control and CH^{2}M Hill (1984).

Site designation	α-chlordane	γ-chlordane	*cis*-nonachlor	*trans*-nonachlor	oxy-chlordane
TO01	<2	<2	<3	<40	<3.8
	<2	<2	<3	<10	<2
TO02	15	30	<10	<40	<7
	13	18	<10	20	<10
TO03	<26	26	<10	20	<10
	<30	<30	<10	<30	<10
TO04	14	11	<10	14	<10
	<10	<10	<10	<15	<10

Results for concentrations of chlordane compounds in *Mytilus californianus* are shown in Table 76. Note that values were reported in units of pg/g (ppt) dry weight.

Table 76. Concentrations of chlordane compounds measured in *Mytilus californianus* collected along the coast adjacent to the site for the San Francisco ocean outfall, 1982 and 1983. Values in picograms/g (ppt), dry weight. Source: City of San Francisco Bureau of Water Pollution Control and CH^{2}M Hill (1984).

Site designation/ Date	α-chlordane	γ-chlordane	*cis*-nonachlor	*trans*-nonachlor	oxy-chlordane
Point San Pedro					
11/12/82	2.9	2.1	0.99	3.0	0.13
11/12/82	2.3	2.1	0.74	4.3	0.15
5/16/83	2.1	1.6	0.98	2.2	0.14
5/16/83	2.9	2.2	--	2.9	0.20
S.E. Farallon Island					
10/17/82	3.2	0.78	0.31	2.8	0.18
10/17/82	1.5	1.0	--	2.4	0.15
6/12/83	1.9	0.69	0.53	2.5	0.10
6/12/83	2.3	0.66	0.71	2.7	0.10
Point Lobos					
11/12/82	8.4	9.6	3.3	11	0.18
5/16/83	6.5	7.5	3.1	10	0.15
5/16/83	6.1	7.4	3.3	8.7	0.13
Mussel Rock					
11/12/82	1.9	3.1	1.2	2.5	0.36
11/12/82	3.5	2.5	0.68	3.7	0.17
5/16/83	3.0	3.0	1.5	5.7	0.23
5/16/83	3.4	3.5	1.8	4.6	0.13

Whole body concentrations of chlordanes in English sole tissues are summarized in Table 77. In contrast to values for sediments and mussels, units reported were ng/g (ppb), dry weight.

Table 77. Concentrations of chlordane compounds measured in English sole collected in the Gulf of Farallones, 1982 and 1983. Values in ng/g (ppb), dry weight. Source: City of San Francisco Bureau of Water Pollution Control and CH^{2}M Hill (1984).

Site designation	α-chlordane	γ-chlordane	*trans*-nonachlor	oxy-chlordane
TO01	1.1	0.29	1.7	0.22
	0.87	0.21	1.5	0.12
TO04	2.6	0.81	1.8	0.38
	2.1	0.78	2.0	0.38

The number of data reported make interpretation difficult. Concentrations of chlordane were uniformly low, relative to other results examined in this review. This was not surprising, given the open coastal environment, lack of industrial activity along this section of coast, and the fact that the San Francisco outfall had not yet begun to discharge. The only discernible geographic trend appeared in mussel tissue samples collected at Point Lobos, at the entrance to San Francisco Bay, which showed higher concentrations of chlordane compounds than the other three sampling sites. The authors noted that higher concentrations of compounds like chlordane found in this area indicated that San Francisco Bay itself was a source of contamination to the coastal environment outside the bay, including the outfall site.

Oregon

Analyses of elutriates, native water, and bottom material in selected rivers and estuaries in western Oregon and Washington, 1980 and 1982. The USGS, in cooperation with the USACOE, sampled water and bottom sediment during 1980 and 1982 from rivers and estuaries between Brookings, Oregon (Chetco River estuary) and Baker Bay, Washington. In addition to direct analyses of the collected water and sediments, the investigation also examined contaminant concentration of elutriates, or the supernatant from mechanically agitated mixtures of sediment and water from same locations. The study areas were designated as either dredging or disposal sites. Chlordane (not defined as to isomeric composition) and heptachlor were among the contaminants targeted. Procedures and results were summarized in Fuhrer and Rinella (1983), and Fuhrer (1984). Buchman (1989) determined that analytical methodologies were comparable in the two studies.

Chlordane and heptachlor were detected infrequently, with detection limits of 0.1 and 0.01 µg/l (ppb) in water and elutriates, and 1 and 0.1 µg/kg in sediment, respectively. Reported results for chlordane and heptachlor are summarized in Table 78.

Table 78. Concentrations of chlordane and heptachlor reported in water, sediments, and elutriates collected along the coast of Oregon and Washington, 1980. Positive values italicized. Source: Fuhrer and Rinella (1983), Fuhrer (1984).

Location	Matrix	Date Sampled	Chlordane ppb[1]	Heptachlor ppb[1]
Pacific Ocean	Water	6/11/80	<0.1	<0.01
Pacific Ocean	Water	7/24/80	<0.1	<0.01
Pacific Ocean	Water	7/24/80	<0.1	<0.01
Pacific Ocean	Water	12/2/80	<0.1	<0.01
Pacific Ocean	Water	7/24/80	<0.1	<0.01
Baker Bay,WA	Water	7/25/80	<0.1	<0.01
Baker Bay, WA	Water	7/25/80	<0.1	<0.01
Columbia River	Water	7/24/80	<0.1	<0.01
Columbia River	Water	12/2/80	<0.1	<0.01
Columbia River	Water	7/24/80	<0.1	<0.01
Baker Bay,WA	Elutriate	7/23/80	<0.1	<0.01
Baker Bay, WA	Elutriate	7/23/80	<0.1	<0.01
Baker Bay, WA	*Sediment*	*7/23/80*	*2.0*	*<0.1*
Baker Bay, WA	*Sediment*	*7/23/80*	*2.0*	*0.2*
Baker Bay, WA	Sediment	7/23/80	<1.0	<0.1
Baker Bay, WA	Sediment	7/25/80	<1.0	<0.1

The only quantifiable levels measured were found in sediments with a range of 1.0-4.0 µg/kg, dry weight. The results suggest that chlordane and heptachlor contamination was not widespread in abiotic substrates of coastal and estuarine areas of Oregon and southwestern Washington during the sampling period between 1980 and 1982.

Oregon Department of Environmental Quality monitoring for priority pollutants in fish and shellfish, 1979-1984. Buchman (1989), in his review of trace contaminant data for coastal Oregon, examined results from the Oregon Department of Environmental Quality monitoring program for selected priority pollutants in fish and shellfish. While it was noted that nearly all of these results were freshwater data, estuarine stations within Coos Bay and Tillamook Bay were also sampled. Chlordane (composition not specified) was among those pesticides analyzed. However, residues were not detected in samples of striped bass, clam, and oyster collected between 1979 and 1984.

Organochlorine residues and shell thinning in Oregon seabirds, 1979. Henny, Blus, and Prouty (1982) reported results of an investigation in which eggs were collected from nests of 11 species of seabirds along the southern Oregon coast. Egg contents were subsequently analyzed for pesticides and PCBs, including chlordane compounds. Although the full suite of contaminants analyzed was not listed explicitly, results for the chlordane constituents *cis*- and *trans*-nonachlor and oxychlordane were reported as data incidental to occurrences of DDT and PCB compounds. Table 79 lists species sampled by Henny, Blus, and Prouty and details of collection and analytical results.

Table 79. Seabird species whose eggs were sampled along the Oregon coast, 1979. Concentrations of chlordane compounds determined in egg contents. Values in ppm wet weight. Source: Henny, Blus, and Prouty (1982).

Bird species	N	Collection Sites	Chlordane compounds ppm wet weight
Black oystercatcher (*Haematopus bachmani*)	5	Whale Cove, Hunter Island	--
Pigeon guillemot (*Cepphus columba*)	5	Island Rock, Haystack Rock	--
Pelagic cormorant (*Phalacrocorax pelagicus*)	4	Gull Island, Island Rock	2 eggs *trans*-nonachlor, 0.09 and 0.18
Tufted puffin (*Lunda cirrhata*)	3	Haystack Rock	--
Brandt's cormorant (*Phalacrocorax penicillatus*)	7	Gull Island, Island Rock	3 eggs *trans*-nonachlor, 0.15, 0.17, and 0.20
Snowy plover (*Charadrius alexandrinus*)	1	Sand Lake	--
Common murre (*Uria aalge*)	8	Gull Island, Island Rock	--
Western gull (*Larus occidentalis*)	8	Gull Island, Haystack Rock, Island Rock	--
Double-crested cormorant (*Phalacrocorax auritus*)	10	Hunter Island	2 eggs oxychlordane, 0.12 and 0.16; 1 egg *cis*-nonachlor, 0.09
Leach's storm petrel (*Oceanodroma leucorhoa*)	11	Hunter Island	5 eggs *trans*-nonachlor, 0.09-0.12
Fork-tailed storm petrel (*Oceanodroma furcata*)	1	Haystack Rock	Oxychlordane, 0.22; *trans*-nonachlor, 0.16

Of the chlordane compounds, *trans*-nonachlor was found most frequently in egg contents (11 eggs, 0.09-0.20 ppm wet weight), followed by oxychlordane (3 eggs, 0.12-0.22 ppm), and *cis*-nonachlor (1 egg, 0.09 ppm).

Chlorinated pesticides and polychlorinated biphenyls in marine species, Oregon/Washington coast, 1970-1973. As part of the International Decade for Ocean Exploration Program of the National Science Foundation, Claeys *et al.* (1975) determined concentrations of PCBs and several chlorinated organic pesticides in marine and estuarine organisms and in one species from an Oregon river. Among the pesticides measured was chlordane (composition not specified). In offshore waters, Claeys *et al.* examined pink shrimp (*Pandalus jordani*), euphausiids (*Euphausia pacifica*), and several flatfish species collected

between Newport, Oregon, and the Strait of Juan de Fuca, Washington. Estuarine bivalves were collected from five Oregon estuaries (Columbia River, Tillamook Bay, Yaquina Bay, Umpqua River, and Coos Bay). The bivalve species collected included cockle (*Clinocardium nuttallii*), eastern soft-shell clam (*Mya arenaria*), bay mussel (*Mytilus edulis*), Asiatic clam (*Corbicula fluminea*), and a species of *Anodonta*. Estuarine fish were collected in Coos Bay and the Columbia River estuary and steelhead trout (*Oncorhynchus mykiss*) in the Rogue River.

Within these ranges of organisms and habitats, chlordane was detected only in the summer-run steelhead trout sampled in the Rogue River in Oregon, at a concentration of 6 µg/kg. Although detection limits were not specified for chlordane, they ranged from 0.3 to 5 µg/kg for other chlorinated organic pesticide compounds.

Washington

Toxicant Pretreatment Planning Study, 1979-1984. In 1979, the Municipality of Metropolitan Seattle (Metro) initiated a major study of toxic substances in its municipal and stormwater waste streams, as well as in the discharge receiving environments, Puget Sound and Lake Washington (Seattle, Washington). Called the Toxicant Pretreatment Planning Study (TPPS), the 5-year, greater than $7 million project consisted of many individual components designed to provide information about the origins and fate of toxic substances in the Seattle and central Puget Sound region. EPA priority pollutants were the primary focus of the Metro study, and among these were chlordane (α- and γ- isomers) and heptachlor. Results were reported in a prolific series of TPPS publications, Romberg *et al.* (1984) among others.

The two compounds were targeted in both dissolved and particulate phases of ambient seawater, as well as in sediments from over 200 samples collected in Puget Sound. Although a number of familiar organochlorines were frequently detected in the water and sediment samples (Aroclor 1254, for example, was found in 94 percent of suspended particulate samples, and 96 percent of sediment samples), both chlordane and heptachlor went undetected throughout a total of 32 water samples and 213 sediment samples (mean detection limits listed for α-chlordane were 0.00075 ppb in water and 0.075 ppb in sediment and tissue; for γ-chlordane, 0.00011 ppb in water and 0.011 ppb in sediment and tissue; and for heptachlor, 0.00015 ppb in water and 0.150 ppb for sediment and tissue).

Chemical contaminants in edible non-salmonid fish and crabs from Commencement Bay, Washington, 1981-1982. Commencement Bay, near Tacoma, Washington, has been recognized as being one of the more contaminated embayments in Puget Sound. Following reports by various agencies that certain waterways in Commencement Bay contained relatively high concentrations of toxic chemicals, the Environmental Services Division of the U.S. EPA Region 10 initiated a study designed to provide the Tacoma-Pierce County Health Department with chemical data needed to assess health risks to persons eating fish and crab from the area. Results were reported in Gahler *et al.* (1982).

A collection of 6 Dungeness crab (*Cancer magister*) and 86 non-salmonid fish was made and edible tissues analyzed for EPA priority pollutants, including metals and organic contaminants. Thirteen species of fish were sampled: walleye pollock (*Theragra chalcogramma*), Pacific hake (*Merluccius productus*), Pacific tomcod (*Microgadus proximus*), Pacific cod (*Gadus macrocephalus*), English sole (*Parophrys vetulus*), rock sole (*Lepidopsetta bilineata*); flathead sole (*Hippoglossoides elassodon*); starry flounder (*Platichthys stellatus*), C-O sole (*Pleuronichthys coenosus*), Pacific staghorn sculpin (*Leptocottus armatus*), whitespotted greenling (*Hexagrammos stelleri*), buffalo sculpin (*Enophrys bison*), and rockfish (*Sebastes sp.*). Among the many organic compounds targeted in the analyses were chlordane (not specifically identified) and heptachlor. However, although other priority pollutants were found in the edible tissues analyzed, the only pesticides detected in any of the tissue samples were three DDT compounds. Reported detection limits for chlordane and heptachlor were 1 ppb, wet weight.

Chemical contaminants and biological abnormalities in central and southern Puget Sound, 1979. In a cooperative effort between two groups within NOAA, the Office of Marine Pollution Assessment and the NMFS, a multidisciplinary program of research oriented toward provision of information on the occurrence, fluxes, transport, fate, and effects of contaminants of concern in Puget Sound was initiated in 1979. A portion of this study consisted of chemical analyses to identify the concentrations and distribution of hydrocarbons and metals in both sediments and tissues of resident biota. Summaries of the results from the first year of analyses were contained in Malins *et al.* (1980). Four urban-associated locations (Elliott Bay, Commencement Bay, Budd Inlet, Sinclair Inlet) were sampled for the study, as were two reference areas (Case Inlet, Port Madison). Among the 50 organic compounds analyzed in both sediments and tissues were

3 constituents of technical chlordane: α-chlordane, *trans*-nonachlor, and heptachlor. Table 80 summarizes sediment results for summed concentrations of chlordane reported by Malins *et al.*

Table 80. Concentrations of Σ chlordane (sum of α-chlordane, *trans*-nonachlor, and heptachlor) measured in sediments sampled at six sites in Puget Sound, 1979. Values in ng/g (ppb), dry weight basis. Source: Malins *et al.* (1980).

Location	N	# Samples w/all bd[1]	Σ Chlordane Range
Elliott Bay	17	12	bd-8.0
Commencement Bay	14	2	bd-50
Budd Inlet	3	3	(all bd)
Sinclair Inlet	4	0	bd-1.7
Port Madison	2	1	bd-0.4
Case Inlet	2	2	(both bd)

[1]bd=below detection, with variable detection limits.

Liver tissue samples were taken from five species of fish: English sole (*Parophrys vetulus*), rock sole (*Lepidopsetta bilineata*), Pacific staghorn sculpin (*Leptocottus armatus*), Pacific tomcod (*Microgadus proximus*), and quillback rockfish (*Sebastes maliger*). Four general classes of invertebrates were also chemically analyzed at many of the six sites. Samples were presumably composites of several individuals. Species included in the invertebrate samples were identified as *Cancer magister*, *C. gracilis*, and *C. productus* for crab; *Pandalus danae*, *P. jordani*, and *Crangon alaskensis* for shrimp; *Macoma carlottensis*, *M. nasuta*, and *Acila castrensis* for clam; and *Glycera capitata*, *Capitella capitata*, and *Prionospio pinnata* for marine worm. Summarized results for tissue samples are shown in Table 81.

Table 81. Concentrations of Σ chlordane (sum of α-chlordane, *trans*-nonachlor, and heptachlor) measured in fish livers and other tissues sampled at six sites in Puget Sound, 1979. Values in ng/g (ppb), dry weight basis. Source: Malins *et al.* (1980).

Location	N	# Samples w/all bd[1]	Σ Chlordane[2] Range
Elliott Bay			
English sole liver (individual)	3	0	bd-80
English sole liver (composited)	6	0	60-220
Rock sole liver (individual)	1	0	60
Rock sole liver (composited)	4	0	80-180
Pacific staghorn sculpin liver (individual)	3	0	50-310
Quillback rockfish liver (individual)	4	0	20-170
Crab	1	0	110
Worms	3	0	2.0-3.0
Shrimp	2	1	bd-2.5
Clams	2	0	1.7-6.0
Commencement Bay			
English sole liver (individual)	5	0	60-190
English sole liver (composited)	4	0	4.0-260
Rock sole liver (individual)	3	0	10-95
Rock sole liver (composited)	5	0	5.0-150
Pacific staghorn sculpin liver (individual)	2	0	71-360
Pacific staghorn sculpin liver (composited)	4	0	170-450
Quillback rockfish liver (individual)	1	0	41
Quillback rockfish liver (composited)	1	0	280

Table 81. Continued

Location	N	# Samples w/all bd[1]	Σ Chlordane[2] Range
Pacific tomcod liver (composited)	1	0	60
Crab	2	0	230-250
Worms	3	2	bd-3.0
Shrimp	2	0	2.5-3.0
Clams	2	0	6.0-22
Budd Inlet			
English sole liver (composited)	1	0	40
Rock sole liver (composited)	1	0	30
Crab hepatopancreas	1	0	8.2
Shrimp	1	0	1.7
Sinclair Inlet			
English sole liver (composited)	1	0	70
Rock sole liver (composited)	1	0	80
Crab hepatopancreas	1	0	20
Worms	1	1	bd
Shrimp	1	0	3.5
Clams	1	0	9.5
Port Madison			
English sole liver (composited)	1	0	45
Rock sole liver (composited)	1	0	25
Worms	1	1	bd
Shrimp	1	0	2.5
Clams	1	0	0.8
Case Inlet			
English sole liver (composited)	1	0	30
Rock sole liver (composited)	1	0	20
Crab hepatopancreas	1	0	4.2
Shrimp	1	1	bd
Clams	1	1	bd

[1]bd=below detection, with variable detection limits.
[2]bd concentrations converted to one half reported detection limit for summary purposes.

Although the total number of samples was not great, some general observations about the data can be made. Ranking on the basis of maximum concentrations found at each general location was consistent in both sediment and tissue analyses: Commencement Bay samples exhibited highest values in both sediments and tissues, with Elliott Bay samples next, followed by Sinclair Inlet collections. Budd Inlet, Port Madison, and Case Inlet showed about equivalent levels. It was not surprising that the highest concentrations were measured in embayments adjacent to the two largest cities on Puget Sound (Seattle and Tacoma). Sinclair Inlet is close to Bremerton, an area with major naval and shipyard activity. Although Budd Inlet is adjacent to the State capital of Olympia and is located at the southernmost end of Puget Sound, concentrations of chlordane compounds encountered there were relatively low and about the same as those found at the geographically removed and rural in character reference sites.

Highest tissue concentrations of chlordane compounds were found in fish liver. The levels of summed chlordane compounds found by Malins *et al.* were low compared to benchmark human health standards and other residue concentrations found elsewhere, even at those Puget Sound locations with highest values. Malins *et al.* reported concentrations on a dry weight basis, and when the maximum tissue levels are considered in the wet weight terms commonly used in assessing health risks, the concentrations are reduced by an approximate factor of five. Furthermore, tissues containing highest concentrations were liver samples, which typically contain much higher concentrations of chlorinated organic compounds than the edible muscle tissues. Therefore, while the results provide insights into comparative aspects of chlordane contamination in fish from various Puget Sound locations, they also enable a degree of comparison to results from other areas and national standards of acceptability from a human health perspective as well.

Compared to chlordane levels measured in fish liver, concentrations in invertebrates sampled at the same site were much lower, generally not detected or an order of magnitude less. This suggests that the bottomfish were concentrating and retaining body burdens of chlordane compounds to which they were exposed.

Puget Sound pollution and its effects on marine biota, 1980. The NMFS continued the work whose initial results were reported in Malins *et al.* (1980), and summarized results for the inclusive period between 1978 and 1981 in Malins *et al.* (1982). Details of chemical analyses were reported in unpublished progress reports to the contracting office, Malins *et al.* (1981a) and (1981b). Those results are summarized in Table 82.

Table 82. Concentrations of Σ chlordane (sum of α-chlordane, *trans*-nonachlor, and heptachlor) measured in sediments sampled at six sites in Puget Sound, 1980. Values in ng/g (ppb), dry weight basis. Source: Malins *et al.* (1981a).

Location	N	# Samples w/all bd[1]	Σ Chlordane[2] Range
Commencement Bay	14	2	bd-13.7
Elliott Bay	2	0	4.0-8.2
Port Susan	2	2	bd
Quartermaster Harbor	1	0	0.12

[1] bd=below detection, with variable detection limits.

[2] bd concentrations converted to one half reported detection limit for summary purposes.

Fewer sediment samples were analyzed and reported in Malins *et al.* (1981a) than in the previous year's report. The two major urban embayments of Elliott Bay and Commencement Bay yielded results for summed chlordane compounds roughly equivalent to the earlier samples, although Commencement Bay sediments showed lower maximum concentrations in 1981.

Fish tissue measurements from samples collected in 1980 were reported in Malins *et al.* (1981a) and (1981b) with English sole (*Parophrys vetulus*) results contained in the former and salmon (*Oncorhynchus tshawtscha* and *O. kisutch*) and Pacific cod (*Gadus macrocephalus*) data in the latter. English sole were sampled in Elliott Bay and Commencement Bay, salmon and cod were collected in Commencement Bay and at the relatively unimpacted Point Jefferson. Data for these collections are summarized in Table 83.

Table 83. Concentrations of Σ chlordane (sum of α-chlordane, *trans*-nonachlor, and heptachlor) measured in fish tissues sampled at three embayments in Puget Sound, 1980. Values in ng/g (ppb), wet weight basis. Source: Malins *et al.* (1981a), (1981b).

Location	N	# Samples w/all bd[1]	Σ Chlordane[2] Range
Elliott Bay/Duwamish estuary			
English sole muscle	5	0	1.31-12.9
English sole liver	5	0	13.0-104
Commencement Bay			
English sole muscle	2	0	1.21-2.8
English sole liver	2	0	17.9-26.6
Salmon muscle	5	0	0.99-2.0
Salmon liver	5	0	1.30-4.1
Pacific cod muscle	3	0	0.27-1.8
Pacific cod liver	3	0	77-102
Point Jefferson			
Salmon muscle	5	0	0.57-5.0
Salmon liver	5	0	0.73-3.3
Pacific cod muscle	3	0	0.07-0.46
Pacific cod liver	3	0	51-109

[1] bd=below detection, with variable detection limits.

[2] bd concentrations converted to one half reported detection limit for summary purposes.

One important difference in the tissue data reported for the 2 years was that the Malins *et al.* (1981b) results were reported in wet weight concentrations, while the earlier data were dry weight. This means that to compare the two sets of results would require conversion of one or the other to an equivalent reporting basis. For example, the 1981 data would have to be multiplied by about a factor of five to give dry weight values for comparison to the 1980 data. On the other hand, the wet weight values could be evaluated as reported in terms of FDA action levels and other health concerns. Measurements of chlordane compounds in edible tissues of English sole, salmon, and Pacific cod were well below the established FDA action level of 300 ppb wet weight.

The more limited data from Malins *et al.* (1981b) suggest differences from the earlier analyses. Highest concentrations (in English sole liver) were found at Elliott Bay sites instead of Commencement Bay sites. Maximum levels measured at Elliott Bay sites were considerably higher in the 1981 report than in the 1980 effort, while highest concentrations in Commencement Bay were lower.

The results for salmon and cod suggested that body burdens of chlordane compounds were not related to collection site characteristics, in that concentrations from Commencement Bay and the relatively rural Point Jefferson were comparable in magnitude. This may have been an indication of the migratory nature of salmon and cod, and that the species did not spend a long enough period of time to reflect contamination levels at specific sites (in contrast to resident bottomfish species).

Assessment of toxic pollutants in English sole and rock sole from Everett Harbor and Port Gardner, 1982. In autumn of 1982, federal (NMFS) authorities alerted Washington State officials that analysis of bottomfish samples from Everett Harbor, in northern Puget Sound, indicated a higher than normal incidence of liver disease associated with environmental contamination. In order to determine whether a problem in fact existed and to provide information relevant to judging health risks to human consumers of fish from the area, the Washington Department of Ecology (WDOE) sponsored a survey in Everett Harbor. Fish (English sole, *Parophrys vetulus* and rock sole, *Lepidopsetta bilineata*) from popular fishing areas were examined for incidence of disease, and chemical contaminants in edible tissues were identified and quantified. Details of the study and summary of results were reported in a WDOE unpublished memorandum (Cunningham, 1982).

Tissue samples were scanned for 114 organic priority pollutants as well as metals. Among the priority organic compounds targeted were technical chlordane and metabolites and heptachlor and metabolites.

However, no residues of chlordane-related compounds were found in liver or muscle tissues from any of the 26 fish analyzed for the study.

Chemical analyses of samples from deceased gray whale specimen, 1984. In June of 1984, a dead gray whale (*Eschrichtius robustus*) was found near Port Angeles, Washington. A pathological investigation funded by the State of Washington concluded that elevated concentrations of synthetic organic compounds was the likely cause of death. However, analysis of samples from the whale which were performed by NOAA, NMFS in Seattle resulted in a different set of values and determined a less certain cause of death. Although the NOAA results were generated under the same analytical protocols as the NS&T Program, the whale tissue measurements were reported on a wet weight basis (NS&T Program results are typically dry weight basis). As in the NS&T Program, three chlordane compounds were measured in the gray whale: α-chlordane, *trans*-nonachlor, and heptachlor. Concentrations found in various tissues of the whale were reported in Malins, Brown, and Chan (unpublished), and are shown in Table 84.

Table 84. Concentrations of chlordane compounds measured in tissues of gray whale found dead near Port Angeles, Washington, 1984. Values reported as wet weight. Source: Malins, Brown, and Chan (unpublished).

Tissue	Sample weight g	α-chlordane ppb wet wt.	*trans*-nonachlor ppb wet wt.	Heptachlor ppb wet wt.
Stomach contents	3.12	1.0	12	<0.86
Liver	3.10	1.7	40	<0.76
Feces	3.13	<0.066	10	<0.11
Blood, from heart	3.21	1.1	24	0.5
Kidney	3.14	0.39	9.7	0.33
Blubber	3.24	<0.13	3.8	<0.20
Blank	-	<0.98	<1.1	<1.2
Blank	-	<0.13	<0.15	<0.20
Spiked blank	-	130	120	91

Interpretation of results from this study is difficult, in that sample size is limited and collection was from a single dead animal. The fact that the whale may have been suffering from pathological conditions either related or unrelated to contaminant burdens also complicates the task. Bearing these *caveats* in mind, some general observations can be made. None of the measured levels of the three compounds appeared to be excessive relative to other studies in fish and marine mammals. However, the low concentrations of chlordanes found in blubber tissue were surprising, as this is usually the tissue reservoir with the highest concentrations measured in a marine mammal. The fact that relatively highest concentrations were found in liver and blood, together with the low levels in blubber, raises the possibility that the animal had been stressed and had tapped fat reserves in blubber as an energy source. This in turn would have mobilized lipophilic organochlorine compounds from blubber into the bloodstream, to be ultimately processed by the liver. It is possible that mobilization of sequestered contaminants may have further contributed to the stressed condition of the animal, but that is speculative.

Reconnaissance survey of environmental conditions in 13 Puget Sound locations, 1988. As part of the U.S. EPA's national program for estuarine studies and pollution abatement, a reconnaissance survey of 13 areas in Puget Sound was undertaken in 1988. The study was multidisciplinary and had multiple goals, including characterization of existing contaminant problems in non-urban and non-industrial embayments and definition of pesticide distribution and concentration. Sediment and fish tissue chemistry were components of the analytical program, and two technical chlordane constituents, α-chlordane and heptachlor, were targeted. Fish species in which contaminant concentrations were measured included English sole (*Parophrys vetulas*), rock sole (*Lepidopsetta bilineata*), starry flounder (*Platichthys stellatus*), and sanddab (*Citharichthys sp.*). Program details and results of analyses were reported in Crecelius, Woodruff, and Myers (1989). General locations of sites sampled are shown in Figure 66. Table 85 summarizes results for the two chlordane compounds in sediment and fish tissues.

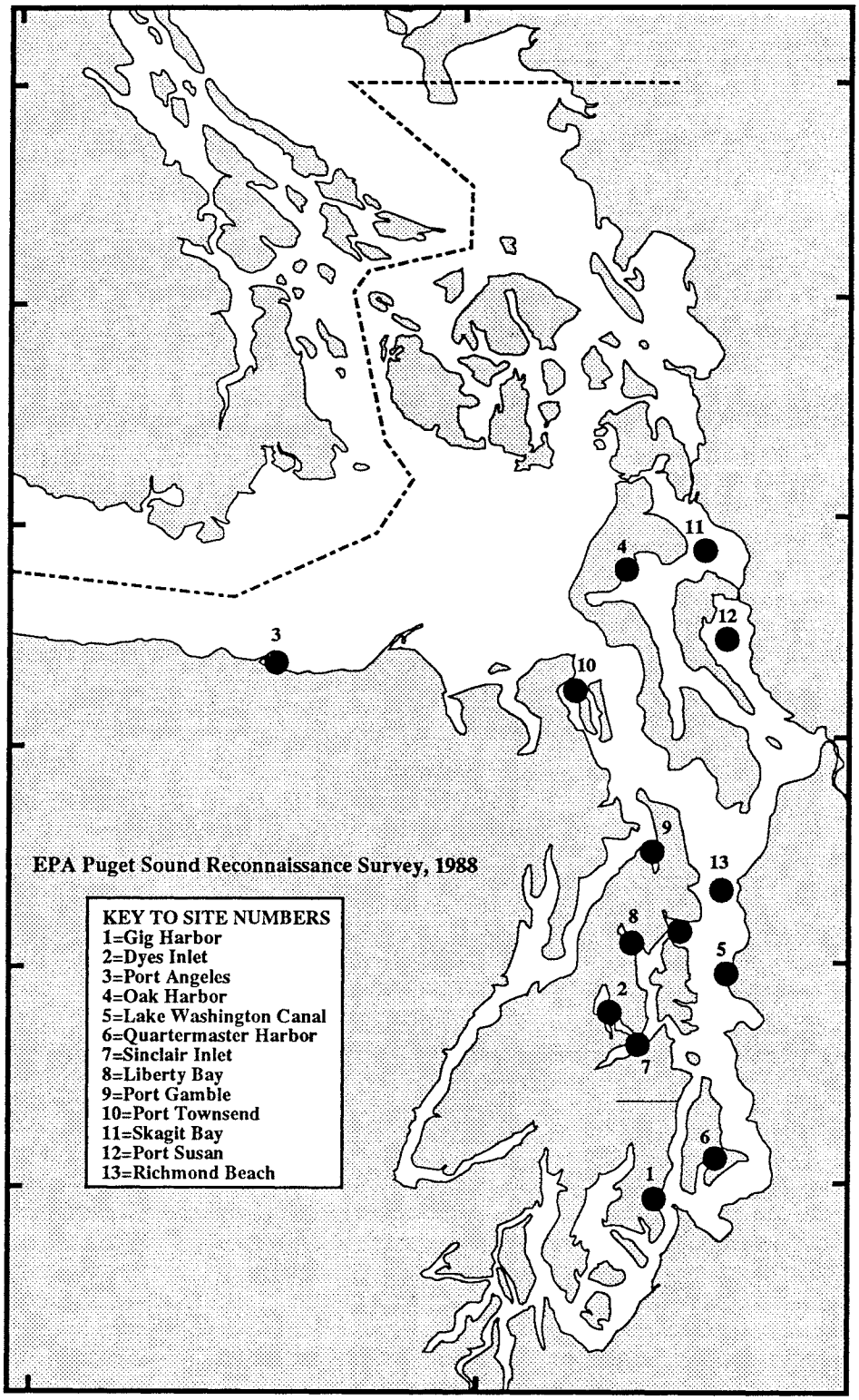

Figure 66. Locations of sites sampled in the 1988 U.S. EPA Puget Sound Reconnaissance Survey. Source: Crecelius, Woodruff, and Myers (1989).

Table 85. Concentrations of α-chlordane and heptachlor measured in tissues of fish sampled at 13 sites in Puget Sound, 1988. Values in μg/kg (ppb). Source: Crecelius, Woodruff and Myers (1989).

Location	Site Number	Matrix/ Species	Tissue	α-chlordane[1]	Heptachlor[1]
Gig Harbor	1	Sediment	-	<0.5	<0.5
		English sole	Muscle	0.6	<0.1
			Liver	<5.0	<5.0
Dyes Inlet	2	Sediment	-	<0.5	<0.5
		English sole	Muscle	0.6	<0.1
			Liver	<5.0	<5.0
Port Angeles	3	Sediment	-	<0.5	<0.5
		Sanddab	Muscle	0.4	<0.1
			Liver	5-25	<5.0
Oak Harbor	4	Sediment	-	<0.5	<0.5
		Starry flounder	Muscle	0.7	<0.2
			Liver	5-25	<5.0
Lake Washington Ship Canal	5	English sole	Muscle	2.0	<0.1
			Liver	5-25	<5.0
Quartermaster Harbor	6	English sole	Liver	<5.0	<5.0
Replicate 1			Muscle	0.4	<0.3
Replicate 2			Muscle	0.3	<0.3
Replicate 3			Muscle	0.7	<0.2
Sinclair Inlet	7	English sole	Muscle	0.8	<0.1
			Liver	<5.0	<5.0
Liberty Bay	8	Rock sole	Muscle	0.2	<0.1
			Liver	<5.0	<5.0
Port Gamble	9	English sole	Muscle	<1.0	<0.1
			Liver	<5.0	<5.0
Port Townsend	10	Rock sole	Muscle	0.5	<0.2
			Liver	<5.0	<5.0
Skagit Bay	11	Starry flounder	Muscle	0.8	<0.1
			Liver	5-25	<5.0
Port Susan	12	English sole	Muscle	0.4	<0.3
			Liver	<5.0	<5.0
Richmond Beach	13	English sole	Muscle	0.4	<0.3
			Liver	<5.0	<5.0

[1] Sediments measured on dry weight basis, tissues on wet weight basis.

The reconnaissance study was designed to focus attention on non-urban bays and inlets in Puget Sound and results for residue levels of chlordane constituents, which in recent years have become identified as primarily urban contaminants, appeared to reflect this. Neither chlordane nor heptachlor was found at concentrations above detection limits in sediments at the four sites where sediments were analyzed for levels of organic compounds. Generally low concentrations were measured in fish tissues as well. While the highest muscle tissue concentration (2.0 ppb) was found at the site with the most urban/industrial character, the Lake Washington Ship Canal, all wet weight concentrations in tissues were two to three orders of magnitude below the established FDA action levels for chlordane constituents. On the basis of these results, it would appear that the non-urban embayments in Puget Sound have not been heavily impacted by chlordane inputs.

Hawaii

Organochlorine pesticide residues in Hawaiian water, sediment, algae, and fish, 1970-71. Bevenue *et al.* (1972) analyzed Hawaiian rainwater, drinking water, nonpotable waters, sediments, and biota (algae and fish) collected between August 1970 and February 1971 for a number of organochlorine pesticides. Chlordane was not included in the full suite of analyses, but results were reported for waters and sediments of two urban canals located in Honolulu. These are summarized in Table 86.

Table 86. Concentrations of chlordane measured in water and sediments of two urban canals in Honolulu, Hawaii, February 1971. Source: Bevenue *et al.* (1972).

Location	Matrix	Date Sampled	Chlordane ppb[1]
Ala Wai Canal, Sta.1	Water	2/18/71	0.0094
	Sediment	2/18/71	720
Ala Wai Canal, Sta. 2	Water	2/26/71	0.013
	Sediment	2/18/71	290
Ala Wai Canal, Sta. 3	Water	2/18/71	0.0048
	Sediment	2/18/71	125
Kapalama Canal, Sta. 1	Water	2/16/71	0.0030
	Water	3/1/71	0.0044
Kapalama Canal, Sta. 2	Sediment	2/9/71	255
Kapalama Canal, Sta. 3	Sediment	2/19/71	125

[1]Sediment concentrations on dry weight basis.

Unfortunately, because chlordane was not included in all analyses performed for the study, comparisons to other estuarine and freshwater environments sampled at that time are not possible. Chlordane residues were not measured in biota in this study. However, available results do provide a general basis for comparison to environmental residues measured by others in the same and in other parts of the State.

Organochlorine pesticides in the Hawaii Kai Marina, 1970-74. Tanita *et al.* (1976) monitored levels of chlordane (α- and γ- isomers) in water, sediments, and oysters (of unspecified species) in the Hawaii Kai Marina region, off Maunalua Bay on the southeastern side of the island of Oahu. The area had been utilized for agriculture and livestock rearing prior to the 1960s, but subsequent to 1963 was considered to be fairly free of industrial and agricultural pesticide inputs. In 1962, construction of a suburban residential housing development and a large marina began there. All home sites in the vicinity were treated for subterranean termites prior to construction.

Tanita *et al.* could not quantify levels of α-chlordane and γ-chlordane in water samples from the marina area due to interference with other compounds, but found that residues were detected in 43 percent and 68 percent, respectively, of the samples analyzed. In sediment samples, they determined an average concentration of 2.97 ppb dry weight (range 0.40-5.27 ppb) for α-chlordane, and 2.30 ppb (range 1.33-5.12 ppb) for γ-chlordane. The former was detected in 97 percent of samples analyzed, and the latter in 93 percent. For oyster tissue analyses, Tanita *et al.* divided the sample into a small organism fraction (< 1 g) and a large size fraction. Oyster results are shown in Table 87.

Table 87. Concentrations of chlordane measured in oysters collected in the Hawaii Kai Marina region of Oahu, Hawaii, April-June 1974. Values in ppb, wet weight. Source: Tanita *et al.* (1976).

Analyte	Mean, Small	Range, Small	Mean, Large	Range, Large	% Occur, Small	% Occur, Large
α-chlordane	18.6	2.34-57.6	8.28	1.58-23.0	33.3	100
γ-chlordane	8.17	ND-8.17	7.86	1.35-23.4	13.3	63.6

These results would suggest bioaccumulation of chlordane compounds by oysters, with concentrations in oyster tissue exceeding those in water by three orders of magnitude, and in sediment by a factor of about three. The measurements in oyster tissue also indicated that although smaller oysters contained higher mean concentrations of both chlordane isomers, the frequency of occurrence for both was considerably higher in large organisms. The range of values obtained in oysters showed that in 1974, some individuals from the Hawaii Kai area exceeded the present FDA action level for chlordane in edible seafood.

Distribution of heavy metals, chlorinated pesticides, and PCBs in the Hawaiian estuarine environment, 1977-1978. As part of a Hawaii Department of Health water monitoring program that targeted intensive surveys on issues of concern, the occurrence and distribution of selected contaminants in the Hawaiian estuarine environment were examined. Sediments, shellfish, and finfish from ten major estuaries (Ala Wai Canal, Honolulu Harbor/Kapalama Canal, Keehi Lagoon, Kaneohe Bay, and Kahana Bay on Oahu; Nawiliwili Bay and Port Allen Bay on Kauai; and Hilo Bay on Hawaii) were sampled and analyzed for levels of metals and organic compounds. Included among the targeted analytes were three chlordane compounds: α- and γ-chlordane, and *trans*-nonachlor. Results were reported in Hawaii Department of Health (1978), and are summarized in Tables 88, 89, and 90.

Table 88. Summary of sample analyses for ten estuarine locations in the State of Hawaii, 1977-1978. Sediment values reported as dry weight, biotic values as wet weight. Source: Hawaii Department of Health (1978).

Compound	Matrix	Detection Limit, ppb	N	Number Quantifiable	Mean ppb	Range ppb
α-chlordane	Sediment	10	91	17	100	20 - 353
	Biota	10	27	7	50	6 -150
γ-chlordane	Sediment	10	91	19	77	15 - 347
	Biota	10	27	6	98	6 - 270
trans-nonachlor	Sediment	15	91	0*	76	19 - 264
	Biota	10	27	7	101	6 - 310

* This is possibly a tabular error in the report from which this is reproduced, as mean and range are well above detection limit.

Table 89 shows that sediment residues of the three chlordane compounds were not detected in six of the ten sampling sites. In Hilo Bay on Hawaii and in Nawiliwili Bay on Kauai, relatively low quantified concentrations of chlordane were found in sediments (Σ chlordane compounds 44 and 84.2 ppb dry weight, converting below detected concentrations to one half the detection limit of 10 ppb). Relatively elevated sediment concentrations (296 and 567 ppb dry weight) were found in Kapalama and Ala Wai Canals, located along the urban waterfront areas of Honolulu.

Table 89. Summary of sediment analyses by site for ten estuarine locations in the State of Hawaii, 1977-1978. Values reported as dry weight. Source: Hawaii Department of Health (1978).

Location	Island	α-chlordane ppb dry wt.	γ-chlordane ppb dry wt.	*trans*-nonachlor ppb dry wt.
Kaneohe Bay	Oahu	<10	<10	<10
Pearl Harbor	Oahu	<10	<10	<10
Ala Wai Canal	Oahu	98.7	116.6	80.5
Kapalama Canal	Oahu	147.0	156.3	264.0
Kaika Bay	Oahu	<10	<10	<10
Kahana Bay	Oahu	<10	<10	<10
Nawiliwili Bay	Kauai	19	20	<10
Hanapepe Bay	Kauai	<10	<10	<10
Manele/Hulopoe	Lanai	<10	<10	<10
Hilo Bay	Hawaii	25.6	30.0	28.6

Table 90. Summary of fish and shellfish analyses for ten estuarine locations in the State of Hawaii, 1977-1978. Values reported as wet weight. Source: Hawaii Department of Health (1978).

Species	Location	Tissue	N	α-chlordn ppb wet wt.	γ-chlordn ppb wet wt.	*t*-nonachlor ppb wet wt.
Mullet						
(*Mugil cephalus*)	Hilo, Waiakea Pond	Flesh	2	60	40	60
	Hilo, Waiakea Pond	Viscera	4	220	120	290
	Hilo, Waiakea Pond	Flesh	3	30	20	30
	Hilo, Waiakea Pond	Viscera	3	270	150	310
	Nawiliwili Bay	Flesh	3	<10	<10	<10
	Nawiliwili Bay	Viscera	3	<10	<10	<10
	Molokai (Kawela)	Flesh	1	<10	<10	<10
	Molokai (Kawela)	Viscera	1	<10	<10	<10
Crab, Hawaiian						
(*Podophthalmus vigil*)	Kaneohe Bay-STP outfall		12	<10	<10	<10
	Kaneohe Bay-Kahaluu		21	<10	<10	<10
	Ala Wai Canal		12	6	<10	9
	Pearl Harbor, West Loch		14	<10	<10	<10
	Nawiliwili Bay		36	<10	<10	<10
Crab, white						
(*Portunus sanguino-*	Kaneohe Bay-STP outfall		15	<10	<10	<10
lentus)	Pearl Harbor, West Loch		7	<10	<10	<10
	Hilo Bay		38	<10	<10	7
Crab, blue claw						
(*Thalamita crenata*)	Kapalama Canal		25	<10	<10	<10
	Pearl Harbor, West Loch		12	<10	<10	<10
	Kaiaka Bay		20	<10	<10	<10
	Ala Wai canal		24	4	<10	7
	Nawiliwili Bay		12	<10	<10	<10
Clam						
(*Tapis phillipinarium*)	Kaneohe Bay		153	<10	<10	<10
	Kaneohe Bay		142	<10	<10	<10
	Pearl Harbor, Middle Loch		103	<10	7	<10
	Pearl Harbor, Middle Loch		123	<10	6	<10
	Pearl Harbor, Middle Loch		122	<10	6	<10
Opihi						
(*Cellana exarata*)	Hulopoe		134	<10	<10	<10

Summed chlordane concentrations were uniformly low in mollusks and crabs sampled at all locations, including those in urban embayments and harbor areas. Of 19 analyses for α-, γ-chlordane, and *trans*-nonachlor, 13 yielded concentrations below detection for all three compounds, while the other six ranged from 16 to 20 ppb wet weight (again using one half the listed detection limits for below detected values).

Mullet sampled at Nawiliwili Bay and Kawela (Molokai) contained the chlordane compounds at concentrations below detection limits in both muscle and viscera. At Waiakea Pond on Hilo Bay, however, mullet contained summed levels of chlordane at 80 and 160 ppb wet weight in muscle tissue and 630 and 730 ppb in viscera. The source report noted that Waiakea Pond had been plagued with a variety of pollution-related problems as early as 1935, so the relatively higher concentrations measured in fish at this site should not have been surprising; even so, levels in edible tissues were well below the FDA action level concentration of 300 ppb.

Chlorinated hydrocarbons in sediments and fish from nearshore marine waters in Hawaii, 1977-1980. Between 1977 and 1980, the Hawaii Department of Health surveyed 16 sites in Hawaii and analyzed approximately 250 sediment samples. Akazawa (1981) reported these results, as well as those of analyses for pesticides and PCBs in fish and crab from selected sites. It is possible that some of these results were

redundant with those summarized in Hawaii Department of Health (1978), but this was not stated explicitly by Akazawa.

As in Hawaii Department of Health (1978), α- and γ-chlordane and *trans*-nonachlor were among those chlorinated organics targeted in Akazawa (1981). Tables 91 and 92 summarize results for sediments and biota, respectively.

Table 91. Ranges of sediment chlordane (α-, γ-chlordane, and *trans*-nonachlor) concentrations by site for nearshore locations in the State of Hawaii, 1977-1980. Values reported as ppb dry weight. Source: Akazawa (1981)

Location	Island	α-chlordane ppb dry wt.	γ-chlordane ppb dry wt.	*trans*-nonachlor ppb dry wt.
Ala Wai Canal	Oahu	15 - 370	28.54	15 - 303
Kaneohe Bay		<10 - 14.6	<10 - 24.5	<15
Kaiaka Bay		<10	<10	<15
Kapalama Canal		18 - 347	28 - 353	<10 - 264
Keehi Lagoon		11 - 119	21 - 141	22 - 31
Kahana Bay		<10	<10	<15
Pearl Harbor, W. Loch	Oahu	<10	<10	<15
Pearl Harbor, Middle Loch	Oahu	<10 - 23	21 - 53	<15
Pearl Harbor, East Loch	Oahu	<10	<10 - 23	<15
Nawiliwili Bay	Kauai	<10	<10	<15
Hanapepe Bay		<10	<10	<15
Kahului Bay		<10	<10	<15
Manele/Hulopoe		<10	<10	<15
Hilo Bay	Hawaii	<10	<10	<15
Waiakea Pond	Hawaii	21 - 40	25 - 46	21 - 41
Kaunakakai Harbor		<10	<10	<15

Table 92. Chlordane (α-, γ-chlordane, and *trans*-nonachlor) residues in fish and shellfish from nearshore locations in the State of Hawaii, 1977-1979. Values reported as ppb wet weight. Source: Akazawa (1981).

Species	Location	Tissue	N	α-chlordn ppb wet wt	γ-chlordn ppb wet wt	*t*-nonachlor ppb wet wt
Mullet						
(*Mugil cephalus*)	Pearl Harbor, Middle Loch	Flesh		20	ND	20
		Viscera		30	ND	60
	Waiakea Pond, Hilo	Flesh		37	21	55
		Viscera		230	130	360
Tilapia						
(*Tilapia sp.*)	Pearl Harbor, Middle Loch	Flesh		ND	ND	ND
		Viscera		ND	ND	ND
Awa awa						
(*Elops hawaiiensis*)	Pearl Harbor, Middle Loch	Flesh		ND	ND	ND
		Viscera		30	ND	50
Crab, Hawaiian						
(*Podophthalmus vigil*)	Keehi Lagoon	Flesh		ND	ND	ND
		Viscera		ND	ND	82
	Kaneohe Bay	Flesh		ND	ND	ND
		Viscera		ND	ND	22

Table 92. Continued

Species	Location	Tissue	N	α-chlordn ppb wet wt	γ-chlordn ppb wet wt	*t*-nonachlor ppb wet wt
Crab, white						
(*Portunus sanguino-*	Kaneohe Bay	Flesh		ND	ND	ND
lentus)		Viscera		ND	ND	18
	Hilo Bay	Flesh		ND	ND	ND
		Viscera		ND	ND	49
Crab, blue claw						
(*Thalamita crenata*)	Keehi Lagoon	Flesh		ND	ND	ND
		Viscera		ND	ND	25

In sediments, it was found that for the 16 sites sampled and over the period 1977-1980, the three chlordane compounds were detected more frequently than any other of the 19 chlorinated hydrocarbons targeted: γ-chlordane was detected in 28.3 percent of the samples, a-chlordane in 26.4 percent, and *trans*-nonachlor in 19.3 percent. This compared with 16.4 percent for PCBs and 12.3 percent for *p,p'*-DDE, which are usually among the most frequently occurring chlorinated hydrocarbons in environmental samples.

Most of the concentrations for chlordane compounds in crab and fish summarized in Akazawa (1982) were not detectable. Similar to results obtained in Hawaii Department of Health (1978), mullet sampled in Waiakea Pond, Hilo, contained the highest concentrations in both flesh and viscera (113 ppb wet weight and 720 ppb, respectively). Among the three species of crab sampled, *trans*-nonachlor was the only chlordane compound detected, and only in viscera. No concentrations in any edible tissues of either fish or crabs exceeded the FDA action level of 300 ppb wet weight.

Other Areas and Media

Obviously, marine environments in the United States are not the only media or areas potentially impacted by use of persistent compounds such as chlordane. With the widespread use of chlordane as a pesticide worldwide, other media (*e.g.*, fresh water, soils) and other nations could be expected to show residues of varying concentrations and impacts. A large body of data exists for those other media and nations. The following are summaries of some studies in the United States that targeted chlordane outside of the marine environment and investigations in other countries that have measured chlordane in various media. While generally not directly comparable to U.S. coastal and estuarine chlordane results, these other efforts may provide a larger perspective within which the former may be placed.

United States

National Pesticide Monitoring Program/National Contaminant Biomonitoring Program, 1967-. In addition to the estuarine and marine sampling efforts discussed previously in this review, the NPMP sampled organochlorine pesticide residues in freshwater fish from sites across the United States. This aspect of the program began in 1967 and has continued into the 1980s as the National Contaminant Biomonitoring Program (NCBP). A brief discussion of these studies may facilitate insights into broader temporal and spatial patterns of chlordane contamination across the United States, and could help to explain distributions in the estuarine and marine environments.

From 1967 to 1969, composited whole fish samples from 50 sites were analyzed by commerical laboratories for 11 organochlorine pesticides, including chlordane and heptachlor (compositions not specified). A total of 62 fish species were represented in 590 composite samples in the 1967-1968 collections, and 44 species in 147 composites in 1969. Results were summarized in Henderson, Johnson, and Inglis (1969), and Henderson, Inglis, and Johnson (1971). In 1970, station coverage for the NPMP was expanded to 100 locations, including the original 50 stations. However, although results for the 1970-1974 period were summarized in Schmitt, Ludke, and Walsh (1981), tissue concentrations for chlordane were not reported because of analytical and interpretive problems stemming from differential metabolism of chlordane compounds. Instead, samples were "screened" for technical chlordane, presumably meaning that they were analyzed only as to presence or absence of detectable chlordane constituents. In contrast, 1976-1979 NPMP

results discussed in Schmitt *et al.* (1983) reported chlordane concentrations in much greater detail than previously: five chlordane compounds, including *cis*-, *trans*-, and oxychlordane, *cis*- and *trans*-nonachlor, and heptachlor were individually targeted.

Because the sampling program ranged over many species, it is difficult to directly compare results; however, it may be instructive to note where elevated chlordane levels occurred in the freshwater fish collected and analyzed. Figures 67 and 68 illustrate maximum chlordane concentrations measured at a given location in the 1967-1968, and 1969 collections, respectively. These plots represent the highest chlordane concentrations measured, without regard to species. Figure 67 shows that for the 1967-1968 time period, relatively elevated concentrations were measured in the northeast and in the central sections of the United States. Chlordane was detected in fish from nearly every site along the eastern coast of the country. Figure 68 shows markedly different results for the 1969 collection, with chlordane at detectable concentrations in far fewer locations. In the 1967-1968 collection, detectable chlordane residues (with detection limits ranging from 0.001 to 0.010 ppm wet weight) were reported in 22 percent (128) of the 590 samples. Heptachlor was measured in 12 percent (70 samples). In the 1969 collection, chlordane was detected in only 11 percent (16) of the 147 composite samples and heptachlor in only 2 percent samples (3%) from the same location.

In the 1976 to 1979 period, chlordane replaced dieldrin as the most ubiquitous of the cyclodiene pesticides in the NPMP sampling effort. *Cis*-chlordane was detected at approximately 93 and 94 percent of the locations sampled in 1976-1977 and 1978-1979, respectively, while *trans*-nonachlor was detected at 70 and 93 percent. The other chlordane compounds measured were measured less frequently. Highest residues were found in fish from Hawaii (Manoa Stream, Honolulu), in the Corn Belt, and in the Great Lakes. Schmitt *et al.* (1983) analyzed station-year data for the 1976 to 1979 period, but found no consistent temporal trends for chlordane.

In the 1980s, the suite of chemicals analyzed under NPMP was expanded to include industrial chemicals and metals as well as pesticides. In recognition of this fact, the program was renamed the NCBP. Jacknow, Ludke, and Coon (1986) summarized broad trends from 20 years of monitoring, although results for chlordane compounds were a relatively recent addition to the list of target compounds. Jacknow, Ludke, and Coon commented that *cis*- and *trans*-chlordane were the most abundant and persistent of the chlordane compounds in 1980-1981, in addition to the preceding sampling cycles mentioned above. However, they also noted that *cis*-chlordane concentrations in fish declined 50 percent between 1976 and 1981.

Figure 69, adapted from Jacknow, Ludke, and Coon, shows distribution of results for chlordane residues in freshwater fish in 1980-1981. As with past sampling under the NPMP, the NCBP measured whole body residues and results were not directly comparable with human health standards established by the FDA. The ranges of results obtained, however, center around the FDA action level of 0.3 ppm and could warrant an examination from the perspective of residue levels in edible portions of the fish. Generally higher concentrations were measured in the midwestern and eastern portions of the country, while western states showed relatively lower residues. Fish from southwestern states contained uniformly low levels.

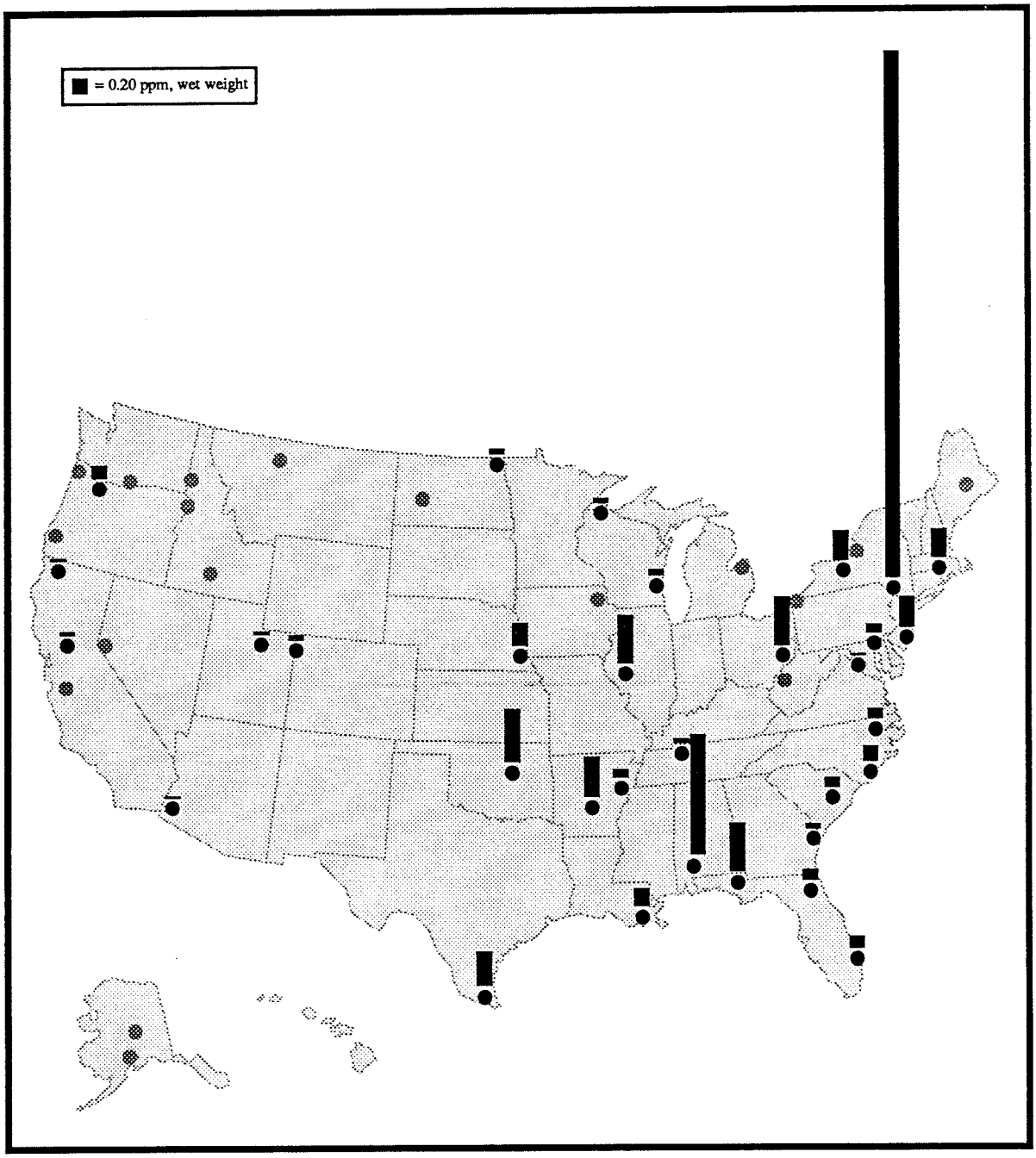

Figure 67. Maximum concentrations of chlordane measured in freshwater fish collected at sites around the United States, 1967-1968. Source: Henderson, Johnson, and Inglis (1969).

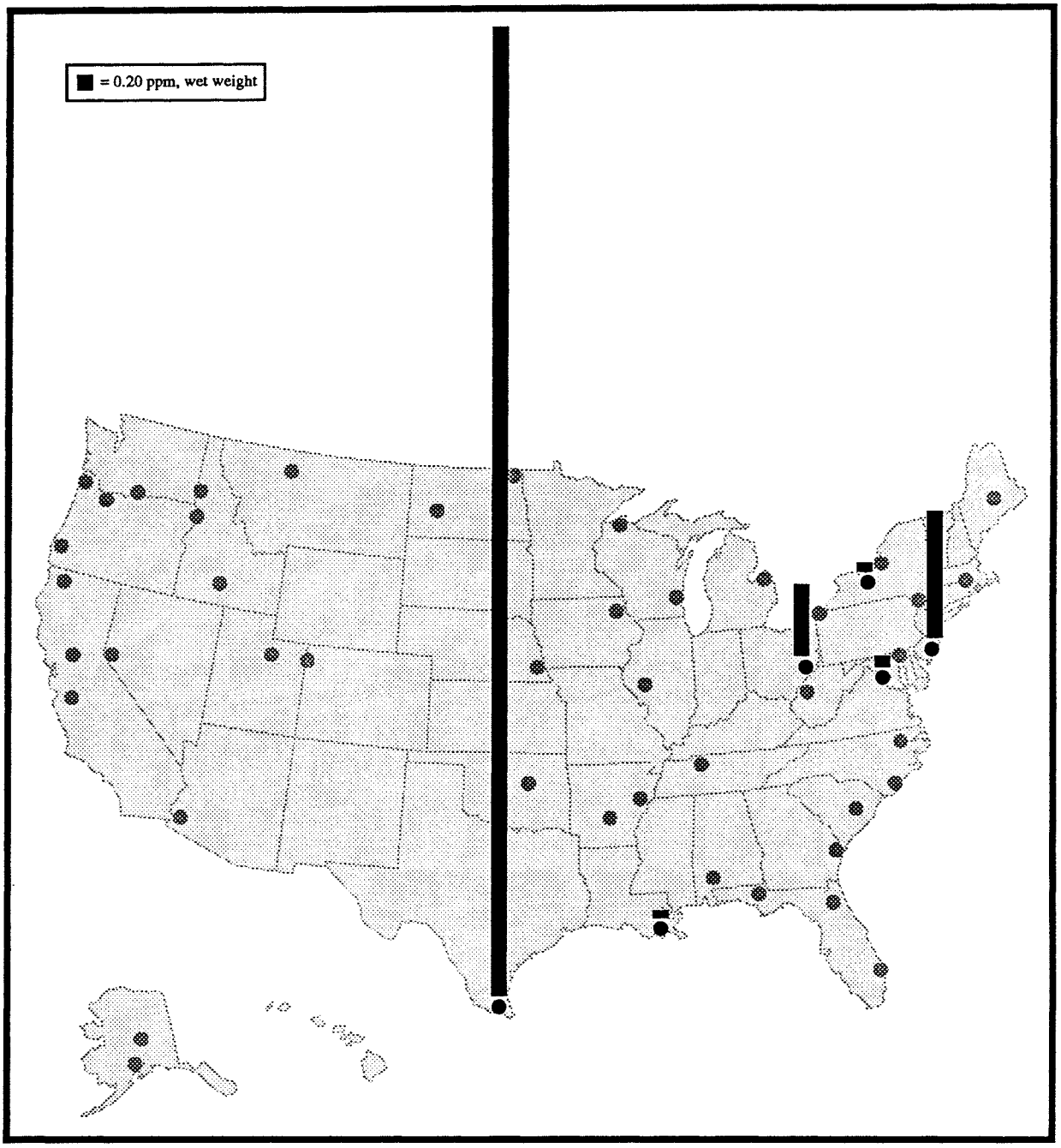

Figure 68. Maximum concentrations of chlordane measured in freshwater fish collected at sites around the United States, 1969. Source: Henderson, Inglis, and Johnson (1971).

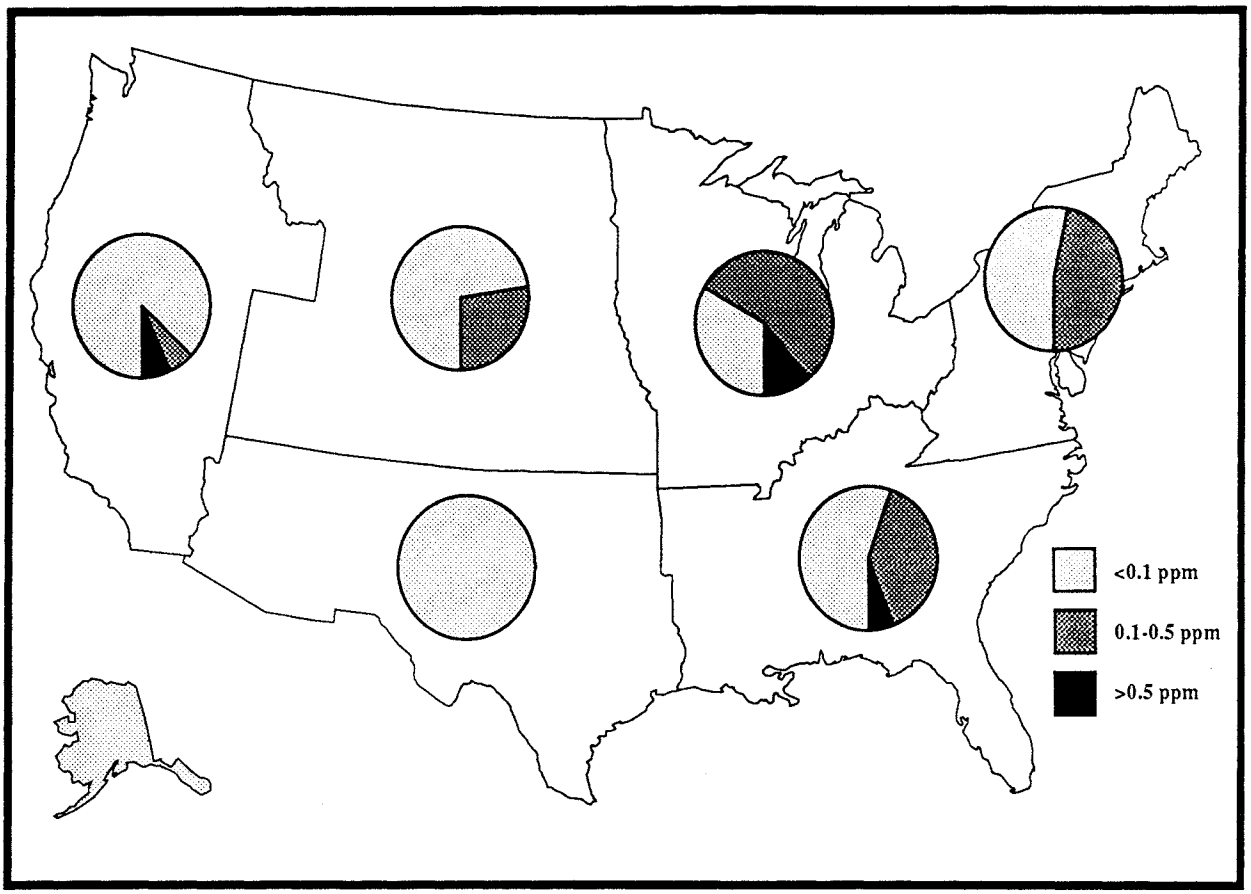

Figure 69. Distributions of chlordane concentrations measured in freshwater fish collected in different regions of the United States, 1980-1981. Source: Jacknow, Ludke, and Coon (1986).

Toxic substances in New York fish and wildlife. The Toxic Substances Monitoring Program of the NYDEC has been discussed elsewhere in this report in the context of results from estuarine and marine fish species. However, the program encompasses a much broader set of samples from the environment of New York state, with many collections targeted on specific identified and perceived environmental problems. Freshwater fish, birds, reptiles, and mammals have also been collected and analyzed for contaminant residues, and in many cases included analysis of chlordane compounds. While a detailed discussion of results is beyond the scope of this report, the reader is referred to publications from the New York DEC, such as NYDEC (1978), Sloan (1981a), Sloan (1981b), and Sloan (1987) for comprehensive summaries of results.

Most reported concentrations of chlordane compounds in freshwater fish collected between 1978 and 1980 were below detection (<0.01 ppm) or relatively low. For example, fish from the St. Lawrence River and the Valatie Kill drainage sampled in 1979 contained undetectable levels of heptachlor and its epoxide, and chlordane concentrations ranging between <0.01 to 0.04 ppm wet weight. The only elevated concentrations of chlordane were reported in a study of contaminant levels in fish from selected urban waterways. Creeks, ponds, and lakes near Buffalo, Rochester, Capital District, and New York City/Long Island were sampled. NYDEC (1978) reported chlordane levels below detection for most fish sampled in the Rochester and Buffalo areas (1 of 15 detectable), a few elevated concentrations in the Capital District (4 of 18 detectable, ranging to 1.25 ppm wet weight), and relatively high levels found on Long Island (three of four with detectable levels ranging from to 1.27 to 1.69 ppm). Results reported in Sloan (1981a) showed that all fish from the Buffalo and Rochester areas contained chlordane in edible tissues at concentrations below the detection limits. Similar to results from the earlier urban waterways collection, fish collected near New York City and Long Island contained consistently higher concentrations, ranging as high as 0.42

ppm in brown bullhead and 0.59 ppm in carp. Sloan (1981a) noted that chlordane had been used extensively on Long Island for the control of turf pests.

Because the New York City/Long Island area yielded relatively high concentrations of chlordane in edible portions of some fish sampled in the urban waterways project, several waters and species were included in a health advisory listing on restrictions for fish consumption. Additionally, a more extensive study focusing on chlordane was undertaken in 1982, 1984, and 1985. In 1982, relatively high concentrations were found at a few locations. For example, a composited sample of goldfish (*Carassius auratus*) from Spring Lake contained 6.75 ppm chlordane (an order of magnitude greater than the FDA action level). Largemouth bass (*Micropterus salmoides*) and goldfish from Massapequa Reservoir yielded mean concentrations of 0.55 and 1.60 ppm, respectively. In 1984, the maximum concentration found was in goldfish, 2.90 ppm at Smith Pond, Rockville Centre. Levels in six species at Massapequa Reservoir ranged from 0.01 to 1.00 ppm. In 1985, chlordane levels were much lower than in previous collections, with a maximum measured concentration of 0.33 ppm in brown bullhead (*Ictalurus nebulosus*) from East Lake. Resident species were not sampled in Massapequa Reservoir, but brown trout (*Salmo trutta*) were held in that body of water for 49 to 66 days. Chlordane was not significantly accumulated in these fish over hatchery controls, and the maximum measured concentration was 0.12 ppm. These results suggested that chlordane levels in freshwater fish on Long Island steadily declined over the collection period.

Sloan (1981b) reported analytical results from various wildlife species, and among these were concentrations measured in several tissues of bald eagle (*Haliaetus leucocephalus*), and in brain tissue of a great horned owl (*Bubo virginianus*). These are shown below in Table 93.

Table 93. Concentrations of chlordane compounds measured in tissues of a bald eagle and a great horned owl from New York State. Values reported as ppm wet weight. Source: Sloan (1981b).

Tissue	Oxy-chlordane	γ–chlordane	α–chlordane	Compound "C"	β–nonachlor	*trans*-nonachlor
BALD EAGLE						
Subcutaneous fat	0.69	0.32	-	-	1.88	0.25
Thoracoabdominal fat	0.61	0.31	-	-	1.78	0.16
Brain	<0.01	0.01	0.02	-	10.01	<0.01
Muscle	0.08	0.11	0.16	0.02	0.41	0.18
Kidney	-	-	-	-	-	-
Liver	<0.01	-	0.02	-	0.02	0.01
GREAT HORNED OWL						
Brain	3.14			0.05	0.11	0.21

The concentrations found in the owl brain tissue were unusually high, but Sloan (1981b) commented that the overwhelming contaminant finding was the presence of PCBs at a concentration of 1600 ppm, a concentration high enough to have been the cause of death. The bald eagle results were derived from an individual electrocuted in high voltage power lines.

Risebrough (1987) mentioned previously in the context of contamination found in fish and other biota of the Southern California Bight also discussed possible reproductive impacts of organochlorine residues found in bald eagles sampled on offshore islands of that region. He indicated that environmental levels found there were still probably causing failure in the recently introduced populations. Unfortunately, Risebrough examined blood and eggs of eagles, and not the same tissues analyzed by Sloan.

Statistical analysis of fish tissue toxics data for the District of Columbia, 1988. In late 1988, the Department of Consumer and Regulatory Affairs of the District of Columbia sponsored the collection and subsequent analysis of fish muscle tissue from the Anacostia and Potomac Rivers. The study was implemented by the Interstate Commission on the Potomac River Basin. Compounds targeted in this study were two PCB formulations (Aroclor 1242 and 1260), and chlordane (composition not specified, but likely technical chlordane). The freshwater fish species channel catfish (*Ictalurus punctatus*), largemouth bass (*Micropterus salmoides*), pumpkinseed sunfish (*Lepomis gibbosus*), redbreast sunfish (*Lepomis auritus*), and bluegill (*Lepomis macrochirus*). Results were summarized and analyzed in Sommerfield and Cummins (1989).

Of the 93 fish of the five species examined, 16 (17%) were found to contain quantifiable levels of chlordane in fillets. All five catfish collected from a single site on the Anacostia River showed fillet chlordane concentrations in excess of the FDA action level of 0.3 ppm wet weight. Concentrations in these five ranged from 0.6 ppm, to 1.00 ppm. Four other fish from this site also contained measurable levels of chlordane, but concentrations were an order of magnitude lower than those found in catfish. Fillets from three fish collected at other locations exceeded the FDA action level, all of which were channel catfish.

These results suggested that feeding habits and lipid content of fish play significant roles in determining muscle tissue burdens of chlordane. That is, channel catfish, which are characterized as bottom feeders with higher lipid content, contained highest levels of chlordane. This general observation held true at those sites where other species were collected.

Based on the results of this study, in July of 1989, the District of Columbia Commissioner of Public Health issued an advisory (District of Columbia Department of Consumer and Regulatory Affairs, 1989) on consumption of channel catfish and other fish caught in the Potomac and Anacostia Rivers. Specifically, it was recommended that individuals not consume more than one meal of fish (channel catfish, carp, or American eel) per week, that only boneless, skinless fillets with fatty portions removed be eaten, and that women of childbearing age, nursing women, and preschool children should be discouraged from eating any of the fish.

Concurrent with these collections in the District of Columbia, the Maryland Department of the Environment (MDE) also surveyed PCB, organochlorine pesticide, and metals levels in four resident fish species of the Potomac River: channel catfish, American eel, brown bullhead, and white perch. PCB and chlordane results for channel catfish were expedited to provide a worst case assessment. However, in contrast to the situation in the District of Columbia, MDE found that its results did not warrant a health advisory (D. Murphy, MDEC, pers. comm., 8 June 1990).

Georgia Toxic Streams Monitoring Program, Trends Analysis Program, 1977-. The Georgia Department of Natural Resources (DNR) has monitored levels of contaminants in the aquatic environment of the State since 1977. Data obtained from the Environmental Protection Division of DNR include results from a program of discharge monitoring authorized under section 205J of the Clean Water Act, and results from a longer term monitoring project to assess contaminant trends at a series of core sites. Included in these data are measurements of chlordane (defined as technical chlordane and metabolites) concentrations in water, sediments, and fish tissue; results for the period 1977-1988 were obtained from the Georgia DNR for inclusion in this review.

No estuarine or marine fish species were sampled. Specimens were composited to give samples representative of broad ecosystem groupings, such as bottomfeeders, predators, or minnows. No species distinctions were made during the sample compositing process (S. Sneed, Georgia DNR, pers. comm. 17 July 1989; D. Kamps, Georgia DNR, pers. comm., 24 August 1989).

Table 94 summarizes chlordane results held in the Georgia DNR database for the period 1977 to 1988. Chlordane was not quantified in any of the 48 water samples analyzed. In sediment samples, chlordane was measured in 18.4 percent of the samples, and in fish, 16.8 percent. Of the 143 analyses of composited whole fish tissues, 18 (12.6%) were greater than or equal to the FDA action level for edible fish tissue. It should be remembered, however, that DNR analyzed whole fish, not simply fillets (see next discussion of Chattahoochee River investigations). Moreover, the Georgia program was not designed as a random, or representative, sampling effort. Emphasis in site selection, for the Toxic Streams Monitoring Program in particular, was placed upon known dischargers. It could therefore be expected that these results reflected more impacted areas of state waters.

Table 94. Summary for aquatic chlordane data collected for Georgia Toxic Streams Monitoring Program, 1977-1988. Source: Data supplied by Georgia DNR (1989b).

Matrix	N	Number Quantifiable	Units, wet/dry	Chlordane, Range of Values
Water	48	0	µg/l total	<0.050 - <0.60
Sediment	348	64	µg/kg dry	<5.0 - 891
Whole fish	143	24	mg/kg wet	<0.050 - 3.6

Georgia Study of Toxins in Fish from the Chattahoochee River, 1989. Because of concerns about levels of contaminants, including chlordane, in fish in the Chattahoochee River near Atlanta, the Georgia Department of Natural Resources sponsored the collection and analysis of common freshwater species in and around that body of water. Eighteen fish were collected in May and June 1989, from two locations on the Chattahoochee. One site, near Morgan Falls Dam, was upstream from the city of Atlanta. The other, at Georgia Highway 92, was downstream from the Atlanta metropolitan area and its major sewage treatment plants. At the Morgan Falls site, largemouth bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*), and common carp (*Cyprinus carpio*) were sampled. At Highway 92, largemouth bass, channel catfish (*Ictalurus punctatus*), and common carp were collected. Edible portions (*i.e.*, fillets) from individual fish were analyzed. Analytical results are shown below as Table 95.

Table 95. Concentrations of chlordane measured in fillets of fish sampled in the Chattahoochee River near Atlanta, 1989. Source: Georgia DNR (1989a).

Location	Species	Weight, lbs.	Chlordane ppm, wet wt.
Morgan Falls Dam (above Atlanta)	Largemouth bass	2.0	<0.045
		1.8	<0.045
		1.8	<0.045
	Bluegill	0.4	<0.045
		0.5	0.073
		0.5	<0.045
	Common carp	4.6	0.080
		3.5	0.074
		3.8	0.140
Highway 92 (below Atlanta)	Largemouth bass	1.9	<0.045
		1.4	<0.045
		1.1	<0.045
	Channel catfish	0.6	0.24
		0.7	0.27
		0.6	<0.045
	Common carp	4.5	0.45
		4.4	0.72
		4.3	0.91

Although the sample size was limited, the elevated concentrations encountered in fish downstream from the city of Atlanta support the characterization of chlordane as a contaminant most recently of urban origin. Common carp at the Highway 92 site showed chlordane concentrations nearly two orders of magnitude greater than individuals of the same species collected upstream from Atlanta. Moreover, all three Highway 92 individuals analyzed exceeded the FDA action level concentration of 0.3 ppm wet weight. Two of three channel catfish at the site approached the action level. Based on these results, the Georgia DNR advised the public to use its own discretion in deciding whether or not to consume carp from the downstream location.

Georgia comparison of fillet and whole fish analyses for PCBs, chlordane, and DDE, 1989. Among the many difficulties that arise in the interpretation of fish contaminant residue results are differences in tissues used for analysis. In order to more clearly define those differences in specimens collected for the preceding study of Chattahoochee River fish, the Georgia DNR conducted a direct comparison of fillet and whole fish concentrations of selected organochlorine compounds, including chlordane. Data resulting from this effort were acquired for this report from the Department of Natural Resources. Those results are shown below in Table 96.

Table 96. Concentrations of chlordane measured in fillets and whole body sample composites of fish sampled in the Chattahoochee River near Atlanta, 1989. Source: Georgia DNR (1989b).

Location	Species	Sample Date	Composite No.	Fillet, ppm, wet wt.	Whole body ppm, wet wt.
At U.S. Highway 92	Largemouth bass	5/2/89	3	<0.045	<0.2
	Largemouth bass	5/2/89	2	0.066	0.57
	Channel catfish	5/2/89	3	<0.045	0.49
	Common carp	5/2/89	5	0.33	1.1
Whitesburg @ Hwy 16	Largemouth bass	5/3/89	3	<0.045	0.34
	Channel catfish	5/3/89	4	0.15	0.36
	Common carp	5/3/89	5	0.17	1.15
Franklin @ Hwy 27	Largemouth bass	5/3/89	5	0.045	<0.1
	Channel catfish	5/3/89	7	<0.045	0.18
	Common carp	5/3/89	5	<0.045	0.21
LaGrange Water Intake	Largemouth bass	5/3/89	5	<0.045	1.03
	Channel catfish	5/3/89	5	<0.045	0.43
	Common carp	5/3/89	5	<0.045	0.066

Although again the sample size was relatively small, it is apparent from the results that fillet concentrations in all species analyzed were substantially less than whole body levels. While 11 of 13 (85%) of whole body measurements yielded quantitated concentrations of chlordane, only 3 (23%) of the fillet concentrations were quantifiable. Of the three measurable fillet concentrations, one composite of common carp at 0.33 ppm exceeded the FDA action level of 0.3 ppm for chlordane in edible tissues. In comparison, eight whole body composites (62%) exceeded the action level.

Mississippi River Fish Contamination Study, 1982. In 1980-1981, as part of a statewide ambient monitoring program, the Tennessee Department of Public Health collected and analyzed fish from the Memphis vicinity: the Mississippi River at the I-40 bridge and at McKellar Lake (an industrial port area feeding into the river). Fish collected at both of these locations showed concentrations of chlordane (composition not specified) in excess of the FDA action levels. As a result, in March of 1982, the Tennessee Commissioner of Public Health issued a health advisory warning against consumption of fish from Shelby County. In addition, the Tennessee Wildlife Resources Commission banned commercial fishing in the area.

These findings and the ensuing events led to the formation of an interstate and interagency group to plan and implement a joint study of Mississippi River contamination. The group included representatives from the city of Memphis, the states of Tennessee, Arkansas, and Mississippi, the U.S. EPA, and the FDA. The cooperative plan developed by the committee called for the collection of fish from 12 stations along over 300 miles of the Mississippi River between northwest Tennessee and northwest Mississippi. Samples were collected in August and September of 1982. Details of the program, including methods and analytical results, were reported in Collins, Robinson, and Sinclair (1989).

Fish sampled included blue catfish (*Ictalurus furcatus*), channel catfish (*Ictalurus punctatus*), flat bullhead (*Ictalurus platycephalus*), gar (*Lepisoteus sp.*), smallmouth buffalo (*Ictiobus bubalus*), carp (*Cyprinus carpio*), river carpsucker (*Carpiodes carpio*), bigmouth buffalo (*Ictiobus cyprinellus*), and black buffalo (*Ictiobus niger*). Six composite samples were collected at each station, with each composite consisting of skinless, boneless fillets. A number of organic compounds and metals were targeted for analysis in the study, and among these were four chlordane compounds: α- and γ-chlordane, and *cis*- and

trans-nonachlor. The Memphis-Shelby County Health Department Laboratory analyzed all samples collected, while the Tennessee and Mississippi labs were responsible for analysis of samples collected by their respective states as well as for about 10 percent of samples collected for quality control. The Analytical Support Branch of EPA Region IV analyzed samples from selected stations and also for quality control purposes. Table 97 illustrates results reported by the four laboratories in Collins, Robinson, and Sinclair (1989).

Table 97. Levels of chlordane (Σ α- and γ–chlordane, and *cis*- and *trans*-nonachlor) in fish collected in the Mississippi River between northwest Tennessee and northwest Mississippi. Concentrations in µg/kg (ppb), wet weight. Source: Collins, Robinson, and Sinclair (1989).

	LABORATORY REPORTING RESULTS				
Location	**Memphis**	**Tennessee**	**Mississippi**	**EPA**	**Ave.**
Tiptonville, Tennessee	37.3	8	–	–	23
River Mile 875	6.5	17	–	–	12
	0.1	24	–	–	12
	111.6	16	–	–	64
	50.3	11	–	–	31
	12.9	17	–	–	15
Blythville, Arkansas	123.2	8	103	70	76
River Mile 820	0.1	59	110	100	67
	71.1	–	–	–	71
	21.1	–	–	200	111
	98.3	–	–	–	98
	0.1	–	–	20	10
Osceola, Arkansas	126.7	–	–	200	163
River Mile 785	135.5	–	–	80	108
	63.3	–	–	100	82
	87.1	–	–	60	74
	0.1	–	–	200	100
	41.4	–	–	60	51
Tipton Co. Line, Tennessee	65.7	59	–	70	65
River Mile 754	16.9	11	–	70	33
	127	65	–	100	97
	126.8	70	–	NA*	98
	53.3	33	–	200	95
	157.1	46	–	200	134
I-40 Bridge, Tennessee	55.6	129	–	300	162
River Mile 735	–	131	–	200	166
	334	250	–	300	295
	451.3	263	–	700	471
	72.3	51	–	80	68
	451.3	706	–	400	519
McKellar Lake, Tennessee	73.1	101	–	100	91
River Mile 725	58.6	78	–	100	79
	116.8	82	<160	100	115
	112.8	85	–	100	99
	146	130	–	200	159
	81.1	56	–	100	79
Head of McKellar, Tennessee	209.4	123	–	–	166
	257.6	74	–	–	166
	135.5	228	–	–	182
	404	102	–	–	253
	361	123	–	–	242
	310	132	–	–	221

Table 97 (continued)

Location	Memphis	Tennessee	Mississippi	EPA	Ave.
Tennessee-Mississippi Line	651	–	244	300	398
River Mile 712	84.8	–	80	80	82
	37.51	–	NA*	20	29
	242.7	–	240	300	261
	65.3	–	189	200	151
	2.8	–	42	70	38
Helena, Arkansas	82.9	–	–	–	83
River Mile 661	164.2	11	73	40	72
	104.9	–	–	–	105
	135.8	–	–	–	136
	94.2	–	–	–	94
	116.6	29	57	70	68
Rosedale, Mississippi	111.1	–	34	–	73
River Mile 588	99.5	–	56	–	78
	323.5	45	125	200	173
	46.1	–	68	–	57
	17.6	–	108	–	63
	66.8	–	43	–	55
Arkansas City, Arkansas	55.6	–	–	–	56
River Mile 554	139.5	–	–	–	140
	9.6	–	–	–	10
	122.2	–	–	–	122
	39.5	–	–	–	40
	38.6	–	–	–	39
Greeneville, Mississippi	366.8	23	68	70	132
River Mile 536	99.9	–	57	–	79
	4.0	–	48	–	26
	11.5	–	NA*	–	12
	1.7	–	28	–	15

*Not available? The designation "NA" was not defined in the reference.

Table 97 shows a large amount of variability, both among samples collected at the same site and analyzed by the same laboratory (with concentrations in some cases ranging over 2 orders of magnitude), as well as among the different laboratories analyzing the same sample (concentrations in some cases showing order of magnitude differences). This variability suggests the difficulties inherent in interpreting limited environmental sampling results; nevertheless, it is clear that at least some tissue samples exceeded the FDA action level of 300 ppb. Collins, Robinson, and Sinclair commented that previous studies had determined relatively high concentrations of cyclodiene compounds in the Memphis area, with the U.S. Fish and Wildlife Service reporting some of the highest concentrations in the nation for whole fish in that vicinity. Several references cited by Collins, Robinson, and Sinclair identified the Velsicol Corporation Memphis facility as the source for cyclodiene contamination in the area. However, the authors also suggest other sources for contamination of the Mississippi River near Memphis, including Hollywood Dump and Cypress Creek, the Memphis North (Sewage) Treatment Plant, generalized urban runoff, and upstream sources.

The controversy over the fishing ban along this portion of the Mississippi River in Tennessee was discussed by Collins, Robinson, and Sinclair. Arkansas, which shares the Mississippi with Tennessee, did not join in the ban on commercial fishing. In 1985, collections made by the Arkansas Game and Fish Commission and the FDA yielded chlordane concentrations an order of magnitude less than Tennessee results obtained from the Arkansas side of the river in 1984. Although it was not possible to directly compare results, it was noted that the Arkansas/FDA results were obtained using an FDA methodology

(Pesticide Analytical Manual), while Tennessee used an EPA method (ultrasonication and gel permeation chromatography).

The state of Tennessee has apparently continued in its efforts to convince the state of Arkansas to extend the fishing ban, but to little avail. Collins, Robinson, and Sinclair, writing for the Tennessee Department of Health and Environment, stated that the commercial fishing bans near Memphis will stay in place until chlordane levels drop below FDA action levels as measured by the method used by Tennessee.

Organochlorine chemical residues in bluegills and common carp from the irrigated San Joaquin Valley floor, California, 1981. In this study, samples of bluegill (*Lepomis macrochirus*) and common carp (*Cyprinus carpio*) collected from the San Joaquin River and two tributaries (Merced River, Salt Slough) in California were analyzed for 21 organochlorine residues to determine if both irrigated and nonirrigated sites exhibited chemical contamination. Four constituents of technical chlordane were among the organochlorines analyzed: *cis*- and *trans*-chlordane, *trans*-nonachlor, and heptachlor. Results were reported in Saiki and Schmitt (1986) and are summarized below as Table 98.

Table 98. Results for chlordane analyses in bluegill and carp (whole body) collected in the San Joaquin Valley, 1981. Source: Saiki and Schmitt (1986).

	Sites[1]							
	SJR-1	SJR-2	SJR-3	SJR-4	SJR-5	SS	MR-1	MR-2
Bluegill								
Total length (mm)	154.9	111.5	121.2	87.8	103.2	107.8	100.2	143.6
Weight (g)	79.7	28.9	38.1	11.9	20.4	27.6	18.3	60.4
Moisture (%)	75.1	75.3	75.1	77.3	76.7	76.0	78.3	75.3
Lipid (%)	3.4	2.3	2.5	1.8	1.9	3.2	1.8	2.9
Chlordane[2] (mg/kg)								
Wet wt.	0	0.021	0	0.014	0.002	0.006	0	<0.001
Lipid wt.	0	0.912	0	0.759	0.011	0.188	0	0.043
Common carp								
Total length (mm)	494.8	237.5	472.6	260.7	475.7	455.9	449.0	491.9
Weight (g)	1510.0	211.3	1329.5	259.5	1572.6	1303.3	1240.5	1580.0
Moisture (%)	74.1	76.1	74.6	77.7	71.6	72.9	70.7	71.5
Lipid (%)	6.4	3.9	3.7	2.6	7.7	6.6	8.7	7.7
Chlordane[2] (mg/kg)								
Wet wt.	0.015	0.012	0.082	0.030	0.273	0.073	0.035	0.096
Lipid wt.	0.088	0.090	2.542	1.212	3.578	1.107	0.403	1.297

[1] SJR-1 through SJR-5 = San Joaquin River, SS = Salt Slough, MR-1 and MR-2 = Merced River.
[2] Sum of *cis*-chlordane and *trans*-nonachlor.

Highest wet weight concentrations of chlordane (sum of *cis*-chlordane and *trans*-nonachlor) were found by Saiki and Schmitt along downstream portions of the San Joaquin River system, while the lowest were measured at the uppermost sections that were least likely to be influenced by irrigation return flows. Concentrations ranged from below detection (<0.004 mg/kg) in bluegill from three sites, to 0.273 mg/kg in carp from the most downstream site in the San Joaquin River. However, none of the mean wet weight concentrations in either species exceeded the FDA action level of 0.300 mg/kg. It is possible that individual specimens did so, but Saiki and Schmitt reported whole body concentrations, as opposed to those in edible tissues. Interestingly, concentrations of chlordanes at a site were not significantly correlated to lipid content of either species of fish.

Although measurable concentrations of chlordane compounds were fairly consistently determined by Saiki and Schmitt, the fact that extremely elevated levels were not found, especially in carp, is notable. The San Joaquin Valley is and has been an area of intensive irrigated agricultural activities for many years, and large quantities of agricultural chemicals have been employed there. That carp in the lower San Joaquin River did not carry highly elevated body burdens of chlordane, while carp in urban-suburban water bodies of New York and Georgia have been found to contain much heavier burdens, serves to emphasize that chlordane residues are manifested most typically as contaminants of urban centers, as opposed to agricultural regions.

Canada

Evidence of atmospheric transport and deposition of organochlorine pesticides and PCB in Canadian Arctic snow, 1986. Gregor and Gummer (1989) reported concentrations of organochlorine pesticides and PCBs measured in samples of snow collected at twelve remote sites in the Northwest Territories of Canada during 1986. Among chlordane constituents, the *cis-* and *trans-* isomers were selected for analysis. Results from Gregor and Gummer (1989) are shown below in Table 99.

Table 99. Concentrations of chlordane compounds measured in Canadian Arctic snow, 1986. Values in ng/l (ppt). Source: Gregor and Gummer (1989).

Collection Site Name	*cis*-chlordane ng/l	*trans*-chlordane ng/l
Hayes River, laboratory replicate A	0.16	0.16
Hayes River, laboratory replicate B	0.15	0.18
Brown River	0.23	0.22
Ferguson River, laboratory replicate A	0.06	0.07
Ferguson River, laboratory replicate B	0.05	0.08
Lorillard River, laboratory replicate A	0.09	0.09
Lorillard River, laboratory replicate B	0.07	0.12
Baker Lake, laboratory replicate A	0.02	nd
Baker Lake, laboratory replicate B	nd	nd
Resolute Bay, laboratory replicate A	0.39	0.34
Resolute Bay, laboratory replicate B	0.17	0.19
Agassiz-1, laboratory replicate A	0.37	0.37
Agassiz-1, laboratory replicate B	0.40	0.48
Agassiz-2, laboratory replicate A	0.31	0.31
Agassiz-2, laboratory replicate B	0.28	0.37
Alexandra Fjord	0.08	0.09
Ice Island	0.11	0.05
Gascoyne	0.15	0.15
Devon	0.04	0.03
Pedder	0.10	0.06
Bag blanks	nd	nd
Theoretical detection limit	0.13	0.13

According to Gregor and Gummer, these results confirmed that toxic organic compounds, such as chlordane constituents, occur in widespread portions of the Canadian Arctic. Furthermore, such occurrences were not explained by proximity to identifiable sources, and the only reasonable explanation for the presence of measurable concentrations would have been long-range atmospheric transport. Although the relative importance of this source to contamination of indigenous biota was not determined, it was thought by the authors to be significant because rapid annual snow melts across very large geographic areas would represent large inputs.

Nonachlor and chlordane in aquatic fauna of the eastern Canadian coast, 1971-1977. Zitko (1978) performed GC-MS analyses for several organochlorine compounds, including *cis-* and *trans-* nonachlor, and chlordane (sum of *cis-* and *trans-* isomers), in tissues and oils of several marine organisms and birds reared in the laboratory and collected along the east coast of Canada. Results of the analyses are summarized below in Table 100.

Table 100. Concentrations of nonachlor and chlordane in tissues and oils of various aquatic fauna. Values in μg/g (ppm) lipid weight basis. Source: Zitko (1978).

Organism	Matrix	Collection year	Comments	*trans*-nonachlor	*cis*-nonachlor	Chlordane (*cis* + *trans*)
Lobster (*Homarus americanus*)	Hepatopancreas	1977	Laboratory reared	0.440	<0.01	0.100
Lobster (*Homarus americanus*)	Hepatopancreas	1977	Commercially caught	0.383	0.032	0.078
Cod (*Gadus morhua*)	Liver	1977	Freshly caught	0.057	<0.01	<0.01
Cod (*Gadus morhua*)	Liver	1977	Held in lab several months	1.88	0.074	-
White shark (*Carchardon carcharias*)	Liver	1971		8.52	1.68	2.60
Herring (*Clupea harengus*)	Commercial oil	1971	-	0.136	0.018	0.143
Herring (*Clupea harengus*)	Commercial oil	1977	-	0.170	0.025	0.114
Herring (*Clupea harengus*)	Commercial oil	1977	-	0.096	<0.01	0.039
Redfish (*Sebastes marinus*)	Commercial oil	1977	-	0.187	0.030	0.179
Atlantic salmon (*Salmo salar*)	Eggs	1976	-	0.210	0.063	0.151
Herring gull (*Larus argentatus*)	Eggs	1973	-	0.521	0.024	0.226
Double-crested comorant (*Phalacrocorax auritus*)	Eggs	1973	-	0.344	0.415	0.373
Double-crested comorant (*Phalacrocorax auritus*)	Eggs	1975	-	0.454	0.620	0.396

It should be noted that the concentrations expressed in Zitko (1978) were normalized to lipid content, and unfortunately, corresponding lipid content values were not given. This makes general comparisons to other results and FDA action levels difficult. In addition, Zitko commented that in at least one sample, the liver of cod (*Gadus morhua*) held in the laboratory for several months before analysis, the relatively low lipid content was at least partially responsible for comparatively high concentration of *trans*-nonachlor.

Environmental contaminants in Canadian seabirds, 1968-1985. Noble and Elliott (1986) reviewed available literature to assess levels of environmental contaminants in Canadian marine birds, to determine if adverse impacts could be discerned, and to evaluate the reliability of seabirds as indicators of broader environmental health. This extensive review effort summarized available Canadian Wildlife Service data for concentrations of several organochlorine and metal contaminants in tissues and eggs of 24 seabird species collected from the Atlantic, Pacific, and arctic coasts. Oxychlordane, *cis*-chlordane, and nonachlor were among those compounds discussed.

Noble and Elliott found that concentrations in tissues and eggs varied considerably with species and with coast. On the Pacific coast, measured concentrations of chlordane compounds were relatively low. Highest concentrations of total chlordane, most of it oxychlordane, were found in eggs of fork-tailed storm petrels sampled in the Queen Charlotte Islands. Leach's storm petrels sampled concurrently contained much lower levels, and Noble and Elliott attributed this finding to dietary differences, *i.e.*, the inclusion of offal in the diet of the former and the exclusively planktivorous diet of the latter.

Chlordane compounds were found to be more prevalent in birds sampled in arctic regions than along the Pacific coast. In that region, ivory gulls contained chlordane compounds (mostly oxychlordane) at concentrations up to 0.09 ppm, while northern fulmars had levels up to 0.26 ppm. The authors speculated that these high levels (relative to Pacific coast birds) resulted from the scavenging behavior of the species, with long-range transport of volatilized organics introducing residues into the far northern environment.

Atlantic coast results were highly variable, ranging from below detection in eiders, to 0.50 ppm in northern gannets and razorbills. The high concentration in northern gannets declined to 0.25 ppm in 1984. Noble and Elliott felt that the high concentrations were attributable to contamination encountered at Gulf coast (for razorbills, the New England coast) wintering grounds, contamination of prey species, or inability to metabolize the chlordane compounds. *Cis*-chlordane, which had been found by previous researchers to be quickly metabolized, was determined to be a major constituent in gannet eggs and in razorbills. This suggested recent exposure to the parent chlordane mixture.

Noble and Elliott felt that the stability of chlordane compounds in seabirds warranted further investigation of possible biological effects of exposure.

Organochlorine pollutants in harp seals (*Phoca groenlandica*), 1976-1978. Ronald *et al.* (1984) reported results of a study that sampled 248 harp seals (*Phoca groenlandica*) from five locations in the northwest Atlantic and arctic during the period between 1976 and 1978. Blood, kidney, brain, muscle, and blubber samples were analyzed for a suite of organochlorine compounds, including the three chlordane compounds, α-, β-, and oxychlordane. Table 101 illustrates chlordane results for summed chlordane concentration in blubber, by sex and by age class; Table 102 shows results for other tissues, grouped by age class, sex, and area of collection.

Table 101. Mean summed chlordane (Σ α-, β-, and oxychlordane) in blubber tissue from harp seals sampled in the northwest Atlantic and arctic. Values in mg/kg (ppm), wet weight basis. Source: Ronald *et al.* (1984)

Location/Sex	Age	N	Mean ±sdev Σ Chlordane, ppm wet wt.
Gulf of St. Lawrence			
Male	Pups	29	0.08±0.05
Female	Pups	18	0.09±0.06
Male	Juveniles	12	0.20±0.11
Female	Juveniles	12	0.23±0.11
Male	Adults	10	0.20±0.09
Female	Adults	56	0.11±0.14
Front ice off Newfoundland-Labrador			
Male	Pups	5	0.09±0.08
Female	Pups	2	0.09±0.04
Male	Juveniles	22	0.09±0.06
Female	Juveniles	14	0.12±0.07
Male	Adults	4	0.18±0.14
Female	Adults	1	0.08
Northern (Grise Fjord, Pangnirtung, northwest Greenland)			
Male	Pups	2	0.18±0.04
Female	Pups	4	0.05±0.04
Male	Juveniles	11	0.24±0.13
Female	Juveniles	10	0.24±0.11
Male	Adults	9	0.19±0.15
Female	Adults	6	0.17±0.12

Table 102. Mean summed chlordane (Σ α-, β-, and oxychlordane) in various tissues from harp seals sampled in the northwest Atlantic and arctic. Values in mg/kg (ppm), wet weight basis; number of animals sampled in parentheses. Source: Ronald *et al.* (1984)

	Mean Summed Chlordane ±sdev					
Age/sex/area	**Blubber**	**Blood**	**Brain**	**Kidney**	**Liver**	**Muscle**
Age						
Newborn pups	0.08±0.05 (60)	0.00±0.0 (4)	0.00±0 (22)	0.00±0.0 (27)	0.00±0.0 (34)	0.00±0.01 (36)
Juveniles	0.17±0.11 (81)	0.00±0.0 (31)	0.00±0.0 (21)	0.00±0.0 (28)	0.00±0.0 (27)	0.00±0.0 (28)
Adults	0.13±0.14 (86)	0.00±0.0 (6)	0.00±0.0 (23)	0.00±0.0 (37)	0.00±0.01 (48)	0.00±0.01 (47)
Sex						
Males	0.14±0.11 (104)	0.00±0.0 (26)	0.00±0.0 (31)	0.00±0.0 (42)	0.00±0.01 (50)	0.00±0.01 (52)
Females	0.13±0.12 (123)	0.00±0.0 (15)	0.00±0.0 (35)	0.00±0.0 (50)	0.00±0.0 (59)	0.00±0.0 (59)
Area						
Gulf	0.12±0.12 (137)		0.00±0.0 (35)	0.00±0.0 (51)	0.00±0.01 (70)	0.00±0.01 (69)
Front	0.10±0.07 (48)	0.00±0.0 (40)	0.00±0.0 (30)	0.00±0.0 (40)	0.00±0.0 (39)	0.00±0.01 (40)
Grise Fjord	0.23±0.13 (20)					
Pangnirtung	0.19±0.12 (16)					
NW Greenland	0.09±0.06 (6)	0.00 (1)	0.01 (1)	0.01 (1)		0.02±0.01 (2)

Results in the tables above illustrate a substantial amount of variability in chlordane concentrations measured in subpopulations of harp seals. The most consistent aspect of the data was the nearly complete absence of detectable concentrations of chlordane in all tissues analyzed except for blubber.

No significant correlations were found between chlordane levels and age in any of the harp seals sampled. Helle *et al.* (1983) determined a similar lack of correlation between chlordanes and age in blubber of Saimaa ringed seals. Ronald *et al.* also found that chlordane concentrations correlated with concentrations of other organochlorines to the smallest degree of any of the compounds.

Chlorinated hydrocarbons in cetaceans, 1971-1975. Taruski, Olney, and Winn (1975) reported results of analyses for chlorinated organic compounds in blubber of whales and dolphins from the coasts of the United States, Canada, and the West Indies. Among the analytes targeted was α-chlordane. The study included 18 samples of 10 different species of odontocetes and mysticetes. Eleven samples were taken from stranded animals, one from a whale fishery, two from animals that died in captivity, and four from biopsy darts fired into free-swimming animals. Table 103 summarizes results for concentrations of α-chlordane measured in blubber samples.

Table 103. Concentrations of α-chlordane residues in blubber of cetaceans. Values in ppm wet weight. Source: Taruski, Olney, and Winn (1975).

Common Name	Species Name	Sex, Maturity	Collection Location	α-chlordane
Humpback whale	*Megaptera novaeangliae*	Female, pregnant	Nova Scotia	ND*
Humpback whale	*Megaptera novaeangliae*	Male, juvenile	New Jersey	0.2
Humpback whale	*Megaptera novaeangliae*	Male, juvenile	Antigua	ND
Humpback whale	*Megaptera novaeangliae*	Male, mature	Saint Kitts	0.1
Sperm whale	*Physeter catodon*	Female, mature	Anegada Passage	ND*
Sperm whale	*Physeter catodon*	Male, mature	Anegada Passage	ND*
Sperm whale	*Physeter catodon*	Female, neonate	Massachusetts	0.3
Dense-beaked whale	*Mesoplodon densirostris*	Male, mature	South Carolina	0.1
Dense-beaked whale	*Mesoplodon densirostris*	Male, juvenile	New Jersey	0.3
Atlantic pilot whale	*Globicephala melaena*	Female, old	Rhode Island	1.4
Atlantic pilot whale	*Globicephala melaena*	Male, juvenile	Maine	0.6
Atlantic white-sided dolphin	*Lagenorhynchus acutus*	Male, mature	Nova Scotia	ND*
Saddleback dolphin	*Delphinus delphis*	Female, mature	Rhode Island	1.2
Striped dolphin	*Stenella caeruleoalba*	Female, suckling	Maryland	2.7
Striped dolphin	*Stenella caeruleoalba*	Male, mature	Rhode Island	1.4
Harbor porpoise	*Phocoena phocoena*	Female, mature	Rhode Island	2.6
Pacific white-sided dolphin	*Lagenorhynchus obliquidens*	Female, mature	California (held in New York)	5.0
Pacific pilot whale	*Globicephala scammoni*	Female, juvenile	California (captive)	ND*

*Detection limits not specified.

Because a diversity of species were sampled from a number of locations, it would be difficult to define trends in the data. In general, however, it may be observed that residue levels in both toothed and baleen whales were much lower than in dolphins. Taruski, Olney, and Winn attributed most of the variability in the results to differences in the degrees of contamination at the locations frequented by the whales and dolphins, but differences in diet (*e.g.*, selective ingestion of higher lipid content fish) were likely to have influenced subsequent blubber concentrations. Interestingly, the animal whose blubber contained the highest level of α-chlordane (5.0 ppm) was a captive dolphin held for over 3 years that had been fed primarily mackerel and herring shipped from eastern Canada.

The authors acknowledged the difficulty in evaluating the significance of measured organochlorine levels in cetaceans, noting both the lack of information on direct effects as well as more basic information on reproductive processes and population biology of the species involved.

Organochlorine contaminants in a Canadian arctic ecosystem. Muir, Norstrom, and Simon (1988) analyzed muscle tissue of arctic cod (*Boreogadus saida*), blubber of ringed seals (*Phoca hispida*), and adipose tissue of polar bears (*Ursus maritimus*) for concentrations of chlorinated organic compounds, including a series of chlordane constituents. Twelve chlordane-related compounds, including seven not previously reported in seal blubber, were observed by GC-MS. Muir, Norstrom, and Simon also analyzed samples of ringed seal blubber that had been collected in 1972, and 1975-76, in order to generally evaluate temporal trends. They found that concentrations of summed chlordane-related compounds in the three sample groups were similar (although sample sizes for the earlier years were small), suggesting that contamination of the arctic environment by chlordane was not a new phenomenon. Figure 70 shows the results for chlordane compounds in ringed seal blubber.

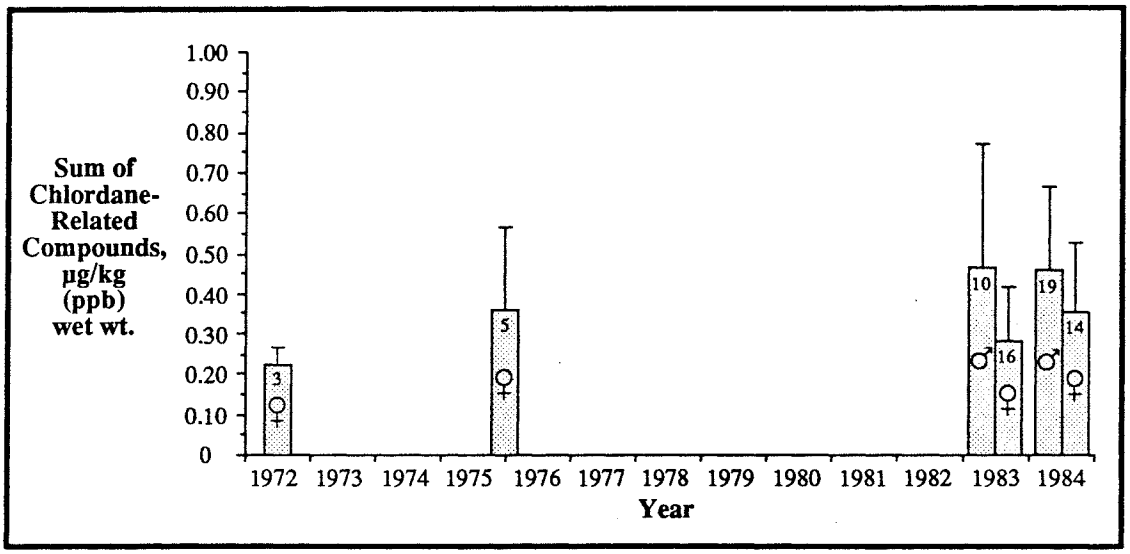

Figure 70. Concentrations of chlordane-related compounds measured in blubber of ringed seals (*Phoca hispida*) sampled in the Baffin Bay region of the Canadian Arctic, 1972-1984. Sample number and sex indicated in histogram. Source: Muir, Norstrom, and Simon (1988).

Muir, Norstrom, and Simon measured chlordane-related compounds at low levels in arctic cod muscle tissue. Results for fish sampled in Admiralty Inlet and Barrow Strait, near Baffin Bay, showed summed chlordane compound concentrations of 0.002 µg/kg (ppb) wet weight and 0.003 µg/kg, respectively. The authors noted that on a lipid weight basis (lipid content averaged 1.5% for arctic cod), these concentrations were two to five times lower than measured levels in fish from the north Atlantic, Baltic, and northwest Pacific; but were higher than fish from the Antarctic region.

Polychlorinated biphenyls in harbor porpoises from the Bay of Fundy, Canada, and adjacent waters, with some information on chlordane and hexachlorobenzene levels, 1975-1977. Gaskin, Frank, and Holdrinet (1983) measured PCBs in tissues of harbor porpoises (*Phocoena phocoena*) collected between 1971 and 1977 in the Bay of Fundy, off the southeastern coast of Canada. In addition to these analyses, they also examined chlordanes (*cis*-, *trans*-, and oxychlordane) in blubber, liver, and muscle tissue in specimens collected in the 1975-1977 period. These results for chlordane compounds are summarized below in Table 104.

Table 104. Concentrations of chlordanes measured in harbor porpoises sampled in the Bay of Fundy, Canada, 1975-1977. Values in µg/g (ppm) wet weight basis. Source: Gaskin, Frank, and Holdrinet (1983).

		Blubber		Liver		Muscle	
Sex	Age	*cis*-chlordane	*oxy*-chlordane	*cis*-chlordane	*oxy*-nonachlor	*cis*-nonachlor	*oxy*-chlordane
Male	1	7.48	2.58	0.03	ND[a]	0.12	ND
Male	5	2.55	1.35	0.03	ND	0.11	ND
Male	6	6.73	4.78	0.16	ND	0.06	ND
Male	8	8.56	5.30	0.06	ND	0.07	ND
Female	1	6.78	3.35	0.02	ND	0.02	ND
Female	2	2.92	1.58	0.03	ND	0.06	ND

[a] Not detected; detection limits not specified.

A total of 18 tissue samples were analyzed. Levels of *trans*-chlordane were too low to be quantified in all 18. As can be noted from results shown in Table 103, relatively high concentrations of both *cis*- and oxychlordane were measured in all blubber samples, consistent with the demonstrated lipophilic nature of the chlordane compounds. The paucity of data, however, precluded determination of consistent trends across sex and age.

Gaskin, Frank, and Holdrinet noted that although the levels of chlordane compounds in blubber were significant, they were nowhere near the levels of PCBs and DDTs measured in the same species sampled in the region (mean PCB concentrations in blubber reported by Gaskin, Frank, and Holdrinet ranged from 22.5 to 55.6 ppm).

Toxaphene and other organochlorines in the Arctic Ocean environment, 1986 and 1987. In a study that targeted toxaphene and other organochlorines, Bidleman *et al.* (in press) determined residue levels in air, snow, seawater, zooplankton, and benthic amphipods collected from an ice island in the Canadian Arctic. Among the organochlorines analyzed were the chlordane compounds *cis*- and *trans*-chlordane, and *cis*- and *trans*-nonachlor. The nearly simultaneous sampling in all five substrates provided evidence of a link between atmospheric inputs of contaminants and polar biota.

Concentrations of chlordane compounds measured in the Canadian Arctic substrates are summarized in Table 105.

Table 105. Concentrations of chlordane and nonachlor isomers in abiotic and biotic substrates sampled in 1986 and 1987. in the Canadian Arctic. Values as specified. Source: Bidleman *et al.* (in press).

Substrate	Date	Units	Σ Chlordane[1]	Σ Nonachlor[2]
Air[3]	8/86	pg/m^3	3.9	1.9
Air[3]	6/87	pg/m^3	6.3	4.4
Snow (Devon Island)	5/87	pg/l	181	81
Snow (Resolute Bay)	5/87	pg/l	69	55
Seawater (10 m)	8/86	pg/l	4	<0.7
Seawater (270 m)	6/87	pg/l	3	0.6
Zooplankton[4], dry weight	8/86	ng/g	3.1	1.6
Zooplankton[4], lipid weight	8/86	ng/g	6.3	3.3
Zooplankton[4], dry weight	6/87	ng/g	7	5.3
Zooplankton[4], lipid weight	6/87	ng/g	29	22
Amphipods[5], dry weight	8/86	ng/g	45	220
Amphipods[5], lipid weight	8/86	ng/g	150	750
Amphipods[5], dry weight	6/87	ng/g	130	390
Amphipods[5], lipid weight	6/87	ng/g	620	1900

[1]Sum of *cis*- and *trans*-chlordane.
[2]Sum of *cis*- and *trans*-nonachlor.
[3]Mean of 7 samples from 1986 and 6 samples in 1987.
[4]*Calanus hyperboreus, Metridia langa,* and *Xanthocalanus borealis.*
[5]*Anoxy sarsi* collected at 190 m in 8/86; *Tmetonyx cicada* at 310 m in 6/87.

Bidleman *et al.* noted that between November and May, the arctic region is typically enshrouded with a pervasive haze, determined by researchers to be composed of sulfate, ammonium, soot, trace elements, other anthropogenic compounds, and crustal and sea salt elements. They also pointed out that other studies had demonstrated the promotion of high molecular weight vapor adsorption to atmospheric particulate matter at low temperatures. The mechanism of adsorption of organic compounds like chlordane to particulate matter subsequently deposited by dry deposition and snow was postulated as a potential input route into the arctic marine environment. Direct deposition from the atmosphere, as well as precipitation into the water, were also listed as possibilities. Transport mechanisms to deeper waters may have included sinking particles, as well as the movements and fecal material of zooplankton.

Results reported by Bidleman *et al.* demonstrated the ubiquitous nature of chlordane contamination in the Canadian Arctic. Although concentrations were low in such substrates as air, snow, and water, the more elevated levels found in amphipods were taken as an indication that such compounds as the organochlorines may reach the bottom and be incorporated into the arctic food web at higher concentrations. According to Bidleman *et al.*, the greater than 20 percent lipid content of the amphipods was typical of Arctic Ocean fauna than must endure long periods of starvation, and because of this, it was noted that potential existed for bioaccumulation of organochlorine compounds like the chlordanes.

Organochlorine contaminants in polar bears. The known propensity for chlorinated organic compounds to volatilize into the atmosphere provides a ready mechanism for global transport and distribution of those materials. Oloffs, Albright, and Szeto (1972) demonstrated that chlordanes, DDTs, and PCBs moved readily from water to the atmosphere if contact with sediments did not occur. The universal occurrence of organochlorine compounds of anthropogenic origin in remote ecosystems previously considered to be pristine is therefore not unexpected, and has been well-documented. For example, Norstrom *et al.* (1988) analyzed liver and adipose tissues of polar bears (*Ursus maritimus*) in the Canadian arctic and subarctic, and found that technical chlordane compounds and metabolites were the second most abundant group of organochlorines, behind PCB congeners. Eight individual technical chlordane constituents and metabolites were identified by Norstrom *et al.* in polar bear tissues, and the relative prominence of this group was an unexpected finding of the study.

Central and South America

Pesticides and PCBs in oysters from Mazatlán, Sinaloa, Mexico, 1988. In early 1988, Martin and Gutierrez-Galindo (1989) collected oysters (*Crassostrea corteziensis*) from the Estero de Urías/Estero de Sirena tropical estuarine system adjacent to the city of Mazatlán in Mexico. These oyster samples were analyzed for a suite of 46 synthetic organic chemicals, including the chlordane compounds *cis*- and *trans*-chlordene, heptachlor, *cis*- and *trans*-nonachlor, *cis*- and *trans*-chlordane, and oxychlordane. Of 110 samples collected at two sites in the estuarine system, none of the chlordane compounds were detected, with minimum reporting concentrations for all compounds at 1 ng/g (ppb), dry weight. Martin and Gutierrez-Galindo noted that concentrations of both pesticides and PCBs were considerably lower in Mazatlán Harbor than along most of the California coast.

Chlorinated pesticide residues in waters of the Broa Reservoir (Brazil) and its tributaries. Cáceres, Tundisi, and Castellan (1980) analyzed chlorinated pesticide residues in the Broa Reservoir, and four of its main tributary systems, near São Carlos, in the state of São Paulo, Brazil (located in the southeastern portion of the country). Chlordane (not identified as to targeted constituents), and heptachlor were among the chlorinated organic pesticides analyzed in the study. Table 106 shows results for those two compounds reported by Cáceres, Tundisi, and Castellan.

Table 106. Concentrations of chlordane and heptachlor measured in waters of the Broa Reservoir and tributaries, Brazil. Sample dates not specified. Values in µg/l (ppb). Source: Cáceres, Tundisi, and Castellan (1980).

	Rivers				Reservoir	
Compound	Perdizes	Itaqueri	Lobo	Geraldo	Centro	Barragem
Chlordane	0.020	0.077	0.012	0.010	0.050	0.060
Heptachlor	0.080	0.064	0.033	0.045	0.432	0.680

Concentrations measured in the reservoir and its tributary rivers were higher than values obtained in U.S. waters. For example, Bevenue *et al.* (1972) reported concentrations of chlordane in Hawaiian waters ranging from 0.003 to 0.009 ppb. Although Fuhrer and Rinella (1983), and Fuhrer (1984) reported none of twenty samples taken in estuarine and coastal waters off Washington and Oregon contained chlordane or

heptachlor above detection limits, those reported limits exceeded the quantified values found by Cáceres, Tundisi, and Castellan.

Concentrations of heptachlor in the reservoir itself were surprisingly elevated. While heptachlor levels measured in the tributary rivers were about equal to chlordane concentrations, heptachlor at the two reservoir sites was found at levels about an order of magnitude greater than chlordane.

Brazilian drinking water standards for maximum permissible concentrations of chlordane and heptachlor in potable water were reported by Cáceres, Tundisi, and Castellan as 3.0 µg/l and 0.10 µg/l, respectively. Table 105 shows that measured heptachlor concentrations in the reservoir were well above the standard, while chlordane concentrations were well below. The results suggest a much heavier use of heptachlor in this particular area than in others. For example, in the U.S., heptachlor concentrations, especially in water samples, have generally not been found at quantifiable levels.

Europe

Pesticide and PCB levels in the eggs of shag *Phalacrocorax aristotelis* and cormorant *P. carbo* from Ireland,1982. Wilson and Earley (1986) investigated levels of pesticides and PCBs in the eggs of two species of coastal seabirds (shag, *Phalacrocorax aristotelis*, and cormorant, *Phalacrocorax carbo*) sampled at three sites along the coast of Ireland. Although 24 compounds were targeted by Wilson and Earley, only 13 pesticides and Aroclor 1254, were detected at quantifiable levels. Four constituents of technical chlordane (α-, γ-, and oxychlordane, and heptachlor) were included; of these, three were detected (α- and oxychlordane, and heptachlor). Results of analyses for chlordane compounds are summarized in Table 107.

Table 107. Concentration of chlordane compounds measured in eggs of shag and cormorant collected on the Irish coast, 1982. Values in ng/g (ppb). Geometric means ± one standard error. Source: Wilson and Earley (1986).

Location/species	α-chlordane	oxy-chlordane	heptachlor
West Cork			
Cormorant	0.05 (0.00-0.10)[1]	2.01 1.26-3.02)	1.16 (0.66-1.82)
Shag	0.79 (0.64-0.96)	7.47 (6.55-8.51)	nd[2]
Saltees			
Cormorant	0.10 (0.00-0.20)	2.56 (1.81-3.52)	1.11 (0.74-1.55)
Shag	0.53 (0.41-0.67)	3.97 (3.29-4.75)	1.06 (0.65-1.57)
Skerries			
Shag	0.43 (0.18-0.73)	8.02 (5.27-11.97)	0.89 (0.46-1.45)

[1] Zero values included by adding 1.0 prior to taking log of each value, then subsequently subtracting 1.0 from final means and upper and lower limits obtained.
[2] Not detected.

Wilson and Earley noted that a considerable amount of variability was evident even within individual pesticides measured at given sites. However, they did find significant ($p = 0.05$) interspecies differences between shag and cormorant for levels of α-chlordane and oxychlordane. Correlations between shell thickness and organochlorine residues were also calculated by Wilson and Earley, but concentrations of chlordane compounds did not show significant positive or negative correlations with the shell thickness parameter.

Based on these results and those obtained from bird censuses and other research efforts, Wilson and Earley indicated that levels found in the Irish seabird eggs were the lowest found in Europe, were among the lowest reported anywhere for any seabird, and as such could be taken as baseline concentrations. Populations of the two species appeared to be increasing, and there were no other indications of lethal or sublethal effects associated with egg burdens of the organochlorines. It was, therefore, concluded that the Irish coast was not suffering from undue environmental contamination by organochlorine compounds.

Application of the Mussel Watch concept in studies of hydrocarbon distribution in the coastal zone of the Ebro Delta (Spain), 1980. In a collaborative effort between American and Spanish researchers,

Risebrough *et al.* (1983) applied the concept of the use of bivalves as environmental indicator organisms to study human-induced changes in the Ebro Delta, along the northeastern coast of Spain. Mussels (*Mytilus galloprovincialis*), oysters (*Ostrea edulis*), and clams (*Venus gallinae*) were collected in summer of 1980 and analyzed for their hydrocarbon content. Among the compounds identified were α- and γ-chlordane. Quantified results were reported for γ-chlordane only, although a representative chromatogram was reproduced in Risebrough *et al.* (1983) and some discussion of relative amounts of the α- and γ- isomers was included. Figure 71 illustrates the location of the Ebro Delta within Spain, as well as the location of sites sampled by Risebrough *et al.* Results for γ-chlordane measured in bivalves of the Ebro Delta region are shown in Table 108.

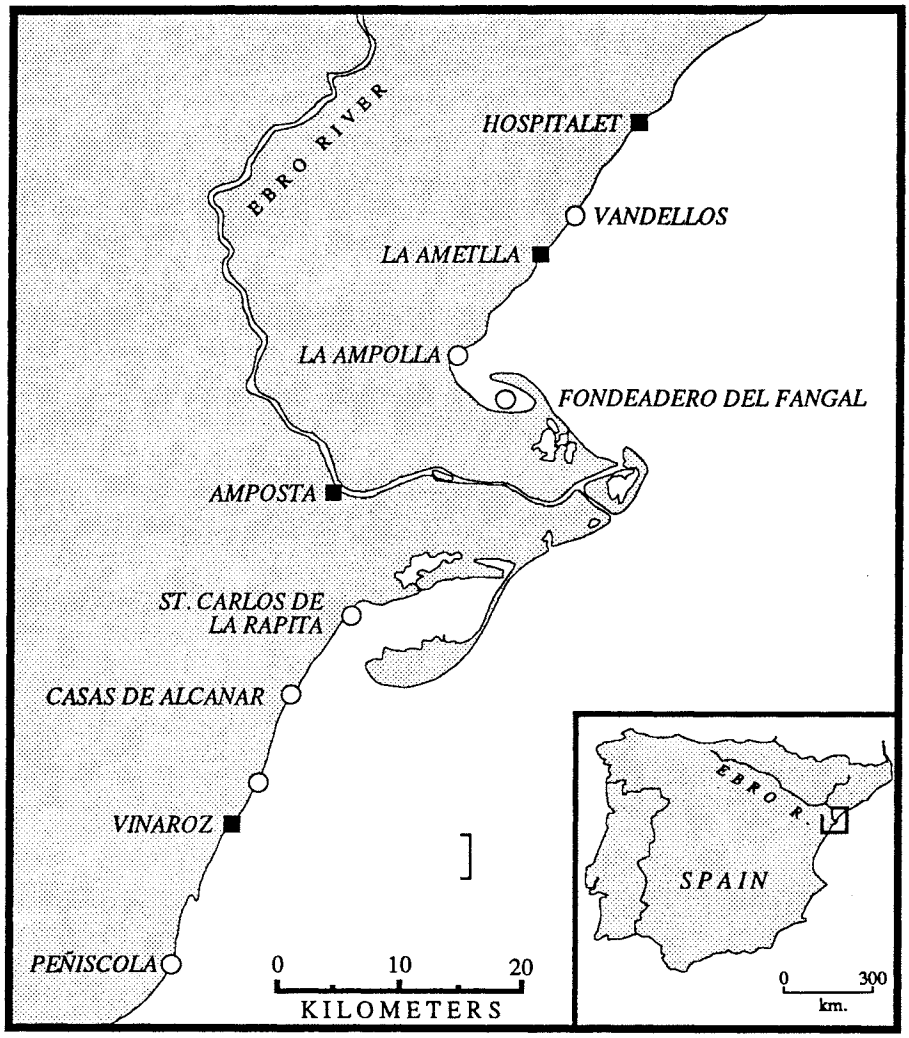

Figure 71. Map showing sites sampled by Risebrough *et al.* along the Mediterranean coast of Spain, 1980. Source: Risebrough *et al.* (1983).

The data in Table 108, especially those for mussels, suggest that the sites located to the north of the mouth of the Ebro River were less contaminated by γ-chlordane than were those at the delta or to the south. Other contaminants (endrin, hexachlorobenzene) in bivalves showed decreasing gradients both to the north and south of the river mouth, which the authors interpreted as an indication that the river was a primary source to that portion of the Spanish coast. Results for γ-chlordane did not illustrate such a clear geographic trend, since concentrations in mussels from sites south of the river were generally equivalent to those at the river.

Table 108. Concentration of γ-chlordane measured in bivalves collected in the Ebro Delta region of Spain, 1980. Values in ng/g, dry weight basis. Source: Risebrough *et al.* (1983).

Site	Species	γ-chlordane
Vandellos	*Mytilus galloprovincialis*	<1.0
Ampolla	*Mytilus galloprovincialis*	<1.0
Fondeadero del Fangal	*Mytilus galloprovincialis*	18
	Venus gallinae	3.1
	Ostrea edulis	1.0
San Carlos de la Rapita	*Mytilus galloprovincialis*	9.1
Casa de Alcanar	*Mytilus galloprovincialis*	13
North of Vinaroz	*Mytilus galloprovincialis*	13
Peñiscola	*Mytilus galloprovincialis*	21

An electron capture chromatogram of the aromatic fraction of mussel extract analyzed for the study was reproduced in Risebrough *et al.* (1983), and this indicated (with confirmation by GC/MS) that the γ- isomer of chlordane was present in much greater concentrations than was the α- isomer. It was noted that in mussels collected along the coast of California, the two isomers were found in approximately equivalent amounts. While the significance of this disparity was not discussed, it could indicate a species difference in metabolism between the mussel collected in Spain and that sampled in California. Metabolic differences among more distantly related species were suggested by results at the Fondeadero del Fangal site for mussel, clam, and oyster: the range of values for γ-chlordane spanned an order of magnitude.

Several of the authors also participated in the California Mussel Watch Program, and results from the Spanish study were directly compared to those from California effort. If the γ-chlordane concentrations found in *Mytilus galloprovincialis* near the Ebro River are presumed to be comparable to those measured in *M. edulis* and *M. californianus* along the coast of California, then the results from the Mediterranean coast suggest a relatively low level of contamination by chlordane compounds. The maximum concentrations of γ-chlordane measured in the CMW Program approached 500 ppb dry weight, at least an order of magnitude greater than the highest values measured around the Ebro Delta.

Organochlorine and metal pollution in aquatic organisms sampled in the Doñana National Park (Spain), 1983-1986. Rico *et al.* (1987) collected a number of freshwater organisms in the Doñana National Park, located on the southwestern coast of Spain, between 1983 and 1986. Four sites along the Brazo de la Torre ("Arm of the Tower"), a tributary to the Guadalquivir River, which in turn flows into the Atlantic Ocean, were sampled, and American crayfish (*Procambarus clarkii*), carp (*Cyprinus carpio*), barbel (*Barbus barbus*), grey mullet (*Mugil capito*), eel (*Anguilla anguilla*), and frog (*Rana perezi*) were analyzed for concentrations of organic compounds and metals in muscle tissues.

Chlordane was apparently not a target compound. Although heptachlor was among the pesticides targeted by Rico *et al.*, it was not measured above the limit of detection (0.01 ppm wet weight) in any of the 39 samples. In contrast, DDT and PCBs were found in all samples.

Chlordane components in the North Sea, 1977-1982. Kerkhoff and de Boer (1982) studied unidentified organochlorine compounds in blubber of harbor seals (*Phoca vitulina*) sampled in The Netherlands, and combined capillary gas chromatographic mass spectrometric analysis determined them to be chlordane constituents. This was the first observation of chlordane constituents in the Dutch marine environment. In a subsequent paper, Kerkhoff, Otte, and de Boer (1982) extended the scope of the previous analysis, and measured residues of chlordane components (identified as including *trans*-nonachlor, *cis*-, *trans*-, and oxychlordane) in harbor seals, white-beaked dolphins (*Lagenorhynchus albirostris*), great cormorant (*Phalacrocorax carbo*), and fish (cod, *Gadus morhua*; hake, *Merluccius merluccius*; sprat, *Sprattus sprattus*; and mackerel, *Scomber scombrus*) collected in the North Sea and Atlantic Ocean between 1977 and 1982. Results reported in this study are summarized as Table 109.

Table 109. Concentration of chlordane compounds in biota of the North Sea collected between 1977 and 1984. Values in µg/g (ppm) lipid weight basis. Source: Kerkhoff, Otte, and de Boer (1982).

Organism/Tissue	Year	*cis*-chlordane	*trans*-chlordane	*trans*-nonachlor	oxy-chlordane	Σ chlordanes
Seal - North Sea coast						
-blubber	1978	ND[1]	ND	2.7	3.0	5.7
-liver	1978	ND	ND	1.2	1.1	2.3
-brain	1978	ND	ND	0.54	0.40	0.94
Seal - Wadden Sea						
-blubber	1982	ND	ND	0.31	0.29	0.60
Seal - North Sea coast						
-blubber	1982	ND	ND	0.25	0.26	0.51
Dolphin - North Sea						
-blubber	1977	2.4	0.64	4.6	1.8	9.4
Dolphin - North Sea						
-blubber	1977	3.0	0.83	5.2	2.3	11.3
Dolphin - North Sea						
blubber	1977	1.4	0.22	2.3	1.5	5.4
Cormorant - Rhine/Meuse Delta						
-liver	1981	int[2]	int	int	7.3	
-kidney	1981	int	int	int	7.4	
Cod - northern North Sea						
-liver	1981	0.13	0.04	0.18	0.07	0.42
Cod - central North Sea						
-liver	1981	0.19	0.05	0.22	0.08	0.54
Cod - southern North Sea						
-liver	1980	0.07	0.02	0.10	0.08	0.27
Cod - southern North Sea						
-liver	1981	0.05	0.02	0.09	0.06	0.22
Hake - Atlantic Ocean						
-liver	1981	0.09	0.03	0.08	0.02	0.22
Herring - Atlantic Ocean						
-muscle	1981	0.03	0.02	0.04	na[3]	0.09
Herring - North Sea						
-muscle	1981	0.05	0.03	0.04	na	0.12
Sprat - Atlantic Ocean						
-muscle	1981	0.03	0.02	0.04	na	0.09
Sprat - North Sea						
-muscle	1981	0.06	0.03	0.05	na	0.14
Mackerel - Atlantic Ocean						
-muscle	1981	0.04	0.02	0.04	na	0.10
Mackerel - North Sea						
-muscle	1981	0.03	0.02	0.05	na	0.10

[1]Not detected
[2]Interference by PCBs
[3]Not analyzed

Fatty tissues of marine mammals were, not surprisingly, tissue matrices containing highest concentrations of summed chlordane constituents. However, there was a high degree of variability between the two species, as well as within the same species. Whether this reflected aspects of a temporal trend--1982 measurements in seal blubber were considerably lower than in 1977--or was indicative of other factors influencing body burdens, is not clear from the limited number of samples analyzed.

As a rule, *trans*-nonachlor was the predominant chlordane compound measured by Kerkhoff, Otte, and de Boer. In seals, *trans*-nonachlor was followed by oxychlordane; in dolphins and fish tissue, it was followed by *cis*-chlordane. Neither *cis*- nor *trans*-chlordane were detected in seal tissue. Percent occurrence of the various constituents is graphically portrayed in Figure 72.

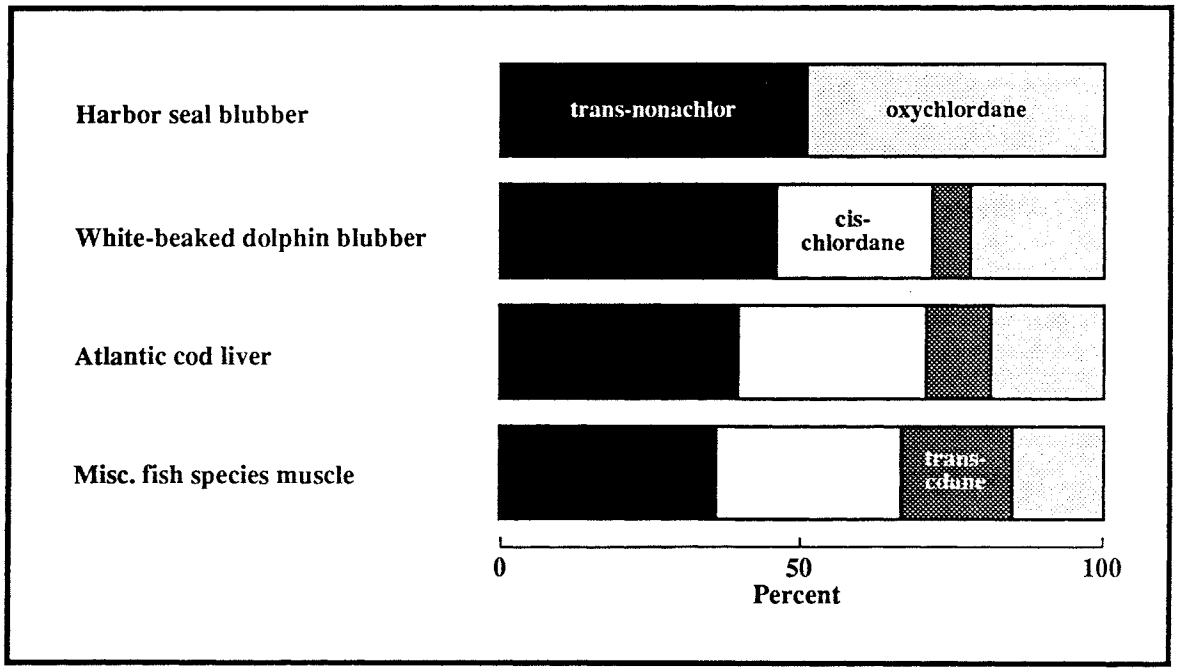

Figure 72. Percent occurrence of chlordane compounds in biota of the North Sea and Atlantic Ocean. Source: Kerkhoff, Otte, and de Boer (1982).

Kerkhoff, Otte, and de Boer pointed out that chlordane use in the Netherlands has been rare. While they acknowledge atmospheric transport as a plausible source to the North Sea environment, they also suggest inputs from the Baltic Sea, and insecticidal applications of the compounds in Eastern Bloc nations as possible additional contamination sources.

Time trends of organochlorine residues in sedentary fish species from a Norwegian fjord, 1972-1982. In a study that focused on temporal trends in DDT- and PCB-related compounds, Skåre *et al.* (1985) also provided limited data for concentrations of chlordane compounds (oxychlordane and *trans*-nonachlor) in liver tissue of several fish species collected in a Norwegian fjord in 1982. The fish chosen for analysis were cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), sea scorpion (*Myoxocephalus scorpius*), ballan wrasse (*Labrus berggylta*), catfish (*Anarhichas lupus*), lemon sole (*Microstomus kitt*), and flounder (*Platichthys flesus*). Results for the chlordane compounds are shown in Table 110.

Table 110. Concentrations of summed chlordane compounds (oxychlordane + *trans*-nonachlor) measured in liver tissue from fish sampled in a Norwegian fjord, 1982. Values in µg/kg (ppb) wet weight basis. Arithmetic means ± standard deviation. Source: Skåre *et al.* (1985).

Species	N	Mean Weight (g)	Mean % Lipid Content	Range	Mean Σ chlordane	Range[a]
Cod	18	636±367	30.9±12.4	6.7-56.8	91±51	38-220
Haddock	7	383±114	33.3±14.3	-	36±18	18-58
Sea scorpion	6	230±63	6.5±6.0	1.0-17.5	15±8	6-28
Catfish	2	2525±1237	4.5±0.9	3.8-5.1	9±8	3-14
Ballan wrasse	10	499±141	2.7±2.0	1.2-7.7	10±7	nd-18
Lemon sole	10	228±112	2.1±0.9	1.3-4.2	7±6	nd-14
Flounder	10	286±129	3.1±1.6	1.1-6.1	4±4	nd-8

[a] nd=not detected. Reported quantification limits for both oxychlordane and *trans*-nonachlor in cod and haddock were 3 µg/kg wet weight, in other fish species 1 µg/kg wet weight.

Relative to other chlorinated organic compounds like the DDTs and PCBs, chlordanes were found at low levels in the liver tissues of fish. As an example, the summed mean concentration of DDT compounds (Σ *p,p'*-DDT, *p,p'*-DDD, and *p,p'*-DDE) in cod sampled in 1982 was 1110 µg/kg, and the summed concentration of PCBs was 454 µg/kg, while that for the sum of the two chlordane compounds (shown in Table 109) was 91 µg/kg. As the maximum chlordane concentration measured in liver was below the U.S. FDA action level for chlordane compounds, levels in edible muscle tissues were quite likely to have been well below that recognized, if unofficial, benchmark. However, the widespread occurrence of chlordane residues in fish taken waters of Norway, where the compounds had not been used for fifteen years, reflects their persistence and ubiquity in the marine environment.

Chlorinated terpenes and chlordane components found in fish, guillemot and seal from Swedish waters, 1974-1978. Jansson *et al.* (1979) collected and analyzed tissue samples from fish (muscle), birds (muscle), and seals (blubber) collected in the Baltic Sea and in Lake Vättern (central southern Sweden), for concentrations of chlorinated terpenes (*e.g.*, toxaphene) and chlordane compounds. Species collected from the Baltic were herring (*Clupea harengus*), guillemot (*Uria aalge*), and grey seal (*Halichoerus gryphus*); char (*Salvelinus fontinalis*) was sampled in Lake Vättern. Identified chlordane compounds included α- and γ-chlordane, *trans*-nonachlor, and oxychlordane.

Jansson *et al.* reported the following results (µg/g, or ppm, lipid weight basis) for summed concentrations of chlordane compounds in biota: in char (8.5% lipid), 0.5 ppm; in herring (3.2% lipid), 0.6 ppm; in guillemot (12% lipid), 0.7 ppm; and seal (79% lipid), 10 ppm. The authors noted that *trans*-nonachlor and oxychlordane were the most common of the chlordane compounds, although concentrations of each component in each matrix were not reported in the reference. Jansson *et al.* did specify that in fish and seal, but not guillemot, *trans*-nonachlor was the major component. Oxychlordane was an important constituent in guillemot and seal tissue but not in fish. These results are consistent with those found by Kerkhoff, Otte, and de Boer (1982) in North Sea and Atlantic Ocean biota, discussed previously.

It was noted that the concentrations reported in Jansson *et al.* (1979) were in the same range as those reported in Canadian biota. However, it was pointed out that use of chlordane in Sweden has been negligible, making the Swedish results rather surprising. It was surmised that atmospheric transport of chlordane compounds could have been the input mechanism for the resultant measured residues.

Spatial and temporal trends of organochlorine compounds in biota from the northwestern hemisphere, 1970-1979. Andersson *et al.* (1988) performed a study to learn more about the contamination of the Swedish environment by chlorinated camphenes (*i.e.*, toxaphene) and chlordane compounds (expressed as the sum of *cis*- and *trans*-chlordane, oxychlordane, and *trans*-nonachlor). For the purposes of comparison, however, a number of marine and freshwater locations from Canada to the Caspian Sea were sampled. Tissues from fish, birds, and mammals were collected for the study as a whole, but chlordanes were measured and reported in fish only. Fish collected included Atlantic salmon (*Salmo salar*), trout (*Salmo trutta*), Alpine char (*Salvelinus alpinus*), mackerel (*Scomber scomber*), and herring (*Clupea harengus*) Results for both fresh and salt water are reported as Table 111.

Table 111. Concentrations of summed chlordane compounds measured in muscle tissue from fish sampled in the northwestern hemisphere, 1970-1979. Values in mg/kg (ppm) lipid weight basis. Source: Andersson *et al.* (1988).

Location	Species	N	Collection Date	% Lipid Content	Σ chlordane	Range
MARINE						
Canada, Atlantic Ocean	Salmon	2	5/79	7.0	0.06	0.04-0.08
Greenland	Salmon	2	5/79	15	0.10	0.09-0.11
Sweden, west coast	Salmon	2	5-8/79	2.3	0.16	0.16-0.17
	Trout	3	6-8/79	4.8	0.08	0.05-0.11
	Mackerel	5	8/79	4.7	0.20	0.09-0.49
	Herring	25	7/79	3.1	0.24	(homogenate)
Baltic Sea	Salmon	5	9/79	8.2	0.34	0.30-0.39
	Herring	24	6/70	1.6	0.52	(homogenate)
	Herring	25	9/79	4.2	0.20	(homogenate)
	Salmon	14	9/71	13	0.28	(homogenate)
Gulf of Bothnia	Herring	25	6/79	1.9	0.78	(homogenate)
FRESHWATER						
Lake Vättern	Salmon	4	6-8/79	3.3	0.35	0.26-0.50
	Trout	3	6-8/79	2.6	0.22	0.14-0.33
	Alpine char	4	10/79	3.9	0.24	0.15-0.44
Lake Vänern	Salmon	10	5/79	2.6	0.31	0.20-0.49
Lake Ören	Alpine char	2	5/79	4.1	0.16	0.15-0.16
Lake Stora Luleälv	Alpine char	2	9/79	1.6	0.23	0.20-0.26
Lake Fatsjön	Alpine char	2	10/79	1.5	0.06	0.05-0.07
Lake Långvattnet	Alpine char	3	9/79	1.2	0.19	0.13-0.28

Results for both fresh and salt water environments are included here for two reasons: first, since analyses were performed by intercalibrated laboratories using the same methods (referenced within the article), and, in some cases, using the same fish species, concentrations from the two aquatic environments can be compared directly; second, according to the authors, chlordane compounds have never been used in Sweden to a significant extent. Residues occurring there were likely to have originated from indirect inputs, such as airborne fallout or, in the case of marine waters, from oceanic transport and migration of living resources. Concentrations were reported as lipid-normalized values in order to reduce the variability introduced by the different matrices analyzed in the full study.

Highest concentrations in marine fish were found in enclosed coastal waters off the east coast of Sweden, *i.e.*, the Baltic Sea and the Gulf of Bothnia (the latter extends north from the Baltic Sea and is bordered on three sides by Sweden and Finland). Lowest concentrations were measured in salmon collected in relatively remote waters of the Atlantic near Baffin Island and Greenland.

In freshwater species, no significant differences were found among the various regions and lakes sampled in Sweden. Concentrations in the lakes fell into approximately the same range as higher values found off the west coast of Sweden, and lower values found in the Baltic Sea.

The data presented by Andersson *et al.* suggest a geographic pattern of marine environmental contamination by chlordane that is associated with water circulation: lowest concentrations were found in remote open waters, intermediate levels in fish from the inland Baltic Sea, and highest concentrations in the relatively enclosed Gulf of Bothnia. This is similar to patterns observed elsewhere. The fact that this spatial pattern of distribution exists in an area where chlordane has never been registered for use and has been applied only sparingly raises the question of source inputs and mechanisms for the compounds to reach the adjacent marine environments, although these were not addressed in this study.

Contents of chlordane-, PCB-, and DDT-compounds and the biotransformation capacity of fishes in the lake area of eastern Finland. Pyysalo, Wickström, and Litmanen (1981) examined levels of chlorinated hydrocarbons, including chlordane compounds (α- and γ-chlordane, heptachlor, *trans*-nonachlor, and chlordenes) in four species of freshwater fish collected in 1980 in Finnish lakes considered to be clean. The fish species sampled were vendace (*Coregonus albula*), perch (*Perca fluviatilis*), roach (*Rutilus rutilus*) and rainbow trout (*Onchorhynchus mykiss*). Table 112 presents tissue concentration data for chlordane measured in the fish collected by Pyysalo, Wickström, and Litmanen.

Table 112. Concentrations measured in fish samples from Finnish lakes, 1980. Values in μg/kg (ppb); reporting basis not made explicit. Source: Pyysalo, Wickström, and Litmanen (1981).

Species, tissue, Date	Lake	Weight g	Length cm	Sex	α-chlordane	γ-chlordane	*trans*-nonachlor
Vendace, muscle	Kallavesi	53.4	17.7	F	520	69	-
12/10/80		54.9	17.9	F	143	52	-
		45.1	17.1	F	281	54	-
		38.0	15.7	F	349	107	-
		34.1	15.0	F	-	-	-
		46.0	16.4	M	175	33	-
		32.9	14.6	M	392	112	52
		48.3	16.7	M	-	-	-
Vendace, liver	Kallavesi	27.5	14.0	F			
12/10/80		29.1	15.2	F	69	-	-
		20.2	20.2	F			
		38.0	15.7	M	589	126	-
		28.1	14.7	M			
Vendace, muscle	Suvavesi	54.2	18.0	F	953	177	83
12/19/80		33.0	15.4	F	876	171	-
		33.8	16.0	F	732	227	69
		23.4	14.0	F	828	199	-
		24.8	14.3	F	-	-	-
		19.6	13.2	F	548	132	-
		24.0	13.7	M	247	56	-
		34.8	17.5	M	906	176	-
		38.2	15.5	M	114	24	45
		27.3	13.8	M	156	-	-
Roach, muscle	Kallavesi	59.4	15.5	F	-	-	-
12/11/80		59.7	15.4	F	-	-	-
		62.4	15.5	F	9	-	-
		48.1	14.6	F	23	-	-
		56.2	15.0	F	7	-	-
		48.7	14.1	F	66	-	-
		47.0	14.6	M	-	-	-
		39.7	13.2	M	96	-	-
Roach, liver	Kallavesi	56.2	15.0	F	400	-	-
12/11/80		59.7	15.0	F			
		62.4	15.5	F	69	-	-
		48.0	14.3	F			
Trout, muscle	Tervo (fish farm)	190	23.1	M	-	-	-
12/17/80		456	30.2	M	-	-	-
		584	33.0	M	-	-	-
Trout, liver	Tervo (fish farm)	456	31.2	M	-	-	-
12/17/80		571	34.7	M	-	-	-
		584	33.0	M	-	-	-
		370	28.5	M	-	-	-
Perch, muscle	Kallavesi	33.5	12.8	F	-	-	-
12/11/80		24.1	12.0	F	109	-	-
		28.3	12.8	F	350	95	-
		31.8	13.1	F	62	-	-
		28.2	13.2	M	-	-	-
		35.7	13.3	M	93	-	-
		36.8	13.8	M	-	-	-
Perch, liver	Kallavesi	24.1	12.0	F	250	84	39[a]
12/11/80		31.8	13.1	F			
		33.5	12.8	F	84	-	-
		36.8	13.8	F			

[a]Oxychlordane @ 16 μg/kg also measured.

Although the above results displayed a large degree of variability, one consistency was the below detection levels of chlordanes in trout tissues, both muscle and liver. Roach muscle tissue was also uniformly low in concentration, with some positive measurements made for α-chlordane, but none for γ-chlordane or *trans*-nonachlor. Perch showed higher levels and more consistent detections of chlordane compounds other than α-chlordane. Vendace contained highest levels of α- and γ-chlordane among the four species, the latter compound being detected in 15 of 20 tissue samples.

In addition to these body burden measurements, Pyysalo, Wickström, and Litmanen also evaluated levels of biotransformation enzymes (cytochrome P-450, benzo(a)pyrene hydroxylase, 7-ethoxycoumarin deethylase, UDP-glucuronosyltransferase, and glutathione S-transferase) in an attempt to link tissue contaminants to ability to process toxic compounds. Those results are shown below in Table 113.

Table 113. Biotransformation activities in the hepatic microsomal fractions of four fish species collected in Finnish lakes, 1980. Source: Pyysalo, Wickström, and Litmanen (1981).

Enzyme system	Season	Rainbow trout	Vendace	Perch	Roach
MONOOXYGENASE REACTIONS					
Cytochrome P-450 content (pmol/mg prot)	Summer	80.1±35.7 n=12	54.0[a] n=30	50.6[a] n=29	89.3[a] n=28
	Winter	74.8±48.5 n=12	57.2±5.8 n=3		
Benzo(a)pyrene hydroxylase (pmol/min x mg prot)	Summer	14.2±4.3 n=12	5.3±2.9 n=12	20.1[a] n=29	4.1[a] n=28
	Winter	38.2±9.5 n=12	8.5±1.4 n=48		
7-Ethoxycoumarin-O-deethylase (pmol/min x mg prot)	Summer	45.6±11.7 n=12	9.9±3.4 n=12	28.5[a] n=29	19.2[a] n=28
	Winter	71.2±17.4 n=12	15.2±6.8 n=48		
CONJUGATION REACTIONS					
UDP Glucuronosyltransferase (pmol/min x mg prot)	Summer	212.8±74.1 n=12	91.3±19.3 n=12	93.5[a] n=29	108.4[a] n=28
	Winter	293.4±74.8 n=12	99.0±3.2 n=48		
Glutathione-S-transferase (nmol/ml x min)	Summer	219.2[a] n=2	20.8[a] n=10	83.3[a] n=8	171.7[a] n=7
Amount of reduced glutathione (μg/ml)[b]	Summer	195.2[a] n=2	15.8[a] n=10	12.9[a] n=8	124.1[a] n=7
	Winter	108.0±16.9 n=12	56.3±1.5 n=48		

[a]Pooled livers.
[b]Determined from 105 000 x g supernatant fraction.

It was noted by the authors that the biotransformation capabilities of the four fish species appeared to negatively correlate with tissue concentrations of chlordane compounds. That is, vendace, which had the lowest level of monooxygenase and conjugation activity, contained the highest concentrations of chlordane. Perch and roach displayed enzymatic activities intermediate between vendace and trout, and accordingly also bore intermediate concentrations of chlordane. Trout had the highest biotransformation capabilities as reflected by the enzyme systems, and also contained no detectable levels of chlordanes. While acknowledging the roles of food sources and fat content in influencing resultant body burdens of contaminants like chlordane, Pyysalo, Wickström, and Litmanen felt their results indicated the importance of metabolic transformation capabilities as a major determinant of those concentrations. Interestingly, concentrations of other chlorinated hydrocarbon contaminants like DDTs and PCBs were less clearly correlated to biotransformation enzyme activity than were those for chlordane, which was taken as a general indication that fish were not able to metabolize the former compounds as readily as the latter.

Time trends of chlordane, DDT, and PCB concentrations in pike and herring from the northern Baltic Sea, 1971-1982. Moilanen *et al.* (1982) measured concentrations of chlordane (identified as α- and γ-chlordane, oxychlordane, and *trans*-nonachlor), DDT- and PCB-compounds in two species of fish collected in the waters of Finland. One species, the pike (*Esox lucius*) was a freshwater-dweller, while the other, Baltic herring (*Clupea harengus*) was collected in the northern Baltic Sea. Samples to cover the period from 1971 to 1982 were available for pike, and from 1978 to 1982 for herring. Figure 73 shows three time series plots of data presented in Moilanen *et al.* (1982): pike muscle tissue chlordane, pike liver chlordane, and herring muscle chlordane. These plots differ from those in the reference article in that lipid weight concentrations, as reported in original tabular data, are used instead of the wet weight values graphed by Moilanen *et al.*

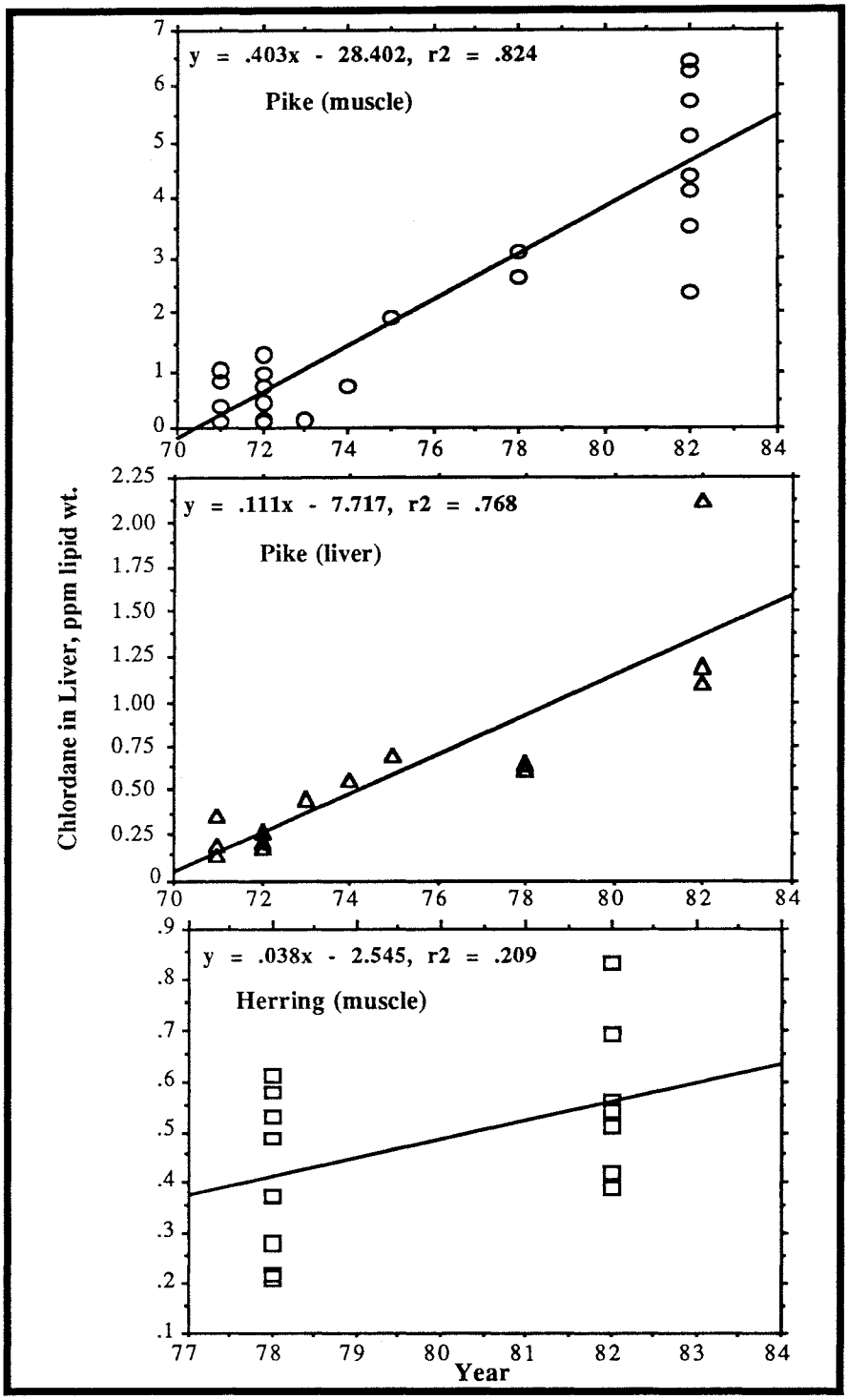

Figure 73. Time series of chlordane concentrations in pike and herring sampled in Finnish waters, 1971-1982. Values in mg/kg (ppm) lipid weight. Source: Moilanen *et al.* (1982).

In all three cases, levels of chlordane in fish tissues appeared to show increases with time. This was in contrast to results in Moilanen *et al.* for DDT and PCB compounds (not repeated here) that clearly indicated concentration declines over the same period of time.

Levels of organochlorine compounds in an inland seal population in eastern Finland, 1977-1981. In a study intended to shed light on Saimaa ringed seal (*Phoca hispida saimensis*) population declines, Helle *et al.* (1983) analyzed several tissues from fourteen specimens collected in eastern Finland. A number of organic compounds were identified and quantified, including the chlordane compounds, α-, γ-, and oxychlordane, and *trans*-nonachlor. Overall results for summed chlordane in tissues of undifferentiated specimens are portrayed below in Table 114. A breakdown of these results by age of individuals is provided in Table 115.

Table 114. Concentrations of summed chlordanes (Σ α-, γ-, oxychlordane, and *trans*-nonachlor) measured in tissues of Saimaa ringed seals from southeastern Finland, 1977-1981. Values in mg/kg (ppm) lipid weight basis. Source: Helle *et al.* (1983).

Tissue	N	Chlordane Mean ±sdev	Chlordane Range
Blubber	14	0.59±0.48	0.11-1.67
Liver	7	0.19±0.17	0.01-0.43
Muscle	4	0.02±0.007	0.01-0.03

Table 115. Concentrations of summed chlordanes (Σ α-, γ-, oxychlordane, and *trans*-nonachlor) measured in blubber of Saimaa ringed seals from southeastern Finland, 1977-1981 by age of specimen. Values in mg/kg (ppm) lipid weight basis. Source: Helle *et al.* (1983).

Age	N	Chlordane Mean ±sdev	Chlordane Range
Fetus/newborn	1	0.11	--
2-4 months	4	0.41±0.48	0.10-1.12
5-15 months	6	0.71±0.55	0.12-1.67
Mature, 4-12 years	3	0.74±0.39	0.45-1.18

In contrast to results obtained with PCBs and DDTs in blubber tissue, Helle *et al.* found that chlordane concentrations were not significantly correlated to either age or weight.

The percent compositions of the summed chlordane total were *trans*-nonachlor, 43.8 percent; oxychlordane, 37.5 percent; γ-chlordane, 10.3 percent; and α-chlordane, 8.4 percent. Referring to other studies that had shown the oxychlordane contribution to the total to be relatively much lower in fish tissues, Helle *et al.* commented that this was an indication that oxychlordane concentrations in seal tissues originated from metabolic processes.

Chlordanes were measured in all samples analyzed. The significance of this finding stems from the fact that chlordane has never been used in Finland. Occurrence of the compound in biota there was thought by Helle *et al.* to suggest that residues were transported by air or possibly formed in industrial processes.

There were no apparent indications of direct impacts on the Saimaa ringed seal population from measured body burdens of chlordane or any other contaminant. Helle *et al.* used results from the Baltic Sea, where the lowest concentrations in normal pregnant females exceeded the highest found in Finland, as a basis for this observation. However, without commenting on the implications to the health of the seal

population, they did note the maternal transfer of organic compounds like chlordane to fetal or newborn animals.

Asia

Levels of chlordane in water and sediment of rivers around Saga, Japan, 1987-1988. Hirai and Tomokuni (1989) measured residues of *cis*- and *trans*-chlordane (identified as 1,2,4,5,6,7,8,8-octachloro-3*a*,4,7,7*a*-tetrahydro-4,7-methanoindane) in river water and sediments in the vicinity of Saga (Kyushu Island), Japan. They found that concentrations of the compounds were not detectable (<0.2 ng/l) in mountain waters above the city, but were detected at levels ranging from 1 to 9 ng/l in all drainage waters closer to Saga. Within the city itself, concentrations ranged from not detected to 20 ng/l, with a median of 3 ng/l. Sediments were also collected and analyzed in the city, and concentrations ranged from 0.5 to 400 ng/g dry weight, with a median of 20 ng/g. These findings are consistent with the characterization of chlordane as an urban contaminant, and one more likely to be associated with particulate matter than water.

Chlordanes and related compounds in goby fish from Tokyo Bay, 1978. Miyazaki *et al.* (1980) measured several constituents and metabolites of chlordane in fish tissue in an early assessment for that country. In that effort, Miyazaki *et al.* collected goby fish (*Acanthogobius flavimanus*) in and around Tokyo Bay in August of 1978, and found residues of oxychlordane, *trans*-chlordane, *cis*-chlordane, *cis*-nonachlor, *trans*-nonachlor, and heptachlor epoxide. The levels of total chlordane (45 ppb) were comparable to those of *p,p'*-DDE (29 ppb), although less than those of PCBs (670 ppb) by an order of magnitude.

Chlordane residues in fish and shellfish from Tokyo Bay, 1977-79. In a study related to the previous effort, Yamagishi *et al.* (1981) measured chlordane (*cis*-, *trans*-, oxychlordane, *cis*- and *trans*-nonachlor) in seawater, clams (*Tapes philippinarum*), and three species of marine fish (*Acanthogobius flavimanus*, *Konosirus punctatus*, and *Lateolaborax japonicus*)collected in Tokyo Bay. They also analyzed freshwater fish (*Zacco platypus*) collected in rivers feeding into the bay.

Chlordanes were not detected in seawater (with a detection limit of 0.0002 ppb), but were detected (with detection limits of 0.05 ppb) at varying levels in all biota. Results by species are shown in Table 116 below. Figure 74 compares summed chlordane concentrations in gizzard shad viscera *vs.* muscle tissue.

Table 116. Concentration ranges for chlordane compounds measured in organisms sampled in and near Tokyo Bay, 1977-79. Values in ng/g (ppb) wet weight basis. Source: Yamagishi *et al.* (1981).

Organism	*cis*-chlordane	*trans*-chlordane	oxy-chlordane	*cis*-nonachlor	*trans*-nonachlor	Σ chlordanes
Short-necked clam (*Tapes japonica*)	0.2-6.8	0.1-5.7	<0.05-0.3	>0.05-3.7	0.1-4.4	0.4-21
Gobyfish (whole) (*Acanthogobius flavimanus*)	1.1-62	0.3-15	1.0-25	2.5-21	6.5-120	11.5-221
Freshwater fish (whole) (*Zacco platypus*)	2.5-230	>0.05-77	<0.05-25	1.4-39	1.6-140	9.3-489

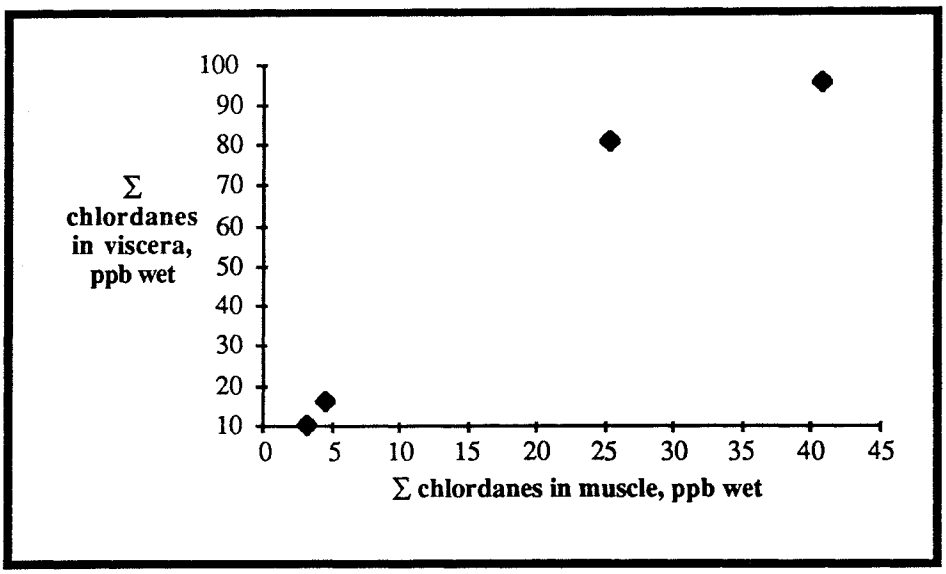

Figure 74. Concentrations of summed chlordane concentrations (*cis*-, *trans*-, oxychlordane, *cis*-, *trans*-nonachlor) in viscera (liver, kidney and stomach) *vs.* muscle tissue in gizzard shad collected from four Japanese bays in autumn, 1978. Triplicate analyses of pooled sample of three individuals. Source: Yamagishi (1981).

Concentrations about an order of magnitude greater were measured in fish samples relative to the clam samples. In both the marine and freshwater fish, the *cis*-chlordane and *trans*-nonachlor fractions were the most important in terms of the percent composition of the summed total. In short-necked clam samples, distribution appeared to be less defined across the five compounds. Only oxychlordane was comparatively minor in its occurrence.

The plot of viscera *vs.* muscle tissue concentrations of summed chlordane concentrations is a representation of limited data; nevertheless, the plot is consistent with results obtained in other studies such as Murphy (1988a) or Risebrough (1987).

Australia and the Antarctic

Organochlorine pesticide residues in grain storage areas of New South Wales, Australia, 1987-1988. Allender (1989) reported results of analysis of swabs taken from grain storage areas in the state of New South Wales (southeastern state in Australia). Although this substrate is rather far removed from the coastal and estuarine environment of the United States., this study is included here as an indication of the widespread use of chlordane compounds. A number of organochlorine pesticides were targeted, including chlordane (not specified as to composition or definition). For four regions within New South Wales, chlordane was detected in 33.1 percent of 1,470 samples collected, although concentrations were not specified.

Pesticide residues in drinking water in the North Coast region of New South Wales, Australia, 1986-1987. Ang, Meleady, and Wallace (1989) investigated the concentrations of several pesticides in drinking water supplies (wells, dams, rooftop tanks) in an area of Australia with a relatively high rural population and intensive agricultural activities. Among the compounds targeted was chlordane (not defined as to specific composition). Chlordane was detected above trace levels (defined as >0.05 µg/l) relatively infrequently relative to another cyclodiene pesticide, dieldrin: chlordane was measured in 2 of 659 samples, at levels of 0.80 and 0.13 µg/l; dieldrin was found in 19 samples, at concentrations ranging from 0.06 to 1.90 µg/l.

Chlordane compounds found in the marine atmosphere of the southern hemisphere, 1983-1984. Having noted the paucity of information on chlordanes in the southern hemisphere, Kawano *et al.* (1985) collected organochlorine compounds on adsorbent material in the low latitudes of the southern hemisphere between the Indian Ocean and the Southern Ocean from late 1983 to early 1984. Concentrations of four chlordane constituents were determined: *cis*- and *trans*-chlordane, and *cis*- and *trans*-nonachlor. Table 117 lists summarized results reported by Kawano *et al.* (1985).

Table 117. Concentrations of chlordane and DDTs measured in the marine atmosphere of the southern hemisphere, 1983-84. Values in pg/m^3. Source: Kawano *et al.* (1985).

Location	Collection Date	Σ chlordane[1]	DDTs[2]
Indian Ocean			
Eastern Indian Ocean	12/83	19	93
Off western Australia	12/83	18	60
Southern Ocean			
45°14'S 115°30'E - 49°51'S 116°05'E	1/84	7.5	8.4
62°22'S 119°59'E - 64°50'S 122°00'E	1/84	4.8	11

[1]Σ chlordane = *cis*- + *trans*-chlordane + *trans*-nonachlor.
[2]DDTs = *o,p'*-DDT + *p,p'*-DDT.

Although higher concentrations of summed chlordane compounds were obtained in the northern portion of the study area, Kawano *et al.* noted that chlordane levels obtained for the marine atmosphere of the southern hemisphere were roughly equivalent to those obtained for the northern hemisphere. The authors attributed this to comparable levels of use in both hemispheres, as well as long-range transport of volatilized constituents.

Bioconcentration and residue patterns in north Pacific, Bering Sea, and Antarctic ecosystems, 1980-82. Kawano *et al.* (1988) reported results from a study of bioaccumulation properties and residual patterns of individual chlordane compounds (identified as *cis*-chlordane, *trans*-chlordane, oxychlordane, *cis*-nonachlor, and *trans*-nonachlor) in marine organisms ranging from zooplankton to marine mammals collected in three regions of the world. In the north Pacific and the Bering Sea, they analyzed zooplankton, squid (*Gonatopsis borealis*), walleye pollock (*Theragra chalcogramma*), chum salmon (*Oncorhynchus keta*), thick-billed murre (*Uria lomvia*), and Dall's porpoise (*Phocoenoides dalli*). In the Antarctic, krill (*Euphausia superba*), a benthic fish species (*Trematomus bernacchi*), Adelie penguin (*Pygoscelis adeliae*), and Weddell sea (*Leptonychotes weddelli*) were sampled. Table 118 summarizes analytical results for concentrations of chlordane compounds for the organisms sampled.

Table 118. Concentrations of chlordane measured in organisms sampled in the north Pacific, Bering Sea, and Antarctic marine ecosystems. Values in ng/g (ppb) lipid weight basis. Source: Kawano *et al.* (1988).

Organism[a]	N	% Lipid	*cis*-chlordane	*trans*-chlordane	*cis*-nonachlor	*trans*-nonachlor	oxy-chlordane
Zooplankton		1.89 (1.45-2.36)	19 (13-27)	13 (7.1-20)	5.1 (3.2-8.7)	14 (12-15)	2.9 (2.3-3.8)
Squid	3	8.51 (7.24-10.8)	15 (11-18)	8.1 (6.3-9.9)	2.4 (2.2-2.8)	18 (14-20)	1.2 (0.77-1.6)
Walleye pollock	3	3.86 (2.74-5.21)	44 (34-54)	17 (16-20)	10 (6.4-12)	62 (47-92)	8.3 (5.6-110
Chum salmon	3	9.11 (5.67-12.8)	9.5 (8.2-11)	5.2 (5.1-5.9)	2.1 (1.6-2.7)	17 (13-21)	2.5 (2.4-2.6)
Thick-billed murre	3	48.2 (39.4-64.5)	2.8 (1.4-3.9)	<0.05	10 (2.9-15)	2.7 (1.7-4.5)	82 (63-130)
Dall's porpoise	3	83.9 (81.4-88.0)	440 (360-550)	63 (53-73)	270 (240-310)	1800 (1600-2000)	250 (160-340)
Krill		2.76	0.58	0.51	0.22	0.80	0.11
Benthic fish	5	3.50 (1.15-6.01)	4.4 (1.8-8.0)	1.7 (0.81-3.4)	3.0 (0.81-7.0)	11 (2.3-29)	0.98 (0.18-1.8)
Adelie penguin	1	89.9	0.91	<0.05	1.7	15	16
Weddell seal	1	87.6	6.8	<0.05	8.3	41	13

[a] Tissues analyzed: zooplankton, squid, walleye pollock, chum salmon, krill, and benthic fish = whole body; thick-billed murre, and Adelie penguin = subcutaneous fat; Dall's porpoise and Weddell seal = blubber.

Kawano *et al.* also measured seawater concentrations of chlordane compounds in the north Pacific and the Bering Sea. They reported concentrations of *cis*-chlordane of 4.7±0.99 pg/l (parts per quadrillion), *trans*-chlordane at 4.2±0.64 pg/l, and *trans*-nonachlor at 1.4±0.14 pg/l. Oxychlordane and *cis*-nonachlor were not detected, with levels <0.2 pg/l.

Kawano *et al.* used their results to evaluate bioconcentration, biomagnification, and relative differences in metabolism of chlordane compounds. They noted that although variation in residue patterns in lower trophic organisms was slight, differences in higher animals were much greater. For example, in thick-billed murre, chlordane residues were dominated by oxychlordane, while in Dall's porpoise, *trans*-nonachlor was the major chlordane compound. Similar results were observed in Adelie penguin and Weddell seal. This would suggest that these bird species metabolize chlordanes to oxychlordane (a known chlordane metabolite that was not detected in waters sampled with the organisms) more efficiently than do marine mammals.

Fate of DDTs, PCBs and chlordane compounds in the Antarctic marine ecosystem, 1981-1982. Hidaka *et al.* (1984) measured concentrations of DDT compounds, PCBs, and chlordane compounds (not identified within this article, but likely to be *cis*-chlordane, *trans*-chlordane, oxychlordane, *cis*-nonachlor, *trans*-nonachlor, α-chlordene, and γ-chlordene, as measured in the referenced paper by Kawano *et al.*, (1985) in seawater, krill, benthic fish, and Weddell seal tissues. Hidaka *et al.* determined that the Japanese base station in the Antarctic itself (Syowa Station) was a point source of chlorinated organic contamination, and so targeted a more isolated location (Tottuki Point) both upwind and upcurrent from Syowa for sample collections.

Chlordane concentrations in the benthic fish, *Trematomus bernacchii*, were compared to body length. It was found that no significant positive correlation existed. Hidaka *et al.* commented that this implied that chlordane compounds reach equilibrium concentrations in that species at an earlier growth stage.

Patterns of bioaccumulation in the Weddell seal (*Leptonychotes weddelli*) collected at Tottuki Point were also examined. Both whole body burdens and concentrations of chlordane increased with age, and were about equivalent to those for PCBs. Concentrations and body burdens for DDT compounds were considerably higher, which the authors attributed to the more biodegradable nature of chlordanes and PCBs.

Bioconcentration of chlordanes in the Antarctic food web was investigated by Hidaka *et al.*, based on concentrations in four media: seawater, krill, fish, and seal. These results, and those for ΣDDT and ΣPCB for comparison, are summarized below in Table 119.

Table 119. Bioconcentration of chlordanes, DDTs, and PCBs in the Antarctic marine ecosystem. Units as specified; biota values wet weight basis. Source: Hidaka *et al.* (1984).

Matrix	Concentration/ Bioconcentration factor	Σ Chlordane	Σ DDT	Σ PCB
Seawater	Concentration (μg/l)	5.4×10^{-6}	1.0×10^{-6}	5.4×10^{-5}
Krill	Concentration (ng/g)	6.1×10^{-2}	2.6×10^{-2}	2.1×10^{-2}
	Bioconcentration factor	1.1×10^{4}	2.6×10^{4}	3.9×10^{2}
Fish	Concentration (ng/g)	4.3×10^{-1}	6.3×10^{-1}	2.5×10^{-1}
	Bioconcentration factor	7.9×10^{4}	6.3×10^{5}	4.7×10^{3}
Seal	Concentration (ng/g)	1.9×10	5.2×10	1.2×10
	Bioconcentration factor	3.5×10^{6}	5.2×10^{7}	2.2×10^{5}

As shown in the table, concentrations and bioconcentration factors for all three compounds increased with increasing trophic level. The bioconcentration factor for chlordane compounds was found to be midway between DDTs and PCBs.

DISCUSSION AND RECOMMENDATIONS

Summary of Acute Toxicity Results

A number of the laboratory studies discussed previously reported toxicity results as LC_{50}s, or exposure concentrations at which half of a specified test group were not able to survive. These values have been summarized into a single table, Table 120.

Table 120. LC_{50} values for chlordane and heptachlor reported in studies referenced in this manuscript. Values in µg/l (ppb). Results reported for oysters by Parrish *et al.* (1976) and Schimmel, Patrick and Forester (1976) were EC_{50s}, as detailed below.

Organism	Time Basis, hrs	LC_{50}, µg/l	Reference
CHLORDANE			
Carp larvae	96	16	Verma, Tonk & Dalela (1981)
Cyprinus carpio			
Freshwater catfish	24	3.5	Mishra & Srivastava (1984)
Heteropneustes fossilis	48	1.25	
	72	0.29	
	96	0.275	
Polychaete worm	288 (seawater)	220	McLeese, Burridge & Van Dinter (1982)
Nereis virens	288 (sediment)	<5800 (µg/kg)	
	288 (overlying water)	190	
Shrimp	96 (seawater)	2.0	McLeese & Metcalfe (1980)
Crangon septemspinosa	97 (sediment)	120	
Freshwater catfish	96	6300	Verma & Tonk (1984)
Heteropneustes fossilis			
Fathead minnow, soft water	24	69	Henderson, Pickering & Tarzwell (1959)
Pimephales promelas	48	69	
	96	52	
Fathead minnow, hard water	24	98	
Pimephales promelas	48	69	
	96	69	
Bluegill	24	36	
Lepomis macrochirus	48	32	
	96	22	
Goldfish	24	166	
Carassius auratus	48	87	
	96	82	
Guppy	24	560	
Lebistes reticultaus	48	190	
	96	190	
Plankton	24?	3.45	Mani & Konar (1986)
Diaptomus forbesi			
Worm	24?	22	
Branchiura sowerbyi			
Tilapia	24?	28	
Tilapia mossambica			
Oyster	96	6.2*	Parrish *et al.* (1976)
Crassostrea virginica			
Pink shrimp	96	0.4	
Penaeus duorarum			
Grass shrimp	96	4.8	
Palaemonetes pugio			
Sheepshead minnow	96	24.5	
Cyprinodon variegatus			
Pinfish	96	6.4	
Lagodon rhomboides			

Table 120. Continued

Organism	Time Basis, hrs	LC_{50}, µg/l	Reference
HEPTACHLOR			
Tilapia *Tilapia mossambica*	48	150	Radhaiah, Girija, and Rao (1987)
Oyster *Crassostrea virginica*	96	1.5*	Schimmel, Patrick, and Forester (1976)
Pink shrimp *Penaeus duorarum*	96	0.11	
Grass shrimp *Palaemonetes pugio*	96	1.06	
Sheepshead minnow *Cyprinodon variegatus*	96	3.68	
Pinfish *Lagodon rhomboides*	96	3.77	
Spot *Leiostomus xanthurus*	96	0.85	

*Effect expressed as percent reduction in shell deposition for oysters.

Interpretation of values reported in this table is limited by a number of factors, including the chlordane formulations employed (and often not specified), the different species tested, and the lack of detailed information on whether the toxic concentrations determined were nominal or measured. The fact that both freshwater and saltwater organisms are included in the table also compounds difficulties in comparison. Recalling that studies such as Radhaiah, Girija, and Rao (1987) and Verma and Tonk (1984) suggested impacts from chlordane exposure on kidney function, differences in osmoregulation between freshwater and saltwater organisms could result in toxicity differences as well. Tachikawa *et al.* (1987) found substantial differences in tissue burdens of chlordane between freshwater- and saltwater-acclimated fish of the same species exposed to the same concentrations, and it would seem reasonable that differences would be manifested in toxicities.

Table 120 shows the large range of values determined for LC_{50} concentrations. The fact that two separate studies, Mishra and Srivastava (1984) and Verma and Tonk (1984), reported such widely disparate results for 96-hour LC_{50}s in the freshwater catfish (0.275 µg/l and 6300 µg/l, respectively) illustrates some of the problems in comparison of results. In fact, the wide disparity may warrant a closer examination of those results and the methods used to generate them.

However, examination of results that do avail themselves to some degree of comparison suggests the following:

1. Exposure via seawater appeared to be more toxic than exposure through sediments only, although water overlying sediments was more toxic than both.
2. Shrimp species were among the organisms most sensitive to chlordane exposure; worms and some fish species appeared to be much less sensitive.
3. The chlordane mixture used by Parrish *et al.* (1976) was consistently less toxic than the technical mixture of heptachlor used by Schimmel, Patrick, and Forester in similar tests with the same animals.
4. Other constituents present in the water affected toxicity of chlordane compounds, with hard water (*i.e.*, higher mineral content) generally showing a lesser degree of toxicity than soft water. Binding of toxic constituents to ionized impurities in water is a likely explanation for the reduced toxicity in hard water.
5. Tests that examined toxicity as a function of time and concentration suggested that some LC_{50} concentrations in fish may plateau after initial exposure, with only small decreases noted between 48 and 96 hours

Characterization of Chlordane Compounds as Urban or Agricultural Contaminants.

Chlordane is a broad-spectrum pesticide that in early years of use was employed in a great many agricultural applications. However, increasingly more restrictive regulatory requirements imposed on use (shown, for example in use patterns from the state of California in Figure 5), and the resultant patterns of occurrence in the coastal environment indicate that in more recent years, it has become a pesticide and a contaminant associated with population centers. For example, results from the NS&T Program of NOAA show that highest concentrations in both biotic and abiotic coastal and estuarine environments occur in heavily populated regions like the urban northeast Atlantic coast and the southern California coast. Regionally focused studies, such as those of Sloan (1981a, 1981b, 1987) and the NYDEC (1978) also indicated that chlordane was detected most frequently and at highest concentrations in freshwater fish sampled around the most heavily populated urban regions of that state: near New York City and on Long Island. Comparison of NS&T results to those from earlier nationwide sampling efforts (*e.g.*, Butler and Schutzmann, 1978) suggests that a rather marked shift in the occurrence of chlordane as a contaminant occurred with the shift in permitted uses of chlordane from agricultural to structural. In collections made in 1973 and 1974 by Butler and Schutzmann, the only positive detections of chlordane in juvenile estuarine fish occurred in Hawaii, Maryland, and Texas. These likely reflected the considerable use of chlordane as an agricultural pesticide in those regions during the years preceding the study. Subsequent collections and analyses did not show an association between agriculture and elevated concentrations in the coastal environment.

However, the emergence of chlordane as the pesticide of choice for the treatment of building foundations against termites and other insects that damage structures very likely accounts for results from more recent environmental assessments in coastal regions and elsewhere that show chlordane to be associated with population density. Virtual elimination of permitted application of chlordane compounds probably will result in measurable declines in occurrences and concentrations of environmental residues. Apparent increases in chlordane use just before most applications were banned (as in the State of California) may result in residue peaks occurring in the environment, after which declines should become common. The demonstrated resistance to breakdown after application, however, suggests that declines will be slow and that residues could be expected to linger in the environment for some time to come.

Trophic Level Differences in Abilities to Metabolize Chlordane Isomers.

Several researchers demonstrated that organisms from different trophic levels appeared to process environmental chlordane mixtures differently. Gardner and Pruell (1988) found concentrations of α- and γ-chlordanes in sediments and biota of Quincy Bay, Massachusetts that seemed to define three distinct residue patterns: one for sediments, another for bivalves, and a third for lobsters and winter flounder. Results from NOAA's NS&T Program Benthic Surveillance and Mussel Watch projects found similar distinctions in proportions of α-chlordane, *trans*-nonachlor, and heptachlor among sediments, mussels, and fish. The relatively higher percentage of *trans*-nonachlor in fish liver tissues indicates that fish either metabolize other compounds to *trans*-nonachlor more readily, or that they have a lesser ability to process *trans*-nonachlor.

Kawano *et al.* (1988) measured chlordane compound residues in biota of the north Pacific, Bering Sea, and Antarctic ecosystems, and determined that bird species metabolized chlordanes to oxychlordane more efficiently than do other higher organisms, including marine mammals.

In the North Sea, Kerkhoff, Otte, and de Boer (1982) found that of the chlordane compounds they targeted for analysis, *trans*-nonachlor predominated in such marine mammals as harbor seals and dolphins. There were, however, differences in the next most prevalent compounds, with *trans*-nonachlor followed by oxychlordane in seals, but *cis*-chlordane in dolphins.

Helle *et al.* (1983) determined a similar set of results in blubber of ringed seals sampled in inland waters of Finland. They found that of a summed total for four chlordane compounds, *trans*-nonachlor and oxychlordane comprised over 80 percent. Because oxychlordane was measured only at relatively low concentrations in fish, the primary food source of the seals, Helle *et al.* took this as an indication that blubber residues of oxychlordane in particular derived from metabolic processes.

Even within the same trophic level, distinct differences in biochemical processing pathways apparently exist and significantly influence levels of chlordane compounds occurring in, for example, fish. Pyysalo,

Wickström, and Litmanen (1981) correlated body burdens of chlordanes in four fish species with biotransformation capabilities as reflected by detoxifying enzyme activities, and determined that some fish species apparently have more efficient enzyme systems that result in lower body burdens of contaminants such as chlordane.

Marine Mammal Blubber Concentrations.

A number of studies have examined concentrations of chlordane compounds in marine mammals sampled around the world, and these were summarized in greater detail in regional discussions of this report. A wide range of mammal species, tissues, chlordane compounds, and reporting bases were used by the various researchers, and as such, direct comparisons are not possible. Table 121 summarizes results for chlordane extracted from studies used in this review.

Table 121. Ranges of concentrations of chlordane compounds measured in tissues of marine mammals, from studies referenced in this report. Note reporting basis of concentrations.

Species	Tissue	Reporting Basis	Target Compounds*	Range, ppb	Reference
Harbor seal *Phoca vitulina*	Blubber	Wet	(not defined)	ND	Anas (1974)
Humpback whale *Megaptera novaeangliae*	Blubber	Wet	α-c	ND-200	Taruski, Olney, Winn (1975)
Sperm whale *Physeter catodon*				ND-300	
Dense-beaked whale *Mesoplodon densirostris*				100-300	
Atlantic pilot whale *Globicephala melaena*				600-1400	
Atl. white-sided dolphin *Lagenorhynchus acutus*				ND	
Saddleback dolphin *Delphinus delphis*				1200	
Striped dolphin *Stenella caeruleoalba*				1400-2700	
Pac. white-sided dolphin *Lagenorhynchus obliquidens*				5000	
Pacific pilot whale *Globicephala scammoni*				ND	
Grey seal *Halichoerus gryphus*	Blubber	Lipid	α-, γ-, oxy-c *t*-n	10000	Jansson *et al.* (1979)
Harbor seal *Phoca vitulina*	Liver	Wet	(chlordane)	<70-<100	New York DEC (1979)
			(heptachlor)	<20	
	Brain		(chlordane)	<60	
			(heptachlor)	<20	
Harbor seal *Phoca vitulina*	Blubber	Lipid	*c*-, *t*-, oxy-c *t*-n	500-5700	Kerkhoff, Otte, DeBoer (1982)
	Liver			2300	
	Brain			940	
White-beaked dolphin *Lagenorhynchus albirostris*	Blubber			5400-11300	

Table 121. Continued

Species	Tissue	Reporting Basis	Target Compounds*	Range, ppb	Reference
Ringed seal	Blubber	Lipid	*c*-, *t*-, oxy-c	110-1670	Helle *et al.* (1983)
Phoca hispida saimensis			*t*-n		
	Liver			10-430	
	Muscle			10-30	
Harbor porpoise	Blubber	Wet	*c*-, oxy-c	3900-13900	Gaskin, Frank, Holdrinet
Phocoena phocoena	Liver		*c*-c, oxy-n	20-160	(1983)
	Muscle		oxy-c, *c*-n	20-120	
Harp seal	Blubber	Wet	α-, β-, oxy-c	80-240	Ronald *et al.* (1984)
Phoca groenlandica	Blood			0	
	Brain			0-10	
	Kidney			0-10	
	Liver			0	
	Muscle			0-20	
Weddell seal	?	Wet?	*c*-, *t*-, oxy-c	19	Hidaka *et al.* (1984)
Leptonychotes weddelli			*c*-, *t*-n, α-, γ-chlordene		
Gray whale	Blubber	Wet	α-c, *t*-n	3.8	Malins, Brown, Chan
Eschrichtius robustus	Gut contents			13	(unpubl.)
	Liver			41.7	
	Feces			10	
	Blood			25.1	
	Kidney			10.2	
Ringed seal	Blubber	Wet	12 chlordane related	0.23-0.47	Muir, Norstrom, Simon (1988)
Phoca hispida					
Weddell seal	Blubber	Lipid	*c*-, *t*-, oxy-c	69.1	Kawano *et al.* (1988)
Leptonychotes weddelli			*c*-, *t*-n		
Bottlenose dolphin	Blubber	Lipid	*t*-n	1000-58000	Geraci (1989)
Tursiops truncatus	Liver	Lipid		0-52000	
	Liver	Wet		0-5500	
Atl. pilot whale	Blubber	Lipid		4000-18000	
Globicephala melaena	Liver	Lipid		0-1500	
	Liver	Wet		0-200	
Harbor porpoise	Blubber	Lipid		5000-12000	Geraci (1989)
Phocoena phocoena	Liver	Lipid		0-8700	
	Liver	Wet		0-400	
Humpback whale	Blubber	Lipid		200-7000	
Megaptera novaeangliae					

*The suffix "-c" represents chlordane, while "-n" represents nonachlor; *c*- and *t*- are abbreviations for *cis*- and *trans*-, respectively.

Table 121 illustrates the diversity of results for chlordane in marine mammal tissues. While direct comparisons must be avoided, some generalizations and observations based on the information in the table can be made:

1. Chlordane compounds are ubiquitous in marine mammal tissues, being found in animals collected in waters both distant and near populated regions, and in waters near and removed from areas where chlordane was used.
2. With few exceptions, concentrations of chlordane compounds were highest in lipid-rich blubber tissue. An exception included results reported by Malins, Brown, and Chan (unpublished) for analysis of tissues from a single deceased gray whale specimen that showed a distribution of levels across tissues with blubber concentrations lowest of six matrices examined.
3. The very high maximum concentrations measured by Geraci (1989) in bottlenose dolphins were verified by analysis of other species and comparison to existing results. The fact that these dolphins were part of the mass mortality occurring on the Atlantic coast of the United States in 1987 and 1988 shows that levels of exposure to chlorinated organic compounds such as chlordanes remain high, with possible pathological implications.
4. High concentrations found in marine mammals could have human health implications if they occur in animals hunted by aboriginal populations. It is possible that at least some tissues consumed by native peoples exceed FDA action levels for chlordane compounds.

Given the demonstrated lipophilicity of chlordane compounds (as well as other organochlorines), and the upper positions occupied by many marine mammal species in marine food webs, the fact that some of the highest biotic levels of chlordane in the marine environment have been found in blubber tissue is not surprising. Moderately elevated concentrations of chlordane would be encountered in primary food items of many marine mammals, *i.e.*, fish, and this dietary exposure would be much higher than that provided by ambient concentrations in the environment. Repeated exposure to these higher concentrations would then result in higher body burdens in the predatory animals.

Although reproductive impacts attributable to body burdens of chlordane have not been demonstrated, elevated concentrations of chlordane found in blubber of marine mammals may have toxicological implications if the animals are stressed by food shortages or other factors. Geraci (1989), for example, speculated that in periods of low food availability , disease, or lactation, energy reserves in the blubber may be utilized, which would also mobilize lipophilic compounds found in those tissues. If sufficient quantities are metabolized, the animal may be subjected to what is equivalent to an acute exposure to chlordane and/or other organochlorine compounds. The very high levels of organochlorine compounds found in bottlenose dolphins analyzed as part of the study of mass mortality along the eastern seaboard in 1987-1988 (Geraci, 1989) suggested this mechanism as a possible contributing factor in the demise of those animals. Measurements (Malins, Brown, and Chan, unpublished) in a gray whale found dead along the coast of Washington state also supported this possibility, as blubber tissue concentrations of chlordane compounds (as well as those for other synthetic organic chemicals) were low, while blood and liver tissue levels were high.

This has potentially serious and long-term implications for evaluating the significance of ambient residues of organic contaminants in the environment. While measured levels in water and sediment may be found to be low, and below concentrations known to be directly associated with acute or chronic impacts in exposed organisms, the propensity of these compounds to accumulate in upper portions of the food webs exemplified by marine mammals suggests that even relatively low levels should be of some concern. Environmental concentrations that would seem to be innocuous may potentially result in marine mammal tissue levels of pathological significance in concert with other stress factors.

Selection of chlordane compounds to monitor. The complexity of the technical chlordane mixture incorporated into pesticide formulations has been detailed by several investigators. As noted previously, Miyazaki, Yamagishi, and Matsumoto (1985) enumerated at least 50 constituent compounds in technical chlordane. This complexity necessitates that researchers and environmental managers operating within fiscal and temporal constraints target only selected compounds for analysis and monitoring. Based on both the primary constituents of the mix, as well as the occurrence and metabolism of individual components, the following four compounds are suggested as potential major analytical targets: α- and γ-chlordane, *trans*-nonachlor, and oxychlordane. According to an analysis of the composition of technical chlordane by the manufacturer (Velsicol Chemical Corporation), the α- and γ- isomers of chlordane comprise about 43 percent of the mixture as marketed. The α- and γ-chlordane isomers are readily metabolized in most

animals (Noble and Elliott, 1986), so their presence in tissues could suggest recent exposure to the parent mixture.

Although it is a relatively minor component in the original chlordane formulation, *trans*-nonachlor is one of the most commonly encountered compounds measured in biotic substrates. This suggests that concentrations of *trans*-nonachlor and their relation to other chlordane constituents and metabolites may be enlightening in terms of defining continued or illegal use, as well as rates of degradation in the environment. (R. J. Sloan, NYDEC pers. comm. 24 May 1990). Similarly, oxychlordane is not even listed by the Velsicol Chemical Corporation in its breakdown of the technical product, but is apparently one of the most important and persistent metabolites of both α- and γ-chlordane (NRCC, 1974), especially in higher animals. Stickel *et al.* (1979) found that oxychlordane was among the most toxic of the chlordane-related compounds. Nonachlor was nearly as persistent, but less toxic.

Heptachlor, a primary pesticide compound in its own right (although at much lower application levels than chlordane), and a component of technical chlordane, has been found only infrequently and at low levels in the marine environment. This might be explained in part by the findings of Peyton, Anderson, and Gantzer (1988), who determined that heptachlor did not desorb from soil particles into water as readily as other chlordane compounds. The relative dearth of quantified measurements in many environmental assessments suggests that analytical effort in broad-scale studies might be better expended in targeting other compounds, such as those mentioned previously. Obviously, if a known heptachlor contamination problem exists, then it would be advisable to include the compound and its primary metabolite, heptachlor epoxide, in environmental surveys.

Because concentrations of contaminants like chlordane compounds are reported in several different ways (*e.g.*, wet weight, dry weight, lipid weight), it is important that the reporting basis in studies be made explicit. Even more helpful is the inclusion in reports and data summaries of normalizing ancillary information such as grain size in sediments, and percent water content or lipid content in tissues. Particularly noteworthy among environmental programs has been the CMW Program, which, in its 10-year data report (Phillips, 1988), presents data as wet weight, dry weight, and lipid weight to facilitate ease of use.

Regional Aspects of U.S. Coastal Chlordane Contamination.

As discussed previously, results from NOAA's NS&T Program depict regions of the nation's coast where relatively more elevated concentrations of chlordane compounds can be found in sediments and resident biota. The Benthic Surveillance Project has identified portions of the urban northeast, and certain areas in southern California, where sediment concentrations of summed chlordanes have been measured at higher levels. In fish liver tissue, embayments in Massachusetts and New York again showed elevated concentrations. In contrast to sediment results, levels in southern California were comparable to those from the urban northeast, while moderately high concentrations were consistently found in the San Francisco Bay region. Available sediment results from the Mussel Watch Project show patterns of chlordane residue concentrations similar to those for Benthic Surveillance sediment results. Bivalve tissue results were consistent with other NS&T measurements, although the greater sampling density appeared to show other regions with higher tissue concentrations. For example, some embayments in states along the eastern portion of the Gulf of Mexico showed more elevated concentrations than were evident in sediments or fish tissue. Particularly unusual were the very high (on a relative basis) levels found by the NS&T Mussel Watch Project in Choctawhatchee Bay, a region removed from major urban and industrial influences, although it is adjacent to Eglin Air Force Base and is located in an area with high tourism. Because high concentrations were measured in both sediments and in oyster tissues, and because no obvious source of these concentrations has been identified, a more intensive survey of the Choctawhatchee Bay region might be warranted. Presently there are three NS&T Mussel Watch sites in the area; with a new site recently added to help define the nature and extent of contamination.

Perhaps just as notable as the regions of the United States where chlordane concentrations were elevated were those areas where levels were *not* high. With the exception of some bivalve tissue measurements in oysters collected from portions of the eastern Gulf Coast states, the southeast Atlantic and the western Gulf of Mexico coasts were remarkably free from contamination. The Texas coast in particular has shown low levels of chlordane contamination, despite the agricultural activities that influence conditions there, and despite the fact that relatively high levels of other organochlorine compounds such as DDE have been found in resident wildlife. The lower concentrations in Texas and elsewhere in the western Gulf of Mexico and southeast Atlantic Coast are reflected not only in recent measurements collected by NOAA and state

agencies (*e.g.*, Ryan *et al.*, 1985; Bugg, Higgins, and Robertson, 1967; Stoker, 1982; Marion, Barker, and Settine, 1987), but also in the relatively few national studies that targeted chlordane compounds in the past.

Factors which have limited the ability to make regional comparisons include incomparability among analytical methods, incompatibility of reporting conventions (*i.e.*, wet weight/dry weight/lipid weight bases, with no ancillary data for converting from one basis to the others), and lack of standardization in the selection of technical chlordane compounds to analyze. The first two considerations are common to all comparative assessments of environmental programs. The complexity of the technical form of chlordane adds another dimension to comparison and interpretation of monitoring and assessment data. Standardization of analytical and reporting protocols would enable more extensive integration of results from different programs, but may not satisfy other program and/or agency requirements. However, to the extent possible, compatibility with other efforts should be considered during design and evaluative phases of environmental programs.

Another set of considerations that hinders direct comparison of results is the potential difference in the ways in which different indicator or monitored species process and retain chlordane compounds. Because no indicator species occurs ubiquitously around the coastline of the United States, regional or national programs must sample multiple species of generally unknown comparability. As a result, it is not clear whether significant differences in tissue burdens from two or more species result from true differences in environmental exposure occurring among habitats, or from metabolic differences among species. As was seen in both the CMW Program and in the NS&T Mussel Watch Project, distinct differences in tissue levels of chlordane appeared to exist between *Mytilus edulis* and *Mytilus californianus*. Although data exist for concentrations in resident *M. edulis* and transplanted *M. californianus* co-located at three sites and collected on the same date, these results were relatively limited in number and scope. The NS&T Mussel Watch Program has also directed effort toward comparative analyses. However, expanded programs of interspecies comparisons should be considered to increase the legitimate intercomparison of results in and among monitoring and assessment programs.

Comparison of freshwater fish data for chlordane concentrations against those found in estuarine or marine fish has suggested that relatively higher levels may occur in freshwater species. Metzger (1987) summarized general aspects of fish residue monitoring data requested by EPA's Office of Pesticide Programs from EPA regional offices, State agencies, and other Federal agencies and noted, that for the entire data set, freshwater fish contained higher average chlordane residues than saltwater fish. Higher trophic level saltwater predators, however, contained slightly more elevated concentrations than their freshwater counterparts. The extent to which this may be attributable to higher levels of exposure in adjacent or nearby environments is not clear. Evidence (Tachikawa *et al.*, 1987) exists that freshwater fish may accumulate chlordane to higher concentrations in some tissues than saltwater fish of the same species exposed to the same concentrations. This has been attributed to osmoregulatory processes in fresh- and saltwater, and complicates comparison of results from the two environments--as well as those from estuarine areas.

Temporal Aspects of U.S. Coastal Chlordane Contamination.

While a few studies have provided enough geographic coverage to enable limited assessment of regional trends in chlordane contamination of the coastal environment, consistent temporal data are much less available. Although the NOAA NS&T Program is designed to provide information on temporal trends in coastal and estuarine environmental quality, the limited data presently available do not permit meaningful analysis with respect to time. However, additional results from recent NS&T Program field collections are expected in the near future, will augment those results already extant, and should enable temporal evaluations to begin in earnest.

The CMW Program is one effort which has been in existence and adhered to consistent collection and analytical methods for a period of time sufficient in length to enable some interpretation of temporal trends and variability (refer to Figures 59 and 60). The temporal plots of these data suggest declines in concentrations measured in the California coastal environment. Many of the most recent measurements (*i.e.*, 1987) in mussels were near or below detection limits. However, Figure 5 showed that a sharp increase in use of chlordane in California was recorded in 1986 and 1987, as remaining stocks were used before the elimination of nearly all permitted applications in 1988. The extent of this use was equivalent in pounds applied to that of the mid-1970s, before regulatory restrictions were imposed. It is possible that monitoring data from 1988 and later may reflect this usage spike. Even so, it is reasonable to presume that monitored media would subsequently show steady declines.

Targeted efforts by Sloan (1987) in the Long Island region of New York suggested levels of chlordane in freshwater fish showed a consistent temporal decline over the period 1982-1985. Again using the FDA action level as a point of reference, the maximum concentration found in 1982 was an order of magnitude higher than the 0.3 ppm action level, while in 1985, the maximum was essentially equal to that concentration.

Such declines with time may or may not be representative of situations in other parts of the world, where the regulatory history for chlordane and other pesticides may be very different from that in this country. For example, one analysis by Muir, Norstrom, and Simon (1988) suggested that levels of chlordane compounds in the blubber of seals in the arctic did not change significantly in animals sampled between 1972 and 1984. Moilanen *et al.* (1982) found that over approximately the same period of time, tissue concentrations of chlordane in Finnish fish showed increases. While current worldwide use levels of chlordane are undoubtedly much lower than they were when almost no use restrictions were in place in the United States and elsewhere, the limited continuing use known to exist in some regions will contribute to global environmental background levels.

Additional Information Needs.

Only recently have biochemical and physiological toxicity mechanisms for chlordane and other cyclodiene pesticides been described in any detail. These toxicity studies have implicated central nervous system effects as a primary impact of acute exposure. Stickel *et al.* (1979) found measurable biochemical changes in brain tissues of birds that died following exposure to chlordane. Eldefrawi and Eldefrawi (1987) identified significant inhibitory effects of cyclodienes on proper functioning of chloride channel proteins than regulate chloride ion transport across neurological cell membranes. Consequences of chronic exposures are typically more difficult to assess, and information on mechanisms of toxicity is not abundant. Indian researchers, such as Mishra and Srivastava (1984) and Radhaiah, Girija, and Rao (1987) have studied the biochemistry of sublethal chlordane exposures in freshwater fish species, and their results suggested a number of broad systemic impacts, including inhibition of carbohydrate metabolism and altered osmoregulatory function. However, the extent to which these changes were due to generalized stress on the fish, or were specific for chlordane intoxication, was not made clear.

There is perhaps a less pressing apparent need for these kinds of data with the virtual elimination of chlordane use in much of the world. However, the continued use of chlordane compounds in some nations, the long environmental lifespan associated with such compounds, and the measurable presence of residues in the environment suggest that chlordane will be a part of environmental assessments for some time into the future. Results such as those of Pyysalo, Wickström, and Litmanen (1981) illustrated that even in countries where chlordane use has not been permitted and should therefore have been negligible, many resident fish contained readily quantifiable concentrations. Better knowledge of mechanisms of toxicity would facilitate a better evaluation of the significance of levels found in the environment and the need for further restriction of remaining uses.

A relatively thorough discussion of the abiotic degradation pathways for major chlordane compounds was included in the review document produced by the NRCC (1974). Related to the need for information on toxicology described above is the need for more information on biochemistry, *i.e.*, how chlordane compounds are metabolized in individual species and ecosystems. Acute toxicity studies that evaluated several different species, and analyses of ratios of chlordane compounds found in bivalves, fish, birds, and mammals indicate that distinct differences exist in the ways in which organisms process chlordane. These result in widely varying degrees of acute toxicity and in distinct tissue residue patterns.

While Menzie (1978) reviewed both abiotic degradation and metabolism of chlordane compounds in several animal species, more detailed investigations in aquatic systems are needed. This would enable more meaningful assessments of the levels of chlordane compounds and metabolites measured in the environment, and provide better direction as to the most important compounds to target in environmental studies. The work of Pyysalo, Wickström, and Litmanen (1981) suggested that chlordane compounds were more easily metabolized than other persistent organic contaminants, but more focused research would help to guide monitoring, assessment, and research activities related to chlordane contamination of the coastal environment.

Beyond collection of fate and effects data for chlordane in the coastal and estuarine environment, more complete information on distribution and use of chlordane worldwide would be helpful in judging the importance of continuing its role as a contaminant of concern. The reliability of such information collected

among states in the United States has, at best, been marginal; in other nations, particularly in many developing countries, this information has been effectively nonexistent, despite the efforts of organizations affiliated with the United Nations. Use data would be valuable in determining the extent and effects of long-range transport of residues and exposure of migratory wildlife. From the perspective of impacts on the U.S. coastal environment, atmospheric transport and deposition may be one of the most important mechanisms for introduction of anthropogenic compounds into "pristine" areas, while migratory fish, birds, and mammals frequenting the U.S. coast during part of the year may be exposed to potentially harmful concentrations elsewhere in their ranges at other times.

Although one manufacturer, the Velsicol Chemical Corporation, has produced and distributed nearly all of the chlordane used worldwide and would be an obvious source of reliable information, the proprietary nature of such data make it unlikely that they would be made directly available for general examination. Alternative sources could include governmental agricultural or port authorities, which oversee the regulatory and export aspects of pesticide usage. These data are critical for understanding contamination processes that extend beyond arbitrary borders, and represent important basic inputs for any global or large-scale modeling efforts that may be undertaken.

Human Health Implications of Chlordane Contamination.

Environmental contamination is often discussed in terms that are not easily comprehended at either literal or intuitive levels. The meaning and significance of residue concentrations measured in units of parts per billion or less are not well understood by environmental toxicologists, much less the general public, and health officials are continually faced with the complex task of determining what is safe and what is not in the face of great uncertainty.

While much of the emphasis in this manuscript has focused on reporting of concentrations measured in various substrates, much less discussion was directed toward human health implications. Comparisons to U.S. FDA action levels were made because those concentrations, despite the fact they are administrative guidelines and not codified, are familiar reference points and are most often used to define acceptable contaminant levels in food items. However, the shortcomings of reliance on FDA action levels are more than a few: for example, the action level for chlordane of 0.3 ppm wet weight, is a benchmark that has endured since it was established in 1955 as a tolerance for residues on fruit and vegetable crops. Although FDA requested an appropriate action level for unavoidable residues of chlordane in fish from the EPA in 1979, the resultant response from EPA in 1979 left the action level at 0.3 ppm. This recommendation was not based entirely upon technical information; it was described as "...a reasonable compromise with economic impact while still maintaining a reasonable level of public safeguard against adverse effects..." (Johnson, 1979).

As has been discussed previously, many uncertainties also exist with more contemporary assessments of risk, as many inherent assumptions are incorporated into the risk level calculation. Nevertheless, refinements both in toxicological assessment and in risk assessment should presumably improve the reliability of current risk estimates.

An interesting comparison and combination of FDA action level concentrations and more recent cancer risk calculations was presented by Connor (1983) in a paper presented at an ocean disposal symposium. Using carcinogenic potencies from the U.S. EPA's Carcinogen Assessment Group, Connor calculated the expected cancer risk that would result from daily ingestion of fish containing contaminants at the action level. Conversely, he also back-calculated what an action level would be to present a lifetime cancer risk of 10^{-5}, a level proposed by EPA in setting water quality criteria. Results for five compounds, including chlordane, are shown below in Table 122.

Table 122. Lifetime cancer risks from fish consumption containing contaminants at the FDA action level and at current levels of contamination. Source: Connor (1983).

Compound	Carcinogenic Potency kd d mg^{-1}	Action Level ppm wet wt.	Cancer Risk* 10^{-6}	Daily Intake µg/day	Cancer Risk* 10^{-6}	Level for 10^{-5} Risk ppb wet wt.
Chlordane	1.6	0.3	45	0.06	1	67
ΣPCB	4.3	2	800	3.6	230	25
ΣDDT	8.4	5	3800	1.5	180	13
Dieldrin	30	0.3	830	0.067	29	3.6
HCB	1.7	0.6	1000	0.03	1	63

*Cancer risk estimates calculated assuming 6.5g/day consumption of fish contaminated at action levels.

In the case of chlordane, the action level that would result in a 10^{-5} level of risk is less than one fourth of the existing FDA action level. However, whether this is more "acceptable" than the $4.5x10^{-5}$ risk that the current action level yields is a question that extends beyond chemistry and toxicology. The notion of "acceptable risk" is a complex one, dependent on a number of factors. Klaassen (1986) listed some of the factors considered in establishing acceptable risk levels, and these included, as beneficial aspects of use of the chemical: economic growth, employment, increased standard of living, increased quality of life, and taxes generated. Detrimental aspects included decreased quality of life, emotional difficulties, health effects, lawsuits, loss of environmental resources, loss of work, and medical payments. It should be apparent that determining acceptable levels of risk is not a clearly defined process.

In the case of chlordane exposure resulting from ingestion of chlordane-contaminated fish, a summary table assembled by Connor (1983) indicated that this dietary route is a relatively minor contribution to total exposure from food items: chlordane exposure in µg/person/day was estimated to be 0.314, while that attributable to fish fell into the range of 0.05-0.10, or approximately 24 percent of the daily total dietary exposure.

This contribution from consumption of fish to *total* chlordane exposure is further diminished when other non-dietary exposure routes are factored in. In particular, much of the recent restrictive regulatory actions involving chlordane resulted from concerns about chronic airborne and respiratory exposure to people living in homes treated for termite prevention, and studies such as those of Fenske and Sternbach (1987) and Louis and Kisselbach (1987) demonstrated that measurable airborne concentrations of chlordane could be found in a large percentage of homes sampled. Fenske and Sternbach found that 74 percent of 133 sampled homes contained detectable (≥0.2 $\mu g/m^3$) chlordane in living spaces, while 88% of 82 sampled homes had detectable concentrations in non-living spaces (*i.e.*, crawl spaces and basements). Louis and Kisselbach measured concentrations following standard treatment for termite control, and determined that one year after treatment, 47 percent of 32 home living spaces contained detectable (≥0.09 $\mu g/m^3$) chlordane (maximum reported concentration 0.81 $\mu g/m^3$) and 81 percent contained detectable (≥0.01 $\mu g/m^3$) heptachlor (maximum concentration 0.57 $\mu g/m^3$). In non-living areas, 92 percent of 13 homes contained chlordane (maximum 1.96 $\mu g/m^3$) and 100 percent contained heptachlor (maximum 3.89 $\mu g/m^3$). Although risk incurred from this route was not discussed in either article, the continuous, low-level nature of the exposure would likely be significant relative to periodic exposures through ingestion of residues in fish.

Chlordane Contamination in a Broader Context.

It has become increasingly more apparent that local, regional, or even national perspectives on environmental problems constitute incomplete evaluative frameworks. Just as fish and other living resources do not respect national boundaries, neither do polluting compounds. This is especially true for many of the chlorinated organic compounds, which tend to readily vaporize and/or adhere to suspended particulate matter in air or water, facilitating long-range transport to virtually any portion of the globe. Studies cited previously in this report confirm that chlordane compounds have been found in the most remote parts of the world: the arctic (Gregor and Gummer, 1989; Ronald *et al.*, 1984; Muir, Norstrom, and Simon, 1988; Bidleman *et al.*, in press; Norstrom *et al.*, 1988) and Antarctic (Kawano *et al.*, 1985;

Kawano *et al.*, 1988; Hidaka *et al.*, 1984). Having been introduced there, they have subsequently been incorporated into the relatively isolated and fragile ecosystems that exist in those environments.

While the presence of chlordanes and other chlorinated hydrocarbons of human origin has been well-documented in nearly every region of the world, the more difficult "so-what" question--that is, the question of significant effects on viability and robustness of ecosystems--has been much less frequently addressed. Patton *et al.* (1989) suggested that because trace pollutants quickly move through the few species comprising food chains in such environments as polar regions, fragile ecosystems, as well as native people who rely on marine animals for food, could be at risk. There are inherent difficulties in performing broader assessments of contaminant effects in biological systems (for example, determining meaningful indicators and measurements, evaluating synergistic effects of several contaminants) anywhere, but in remote environments hostile to human presence, these difficulties are compounded greatly. Among other considerations, simple logistics involving physical acquisition of data become much more problematic. However, to answer the "so-what" question, it will be necessary to confront the attendant problems during design and implementation of studies focusing on environmental contamination.

It is clear that chlordane use has been effectively eliminated among the developed nations of the world. However, this is probably not the case in developing regions. Kohn (1987), in a relevant set of observations published by the American Chemical Society, noted the different set of attitudes that necessarily prevail in much of the Third World. In such regions, higher crop yields resulting from pesticide use can mean the difference between death by starvation, and a minimal level of subsistence; and while environmental problems are recognized, requirements for human sustenance understandably take precedence. Also recognized is that fact newer, less environmentally harmful chemicals developed to replace the organochlorine pesticides are almost always more expensive and more complex to synthesize, and developing nations do not possess the facilities to manufacture the compounds and additionally lack the hard currency to pay for them. As a result, Third World countries either must import low-cost pesticides or manufacture the cheaper, less complex, but more persistent organochlorines. Kohn commented that use of pesticides that contaminate the environment in these situations is not a consequence of ignorance or stupidity. Rather, Kohn observed, it is a result of poverty.

It would be helpful to temper consideration of the continuing use of compounds such as chlordane in developing nations with these reflections. Recent media articles (*e.g.*, *Seattle Post-Intelligencer*, 1989) have detailed reports by environmental advocacy groups that chlordane and heptachlor are currently being exported for use to Third World nations by the sole U.S. manufacturer, despite the nearly total restriction of use in this country. This is likely to be true, although it is difficult to confirm, as such proprietary export figures are not made available to the general public. However, the situation is not only a matter of the ethical propriety involved in selling a compound with known adverse human and environmental consequences to a willing buyer, and it goes beyond the narrowly focused possibility of residues of banned substances being imported back into this country on fruits and vegetables. Just as the chlordane contamination in the U.S. coastal environment is more meaningfully evaluated within a global context, these examples of global environmental contamination are most constructively viewed within the larger framework of world economics and international relations.

Chlordane use clearly has direct and indirect costs. In the United States it took nearly 40 years from initial identification of potentially adverse environmental impacts to virtual elimination of use. While many other nations either have also restricted or eliminated use or are in the process of doing so, for many nations the environmental costs do not yet outweigh economic benefits. The documented past patterns of heavy chlordane use, the likely current (if relatively minor) application in poorer nations, and the characteristic long life in the environment suggest that chlordane compounds will be found in the marine environment of the United States and the rest of the world for many years to come.

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APPENDIX A

Common and Scientific Names of Organisms Discussed in Text

Common Name	Genus and Species
Adelie penguin	*Pygoscelis adeliae*
Albacore tuna	*Thunnus alalunga*
Alewife	*Alosa pseudoharengus*
Alpine char	*Salvelinus alpinus*
American avocet	*Recurvirostra americana*
American crayfish	*Procambarus clarkii*
American eel	*Anguilla rostrata*
American shad	*Alosa sapidissima*
Amphipods	*Anoxy sarsi, Tmetonyx cicada*
Antarctic cod	*Trematomus bernacchii*
Arctic cod	*Boreogadus saida*
Asiatic clam	*Corbicula fluminea*
Atlantic croaker	*Micropogon undulatus*
Atlantic mackerel	*Scomber scombus*
Atlantic menhaden	*Brevoortia tyrannus*
Atlantic rangia	*Rangia cuneata*
Atlantic salmon	*Salmo salar*
Atlantic silversides	*Menidia menidia*
Atlantic tomcod	*Microgadus tomcod*
Awa awa	*Elops hawaiiensis*
Bald eagle	*Haliaetus leucocephalus*
Ballan wrasse	*Labrus berggylta*
Barbel	*Barbus barbus*
Barred sand bass	*Paralabrax nebulifer*
Bay scallop	*Aequipecten irradians*
Bigmouth buffalo	Ictiobus cyprinellus
Black buffalo	*Ictiobus niger*
Black crappie	*Pomoxis nigromaculatus*
Black drum	*Pogonias cromis*
Black oystercatcher	*Haematopus bachmani*
Blue catfish	Ictalurus furcatus
Blue claw crab	*Thalamita crenata*
Blue crab	*Callinectes sapidus*
Blueback herring	*Alosa aestivalis*
Black surf perch	*Embiotoca jacksoni*
Blue mussel	*Mytilus edulis*
Blue rockfish	*Sebastes mystinus*
Bluefish	*Pomatomus saltatrix*
Bluegill	*Lepomis macrochirus*
Bobwhite quail	*Colinus virginianus*
Bocaccio	*Sebastes paucispinis*
Bonito	*Sarda chiliensis*
Brandt's cormorant	*Phalacrocorax penicilliatus*
Brown bullhead	*Ictulurus nebulosus*
Buffalo sculpin	*Enophrys bison*
C-O sole	*Pleuronichthys coenosus*
California mussel	*Mytilus californianus*
California rattail	*Nezumia stegidolepis*
California scorpionfish	*Scorpaena guttata*

Common Name	Genus and Species
Catfish	*Anarhichas lupus*
Central mudminnow	*Umbra limi*
Channel catfish	*Ictulurus punctatus*
Char	*Salvelinus fontinalis*
Chinook salmon	*Oncorhynchus tshawytscha*
Chum salmon	*Oncorhynchus keta*
Clam	*Tapis phillipinarium*
Clam	*Venus gallinae*
Clams	*Macoma carlottensis, M. nasuta, Acila castrensis*
Cockle	*Clinocardium nuttallii*
Cod	*Gadus morhua*
Common carp	*Cyprinus carpio*
Common murre	*Uria aalge*
Common tern	*Sterna hirundo*
Cowbird	*Molothrus ater*
Crab	*Cancer magister, C. gracilis, C. productus*
Crangonid shrimp	*Crangon septemspinosa*
Dall's porpoise	*Phocoenoides dalli*
Diamond turbot	*Hypsopsetta guttulata*
(Dinoflagellate)	*Exuviella baltica*
(Dinoflagellate)	*Ptychodiscus brevis*
Dolly Varden	*Salvelinus malma*
Double-crested comorant	*Phalacrocorax auritus*
Dover sole	*Microstomus pacificus*
Dungeness crab	*Cancer magister*
Eastern oyster	*Crassostrea virginica*
Eel	*Anguilla anguilla*
English sole	*Parophrys vetulus*
Euphausiids	*Euphausia pacifica*
Fathead minnow	*Pimephales promelas*
Flat bullhead	*Ictalurus platycephalus*
Flathead sole	*Hippoglossoides elassodon*
Flounder	*Platichthys flesus*
Fork-tailed storm petrel	*Oceanodroma furcata*
Four-horn sculpin	*Myoxocephalus quadricornis*
(Freshwater bivalve)	*Lamellidens marginalis*
(Freshwater fish)	*Zacco platypus*
Frog	*Rana perezi*
Gar	*Lepisoteus* Sp.
Gizzard shad	*Konosirus punctatus*
Gobyfish	*Acanthogobius flavimanus*
Goldfish	*Carassius auratus*
Grackle	*Quiscalus quiscula*
Grass shrimp	*Palaemonetes pugio*
Gray seal	*Halichoerus gryphus*
Gray whale	*Eschrichtius robustus*
Great cormorant	*Phalacrocorax carbo*
Great horned owl	*Bubo virginianus*
Grey mullet	*Mugil capito*
Guillemot	*Uria aalge*
Guppy	*Lebistes reticulatus*
Haddock	*Melanogrammus aeglefinus*

Common Name	Genus and Species
Hake	*Merluccius productus*
Harbor porpoise	*Phocoena phocoena*
Harbor seal	*Phoca vitulina; Phoca vitulina richardii*
Hard-shell clam	*Mercenaria mercenaria*
Harp seal	*Phoca groenlandica*
Hawaiian crab	*Podophthalmus vigil*
(Hawaiian oyster-*Hoolepe*)	*Ostrea sandvicensis*
Herring	*Clupea harengus*
Herring gull	*Larus argentatus*
Hornyhead turbot	*Pleuronectes verticalis*
Humpback whale	*Megaptera novaeangliae*
(Indian catfish)	*Heteropneustes fossilis*
(Isopod)	*Limnoria tripunctata*
(Isopod)	*Sphaeroma terebransi*
Jack mackerel	*Trachurus symmetricus*
Kelp bass	*Paralabrax clathratus*
Killifish	*Fundulus* (?) *sp.*
Krill	*Euphausia superba*
Largemouth bass	*Micropterus salmoides*
Leach's storm petrel	*Oceanodroma leucorhoa*
Lemon sole	*Microstomus kitt*
Ling cod	*Ophiodon elongatus*
Lizardfish	*Synodus lucioceps*
Lobster	*Homarus americanus*
Long-billed dowitcher	*Limnodromus scolopaceus*
Long-spined thornyhead	*Sebastolobus altivelus*
Longhorn sculpin	*Myoxocephalus octodecemspinosus*
Mackerel	*Scomber scombrus*
Macoma	*Macoma sp.*
Marine worms	*Glycera capitata, Capitella capitata, Prionospio pinnata*
Mullet	*Mugil cephalus*
Mussel	*Brachiodontes recurvus*
Mussel	*Mytilus galloprovincialis*
Night smelt	*Spirinchus starksi*
Northern horse mussel	*Modiolus modiolus*
Northern redhorse sucker	*Moxostoma macrolepidotum*
Ocean whitefish	*Caulolatilus princeps*
Olive rockfish	*Sebastes serranoides*
Olympia oyster	*Ostrea lurida*
Opihi	*Cellana exarata*
Oyster	*Crassostrea corteziensis*
Oyster	*Ostrea edulis*
Pacific cod	*Gadus macrocephalus*
Pacific hake	*Merluccius productus*
Pacific mackerel	*Pneumatophorus diego*
Pacific oyster	*Crassostrea gigas*
Pacific staghorn sculpin	*Leptocottus armatus*
Pacific tomcod	*Microgadus proximus*
Pacific whiting	*Merluccius productus*
Pelagic cormorant	*Phalacrocorax pelagicus*
Perch	*Perca fluviatilis*
(Phytoplankton)	(natural population)
Pigeon guillemot	*Cepphus columba*

Common Name	Genus and Species
Pike	*Esox lucius*
Pile perch	*Rhacochilus vacca*
Pilot whale	*Globicephala melaena*
Pinfish	*Lagodon rhomboides*
Pink salmon	*Oncorhynchus gorbuscha*
Pink shrimp	*Pandalus jordani*
Pink shrimp	*Penaeus duorarum*
(Plankton)	
(Plankton, freshwater)	*Diaptomus forbesi*
Polar bear	*Ursus maritimus*
(Polychaete worm)	
(Polychaete worm)	*Nereis virens*
Pumpkinseed sunfish	*Lepomis gibbosus*
Quillback rockfish	*Sebastes maliger*
Rainbow trout	*Oncorhynchus mykiss*
Red drum	*Sciaenops ocellatus*
Red snapper	*Lutjanus campechanus*
Red-winged blackbird	*Agelaius phoeniceus*
Redbreast sunfish	*Lepomis auritus*
Redfish	*Sebastes marinus*
Redtail surfperch	*Amphistichus rhodoterus*
Ribbed mussel	*Brachidontes demissus plicatulus*
Ribbed mussel	*Modiolus demissus*
Ringed seal	*Phoca hispida*
River carpsucker	*Carpiodes carpio*
Roach	*Rutilus rutilus*
Rock crab	*Cancer antennarius*
Rock sole	*Lepidopsetta bilineata*
Rockfish	*Sebastes sp.*
Rosy rockfish	*Sebastes rosaceus*
Round stingray	*Urolophus halleri*
Sablefish	*Anoplopoma fimbria*
Saimaa ringed seal	*Phoca hispida saimensis*
Sand bass	*Paralabrax nebulifer*
Sanddab	*Citharichthys sp.*
Sardine	*Sardinops caerulea*
Sea bass	*Lateolaborax japonicus*
Sea catfish	*Arius felis*
Sea scallop	*Placopecten magellanicus*
Sea scorpion	*Myoxocephalus scorpius*
Sheepshead	*Archosargus probatocephalus*
Sheepshead minnow	*Cyprinodon variegatus*
Shiner perch	*Amphistichus rhodoterus*
(Shipworm)	*Teredo spp.*
(Shipworm)	*Bankia spp.*
Short-necked clam	*Tapes philippinarum*
Short-spined thornyhead	*Sebastolobus alascanus*
Shrimp	*Pandalus danae, P. jordani, Crangon alaskensis*
Silver salmon	*Oncorhynchus kisutch*
Smallmouth buffalo	*Ictiobus bubalus*
Snowy plover	*Charadrius alexandrinus*
Sockeye salmon	*Oncorhynchus nerka*
Soft-shell clam	*Mya arenaria*

Common Name	Genus and Species
Southern flounder	*Paralichthys lethostigma*
Southern kingfish	*Menticirrhus americanus*
Spanish mackerel	*Trachurus symmetricus*
Spiny dogfish	*Squalus acanthias*
Spot	*Leiostomus xanthurus*
Spotted drum	*Equetus punctatus*
Spotted seatrout	*Cynoscion nebulosus*
Sprat	*Sprattus sprattus*
Squid	*Gonatopsis borealis*
Starling	*Sturnus vulgaris*
Starry flounder	*Platichthys stellatus*
Starry rockfish	*Sebastes constellatus*
Striate piddock	*Martesia striata*
Striped bass	*Morone saxatilis*
Striped mullet	*Mugil cephalus*
Striped surfperch	*Embiotoca lateralis*
Summer flounder	*Paralichthys dentatus*
Surf clam	*Spisula solidissima*
Surf smelt	*Hypomesus pretiosus*
Tautog	*Tautoga onitis*
Thick-billed murre	*Uria lomvia*
Tilapia	*Tilapia sp.*
Tilapia	*Tilapia mossambica*
Tilapia	*Tilapia nilotica*
Treefish	*Sebastes serriceps*
Trout	*Salmo trutta*
Tufted puffin	*Lunda cirrhata*
Vendace	*Coregonus albula*
Vermillion rockfish	*Sebastes miniatus*
Walleye pollock	*Theragra chalcogramma*
Weakfish	*Cynoscion regalis*
Weddell seal	*Leptonychotes weddelli*
Western gull	*Larus occidentalis*
Western sandpiper	*Calidris mauri*
White-beaked dolphin	*Lagenorhynchus albirostris*
White catfish	*Ictalurus catus*
White crab	*Portunus sanguinolentus*
White croaker	*Genyonemus lineatus*
White-faced ibis	*Plegadis chihi*
White perch	*Morone americanus*
White seaperch	*Phanerodon furcatus*
White sturgeon	*Acipenser transmontanus*
White shark	*Carcharodon carcharias*
White sucker	*Castostomus commersoni*
Whitespotted greenling	*Hexagrammos stelleri*
Windowpane flounder	*Scophthalmus aquosus*
Winter flounder	*Pseudopleuronectes americanus*
(Worm)	*Branchiura sowerbyi*
Yellow perch	*Perca flaviscens*
Yellowfin tuna	*Thunnus albacares*
Zooplankton	*Calanus hyperboreus, Metridia langa, Xanthocalanus borealis*

APPENDIX B

Summaries of Analyses in Cited References

Reference	Compounds Analyzed	Matrix	Method	Approx. No. of Analyses
Agrawal (1986)	(not defined)	Mollusc tissues	Referenced, not specified	(not specified)
Akazawa (1981)	α- & γ-chlordane, *trans*-nonachlor	Sediment, crab, fish	(not specified)	~287
Allender (1989)	(not defined)	Swabs from grain storage areas	GLC	1470
Anas (1974)	Unspecified chlordane, heptachlor	Harbor seal blubber	GLC; TLC	13
Andersson *et al.* (1988)	*cis*-, *trans*-, oxychlordane *trans*-nonachlor	Fish muscle	GC w/ECD	54
Ang, Meleady, & Wallace (1989)	(not defined)	Drinking water supplies, Australia	GC w/ECD	659
Belton *et al.* (1983)	α- & γ-chlordane	Bluefish muscle	GC	239
Belton *et al.* (1985a)	α- & γ-chlordane	Lobster tissues	(not specified)	10
Bevenue *et al.* (1972)	(not defined)	Water, sediment	GC w/ECD	11
Bidleman *et al.* (1981)	*cis*- & *trans*-chlordane	Marine air samples	GC w/ECD	29
Bidleman *et al.* (in press)	α- & γ-chlordane; α- & γ-nonachlor	Air, snow, seawater, zooplankton, amphipods	GC w/ECD	(not specified)
Biggs *et al.* (1978)	60% octachloro-4,7-methanotetrahydroindane; 40% "related insecticidal compounds"	Estuarine phytoplankton	Particle concentration; carbon uptake; chlorophyll *a*	279
Bopp *et al.* (1982)	α- & γ-chlordane (*trans*-nonachlor not resolved)	Sediments	GC w/ECD	32
Bugg, Higgins & Robertson (1967)	1,2,4,5,6,7,8,8-octachloro-3*a*,4,7,7*a*-tetrahydro-4,7-methanoindane	Oysters	GC w/ECD	132
Butler (unpublished)	(not defined)	Estuarine fish	GC w/ECD	561
Butler (1973)	(not defined)	Bivalve mollusks	GC w/ECD	8095
Butler, Kennedy, & Schutzmann (1978)	(not defined)	Bivalve mollusks	GC w/ECD	178
Butler & Schutzmann (1978)	(not defined in paper; see discussion)	Estuarine fish	GC w/ECD	1524

Reference	Compounds Analyzed	Matrix	Method	Approx. No. of Analyses
Cáceres, Tundisi, & Castellan (1980)	Chlordane, not defined; heptachlor	Water	GC w/ECD	6
Casper (1967)	1,2,4,5,6,7,8,8-octachloro-3*a*,4,7,7*a*-tetrahydro-4,7-methanoindane	Water, oysters	GC w/ECD	15
Chesmore, Brown, & Anderson (1972)	Heptachlor	Mud, clams	GC	64
City of SF and CH2M Hill (1984)	α-, γ-, oxychlordane; *cis*-, *trans*-nonachlor	Sediment, fish, mussels	GC, MS	27
Claeys *et al.* (1975)	(not defined)	Shrimp, fish bivalves, euphausiids	GLC w/ECD & microcoulometric detector	114
Collins, Robinson, and Sinclair (1989)	α-, γ-chlordane, *cis*-, *trans*-nonachlor	Fish muscle	GC	72
Crecelius, Woodruff, & Myers (1989)	α-chlordane, heptachlor	Sediment, fish tissues	HPLC w/UVD, GCw/ECD, or GC w/NPD	56
Custer & Mitchell (1989)	*cis*-, oxychlordane; *cis*-, *trans*-nonachlor	Bird eggs	Referenced, not detailed	20
Duke & Wilson (1971)	(not defined)	Fish livers	GC w/ECD	73
Eisenberg & Topping (1981, 1984, 1985)	(not defined), heptachlor	Shellfish & finfish	GC w/ECD	1485
Fenske & Sternbach (1987)	(not defined)	Indoor air	GC	300
Fiske, Watson, & Coates (1966)	Heptachlor	Clams	(not specified)	(not specified)
Foehrenbach, Mahmood, & Sullivan (1971)	Heptachlor	Shellfish	GC w/ECD	210
Fuhrer (1984)	Chlordane (not defined), heptachlor	Water, sediment, elutriates	Referenced, not explicitly described	9
Fuhrer & Rinella (1983)	Chlordane (not defined), heptachlor	Water, sediment, elutriates	Referenced, not explicitly described	60
Gahler *et al.* (1982)	Chlordane (not defined), heptachlor	Edible tissues of fish	GC w/ECD	92
Gardner & Pruell (1988)	α- & γ-chlordane	Sediment, clam, oysters, lobster, fish	GC w/ECD	91
Garreis & Murphy (1986a)	*cis*-, *trans*-, oxychlordane, chlordene, nonachlor	Fish muscle	GLC	749

Reference	Compounds Analyzed	Matrix	Method	Approx. No. of Analyses
Garreis & Murphy (1986b)	(not defined)	Crab tissue	GLC	23
Gaskin, Frank, & Holdrinet (1983)	*cis*-, *trans*-, oxychlordane	Porpoise tissues	GC	18
Geraci (1989)	*trans*-nonachlor	Dolphin, whale tissues	GC w/ECD	83+
Gregor & Gummer (1989)	*cis*- & *trans*-chlordane	Snow	GC w/ECD	22
Hawaii Department of Health (1978)	α- & γ-chlordane, *trans*-nonachlor	Sediment, clam, crab, & fish tissue	(not specified)	118
Helle *et al.* (1983)	α-, γ- & oxychlordane, *trans*-nonachlor, heptachlor	Seal tissues	GLC/SIM, GLC w/ECD	14
Henderson, Pickering, & Tarzwell (1959)	1,2,4,5,6,7,8,8-octachloro-2,3,3*a*,4,7,7*a*-hexahydro-4,7-methanoindene	Fish bioassay	24-, 48-, 96-hr. LC_{50}	50
Henny, Blus, & Prouty (1982)	Unspecified; *cis*- & *trans*-nonachlor, oxychlordane mentioned	Seabird eggs	GLC w/ECD	63
Hidaka *et al.* (1984)	"Several" compounds & metabolites, not defined	Seawater, krill, fish, seals	GC/MS, selected ion monitor	(not specified)
Hirai & Tomokuni (1989)	1,2,4,5,6,7,8,8-octachloro-3*a*,4,7,7*a*-tetrahydro-4,7-methanoindane	River water, sediment	GC/MS	89
Iwanowicz, Anderson, & Ketschke (1973)	Heptachlor	Mud, clam, flounder	GC w/ECD	93
Iwanowicz, Anderson, & Ketschke (1974)	Heptachlor	Mud, clam, flounder	GC	90
Jansson *et al.* (1979)	α-, γ-, oxychlordane, *trans*-nonachlor	Fish, birds, seals	HRGC, GC/MS	(not specified)
Jerome *et al.* (1965)	Heptachlor	Clam	GC	14
Jerome, Chesmore, & Anderson (1966)	Heptachlor	Clam, fish	GC	14
Jerome, Chesmore, & Anderson (1968)	Heptachlor	Clam, fish	GC	9
Kapila, Puri, & Orazio (1988)	Not specified; included technical chlordane, heptachlor, chlordenes, metabolites	Soil & water	(not specified)	(not specified)
Kawano *et al.* (1985)	*cis*- & *trans*-chlordane, *cis*- & *trans*-nonachlor	Atmosphere	GC w/ECD, GC/MS	4

Reference	Compounds Analyzed	Matrix	Method	Approx. No. of Analyses
Kawano *et al.* (1988)	*cis*-, *trans*-, oxychlordane, *cis*-, *trans*-nonachlor	Zooplankton, fish, birds, mammals	GC w/ECD, GC/MS	23
Kerkhoff & de Boer (1982)	*trans*-nonachlor, oxychlordane	Seal blubber	GC/MS	(not specified)
Kerkhoff, Otte, & de Boer (1982)	*cis*-, *trans*-, oxychlordane, *trans*-nonachlor	Seals, dolphin, bird, fish	GC w/ECD, GC/MS	21
Kiene & Capone (1984)	chlordane, heptachlor (not defined)	Anaerobic salt marsh sediments	GC w/FID (CH_4); GC w/TCD (CO_2, H_2S)	6
Law & Goerlitz (1974)	(not defined)	Sediment	GC w/ECD, GC/MS	39
Lichtenstein & Polivka (1959)	Chlordane (not defined), heptachlor	Turf application	Referenced, not described	40
Louis & Kisselbach (1987)	(not defined)	Indoor air samples	GC w/ECD	190
Ludke (1976)	Oxychlordane, heptachlor, heptachlor epoxide, nonachlor, compounds "C" & "E", α- & γ-chlordane.	Brain & carcass of quail	GC w/ECD	43
MacLeod *et al.* (1981)	α-chlordane, *trans*-nonachlor, heptachlor	Water, microlayer, sludge, effluent, misc. inverts, fish	GC w/ECD, GC/MS	244
Malins *et al.* (1980, 1981a, 1981b, 1982)	α-chlordane, *trans*-nonachlor, heptachlor	Sediment, invertebrates, fish	GC w/ECD, GC/MS	191
Malins *et al.* (1987)	α-chlordane, *trans*-nonachlor	White croaker tissues	GC w/ECD, GC/MS	18
Malins, Brown, & Chan (unpublished)	α-chlordane, *trans*-nonachlor, heptachlor	Whale tissues	GC w/ECD, GC/MS	6
Magnani *et al.* (1978)	60% octachloro-4,7-methanotetrahydroindane; 74% 1,4,5,6,7,8,8*a*-heptachloro-3*a*,4,7*a*-tetrahydro-4,7-methanoindene	Dinoflagellate	Optical cell counts; fluorometric chlorophyll *a*; photosynthesis; cell size distribution	84
Mani & Konar (1986)	Termex (20% chlordane active ingredient)	Plankton, worm, fish	LC_5, LC_{50}, LC_{95}	9
Marion, Barker, Settine (1987)	Heptachlor	Oyster & clam tissue	GC/MS	(not specified)
Martin & Gutierrez-Galindo (1989)	*cis*- & *trans*-chlordane; *cis*- & *trans*- chlordene; *cis*- & *trans*-nonachlor; oxychlordane; heptachlor	Oyster tissue	GC w/ECD	110

Reference	Compounds Analyzed	Matrix	Method	Approx. No. of Analyses
Mathews (unpublished)	α-, γ-chlordane; α-chlordene	Sediment & oyster	GC w/ECD	(total not specified)
McLeese & Metcalfe (1980)	Technical chlordane	Shrimp bioassay	96 hr LC_{50}, LT_{50}	(not specified)
McLeese, Burridge, & Van Dinter (1982)	Technical chlordane	Polychaete bioassay	96 hr LC_{50}	(not specified)
Mehrle *et al.* (1982)	1,2,4,5,6,7,8,8-octachlor-2,3,3*a*,4,7,7*a*-hexahydro-4,7-methanoindane	Striped bass	GLC	600
Mishra & Srivastava (1984)	1,2,4,5,6,7,8,8-octachloro-2,3,3*a*,4,7,7*a*-hexahydro-4,7-methanoindene	Catfish bioassay	LC_{50}, blood & tissue chemistry	(not specified)
Miyazaki *et al.* (1980)	*cis*-, *trans*-, oxychlordane, *cis*-, *trans*-nonachlor, heptachlor epoxide	Fish tissue	GC w/ECD, GC/MS	(not indicated)
Moilanen *et al.* (1982)	*cis*-, *trans*-, oxychlordane, *trans*-nonachlor	Fish tissue	HRGLC w/SIM MS	38
Muir, Norstrom & Simon (1988)	(12 chlordane-related)	Fish, marine mammals	GC w/ECD	176
Munson (1972)	α- & γ-chlordane bioassay	Clam & oyster	GC, TLC, MS specified)	(not
Munson (1975)	α- & γ-chlordane	Water, suspended sediment, zoo-plankton, shellfish	GLC w/ECD	394
Murphy (1988a)	*cis*-, *trans*-, oxychlordane, chlordene, nonachlor, heptachlor	Striped bass fillets	GLC	35
Murphy (1988b)	*cis*-, *trans*-, oxychlordane, chlordene, nonachlor	Bluefish fillets	GLC	71
Murphy (1989)	(not defined)	Oysters & clams	(not specified)	886
Nisbet & Reynolds (1984)	α-, γ-, oxychlordane	Tern eggs	GLC w/ECD	128
NOAA, FDA, & EPA (1986)	*cis*-, *trans*-chlordane; *trans*-nonachlor	Bluefish fillets	GLC w/ECD	690
Norstrom *et al.* (1988)	(7 chlordane-related)	Polar bear tissues	HRGC w/ECD, HRGC-MS	141
Parrish *et al.* (1976)	1,2,4,5,6,7,8,8-octachloro-3*a*,4,7,7*a*-tetrahydro-4,7-methanoindane	Oyster, shrimp, fish bioassay	GC w/ECD, MS	

Reference	Compounds Analyzed	Matrix	Method	Approx. No. of Analyses
Patton *et al.* (1989)	*cis*-, *trans*-chlordane, *cis*-, *trans*-nonachlor	Air, snow, surface seawater	GC w/ECD, GC/MS	25
Peyton, Anderson, & Gantzer (1988)	"chlordane", *cis*- & *trans*-chlordane, heptachlor	Soil, leachate, runoff	GC w/ECD	(not specified)
Pyysalo, Wickström, & Litmanen (1981)	*cis*-, *trans*-chlordane, heptachlor, chlordenes, *trans*-nonachlor	Fish tissues; detoxification enzyme systems	GC/MS	48
Radhaiah, Girija, & Rao (1987)	Heptachlor	Blood & kidney parameters	Blood & tissue chemistry	240
Rico *et al.* (1987)	Heptachlor	Crayfish, fish, frog	GC w/ECD	39
Risebrough (1987)	*cis*-, *trans*-nonachlor, α-, γ-, oxychlordane	Sediment, mussels fish, eagle	GC w/ECD, GC/MS	??
Risebrough *et al.* (1983)	α- & γ-chlordane	Bivalves	GC w/ECD, GC/MS	9
Roberts, deFrietas, & Gidney (1977)	*cis*-, *trans*-chlordane	Uptake study in fish	GC w/ECD	57
Robinson & Ryan (1986)	α-chlordane, *trans*-nonachlor, heptachlor	Transplanted mussels	GC w/ECD	7
Romberg *et al.* (1984)	α- & γ-chlordane, heptachlor	Seawater, sediment	GC w/ECD, GC/MS	245
Ronald *et al.* (1984)	α-, β-, oxychlordane	Harp seal tissue	GLC w/ECD	248
Ryan *et al.* (1985)	(not defined)	Water, sediments	Referenced, not explicitly described	42
Saiki & Schmitt (1986)	*cis*- & *trans*-chlordane, *trans*-nonachlor, heptachlor	Whole fish	GC w/ECD	(not specified)
Schimmel, Patrick, Forester (1976a)	Technical heptachlor (65% heptachlor, 22% *trans*-chlordane, 2% *cis*-chlordane, 2% nonachlor); 99.8% heptachlor	Oyster, shrimp, fish bioassay	GC w/ECD, MS	42
Schimmel, Patrick, Forester (1976b)	Technical heptachlor (65% heptachlor, 22% *trans*-chlordane, 2% *cis*-chlordane, 2% nonachlor); 99.8% heptachlor	Spot bioassay	GC w/ECD, MS	(not indicated)
Shute (1981)	Chlordane (not specified)	Oysters and clams	(not specified)	(not specified)

Reference	Compounds Analyzed	Matrix	Method	Approx. No. of Analyses
Skåre *et al.* (1985)	Oxychlordane, *trans*-nonachlor	Fish liver tissue	GLC w/ECD	63
Sloan *et al.* (1988)	Chlordane (not specified), nonachlor (not specified)	Striped bass fillets	(not specified)	1112
Sommerfield & Cummins (1989)	Probably technical chlordane	Freshwater fish fillets	(not specified)	93
Stickel *et al.* (1979)	Technical chlordane, oxychlordane, heptachlor	Birds	GLC, TLC, MS	49
Stoker (1986)	Chlordane (not specified), heptachlor	Water, sediment	Referenced, not explicitly described	(not specified)
Tachikawa *et al.* (1987)	α-, γ-, oxychlordane, *cis*- & *trans*-nonachlor,	Fish tissues	GC w/ECD	78
Tashiro and Matsumura	*cis*-, *trans*-chlordane; 12 metabolites	Rat tissue and waste	GLC w/ECD, TLC, IR, NMR	(not specified)
Tanita *et al.* (1976)	α-chlordane, γ-chlordane	Water, sediment, oysters	GC w/ECD	12 water, 12 sediment, 38 oysters
Vigfusson *et al.* (1983)	43.2% 1,2,4,5,6,7,8,8-octachlor-2,3,3*a*,4,7,7*a*-hexahydro-4,7-methano-indane	Mudminnow	*in vivo* bioassay	59
White, Mitchell, & Kaiser (1983)	Chlordane (not specified)	Shorebirds	GLC, MS	165
White, Mitchell, & Swineford (1984)	Chlordane (not specified)	Bird eggs	Referenced, not specified; MS	205
Wilson & Earley (1986)	α-, γ-, & oxychlordane; heptachlor	Seabird eggs	GC	44
Yamagishi *et al.* (1981)	*cis*-, *trans*-, oxychlordane, *cis*-, *trans*-nonachlor	Seawater, clams, fish	GC w/ECD	14 water, 32 clams, 54 fish
Zitko (1978)	*cis*-, *trans*-nonachlor, *cis*- + *trans*-chlordane	Various aquatic fauna & tissues	GC-MS	38

[1] Chlordane compounds are listed here as they are identified in specified reference. It is recognized that nomenclature may vary from reference to reference; for example, by some designations, α-chlordane, ß-chlordane, and *cis*-chlordane may refer to the same isomer (NRCC, 1974).

[2] ECD = electron capture detectors; GC = gas chromatography; GLC = gas liquid chromatography; HPLC = high performance liquid chromatography; IR = infrared; MS = mass spectrometry; NMR = nuclear magnetic resonance; NPD = nitrogen-phosphorus detector; SIM = selective ion monitoring; TCD = thermal conductivity detector; TLC = thin-layer chromatography; UVD = ultraviolet light detector; HR as a prefix = high resolution.

APPENDIX C

ABBREVIATIONS AND ACRONYMS

ANOVA	analysis of variance
BHC	benzohexachloride
CAS	Chemical Abstract Service
cm	centimeter
CMW	California Mussel Watch
DEC	Department of Environmental Conservation
DNR	Department of Natural Resources
EPA	Environmental Protection Agency
FDA	Food and Drug Administration
FIFRA	Federal Insecticide, Fungicide and Rodenticide Act
FWPCA	Federal Water Pollution Control Administration
g	gram
GABA	γ-aminobutyric acid
GC-MS	gas chromatography-mass spectrometry
GLPC	gas liquid partition chromotography
HCB	hexachlorobenzene
IARC	International Agency for Research on Cancer
IPCS	International Programme for Chemical Safety
kg	kilograms
m	meter
MDE	Maryland Department of the Environment
MESA	Marine EcoSystems Analysis
mg	milligram
MRL	maximum residue limit
mt	metric ton
MWRRC	Missouri Water Resources Research Center
NAS	National Academy of Science
NCBP	National Contaminant Biomonitoring Program
NCI	National Cancer Institute
NJDEP	New Jersey Department of Environmental Protection
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NPMP	National Pesticide Monitoring Program
NRCC	National Research Council of Canada
NSSP	National Shellfish Sanitation Program
NS&T	National Status and Trends
NYDEC	New York Department of Conservation

PCB	polychlorinated biphenyl
ppb	parts per billion
ppm	parts per million
ppt	parts per trillion
RCRA	Resource Conservation and Recovery Act of 1976
SESD	South Essex Sewage District
TPPS	Toxicant Pretreatment Planning Study
USACOE	United States Army Corps of Engineers
USGS	United States Geological Survey
WDOE	Washington Department of Ecology
WHO	World Health Organization