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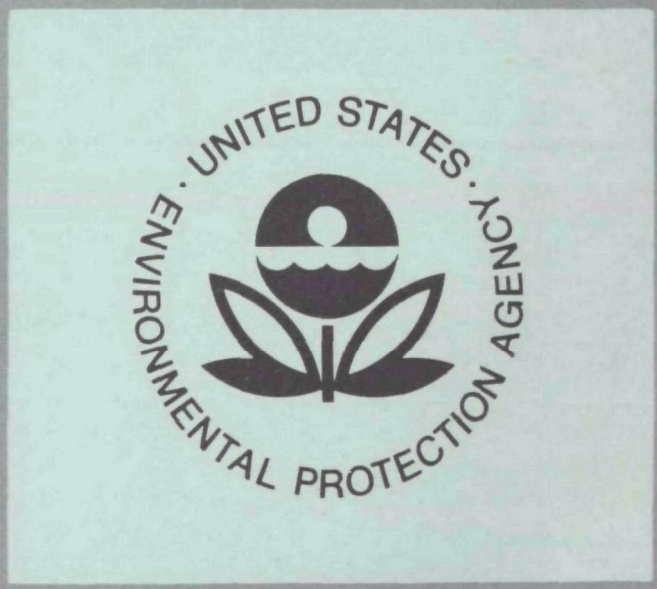
THE ECOLOGICAL IMPACT OF SYNTHETIC ORGANIC COMPOUNDS ON ESTUARINE ECOSYSTEMS



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ABSTRACT

This review and indexed bibliography concerns the presence and effects of pesticides (i.e., insecticides, herbicides, fungicides, etc.) and industrial toxicants in the estuarine ecosystem. The industrial toxicants refer, primarily, to polychlorinated biphenyls, but phthalate esters, polychlorinated terphenyls, chlorinated dibenzodioxins and dibenzofurans are also discussed. The review covers literature of the last decade, with emphasis on the most recent 5 years. However, the 700-plus references in the bibliography span a much wider range. A permuted keyword retrieval system (SPINDEX) is provided to allow practical use of the bibliography by scientists, academicians, and societal decision makers.

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SECTION I

INTRODUCTION

SCOPE AND LIMITATIONS OF THE REVIEW

The objectives of this publication are to provide: (1) a fairly comprehensive state-of-the-art review on the impact of synthetic organic compounds (SOC's) on the estuarine ecosystem, and, (2) a useful (i.e., workable), keyword accessible bibliography dealing with the same subject.

The limitations of such an effort become obvious as soon as one tries to decide exactly where an estuary ends and where the freshwater and marine habitats begin. The vast majority of fin- and shellfishes, which are valuable from a commercial or sport standpoint, require estuarine residence during some critical period of their life cycle. In addition, many organisms (e.g., salmon, trout, lampreys and eels) spend varying periods of time in the estuaries en route to their spawning grounds; a point in their life when sublethal, reproductive effects could be especially critical. For this reason, a broad interpretation of 'estuarine species' was taken and, in several cases, freshwater species, congeneric to their estuarine counterparts, were considered. This was especially the case when little or no data were available for the latter.

Since the objective of this study was a state-of-the-art presentation, emphasis in the review was placed on the literature of the past 5 to 10 years. However, in order to maximize utility, many older references may be found in the BIBLIOGRAPHY (Section VIII) and accessed by subject through the SUBJECT PROFILE INDEX (Section IX).

APPROACH TO THE LITERATURE

Our approach to the literature involved primarily three methods of searching:

1. A core of reprints at the Mote Marine Laboratory (Sarasota Facility) provided initial citations and from these evolved a main list of authors. Abstracts and indexes were consulted continuously throughout the search as they appeared in different locations and from them new references were gleaned. Other relevant resources such as proceedings, transactions, government reports, etc. were also indexed

from these sources. Some of the more productive abstracts and indexes were:

Aquatic Sciences Abstracts: Contains important information oriented toward specific species' problems.

Chemical Abstracts/Index: These report important information regarding the chemical elements involved and additional sources are easily traced.

Commercial Fisheries Abstracts: Current information on specific marine species.

Monthly Catalog (Government Documents): This gives access to government research and recent GPO publications such as bibliographies, indexes concerned with the topic of interest and was extremely important as a secondary source.

Pollution Abstracts: Good source from both chemical and species retrieval.

Science Citation Index: This was an important source for locating authors and published works in the immediate field of interest.

Zoological Record: This proved to be an excellent presentation with taxonomic, topical and geographic indexing; easily handled.

2. Our approach to the literature, however, involved primarily combing the holdings of the major libraries and research institutions in the United States that had some emphasis in the marine area (see Acknowledgement section). The major emphasis was on on-site searching of card catalogs and relevant periodicals which was performed at each location. Both periodical listings and monographs were unique to each location. For example: MBL Library, Woods Hole, is particularly rich in foreign holdings; EPA-Raleigh - industrial health; Univ. of Florida - government documents; UC-Davis - toxicology; NOAA Bureau of Fisheries - fish and fishing; Scripps - oceanography, and; VIMS - coastal development.

3. During the last few months of the study, various data bases were computor searched as a cleanup operation and to check on the completeness of our manual approach. The data bases and searching organizations were as follows:

Organization/System	Data Base
U.S. Dept. Commerce NOAA Environmental Data Service (OASIS) Environmental Science Information Center (D832) Page Bldg. Two 3300 Whitehaven St., NW Washington, D. C. 20235	National Oceanographic Data Center Biological Information Systems (BIRS) Oceanic Index (OI) Toxic Materials (TOXMAT) TOXLINE Selected Water Resources Abstracts (SWRA) National Agricultural Library (CAIN)
U.S. Dept. Commerce National Technical Information Center Springfield, Va. 22151	NTISearch
Environmental Protection Agency Office of Pesticide Programs Technical Services Division Washington, D. C. 20460	On- and Off-Line Data Bases

As secondary references were obtained from computor data bases, during the course of the searching, they were posted along with the other references alphabetically and retrieved wherever the appropriate periodical or book appeared. Since data were still coming in from these bases, providing current up-dated material, a final list was compiled and sent back to the Woods Hole MBL Library for retrieval.

Once the references were obtained, each was read and assigned from 4 to 7 key words using the SPINDEX (Subject Profile Index) system, as suggested by the American Bibliographical Center (Santa Barbara, California). Key words included: subject; taxa; chemicals,and; effects. Geographic location was added in cases of a field study. These key words were then permuted and alphabetized using an IBM 1401 computor and AUTOCODER language. In this form (Section IX), approximately 700 references in the bibliography (Section VIII) become readily available to the user by subject and location.

BACKGROUND

For our purposes here, the term "synthetic organic compounds" refers to manmade compounds and includes, but is not restricted to, pesticides, polychlorinated biphenyls (PCB's), hexachlorobenzene (HCB) and phthalate esters (PAE's) as well as toxic contaminants of some of these, like chlorinated dibenzodioxins and dibenzofurans.

The "estuarine ecosystem" has been variously defined, but for the sake of simplicity it will be considered as that zone of interface where fresh and salt water mix. This estuarine ecosystem serves a vital function in that most marine finfish and shellfish depend on a high quality estuary for some critical portion of their life history (678, 684). In addition, many salmonids, shad and other anadromous fishes spend a variable amount of time in this habitat before ascending the rivers to spawn. The same is true for catadromous species, such as the eel, which migrate from fresh water to the sea to reproduce.

Unfortunately, the oceans are the recipients and ultimate accumulation sites for persistent pollutants like organochlorines (530, 685). In fact, an estimated 25 percent of all DDT applied to the land has found its way to the sea (721). Risebrough and his co-workers (529) indicate that 11 tons of DDT per year is transported down the Mississippi River to the Gulf of Mexico alone. Because of their unique physical and chemical characteristics, estuaries tend to be "toxicant traps". The detritus which forms the base of the estuarine food chain may contain up to 50 ppm total DDT (471), and Woodwell, et al. (656) estimated that total estuarine ecosystem levels as high as 14.7 kg/hectare were possible.

DDT and other synthetic organics are termed "toxic" when, because of their physical or chemical properties, they interfere with normal biological functions. The interference can occur at any level, whether it be as subtle as pesticide-induced decreased growth in oysters or as gross as reproductive failure in bald eagles or mass fish mortality. There are naturally occurring toxic substances, which include such things as the resin from certain plants and the toxin(s) associated with red tide organisms. By far, however, most deleterious substances find their origin with modern day man and his efforts to promote "progress."

A logical breakdown of synthetic organic compounds which are considered in this paper along with available production and/or consumption information follows:

1. Pesticides are chemicals which kill organisms identified as "pests" and include insecticides, fungicides, piscicides, herbicides, miticides, etc. Insecticides are commonly broken down into: (a) organochlorines (chlorinated hydrocarbons), like DDT, aldrin, dieldrin, heptachlor, toxaphene and chlordane; (b) organophosphates, like malathion, parathion, diazinon and guthion; and (c) carbamates, like Sevin and zectran. Fungicides include such things as dithiocarbamates (e.g., ferbam and ziram), nitrogen-containing compounds (e.g., phenylmercuric acetate), triazines, quinones, heterocyclics and inorganics like the heavy metals. Hexachlorobenzene (C_6Cl_6 or HCB) is a fungicide but is, in addition, used in organic synthesis processes. Herbicides are quite varied, with the most common being the phenoxy acids like 2,4-D and 2,4,5-T. Frequently used aquatic herbicides include endothal and diquat, which are often used in combination with a surfactant.

The U. S. production of the major synthetic organic pesticides for 1967 through 1973 is reproduced in Table 1. In 1971, the production of synthetic organic insecticides in the United States climbed nearly 14 percent from the year before, reaching the third highest on record (689). Insecticides accounted for 49 percent of the tonnage of synthetic pesticides produced. As can be seen in Table 1, the trend did not continue but production took a significant leap in 1973 (preliminary data). Although it would have been interesting to follow the trends of specific insecticides, like DDT and parathion, production data were withheld to avoid disclosure. However, the aldrin-toxaphene and organophosphorous groups have shown a continued increase in production since 1970 and 1971, respectively.

Table 2 reveals the domestic disappearance of selected pesticides for the years 1966 through 1973. Except for the aldrin-toxaphene group, and recently, copper sulfate and the 2,4-D group, there is a fairly consistent downward trend. Domestic disappearance of DDT, for instance, was 5.99×10^6 kg in 1971 which was a decrease of approximately 33 percent from 1970. The most obvious decrease in domestic use is reflected by a comparison of the 1972 and 1973 DDT data (1.04×10^7 kg and 4.54×10^5 kg, respectively). Conversely, copper sulfate and the 2,4-D group showed an increase in U. S. use but the aldrin-toxaphene group did not. This is particularly interesting in view of the increased production of the latter group during 1973 over 1972 (approximately 2.27×10^6 kg). In 1973 2.81×10^7 kg of the aldrin-toxaphene group were exported; an increase of 1.45×10^7 kg over 1972 (214).

2. "Industrial Toxicants" is a catchall term that has been variously subdivided. Polychlorinated biphenyls (PCB's) are chlorinated compounds which find use in almost every sector of modern man's world and have recently come under close scrutiny (172, 240, 254, 260, 322, 467, 490, 492, 528, 532, 664). In the past, they have been used in such diverse products as printer's ink to swimming pool paint. Although a voluntary curtailment by Monsanto has restricted their use, a recent renewal of interest has been sparked by PCB's being found in unexplainable parts of the ecosystem and in high levels.

PCB's and PCT's (polychlorinated terphenyls) are produced under the trade name Aroclor® by Monsanto in the United States. PCB production peaked during the period 1967-1970 (Table 3). PCT production shows a similar, but later, production peak during 1970-1971. PCT's are no longer being produced and the manufacture of PCB's is directed exclusively towards the heat transfer, transformer and capacitor sales categories. In an effort to overcome some of the potential environmental problems of existing biphenyls, Aroclor 1016 was produced. Approximately 23.5 million pounds of Aroclor 1016 were sold domestically in 1973. The 1973 sales for Aroclors 1221, 1242 and 1254 were recorded at 0.04, 6.20 and 9.98 million pounds, respectively. All other PCB's showed no domestic sales (personal communication)*.

* Mr. W. B. Papageorge, Monsanto Company, St. Louis, Missouri.

Table 1. U. S. PRODUCTION OF SYNTHETIC ORGANIC PESTICIDES BY CLASS, 1967-1973[a]
(In thousands of pounds)

	1967	1968	1969	1970	1971	1972	1973[b]
Fungicides	177,886	190,773	182,091	168,470	180,270	170,569	193,362
Herbicides	439,965	499,574[c]	423,840	434,241	458,849	481,618	526,109
Insecticides, fumigants, rodenticides[d]	503,796	581,619	580,884	495,432	564,818	569,157	643,115
Total	1,121,647	1,271,966[c]	1,186,815[c]	1,098,143[c]	1,203,937	1,220,926[c]	1,362,586

[a] Fowler and Mahan (214, 689).
[b] Preliminary data.
[c] Revised.
[d] Includes small quantity of synthetic soil conditioners; does not include the fumigants carbon tetrachloride, paradichlorobenzene or inorganic rodenticides.

Table 2. DOMESTIC DISAPPEARANCE OF SELECTED PESTICIDES AT PRODUCERS' LEVEL, UNITED STATES, 1966-1973[a]
(In thousands of pounds)

Pesticide	1966[b]	1967[b]	1968[b]	1969[c]	1970[c]	1971[c]	1972[c]	1973[c]
Aldrin-toxaphene group[d]	86,646	86,289	38,710	89,721	62,282	85,005	105,980	89,362
Calcium arsenate	2,942	2,329	1,992	2,117	2,900	2,457	1,751	1,299
Copper sulfate	104,020	85,274	87,452	99,840	77,344	70,272	72,214	90,878
DDT	45,603	40,257	28,253	25,756	20,457	13,234	23,546	1,053
Lead arsenate	6,944	6,152	4,747	7,721	5,860	4,142	5,024	3,005
2,4-D	63,903	66,955	68,404	49,526	46,942	32,174	23,179	40,011
2,4,5-T	17,080	15,381	15,804	3,218	4,871	1,389	498	2,514

a Fowler and Mahan (214, 689).
b Year ending September 30.
c Year ending December 31.
d Includes aldrin, chlordane, dieldrin, endrin, heptachlor, Strobane® and toxaphene.

Table 3. U. S. PRODUCTION OF POLYCHLORINATED BIPHENYLS (PCB'S) AND POLYCHLORINATED TERPHENYLS (PCT'S) BY MONSANTO INDUSTRIAL CHEMICALS COMPANY 1959-1973[a]
(In thousands of pounds)

Year	PCB's	PCT's
1959	[b]	2,996
1960	37,919	3,850
1961	36,515	2,322
1962	38,353	4,468
1963	44,734	4,920
1964	50,833	5,288
1965	60,480	6,470
1966	65,849	8,190
1967	75,309	9,450
1968	82,854	8,870
1969	76,389	11,600
1970	85,054	17,768
1971	34,994	20,212
1972	38,600	8,134
1973	42,178	[c]

[a] Personal communication, Mr. W. B. Papageorge, Monsanto Company, St. Louis, Missouri.
[b] Data unavailable.
[c] Production terminated in April, 1972.

Phthalate esters (PAE's) were introduced in the 1920's to overcome the problems of camphor in the plasticizer industry. Major uses of PAE's include construction products, automobile and home furnishings, clothing, food coverings and medical products. Phthalates are also found in biochemical pathways and several natural products such as poppies and tobacco leaves (412, 693). The documentation that PAE's were readily extracted into blood from plastic storage bags and other medical devices (671) was the original basis for the fear that the human population might be continuously exposed.

"Plasticizers" are produced by a variety of manufacturers; however, phthalates (DOP, DIOP, DIDP and linear) are the major groups consumed (Table 4). During 1972, production of phthalic anhydride esters totaled 1,145,693 pounds and sales followed closely at 1,138,493 pounds (725).

Table 4. CONSUMPTION OF PLASTICIZERS BY TYPE[a]
(In thousand metric tons)

Plasticizer	1972	1973	1974
Adipates	28.0	28.4	27.3
Azelates	6.8	7.2	7.3
DOP/DIOP/DIDP	345.0	379.5	363.6
Epoxy	50.0	56.8	59.1
Linear phthalates	109.0	125.5	143.2
Polyesters	22.7	25.4	24.1
Trimellitates	8.1	8.5	10.5
Others	110.0	113.0	113.6
Total	679.6	744.3	748.7

[a] Anon. (672).

SECTION II

CHARACTERISTICS OF SYNTHETIC ORGANIC COMPOUNDS IN ESTUARINE ECOSYSTEMS

PERSISTENCE

This characteristic has been discussed at length (426, 427, 544) and specific attention has been given to the organochlorine pesticides (33, 138, 166, 396, 426, 569, 584, 637), organophosphates (405, 580), carbamates (335) and polychlorinated biphenyls (33, 268). Although persistence is critically tied to biological activity, photodecomposition, volatization, transport, state of eutrophication, temperature, detoxification, etc. (33, 166, 427, 544, 569), this discussion will emphasize quantitating the length of time synthetic organic compounds remain in the estuarine and juxtaposed ecosystems.

Menzie (426) provides a generalized table of persistence in water and fish of commonly used pesticides. Although the degrees of persistence in fish range from less than a day for malathion to more than 6 months for DDD and toxaphene, no reference is made to the source(s) of this information. Bourquin and his co-workers (569) indicated that DDT, BHC and chlordane can remain for 10 - 12 years, but no mention is made of the substrate.

Rumker et al. (544) also provide a summary comparison of persistence of organochlorines, organophosphates, carbamates and herbicides in soil and water. Although at first glance, persistence appears much greater in soil, these authors point out that the fate of environmental chemicals is often simply relegated to the sediment or food chains from the water, thereby only apparently being removed. One of these possibilities was investigated by Durant and Reimold (166), who reported on the effect of dredging on the redistribution of toxaphene, which had been present in the sediment at levels approaching 2,000 ppm. They found that dredging operations did not increase the toxaphene residues in downstream oysters and the anticipated sudden kill of fish and shellfish did not occur.

The organochlorine pesticides are most noted for their persistence. Lowe et al. (396) found that if mirex bait (in flow through nylon bags) was suspended in open seawater, approximately one half the original amount

remained after 6 months and one third after 9 months. Stickel and his co-workers reported extreme persistence in the tissues of young grackles, a bird commonly found feeding along the estuarine shores. Birds were fed 750 ppm mirex for 4 days. After 28 weeks, or almost 7 months, only one half of the original body burden had been lost.

In terms of quantitative decay of an organochlorine in seawater, Werner and Waldichuk (637) found that the different isomers of hexachlorocyclohexane (HCH and more popularly called BHC) behaved somewhat differently. The alpha, beta and gamma forms were reduced approximately 43, 60 and 85 percent, respectively. The latter two were similar in showing an initial, rapidly decelerating curve in contrast to the more gradual slope of the alpha isomer.

Less information is available on the persistence of organophosphates and carbamates. This may reflect our concept of their longevity, biased by our limited analytical capabilities with respect to their breakdown products or it may truly reflect their short-term persistence nature. Although generally considered quite short-lived, approximately 0.1 percent of the originally applied parathion remained in a sandy loam soil 16 years after the last application (580). This may be a miniscule amount but interest in this study was only rekindled after the non-specific Averell-Norris colorimetric parathion analysis was replaced by the more discerning flame photometric detection.

Dursban®, applied at 0.057 kg/ha and diazinon, applied at 0.225 kg/ha to control Culicoides larvae on an intertidal zone in Cape Cod, lasted varying periods of time depending on the substrate (405). For instance, the average time to last detectable trace of Dursban® for intertidal sand was 2 days while it was still detectable for 5 days in salt marsh sod and 15 days in salt marsh mud. Similar figures for diazinon were 4, 6 and 10 days respectively. With one minor exception, seawater collected adjacent to each treated plot did not contain any residues even on the day following treatment. Because of potentially great dilution factors, this should come as no surprise.

Karinen et al. (335) reported on the persistence of the carbamate, carbaryl, in estuarine water and mud under laboratory conditions at two different temperatures and under field conditions. They found that, in an aquarium without mud, approximately one half of the carbaryl disappeared in 38 days at 8 C. Most of this was accounted for by the production of 1-naphthol. An increase in temperature resulted in a significantly increased rate of disappearance. At 20 C almost all the carbaryl was gone by day 17. The addition of mud to the system resulted in both the parent compound and 1-naphthol declining to less than 10 percent in the seawater in 10 days. Under field conditions, carbaryl was also applied at levels comparable to those used on oyster pests. It was still detectable in the mud, after 42 days yet the major breakdown product (1-naphthol) persisted essentially only 1 day.

BIOACCUMULATION

The accumulation of synthetic organic compounds by biological entities is a subject touched upon by most investigators dealing with environmental chemicals. Portmann (715) recently discussed this phenomenon with respect to marine animals. Quite often it is only after these chemicals are found in a particular taxon that the question of their potential effects is seriously considered. The reasons why bioaccumulation occurs in any trophic level are not always obvious and the subject is often characterized by controversy (443). This discussion, however, will be restricted to the degree to which various estuarine organisms concentrate synthetic organic compounds over ambient levels. Wherever possible, the subject of residue loss (i.e., depuration) over time will be addressed.

Microogranisms and Plankton

Bourquin and his co-workers (569) found that microorganisms (isolated from a contaminated petroleum-base cutting fluid and maintained in a 100 ppm heptachlor-salts solution) could remove the vast majority of several organochlorines in water during a 4-hour period.

Uptake of DDT and dieldrin by 6 species of marine phytoplankton was found to increase linearly with an increasing concentration of the pesticides. However, with one exception (i.e., Amphidinium), accumulation did not increase linearly with increasing cell concentration (561). This study revealed that various species differ significantly in their ability to accumulate organochlorines. Dieldrin was concentrated in these phytoplankters by a factor of approximately 1,000 times the medium levels. DDT accumulation varied but, in general, exceeded the concentration factor of dieldrin by at least one order of magnitude.

Three species of marine phytoplankton were shown to accumulate low levels of DDT to varying degrees (16 to 54 percent of the original concentrations). Relative partition coefficients ranged from 1.2×10^5 to 2.9×10^5 (135). In another study (342), the marine diatom, Cylindrotheca, was found to adsorb DDT from culture media and on the average concentrate it approximately 265 times. The hypothesis that these and similar marine organisms might serve as storage sites for pesticides was supported.

The ability of microorganisms to accumulate vast quantities of organochlorines was elucidated by the, now classic, Biscayne Bay study by Seba and Corcoran (553) in 1969. They found that some of the surface slicks analyzed contained up to almost 13 ppb total pesticides. Although a minimum concentration factor (CF) of 10^5 was observed for the organochlorine pesticides, the CF's for other organic compounds were quite low. The average concentration of acetone, butyraldehyde and 2-butanone in slicks was never more than threefold greater than that in the water. In a similar study on the microlayer of Narragansett Bay (158), PCB's were concentrated 28 times in the upper 100 - 150 μm over the levels in subsurface water. Depending on how one defines the thickness of this slick, the enrichment factor for PCB's could be as great as 4×10^4.

Keil et al. (341, 344) exposed bacteria to 0.1 and 0.01 ppm of DDT or PCB. E. coli exposed to the lower treatment level accumulated 2.2 and 4.4 ppm DDT and PCB, respectively while those on the higher level accumulated 11.2 ppm DDT and 13.6 ppm PCB. Concentration factors, therefore, ranged between approximately 100 and 500, with a suggestion that they may be inversely related to dosage. The CF may also vary with the particular PCB considered. Preliminary studies carried out at Woods Hole (652) indicated that selective bacterial uptake of some PCB isomers over others did occur.

At least one section of the FAO Technical Conference on Marine Pollution held in Rome in 1970 addressed uptake of industrial toxicants by marine algae (468). Along similar lines, Keil, Priester and Sandifer (343) reported that the marine diatom, Cylindrotheca closterium, could concentrate Aroclor 1242 up to 1,100 times above media levels.

Other Invertebrates

To be sure, the uptake of any chemical will vary with the tissue sampled. Roberts (535) reported on the distribution of Endosulfan in various organs of the mussel (Mytilus edulis). Although it is commonly held that the most likely path of entry of adsorbed pesticides is via the gills, the gills did not assimilate Endosulfan to levels comparable to the digestive gland and the rise to maximum level was slower. However, unlike the study by Schoor (550), the frequency of monitoring intervals was probably too gross (i.e., 10 days) to allow adequate data interpretation.

The estuarine mollusk, Rangia cuneata, was shown to accumulate dieldrin from 200 to 2,000 times over the ambient levels during a 12 to 36 hour exposure period. Although the sample sizes were particularly small, the most important result of this study by Petrocelli, Hanks and Anderson (495) was the measurable accumulation of dieldrin from acceptably low levels in water, which are presumed to be safe.

Lowe and his co-workers (397) showed the comparative pesticide uptake by oysters exposed to 1 ppb DDT, toxaphene or parathion. At week 24, the CF's for the 2 organochlorines were 6.3×10^4 and 2.3×10^4, respectively while that for the organophosphate was at least two orders of magnitude lower (i.e., 2.4×10^2). It took 12 weeks for the toxaphene and parathion oyster groups to purge themselves of their accumulated residues. After the same period, the DDT group had less than one percent left.

Brodtmann (64) suggested that the uptake of DDT by his experimental oysters appeared to be by diffusion rather than by active transport. He suggested that the oyster gills were probably the primary entry site of DDT. Because of the rapid pesticide elimination rate, the value of the oyster as an environmental integrator was challenged.

The subject of DDT accumulation by two marine polychaetes was presented at the 1970 FAO Conference (197). Uptake varied with exposure levels and time. However a concentration factor of 100 was common for one species for as short a period as 2 days and 300 for another (Lanice and Nereis, respectively).

The CF for the euphausiid shrimp (Euphausia pacifica) exposed to 5 to 33 ppt DDT for 2 hours ranged between 1.1×10^3 and 4.4×10^3. Smaller shrimp exhibited higher values, however, larger animals took longer to equilibrate (137). Both uptake from water and assimilation from food were shown to be two important sources of DDT for this shrimp.

Epifanio (194) was able to show the relative importance of dieldrin in seawater versus that in the diet of a larval crab (Leptodius floridanus). If equal concentrations of this synthetic organic were available to the crab larvae, they would accumulate it 8,000 times as fast from the water as from the food. It was calculated, however, that the larvae would accumulate the pesticide only 1.23 times as quickly from water as from food under reasonable field conditions.

Blue crabs (Callinectes sapidus), exposed to 0.22 ppb mirex (nominal) for 15 minutes to 16 hours, accumulate from 0.24 to 0.69 ppb in their hemolymph to 1.6 to 31 ppb in their hepatopancreas (550). Schoor suggested that the gills were the mode of entry, since the gills showed traces of mirex-^{14}C after only 5 minutes exposure while it appeared in the hepatopancreas only after 15 minutes.

Kobayashi, Akitake and Tomiyama (359, 360) discussed the mechanism by which the herbicide, pentachlorophenate (PCP), decreased in the exposure medium when inhabited by a culture of shellfish (Tapes philippinarum). Although a previous study had revealed that the amount taken up by the shellfish was appreciably less than the decrease in its medium, their studies indicated that the decrease did not mean decomposition. It was found that the PCP was transformed by the shellfish to some bound form (360).

The freshwater crab (Uca minax), which is congeneric to many estuarine fiddler crabs, was shown to accumulate the experimental herbicide, bentazon (3-isopropyl-1H-2,1,3-benzothiadiazin - 4(3H)-1-2,2-dioxide) 50 times over the amount found in the water of a model ecosystem (56).

Parrish and his co-workers (483) discussed PCB uptake in oysters. These shellfish showed a concentration factor (of 5 ppb Aroclor 1254) of 8.5×10^4 and lost all but a trace after 32 weeks of depuration. An exposure of 1 ppb 1254 resulted in a 10×10^4 CF, all but a trace of which disappeared after 12 weeks' depuration (395, 483). Oysters exposed to 10 ppb Aroclor 1016 accumulated the chemical 1.3×10^4 times the exposure level and none was detectable after 56 days in PCB-free water (482).

Since Aroclor 1016 has been produced to replace other PCB's for sale to capacitor manufacturers, its accumulation in estuarine organisms is of importance. Hansen, Parrish and Forester (267) reported that the American oyster (Crassostrea virginica) and the brown shrimp (Penaeus aztecus) were similar in their uptake but the grass shrimp (Palaemonetes pugio) displayed a concentration factor of approximately 1/4 to 1/2 that of the above two invertebrates. Based on the nominal exposure concentration for 96 hours, the CF ranged between 440 and 4,000, decreasing at higher exposure levels.

The relative importance of sediment as a source of PCB residues to the grass shrimp, pink shrimp and fiddler crabs was discussed by Nimmo et al. (462, 465). Its importance would be expected to be far greater to invertebrates that are detritus feeders like the fiddler crab (465, 471). Grass shrimp were exposed to several concentrations of Aroclor 1254 for up to 63 days (462). The shrimp continued to accumulate 1254 throughout the experiment, although, after 5 weeks, concentration factors ranged from 200 to 26,000. Within 4 weeks after exposure was discontinued, 60 to 90 percent of the chemical was lost from the shrimp. Other reports, not covered above, also deal with uptake of industrial toxicants (545, 642, 643) or pesticides (534, 594, 619) by invertebrates.

Fishes

In view of the commercial and recreational interest in this taxon, an unexpectedly small amount of uptake work has been carried out on estuarine fishes.

An extensive field monitoring study by Borthwick et al. (58) provided evidence for apparent fish bioaccumulation of mirex in a South Carolina estuary. Under controlled conditions, Hansen and Wilson (272) found that pinfish (Lagodon rhomboides) and Atlantic croakers (Micropogon undulatus), exposed to 0.1 ppb DDT, reached maximum body burdens in two weeks. Maximum concentration of DDT ranged from 1.0×10^4 to 3.8×10^4 times that in the test water, depending on species and exposure. DDT was lost quite slowly with 87 and 78 percent being lost by pinfish and croakers, respectively, after eight weeks in clean water.

Ernst (197) presented some interesting findings relating to the uptake and metabolism of labelled DDT by two species of flatfish. After 28 days, the proportions of DDD:DDE:DDT were 1:2:12 in the flesh. These findings, like those of Hansen and Wilson (272) indicated the importance of pesticide metabolism under field conditions; a phenomenon which is not generally operative during short-term laboratory studies.

The significance of the food chain as a source of DDT accumulation by fish was elucidated by Macek and Korn (399). They found that brook trout (Salvelinus fontinalis) accumulated approximately 10 times more of the available DDT from the food than directly from the water.

Pritchard, Guarino and Kinter (510) reported on the bioaccumulation and tissue distribution of DDT and mirex by the winter flounder (Pseudopleuronectes americanus) over time. Their findings are especially interesting since neither pesticide is initially distributed in accord with the lipid content of recipient tissues.

Initial efforts of Dvorchik and Maren (171) indicated that p,p'-DDT is probably bound to the plasma lipoproteins of the dogfish shark (Squalus acanthias). Their studies also showed that the liver sequestered the compound quantitatively and there was no DDT excreted via the gills and no significant amount by the urine.

PCB's have become ubiquitous in the environment (490, 492, 528) and may even be more widespread than DDT and its breakdown products. Like the relationship for DDT, that between PCB residues in the environment and the organism's body burden can only be accurately characterized under strictly controlled experimental conditions. The concentration factor varies considerably with species and with mode of exposure.

Most PCB residues quantified from field samples are done so based on Aroclor 1254 because of the similarity of their chromatogram profiles with those of the 1254 standard. Aroclor 1254 may, therefore, be one of the most widespread PCB's. Hansen et al. (268) found that spot (Leiostomus xanthurus), exposed to 1 ppb Aroclor 1254, rapidly stored this chemical with maximum concentration levels being reached between 14 and 28 days.

The polychlorinated terphenyls (PCT's) are less commonly found but this may partially be a result of analytical difficulties. Addison et al. (4) tube-fed cod (Gadus morhua) herring oil containing 1 g of Aroclor 5460. In some cases, appreciable amounts of this PCT remained even after 70 days. Their experiments indicated that excretory efficiency seemed poor.

Since Aroclor 1016 has come to replace many of the PCB's in the more controversial 1200 series, its uptake behavior is of interest. Pinfish exposed to 1 ppb Aroclor 1016 accumulated maximum concentration levels by 21 to 28 days. In this study, Hansen, Parrish and Forester found that maximum body burdens (based on wet weight) were 1.7 x 10^4 times the nominal test water concentration (267). In another report by Parrish and co-workers (482) on the same compound and fish species, they found that after 56 days' depuration, 61 percent of the PCB disappeared from the fishes.

Another approach to demonstrating the dynamics of residue uptake over time is to compare the body burdens of fish in various age groups. Parejko et al. (480) did this for a small sampling of lake trout (Salvelinus namaycush) but could not show any correlation between age and PCB concentration. Since the ascending part of the residue uptake curve may occur in the first few years or even months of life, it is unfortunate that the youngest age group was 4 years.

Very little has apparently been done with the bioaccumulation of other industrial toxicants by fish. One study, however, dealt with the uptake of the polycyclic aromatic hydrocarbons, naphthalene and 3, 4 benzopyrene. It was found to be rapid by three marine fish, however, detoxification mechanisms provided efficient removal from body tissues (379).

Birds

Uptake research involving birds has historically revolved around domestic species, which with the singular exception of the duck, have no close relatives inhabiting the estuary. For this reason and the personal biases for certain taxa by many investigators working in estuarine pollution, very little of the avian data seldom reaches the estuarine review level.

In reviewing data from almost four thousand samples, Dindal (151) found that obvious relationships existed between the type of food ingested (plant or animal) and DDT residues in wild mallard ducks (Anas platyrhynchos) and lesser scamp ducks (Aythya affinis). In that, now classic, radio-labeled DDT study of a marsh system during 1964 and 1965, DDE was the predominant metabolite found. After being exposed to a 0.225 kg DDT/ha treatment, maximum residues for the heart, breast and gizzard of the mallards reached approximately 1.55 ppm. Lesser scaup accumulated larger DDT residues but the accumulation was shorter lived.

Robinson and his colleagues (537) reported that dieldrin levels in adult birds were independent of age. Carcass levels of dieldrin in redwinged blackbirds (Agelaius phoeniceus), exposed to 10 ppm aldrin, increased linearly with time and showed no tendency to level off (109). Elevations of the regression lines were, however, significantly different for "stressed" versus "non-stressed" birds. Dieldrin brain levels, on the other hand, were curvilinear over time and asymptotic at about 25 ppm (wet weight basis). This leveling off began approximately 15-20 days after diet began.

Stickel et al. (584) fed grackles (Quiscalus quiscala) 750 ppm mirex for four days and noted depuration over time. Their data indicated mirex to be one of the most persistent organochlorines documented for birds, with only one-half the original amount being lost after 28 weeks or nearly 7 months.

Parslow and Jefferies (484) cleverly tied several loose ends together to come up with a reasonable picture of PCB uptake by guillemots (Uria aalge). They pointed out the limitations of their efforts on this front but suggested that this species (in the Irish Sea) either ingests, on the average, less than 1 ppm in the diet or the rate of storage is considerably less than the 9 percent reported for another, unrelated, species.

Risebrough and de Lappe (528) in their overview of PCB accumulation in ecosystems, discuss bioaccumulation of PCB's and DDT in terms of the

feeding behavior of various oceanic species as it compares to that of more estuarine inhabitants, such as the brown pelican (Pelecanus occidentalis) and the osprey (Pandion haliaetus).

Little additional information is available on the bioaccumulation of other industrial toxicants. However, there is some evidence that DBP and DEHP may not accumulate in avian tissue to within a detectable range, even when fed 10 ppm for 5 months (673).

BIOMAGNIFICATION

Unlike the term "bioaccumulation", which is generally restricted to the uptake and accumulation of synthetic compounds by an organism from its environment, usually water, "biomagnification" relates to the commonly observed increase in body burdens with each ascending trophic level up the food chain. Since neither term can be found in the common dictionaries, their usage suffers the inconsistencies of personal prejudices and convenience.

Undoubtedly the organochlorines (both pesticides and industrial toxicants, like PCB's), because of their lipophilic nature leading to persistence, are the greatest biomagnifiers. The transfer of synthetic organic compounds from one trophic level to the next has been shown by Macek and Korn (399) to be a far more important source than the ambient water, at least for DDT and freshwater fish. Some caution must be exercised since Nakatsugawa and Nelson (452) present some dieldrin examples which remind us that magnification does not necessarily occur at each step in the trophic chain.

The question of how biomagnification occurs is one of controversy. Risebrough and de Lappe (528) and others (157) point out that many things including the amounts and kinds of lipids may affect retention of PCB's thereby modifying the trophic accumulation predicted by the classical food chain concentration theory. Moriarty (443) pointed out some inconsistencies in the data which have supported the argument that organochlorine insecticides concentrate along the food chain. It was suggested that residue differences between species depend largely on specific differences in rates of excretion and metabolism, which are not necessarily related to position in food chain (see also reference 452 for role of detoxication enzymes). Along these lines, Robinson et al. (537) noted that, although organochlorine insecticides tended to be greater in marine organisms of higher trophic levels, the tendency was not found in all food chains. A second assumption that Moriarty challenged was that plateau concentrations are reached. If this were not the case, longer-lived predators would reach higher concentrations because of longevity alone. To a certain extent, the controversy is one based on semantics. Most investigators in the United States, at least, seem to use the term "biomagnification" to describe an observed phenomenon and not the potential mechanisms responsible for it.

Since Woodwell, Wurster and Isaacson documented DDT increasing with trophic levels through more than three orders of magnitude in an Atlantic coast estuary (656), many similar reports have been added to the scientific and popular literature. Dustman and Stickel (169), Menzie (426), and Nakatsugawa and Nelson (452) give overviews of the pesticide biomagnification situation, which include some estuarine species. Menzie points out that "biological magnification...is dependent only on three conditions: (1) the material must persist in the environment; (2) the material must be biologically available; and (3) the material must persist in the biological system which assimilates it...."

Foehrenbach (208) looked at two simple food chains, involving invertebrates, fishes and birds, in a Long Island (New York State) estuary. He reported higher concentrations of DDE, DDD and DDT in upper trophic levels than in lower ones, with two orders of magnitude between the extremes.

Kneip, Howells and Wrenn (358) provided a graphic presentation of food chain biomagnification of DDT and dieldrin by Hudson River biota. Based on a comparison of residues (ppm, wet weight) in the fauna with that in the water, concentration factors increased up the food chain as follows: plankton ($.1 \times 10^4$ - $.7 \times 10^4$); bivalves, fish ($.1 \times 10^4$ - $.5 \times 10^4$); fish ($.5 \times 10^4$ - 5×10^4), and; birds (5×10^4 - 15×10^4).

Metcalf and his group approached what they called "ecological magnification" of DDT and methoxychlor using a model ecosystem approach (432). The ecosystem had a terrestrial-aquatic interface and a seven-element food chain, including the salt marsh caterpillar (Estigmene acrea), Culex mosquito larvae and mosquito fish (Gambusia affinis). When ^{14}C-labeled DDT was applied at a rate equivalent to 1.122 kg/ha, Gambusia accumulated DDT 1×10^4 to 8.4×10^4 times the level in the water. DDE, on the other hand, was accumulated 11×10^4 times. Methoxychlor did not show a similar ecological magnification and the authors suggested that it was more environmentally degradable.

Booth and his co-workers (56) also used a model ecosystem to study possible accumulation of the experimental herbicide, bentazon (3-isopropyl-1H - 2, 1, 3-benzothiadiazin 4(3H)-1-2,2-dioxide). Their conclusion was that bentazon does not accumulate in aquatic food chains. This was a freshwater system but it included Uca minax, which is congeneric with a myriad of estuarine fiddler crabs.

Dustman et al. (170), Peakall and Lincer (492), and Stickel (582) provide reviews which include PCB biomagnification in estuarine ecosystems.

There are unique analytical problems associated with PCB's and unless careful separation and quantitation are made, incorrect conclusions may be drawn from a comparison of residues found in different trophic levels. Zitko, Hutzinger and Choi (668) found that fish from lower and intermediate trophic levels contained relatively lower amounts of hexachlorobiphenyls

(4th, 5th and 6th major Aroclor 1254 peaks) and higher amounts of tetra- and pentachlorobiphenyls (1st and 2nd major peaks) than did the predaceous fish and birds.

Prestt, Jefferies and Moore (507) compared liver PCB levels between birds feeding on obviously different prey. PCB residues ranged up to 40, 50 and approximately 900 ppm for those species feeding on birds, mammals and fish (mainly freshwater), respectively. The last group included bitterns, kingfishers, grebes and herons.

Because of the similarity in distribution of DDT and PCB's it is of interest to compare their biomagnification characteristics. Although a comparison of different studies would risk potential analytical differences leading to incomparable data, a few studies include food chain residue information on both groups of synthetic organics. Munson (447) reporting on PCB and DDT residue levels in the invertebrates of the Laguna Beach (San Diego) area, documented a classic case of biomagnification for both with progression up the food chain from strict herbivores, such as the abalone, to scavengers, such as the whelk and spiny lobster. In another study, organochlorine insecticide residues were 7-fold greater in sediments than in the water and those in fish were 10- to 60-fold greater than in sediments (249). Greichus, Greichus and Emerick (249) reported that adult cormorants (Phalacrocorax auritus) had residues which were 250-fold greater than the fish while the white pelican (Pelecanus erythrorhynchos) showed levels which were elevated 280-fold. Interestingly, PCB's showed less biomagnification, i.e., 60- and 30-fold increases in cormorants and pelicans, respectively, over fish.

METABOLISM

Perhaps the most comprehensive efforts dealing with the metabolism of pesticides are those by Menzie (425, 428), which are arranged by compound and not restricted to the estuarine ecosystem. A partial list of metabolic conversions by estuarine or marine organisms can be found in Table 5. In general, those so listed are not discussed below and vice versa.

Bourquin (60) provided a brief review in 1973 on estuarine microbes and organochlorine pesticides. He very accurately pointed out the importance of understanding the pathways of microbial breakdown of pesticides, since they are probably the main environmental mechanism by which these compounds are reduced. Thom and Agg (724) discuss the biological breakdown of SOC's, especially as it relates to sewage treatment. From the organism's standpoint, Khan, Stanton and Reddy (353) reviewed the literature on the main detoxication and conjugation systems of insects and invertebrates, some of which are marsh/estuary inhabitants.

Table 5. OBSERVED METABOLIC CONVERSIONS OF SYNTHETIC ORGANIC COMPOUNDS BY ESTUARINE/MARINE ORGANISMS UNDER CONTROLLED CONDITIONS

Subject	Organ or other information	Observed metabolism		Reference
		From	To	
Sediment		p,p'-DDT	p,p'-DDD(TDE)	7
Microflora	From intestine of anchovy	DDT	DDD	404
Bacteria	*E. Coli*	DDT	DDD (TDE) and DDE	344
Microbial isolates		DDT	DDOH, DDNS, TDE, DDE, etc.	487
Surface film		Dieldrin	Photodieldrin, etc.	487
Surface film		Aldrin	DIOL	487
Algae	From fish pond	Endrin	Ketoendrin	487
Algae	*Dunaliella* spp.	Dieldrin	Photodieldrin, etc.	487
Diatom	*Cylindrotheca closterium*	DDT	DDE	342
Lobster	Hepatopancreas	Parathion	p-nitrophenol	98
Flatfish	Various	^{14}C-DDT	DDD (TDE) and DDE	198
Trout and skate	?	PCB	2 and 4-hydroxybiphenyl	205
Atlantic salmon	Whole	DDT	DDE and DDD (TDE)	247
Atlantic salmon	Gut	p,p'-DDT	p,p'-DDD (TDE)	106
Mallard duck	Liver and muscle (possibly by microorganisms)	o,p'-DDT	o,p'-DDD (TDE)	371

Adamson and Sieber (2) reviewed the work of several researchers on the metabolism of various xenobiotics, including synthetic organic compounds, by freshwater and marine fishes.

Review-level publications on the metabolism of industrial toxicants are scarce, however, Stickel (582) included a brief review of the metabolism and kinetics of PCB's by estuarine organisms in her 1972 publication.

Sediment, Water and Microorganisms

Biologically active marine sediment can act as a reservoir for organochlorine compounds as shown by Oloffs et al. (477). Their studies indicated that, with the exception of lindane, all other compounds applied (α and γ chlordane, DDT and Aroclor 1260) moved into the sediment within 6 weeks. Although most of the lindane was normally metabolized, sterilization of water and sediment prevented that action; the same was not true for DDT and DDD.

Albone and his associates (7, 8) tested the ability of Severn estuary sediments to degrade DDT, both in situ and in the laboratory, and compared that with the more extensive degradation by anaerobic sewage sludge.

Although heptachlor was not apparently a source of carbon for polymer formation by microorganisms, which formed floc particles (569), there is some evidence that ethion may serve as a source of sulfur for microbial growth (559).

Sikka and his co-workers (560) looked at the metabolism of selected pesticides by various marine microorganisms, including algae, bacteria, fungi and yeast. A carbamate, carbaryl, was shown to be quite resistant. None of the species tested was able to significantly degrade it. In an earlier study by Sikka and Rice (561), several species of unicellular marine algae were shown capable of converting small quantities of DDT to DDE (i.e.,0.103 - 11 percent of the parent compound after 24 days exposure). No species tested (Amphidinium, Isochrysis, Skeletonema, Tetraselmis, Olisthodiscus, nor Cyclotella) was capable of metabolizing dieldrin (524,561). Bousch and Matsumura (62) have also been looking at the ability of major groups of algae to degrade pesticides.

Of about 100 marine microbial isolates from Hawaii and Houston, Texas, 35 appeared to be active in degrading DDT, with TDE (DDD) being the predominant metabolite. Similar exposure of dieldrin to microorganisms indicated that photodieldrin was the main metabolite (487). Perhaps the most interesting finding of Patil and co-workers was that these insecticides were not metabolized in plain, even relatively polluted, estuarine water.

The importance of microorganisms in degrading pesticides should not be restricted to the sediment habitat. They have been shown to play a similarly important role in the intestines of various aquatic organisms, including the Atlantic salmon (106) and the northern anchovy (Engraulis

mordax) (404). Interestingly, some differential activity of the intestinal fungi and bacteria was suggested by the data in the latter study.

In a model ecosystem treated with ^{14}C-bentazon, an experimental herbicide, water samples did not yield appreciable levels of the chemical until hydrolyzed with dilute HCl (56). One possible explanation was that much of this herbicide was rendered relatively unextractable by conjugation.

Little has been published relating to the breakdown of industrial toxicants by this facet of the estuarine ecosystem, however, some preliminary work on PAE's by Johnson (328) indicates that a freshwater hydrosol may be quite active. Marked differences between two common esters, di-2-ethylhexylphthalate (DEHP) and di-n-butyl phthalate (DBP) were noted. Under aerobic conditions, 98 percent of the DBP was degraded within 5 days, however, after 14 days only 50 percent of the DEHP was degraded.

Other Invertebrates

In metabolic experiments dealing with DDT and the marine polychaete (Nereis diversicolor), the unchanged parent compound was the only one recovered except in the case of oral application with a waiting period of 5 days. After repeated application of DDT to another polychaete (Lanice conchilega), DDE and traces of DDD and unidentified polar metabolites were extracted from coelomic fluid and tubes but these were attributable to bacterial degradation (197).

Very few other studies have been carried out on the ability of marine invertebrates to metabolize pesticides but a couple have dealt with organophosphates. One such study by Carlson (98) indicated that the lobster (from the Rhode Island area) should be quite resistant to parathion since its heptopancreas was able to detoxify the chemical to the less toxic p-nitrophenol but apparently not to the toxic paraoxon. Unfortunately, this in vitro study was not supported by Carlson's actual toxicity data and he speculated that the in vitro reaction does not take place in vivo or other organs are capable of the toxic conversion (i.e., "biochemical suicide") to paraoxon.

Nakatsugawa and Nelson (452) in their discussion of the detoxification role of invertebrate metabolism mention that the saltwater clam, Venus mercenaria, is paralyzed and killed by parathion. Although neurotransmitter information is lacking, the implication is that microsomal activation is occurring.

Kobayoski and his co-workers (359, 360, 361) have worked extensively on the metabolism of the herbicide, pentachlorophenate (PCP), by the marine shellfish, Tapes philippinarum. Their studies (360) indicated that this shellfish was capable of transforming the PCP to a bound form by detoxification.

Metabolism of naphthalene has been reported for the crab, Maia squinado, and the mixed-function oxidases of marine invertebrates has also come under recent investigation (Corner et al., 1973 and Pohl et al., both In Khan et al., ref. 353). Along these lines, the quahaug (Mercenaria mercenaria) showed no oxidative metabolism of EPN, p-nitroanisole, aminopyrine or hexobarbital (Carlson, 1972 In Kahn et al., ref. 353).

Fishes

Twenty-eight days after being fed ^{14}C-DDT, the flesh of flatfish, Platichthys flesus, contained DDD, DDE and DDT in a ratio of 1:2:12 (197). In this study, Ernst noted that several unidentified polar metabolites were also found in the gastrointestinal tract and in the feces. Ernst and Goerke (198) found that another flatfish (Solea solea) also possessed the ability to metabolize ^{14}C-DDT and the percentages of metabolites were higher in the liver and gastrointestinal tract than in the skeletal muscle.

Contrary to the above findings, tissues from winter flounder injected with ^{14}C-labeled mirex or DDT contained primarily the unaltered pesticide (510). Urine from the DDT-treated fish was the only exception, showing a preponderance of one or more metabolites. Again, after intravenous injection of DDT into the skate (Raja radiata), significant DDT metabolism was not determined, even after prolonged periods (205). Another experiment by Dvorchik and Maren (171) with DDT and another cartilaginous fish, the dogfish shark (Squalus acanthias), indicated a similar lack of metabolic breakdown ability.

A preliminary report (205) on cod fed Aroclor 5460, a chlorinated terphenyl, indicated that absorptive and excretory efficiency seemed poor. However, some selective adsorption, depositon or excretion of the PCT was observed.

In view of the potential interpretive and analytical problems associated with the commercial biphenyls, Hutzinger and co-workers (312) looked at the metabolic behavior of pure mono-, di-, tetra- and hexachlorobiphenyl isomers. In a comparative study of rainbow trout, pigeons and rats, they found hydroxymetabolites in the mammal's urine and bird's excreta but not in that of the fish.

Lee, Sauerheber and Dobbs (379) tested three species of marine fish for their ability to metabolize polycyclic aromatic hydrocarbons. The main product of 3, 4-benzopyrene metabolism was tentatively identified as 7,8-dihydro-7,8-dihydroxybenzopyrene. Naphthalene was metabolized to 1,2-dihydro-1,2-dihydroxynaphthalene. The parent compounds were rapidly taken up but detoxification mechanisms existed for efficient removal (via the urine) of the compounds from body tissues.

Although phthalates have been shown to be present in estuarine and marine fishes (645), little has been done on the ability of estuarine fish to metabolize this industrial toxicant. One effort by Stalling et al. (574)

addresses the metabolism of two phthalates (DEHP and DBP) by the freshwater channel catfish (Ictalurus punctatus). Their results indicated that at least two separate and distinct microsomal enzyme systems degrade PAE's.

Birds and Mammals

Dindal and Peterle (152), after spraying a marsh with radioactively tagged chlorine-36 (i.e., ring-labeled), technical DDT found primarily DDE as the most abundant metabolite. However, labeled p,p'-DDT, DDD (TDE), and the intermediate metabolite, DDMU were also found. The last was present only in liver, brain and wing tissues.

Work by Lamont, Bagley and Reichel (371) indicated that the mallard is capable of transforming dietary o,p'-DDT to o,p'-DDD. However, they suggested the possibility of postmortem breakdown.

In an effort to shed some light on bird die-offs, such as those in the Irish Sea, Lincer and Peakall (385) showed the ring dove (Streptopelia risoria) capable of differential metabolism of Aroclor 1254 components. This was accompanied by the production of late arising chromatogram peaks, probably as a result of in vivo microsomal transformation.

Residue work by Jansson and co-workers (319) revealed PCB metabolites in the feces of Baltic grey seal (Halichoerus grypus) and guillemot (Uria algae). However, discrepancies in chromatogram profiles indicated different metabolic pathways.

SECTION III

PRESENCE OF SYNTHETIC ORGANIC COMPOUNDS IN ESTUARIES

GENERAL

Pesticides

Considering only the organochlorine pesticides, DDE (the major breakdown product of DDT) is probably the most widely distributed in fish and wildlife (see Tables 6 and 7 for organochlorine pesticides found in estuarine/marine fishes and fish-eating birds, respectively). Being lipophilic (i.e., "fat-loving"), DDE like other organochlorines is not very soluble in water but accumulates in the fat of organisms (for overview, also see Table 1 in reference 687). Organochlorine pesticides are passed from prey to predator with little lost by way of excretion. This "biological magnification" with each transfer from one food level (i.e., trophic level) to the next commonly results in animals at the tops of food chains acquiring inordinate amounts of these compounds (see Section III). For instance, DDE concentration reached 1,100 ppm (parts per million) in the fat of brown pelican eggs collected off the coast of California and 1,000 ppm in the eggs of the white-tailed eagle collected in the Baltic (530).

Organochlorine pesticides are readily accumulated by shellfish and this characteristic has been taken advantage of to characterize the geographic distribution of pesticide contamination (see below).

Although most organophosphate and carbamate pesticides are advertised as short-lived, there is evidence that some may not be. In an application of carbaryl (Sevin) at rates comparable to those used to control oyster pests, the chemical could still be detected in the mud 42 days post-treatment (335). Similarly, 14 days after a standard ground application of malathion, the organophosphate could still be found in the estuarine plant Juncus (722). In addition, in monitoring efforts where organophosphates have been specifically looked for in estuarine fishes, like the sheepshead minnow and the spot, these have been suspected (121, 305).

The presence of other major groups of pesticides in the estuarine ecosystem has not been widely documented. However, the fungicide hexachlorobenzene (HCB), has recently been reported in several species of freshwater and some species of anadromous fishes including coho

Table 6. PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a]		Geog. loc.[b]	Ref.	Date[c]
		DDT[d]	Other			
Hake	Liver	0.11-6.61	Dieldrin -[e]	La Jolla area, Calif.	165	1971
Jack mackeral	Liver	0.14-0.30	Dieldrin -	La Jolla area, Calif.	165	1971
Ocean whitefish	Liver	0.08-0.94	Dieldrin -	La Jolla area, Calif.	165	1971
Spiny dogfish	Liver	228-473	Dieldrin 0.13-0.14	La Jolla area, Calif.	165	1971
Sockeye salmon	Liver	DDT -	Dieldrin -	Seattle, Washington	165	1971
Chum salmon	Liver	DDT -	Dieldrin -	Auke Bay, Alaska	165	1971
Coho salmon	Liver	DDT -	Dieldrin -	Auke Bay, Alaska	165	1971
Sardine	Whole	0.011-0.48	Lindane 0.009-0.027	Mediterranean Coast, Spain	35	1973
Sardine	Whole	e	Cyclodienes <0.001-0.064	Mediterranean Coast, Spain	35	1973
Sardine	Whole	< 0.005-0.084	Lindane 0.004-0.029	Atlantic Coast, Spain	35	1973
Sardine	Whole		Cyclodienes <0.001-0.007	Atlantic Coast, Spain	35	1973
Dogfish	Muscle	0.76-1.01	Lindane -- - 0.01	Mediterranean Coast, Spain	35	1973
Dogfish	Muscle		Cyclodienes -- - <0.001	Mediterranean Coast, Spain	35	1973
Dogfish	Liver	1.91-18.1	Lindane 0.001-0.141	Mediterranean Coast, Spain	35	1973
Dogfish	Liver		Cyclodienes 0.006-0.052	Mediterranean Coast, Spain	35	1973
Dogfish	Muscle	-- - 0.095	Lindane 0.005-0.009	Atlantic Coast, Spain	35	1973
Dogfish	Muscle		Cyclodienes 0.001-0.024	Atlantic Coast, Spain	35	1973
Dogfish	Liver	0.61-3.79	Lindane 0.003-0.111	Atlantic Coast, Spain	35	1973
Dogfish	Liver		Cyclodienes 0.014-0.39	Atlantic Coast, Spain	35	1973
Roughscale rattail	Liver	0.29-7.9		Central California	557	1972

Table 6 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a] DDT[d]		Other	Geog. loc.[b]	Ref.	Date[c]
Sanddob	Liver	0.041-13			Monterey Bay, Calif.	557	1972
Sablefish	Liver	0.28-6.9			Central California	557	1972
Jack mackeral	Liver	0.10-1.6			Southern California	557	1972
English sole	Liver	0.22-6.1			Monterey Bay, Calif.	557	1972
Roughscale rattail	Flesh	0.0026-4.7			Central California	557	1972
Sablefish	Flesh	0.59-6.3			Central California	557	1972
Jack mackeral	Flesh	0.010-0.074			Southern California	557	1972
Winter flounder juvenile	Muscle	< 0.01-0.31	(T)	Heptachlor <0.01-1.10	Buzzards Bay, Mass.	564	1970
Winter flounder juvenile	Muscle	0.03-1.07	(E)	Dieldrin <0.01-0.05 Heptachlor epoxide <0.01-0.44	Buzzards Bay, Mass.	564	1970
Winter flounder	Muscle	0.01	(T)	Dieldrin <0.01 Heptachlor epoxide <0.01-0.56	Buzzards Bay, Mass.	564	1970
Winter flounder	Muscle	0.18-1.07	(E)	Heptachlor 0.62-1.55	Buzzards Bay, Mass.	564	1970
Winter flounder	Ovarian tissue	0.11-0.40	(T)	Dieldrin <0.01 Heptachlor epoxide <0.01-0.65	Buzzards Bay, Mass.	564	1970
Winter flounder	Ovarian tissue	0.02-0.22	(E)	Heptachlor <0.01-0.07	Buzzards Bay, Mass.	564	1970
Salmon, sea trout				Pollutants from sewage and coke oven operation also present	England	568	1932
Atlantic salmon	Muscle	0.02-0.10			Atlantic Coast, Canada	570	1971
Atlantic salmon	Viscera	0.15-0.65			Atlantic Coast, Canada	570	1971
Atlantic mackeral	Whole	0.45-0.77			Atlantic Coast, Canada	570	1971
Atlantic cod	Muscle	0.02-0.08			Atlantic Coast, Canada	570	1971
Atlantic cod	Viscera	0.36-1.16			Atlantic Coast, Canada	570	1971

Table 6 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a] DDT[d]		Other	Geog. loc.[b]	Ref.	Date[c]
White hake	Muscle	0.01-0.08			Atlantic Coast, Canada	570	1971
American smelt	Whole	<0.01-0.06			Atlantic Coast, Canada	570	1971
American smelt	Whole	0.02-0.19			Atlantic Coast, Canada	570	1971
Winter flounder	Muscle	<0.01-0.03			Atlantic Coast, Canada	570	1971
Winter flounder	Viscera	<0.01-0.02			Atlantic Coast, Canada	570	1971
Atlantic tomcod	Whole	DDT -			Atlantic Coast, Canada	570	1971
Anchovy	Whole	0.074	(E)		Washington Coast	586	1968
Anchovy	Whole	0.074	(D)		Washington Coast	586	1968
English sole	Whole	0.009-0.016	(E)		Washington Coast	586	1968
English sole	Whole	0.009-0.016	(D)		Washington Coast	586	1968
Hake	Whole	0.058	(E)		Washington Coast	586	1968
Hake	Whole	0.047	(D)		Washington Coast	586	1968
Hake	Whole	0.074	(E)		Oregon Coast	586	1968
Hake	Whole	0.068	(D)		Oregon Coast	586	1968
Ocean perch	Muscle	0.012	(E)		British Columbia, Can.	586	1968
Ocean perch	Muscle	Trace	(D)		British Columbia, Can.	586	1968
Ocean perch	Muscle	0.013	(T)		British Columbia, Can.	586	1968
English sole	Whole	0.010-0.019	(T)		Washington Coast	586	1968
Hake	Whole	0.090	(T)		Washington Coast	586	1968
Hake	Whole	0.143	(T)		Oregon Coast	586	1968
Starry flounder	Muscle	0.018	(E)		Washington Coast	586	1968
Starry flounder	Muscle	0.026	(D)		Washington Coast	586	1968

Table 6 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a] DDT[d]		Other	Geog. loc.[b]	Ref.	Date[c]
Starry flounder	Muscle	0.013	(T)		Washington Coast	586	1968
Yellowtail rockfish	Muscle	0.017	(E)		British Columbia, Can.	586	1968
Yellowtail rockfish	Muscle	Trace	(D)		British Columbia, Can.	586	1968
Yellowtail rockfish	Muscle	0.004	(T)		British Columbia, Can.	586	1968
Yellowtail rockfish	Remains[f]	0.042	(E)		British Columbia, Can.	586	1968
Yellowtail rockfish	Remains	0.006	(D)		British Columbia, Can.	586	1968
Yellowtail rockfish	Remains	0.021	(T)		British Columbia, Can.	586	1968
Yellowtail rockfish	Muscle	0.119	(E)		Washington Coast	586	1968
Yellowtail rockfish	Muscle	0.222	(D)		Washington Coast	586	1968
Yellowtail rockfish	Muscle	0.480	(T)		Washington Coast	586	1968
Yellowtail rockfish	Remains	0.256	(E)		Washington Coast	586	1968
Yellowtail rockfish	Remains	0.055	(D)		Washington Coast	586	1968
Yellowtail rockfish	Remains	0.104	(T)		Washington Coast	586	1968
Notothenia	Whole	0.002-0.013	(E)	Dieldrin 0.001-0.009 Alpha BHC 0.001-0.007	Antarctica	594	1967
Notothenia	Whole	0.000-0.018	(D)	Beta BHC 0.002-0.008	Antarctica	594	1967
Notothenia	Whole	0.006-0.020		Gamma BHC 0.001-0.004	Antarctica	594	1967
Notothenia	Whole			Heptachlor epoxide 0.002-0.004	Antarctica	594	1967

Table 6 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a] DDT[d]	Other	Geog. loc.[b]	Ref.	Date[c]
Herring	Whole	0.02-0.40	(E)	Atlantic Coast, Canada	667	1974
Perch	Whole	<0.01-0.24	(E)	Atlantic Coast, Canada	667	1974
Northern anchovy	Whole	0.33-0.59		Northern California	526	1969
Northern anchovy	Whole	0.74-14.0		Southern California	526	1969
English sole	Whole	0.19-0.55		Northern California	526	1969
English sole	Whole	0.76		Southern California	526	1969
Northern anchovy	Whole	14.0		Northern California	526	1969
Shiner perch	Whole	1.0-1.4		Northern California	526	1969
English sole	Whole	0.19-0.55		Northern California	526	1969
English sole	Whole	0.76		Southern California	526	1969
Jack mackeral	Whole	0.56		Southern California	526	1969
Hake	Whole	1.8		Channel Island, Washington	526	1969
Bluefin tuna	Muscle	0.56		Southern California	526	1969
Bluefin tuna	Liver	0.22		Southern California	526	1969
Yellowfin tuna	Liver	0.07		Galapagos Archipelago	526	1969
Yellowfin tuna	Liver	0.62		Southern California	526	1969
Skipjack tuna	Liver	0.057		Hawaii	526	1969
Skipjack tuna	Muscle	0.051		Hawaii	526	1969
Skipjack tuna	Liver	0.056		Galapagos Archipelago	526	1969
Hake	Whole	0.18		Puget Sound, Washington	526	1969
Catfish	Whole	0.34-1.74		Pacific Coast, Guatemala	345	1973
Snook	Whole	0.25-0.53		Pacific Coast, Guatemala	345	1973

Table 6 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a]		Geog. loc.[b]	Ref.	Date[c]
		DDT[d]	Other			
Broad headed sleeper goby	Whole	0.11-9.08		Pacific Coast, Guatemala	345	1973
Sleeper	Whole	0.04-0.55		Pacific Coast, Guatemala	345	1973
Mullet	Whole	2.76-36.54	Toxaphene detected	Pacific Coast, Guatemala	345	1973
Jack	Whole	0.06		Pacific Coast, Guatemala	345	1973
Goby	Whole	0.54		Pacific Coast, Guatemala	345	1973
Four eye fish	Whole	4.23-4.84		Pacific Coast, Guatemala	345	1973
Mojarra	Whole	0.23-2.75		Pacific Coast, Guatemala	345	1973
Molly	Whole	2.10-45.17		Pacific Coast, Guatemala	345	1973
Sardine	Whole	15.34-19.44		Pacific Coast, Guatemala	345	1973
Anchovy	Whole	0.69		Pacific Coast, Guatemala	345	1973
Sea trout	Whole	0.64		Pacific Coast, Guatemala	345	1973
Bocaccio	Liver	519.0-590.0		Southern California	402	1974
Bocaccio	Muscle	11.6-12		Southern California	402	1974
Starry rockfish	Liver	1026-1030		Southern California	402	1974
Starry rockfish	Muscle	57-57.6		Southern California	402	1974
Vermillion rockfish	Liver	163		Southern California	402	1974

Table 6 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a]		Geog. loc.[b]	Ref.	Date[c]
		DDT[d]	Other			
Vermillion rockfish	Muscle	161		Southern California	402	1974
Dover sole	Liver	63		Southern California	402	1974
Dover sole	Muscle	13		Southern California	402	1974
Sablefish	Liver	103		Southern California	402	1974
Sablefish	Muscle	23		Southern California	402	1974
California smoothtongue	Whole	0.49		Southern California	402	1974
Eelpout	Whole	5.63		Southern California	402	1974
Hatchetfish	Whole	0.09		Southern California	402	1974
Bocaccio	Fat	115.0		Southern California	402	1974
Herring	Muscle	0.05-0.21	Dieldrin 0.01-0.08	Scottish Coast	301	1971
Sprats	Muscle	0.10-0.21	Dieldrin 0.03-0.09	Scottish Coast	301	1971
Striped mullet	Whole	0.58	Dieldrin 0.39 BHC 1.14	Mississippi River	293	1971
Rainbow trout	Whole	0.73	Dieldrin 0.04 BHC 0.01	Snake River, Idaho	293	1971
Herring	Whole	0.22-0.57		Arch. of Pitea, Baltic Sea	325	1971
Herring	Whole(EF)	14-37		Arch. of Pitea, Baltic Sea	325	1971
Herring	Whole	0.12-1.4		Middle of Baltic Sea	325	1971
Herring	Whole(EF)	3.2-37		Middle of Baltic Sea	325	1971
Herring	Whole	1.6-0.89		Southern Baltic Sea	325	1971
Herring	Whole(EF)	18-72		Southern Baltic Sea	325	1971
Cod	Muscle(EF)	13		Northern Baltic Sea	325	1971
Cod	Muscle	0.086		Northern Baltic Sea	325	1971
Cod	Liver(EF)	30		Northern Baltic Sea	325	1971

Table 6 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a]		Geog. loc.[b]	Ref.	Date[c]
		DDT[d]	Other			
Cod	Liver	8.5		Northern Baltic Sea	325	1971
Cod	Muscle(EF)	12-20		Middle Baltic Sea	325	1971
Cod	Muscle	0.067-0.12		Middle Baltic Sea	325	1971
Cod	Liver(EF)	47-64		Middle Baltic Sea	325	1971
Cod	Liver	13-17		Middle Baltic Sea	325	1971
Cod	Muscle(EF)	8.4-22		Southern Baltic Sea	325	1971
Cod	Muscle	0.064-0.13		Southern Baltic Sea	325	1971
Cod	Liver(EF)	28-59		Southern Baltic Sea	325	1971
Cod	Liver	11-18		Southern Baltic Sea	325	1971
Cod	Muscle(EF)	5.8-15		Mouth of Baltic Sea	325	1971
Cod	Muscle	0.032-0.11		Mouth of Baltic Sea	325	1971
Cod	Liver(EF)	19-47		Mouth of Baltic Sea	325	1971
Cod	Liver	4.4-13		Mouth of Baltic Sea	325	1971
Salmon	Whole(EF)	10-60		Baltic Sea	325	1971
Salmon	Whole	0.51-7.2		Baltic Sea	325	1971
Whitefish	Whole(EF)	0.99-16		Baltic Sea	325	1971
Whitefish	Whole	0.017-0.41		Baltic Sea	325	1971
Vendace	Whole(EF)	0.56-8.9		Baltic Sea	325	1971
Vendace	Whole	0.034-0.13		Baltic Sea	325	1971
Sprat	Whole(EF)	1.6-76		Baltic Sea	325	1971
Sprat	Whole	0.058-4.4		Baltic Sea	325	1971
Flounder	Whole(EF)	1.9-43		Baltic Sea	325	1971
Flounder	Whole	0.024-0.70		Baltic Sea	325	1971
Plaice	Whole(EF)	2.4-8.5		Baltic Sea	325	1971
Plaice	Whole	0.011-0.076		Baltic Sea	325	1971
Pinfish	Whole	0.01-1.11		Northern Gulf Coast, Florida	272	1970

Table 6 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a]		Geog. loc.[b]	Ref.	Date[c]
		DDT[d]	Other			
Atlantic croaker	Whole	<0.01-0.21		Northern Gulf Coast, Florida	272	1970
Spot	Whole	<.01-0.89		Northern Gulf Coast, Florida	272	1970
Pigfish	Whole	.01-1.07		Northern Gulf Coast, Florida	272	1970
Silver perch	Whole	.02-1.26		Northern Gulf Coast, Florida	272	1970
Dwarf perch	Whole	0.094-0.366		San Francisco Bay, California	196	1971
Shiner perch	Whole	0.012-0.281		San Francisco Bay, California	196	1971
Pile perch	Whole	0.033-0.410		San Francisco Bay, California	196	1971
Starry flounder	Whole	0.073-0.127		San Francisco Bay, California	196	1971
White perch	Whole	0.028-0.227		San Francisco Bay, California	196	1971
English sole	Whole	0.020-0.124		San Francisco Bay, California	196	1971
Staghorn sculpin	Whole	0.027-0.140		San Francisco Bay, California	196	1971
Speckled sanddab	Whole	0.025-0.090		San Francisco Bay, California	196	1971
Starry flounder	Whole	0.039-0.126		San Francisco Bay, California	196	1971
Mackeral	Whole	<0.01->2.00(T)		Gulf of St. Lawrence, Canada	159	1968
Mackeral	Whole	<0.01-1.00 (D)		Gulf of St. Lawrence, California	159	1968
Mackeral	Whole	<0.01-2.00 (E)		Gulf of St. Lawrence, California	159	1968

Table 6 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a] DDT[d]	Residues[a] Other	Geog. loc.[b]	Ref.	Date[c]
Salmon	Muscle	<0.01-6.8 (T)		Gulf of St. Lawrence, California	159	1968
Salmon	Muscle	<0.01-2.4 (D)		Gulf of St. Lawrence, California	159	1968
Salmon	Muscle	0.1-20.8 (E)		Gulf of St. Lawrence, California	159	1968
Rainbow trout	Whole		HCB 0.002	Missouri	329	1974
Striped bass	Whole		HCB 0.090	Florida	329	1974
Menhaden	Whole		HCB 0.11	Florida	329	1974
Coho salmon	Whole		HCB <0.010-0.010	Michigan	329	1974
Striped bass	Whole		HCB 0.007-0.050	Maryland, Florida	329	1974
Chinook salmon	Whole		HCB 0.002	Oregon	329	1974
Rainbow trout	Whole		HCB <0.001-0.005	Missouri, New Hampshire	329	1974
Mexican lampfish	Whole	0.014-0.058		Gulf of California, Mexico	133	1970
White croaker	Muscle	0.96-18.56		Los Angeles Coast, California	102	1972
White croaker	Muscle & skin	1.95-30.64		Los Angeles Coast, California	102	1972
Salmon	Whole	4.30-14.6 (E)		Maine	22	1970
Salmon	Whole	0.25-2.72 (D)		Maine	22	1970
Salmon	Whole	0.25-3.61 (T)	Dieldrin 0.011-0.037	Maine	22	1970
Lake trout	Belly tissue		Heptachlor 0.019-0.029	Lake Superior, Mich.	480	1973
Lake trout	Belly tissue		Heptachlor epoxide 0.167	Lake Superior, Mich.	480	1973

[a] Residues (parts per million) expressed on the basis of sample wet weight unless otherwise denoted (i.e., EF=extractable fat weight basis; OD=oven dry weight basis).

Table 6 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN ESTUARINE/MARINE FISHES

[b] Geographic location within USA unless otherwise indicated.
[c] Refers to date of publication and not necessarily date of sample collection.
[d] Total DDT residue unless otherwise denoted (i.e., T=DDT; D=DDD(TDE); E=DDE).
[e] Dash indicates below limits of detection. Space indicates residue was not sought (or not reported).
[f] "Remains" = fish after removal of fillets.

Table 7. PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a]		Geog. loc.[c]	Ref.	Date[d]
		DDT[b]	Other			
Penguin	Liver	0.002-0.028	Dieldrin 0.001-0.006 Heptachlor epoxide 0.000-0.006 BHC 0.002-0.013	Antarctica	594	1967
Penguin	Fat	0.018-0.066	Dieldrin 0.004-0.010 Heptachlor epoxide 0.001-0.003 BHC 0.002-0.010	Antarctica	594	1967
Penguin	Stomach contents	0.002-	Dieldrin 0.001 BHC 0.001-0.005	Antarctica	594	1967
Penguin	Eggs	0.022-0.049	Dieldrin 0.003-0.008 BHC 0.006-0.010	Antarctica	594	1967
Brown skua	Liver	1.12-4.33	Dieldrin - [e] Heptachlor epoxide 0.035-0.100	Antarctica	594	1967
Brown skua	Fat	6.690-28.50	Dieldrin - BHC - 0.050 Heptachlor epoxide 0.120-0.730	Antarctica	594	1967
Blue eyed shag	Liver	0.014-0.024	Dieldrin 0.001-0.002 Heptachlor epoxide 0.002 BHC 0.008	Antarctica	594	1967
Blue eyed shag	Fat	0.072-0.183	Dieldrin 0.004-0.006 BHC 0.009-0.012	Antarctica	594	1967
Penguin	Liver	0.016-0.115	[e]	Antarctica	563	1966
Penguin	Fat	0.024-0.152		Antarctica	563	1966
Crabeater	Fat	0.039		Antarctica	563	1966
Crabeater	Liver	0.013		Antarctica	563	1966
White pelican	Carcass	63.0	Dieldrin 0.00 Toxaphene 4.0	Tule Lake, N.E. Calif.	349	1966
White pelican	Liver	30.7	Dieldrin 0.00 Toxaphene 8.0	Tule Lake, N.E. Calif.	349	1966
White pelican	Kidney	17.0	Dieldrin - Toxaphene 10.3	Tule Lake, N.E. Calif.	349	1966

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a]		Geog. loc.[c]	Ref.	Date[d]
		DDT[b]	Other			
White pelican	Brain	27.1	Dieldrin - Toxaphene 3.0	Tule Lake, N.E. Calif.	349	1966
White pelican	HLKM[f]	23.4-54.0	Dieldrin 3.2 Toxaphene 3.2	Tule Lake, N.E. Calif.	349	1966
White pelican	HLKMB[g]	12.5	Dieldrin - Toxaphene 0.5	Tule Lake, N.E. Calif.	349	1966
Western grebe	Carcass	75.5	Dieldrin - Toxaphene 0.3	Tule Lake, N.E. Calif.	349	1966
Western grebe	Adipose	459.5	Dieldrin - Toxaphene 31.5	Tule Lake, N.E. Calif.	349	1966
Western grebe	HLKM	16.6	Dieldrin - Toxaphene 0.2	Tule Lake, N.E. Calif.	349	1966
Western grebe	Brain	14.5	Dieldrin - Toxaphene -	Tule Lake, N.E. Calif.	349	1966
American egret	Carcass	102.7	Dieldrin - Toxaphene 9.2	Tule Lake, N.E. Calif.	349	1966
American egret	HLKMB	42.8	Dieldrin - Toxaphene 0.00	Tule Lake, N.E. Calif.	349	1966
Great blue heron	Carcass	3.0	Dieldrin - Toxaphene 10.0	Tule Lake, N.E. Calif.	349	1966
Black crowned night heron	Carcass	12.0-18.0	Dieldrin - Toxaphene 15.0	Tule Lake, N.E. Calif.	349	1966
Black crowned night heron	HLKMB	4.0	Dieldrin - Toxaphene 0.00	Tule Lake, N.E. Calif.	349	1966
Double crested cormorant	Carcass	24.0	Dieldrin - Toxaphene 9.5	Tule Lake, N.E. Calif.	349	1966
California gull	HLKM	16.1	Dieldrin - Toxaphene 0.00	Tule Lake, N.E. Calif.	349	1966
Ring-billed gull	HLKMB	1.3-9.6	Dieldrin - Toxaphene 0.00	Tule Lake, N.E. Calif.	349	1966
Blue-winged teal	Whole		Toxaphene 7	Nebraska Lake, Neb.	349	1966

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a]		Geog. loc.[c]	Ref.	Date[d]
		DDT[b]	Other			
Shoveller	Whole		Toxaphene 12	Nebraska Lake, Neb.	349	1966
Sandpiper	Whole		Toxaphene 10	Nebraska Lake, Neb.	349	1966
Black crowned night heron	Whole		Toxaphene 64	Nebraska Lake, Neb.	349	1966
Coot	Whole		Toxaphene 17	Nebraska Lake, Neb.	349	1966
Mallard	Whole		Toxaphene 10	Nebraska Lake, Neb.	349	1966
Black duck	Eggs	0.115	Dieldrin 0.025 Heptachlor epoxide 0.0025	Nova Scotia, Canada	387	1973
Black duck	Eggs	0.24	Dieldrin 0.025 Heptachlor epoxide 0.00	New Brunswick, Canada	387	1973
Black duck	Eggs	0.28-2.10	Dieldrin 0.10 Heptachlor epoxide 0.01-0.02	Quebec, Canada	387	1973
Black duck	Eggs	0.35-1.90	Dieldrin 0.05 Heptachlor epoxide 0.01-0.02	Ontario, Canada	387	1973
Black duck	Eggs	0.36-2.43	Dieldrin 0.08 Heptachlor epoxide 0.01	Maine	387	1973
Black duck	Eggs	0.19-1.33	Dieldrin - Heptachlor epoxide 0.006-0.02	New Hampshire	387	1973
Black duck	Eggs	0.22-1.27	Dieldrin - Heptachlor epoxide 0.007-0.02	Vermont	387	1973
Black duck	Egg	0.44-2.69	Dieldrin 0.07-0.81 Heptachlor epoxide 0.01-0.05	Massachusetts	387	1973
Black duck	Egg	0.29	Dieldrin - Heptachlor epoxide -	Connecticut	387	1973
Black duck	Egg	0.56-3.52	Dieldrin 0.09-0.11 Heptachlor epoxide 0.01-0.08	New York	387	1973
Black duck	Egg	1.15-4.18	Dieldrin 0.06-0.09 Heptachlor epoxide 0.01-0.04	New Jersey	387	1973
Black duck	Egg	2.25-14.56	Dieldrin 0.05-0.44 Heptachlor epoxide 0.02	Delaware	387	1973

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a]		Geog. loc.[c]	Ref.	Date[d]
		DDT[b]	Other			
Black duck	Egg	0.26-0.72	Dieldrin 0.05-0.39 Heptachlor epoxide 0.01-0.16	Maryland	387	1973
Bald eagle	Carcass	34.9	Dieldrin 6.5 Endrin - Heptachlor epoxide 0.07 Dichlorobenzophene 0.5	Florida	520	1969
Bald eagle	Liver	88.0	Dieldrin 15.7 Endrin 0.1 Heptachlor epoxide 0.1 Dichlorobenzophene 0.8	Florida	520	1969
Bald eagle	Brain	31.6	Dieldrin 7.0 Endrin - Dichlorobenzophene 0.3 Heptachlor epoxide 0.07	Florida	520	1969
Bald eagle	Carcass	356.1	Dieldrin 5.3 Endrin 0.0 Heptachlor epoxide - Dichlorobenzophene 3.5	Florida	520	1969
Bald eagle	Liver	247.0	Dieldrin 5.2 Endrin 0.0 Heptachlor epoxide 0.0 Dichlorobenzophene 2.6	Florida	520	1969
Bald eagle	Brain	123.3	Dieldrin 2.3 Endrin 0.00 Heptachlor epoxide 0.0 Dichlorobenzophene 0.0	Florida	520	1969
Bald eagle	Carcass	9.68	Dieldrin .98 Heptachlor epoxide 0.15	USA	519	1969
Great crested grebe	Egg	0.7-22.0(E)	Dieldrin -- - 0.9 BHC 0.4	England	506	1969
Great crested grebe	Egg	0.1-3.1 (D)		England	506	1969
Great crested grebe	Egg	0.0-3.8 (T)		England	506	1969
Great crested grebe	Embryo	14.3	Dieldrin 0.6	England	506	1969

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS.

Species	Sample	Residues[a]		Geog. loc.[c]	Ref.	Date[d]
		DDT[b]	Other			
Great crested grebe	Liver	0.0-81.0 (E)	Dieldrin 0.00-2.8 BHC 0.1-2.1	England	506	1969
Great crested grebe	Liver	0.0-14.2 (D)	Endrin 0.45	England	506	1969
Great crested grebe	Liver	0.0-2.5 (T)	Aldrin 70.02	England	506	1969
Common murre	Egg	2.12 (E)		Farallon Islands, Cal.	250	1971
Common murre	Egg	240-395 EF		Farallon Islands, Cal.	250	1971
White faced ibis	Brain	0.1-0.7 (E)	Dieldrin 0.4-8.0	Gulf Coast, Texas	207	1972
White faced ibis	Body remainder	0.2-0.4 (E)	Dieldrin 0.3-6.7	Gulf Coast, Texas	207	1972
Double crested cormorant	Egg	8.63-29.4(E)		East Coast, Canada	665	1972
Herring gull	Egg	2.83-5.67(E)		East Coast, Canada	665	1972
Black duck	Egg	1.50		East Coast, Canada	665	1972
Bermuda petrel	Egg	10.71-11.02		Bermuda	659	1968
Bermuda petrel	Addled egg	3.61		Bermuda	659	1968
Bermuda petrel	Chick in egg	4.52-6.08		Bermuda	659	1968
Bermuda petrel	Chick brain	0.57		Bermuda	659	1968
Bermuda petrel	Chick 1-2 days old	6.97		Bermuda	659	1968
Great crested grebe	Breast muscle	1.5-4.8 (E)	Dieldrin 0.55-5.0	Great Britain	622	1967
Heron	Breast muscle	0.1-14.4 (E)	Dieldrin 0-15.0	Great Britain	622	1967
Terns	Egg	0.3-1.5 (E)	Dieldrin 0.1-0.5	Great Britain	622	1967

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a] DDT[b]	Residues[a] Other	Geog. loc.[c]	Ref.	Date[d]
Peregrine falcon[h]	Egg	2.6-30.8 (E)	Dieldrin 0.1-1.1	Great Britain	622	1967
Great crested grebe	Egg	1.5-13.3 (E)	Dieldrin 0.1-1.2	Great Britain	622	1967
Heron	Egg	2.4-21.0 (E)	Dieldrin 2.5-12.8	Great Britain	622	1967
Kittiwake	Egg	0.1-0.2 (E)	Dieldrin -	Great Britain	622	1967
Herring gull	Egg	0.2-0.9 (E)	Dieldrin 0.1-0.4	Great Britain	622	1967
Guillemot	Egg	1.5-4.0 (E)	Dieldrin 0.1-2.0	Great Britain	622	1967
Guillemot	Egg	0.5-0.9 (T)		Great Britain	622	1967
Razorbill	Egg	2.9 (E)	Dieldrin 1.6	Great Britain	622	1967
Shag	Egg	3.1-4.3 (E)	Dieldrin 2.5-3.3	Great Britain	622	1967
Shag	Egg	1.1-1.3 (T)		Great Britain	622	1967
Heron	Breast muscle	0.2-14.2 (E)	Dieldrin 0.0-11.0	Great Britain	622	1967
Heron	Breast muscle	0.0-0.3 (T)	Dieldrin -	Great Britain	622	1967
Heron	Breast muscle	0.0-3.6 (D)		Great Britain	622	1967
Heron	Liver	0.4-190 (E)	Dieldrin 0.0-20.3 Heptachlor epoxide 0.0-1.7	Great Britain	622	1967
Heron	Liver	0.0-0.2 (T)		Great Britain	622	1967
Heron	Liver	0.0-12.0 (D)		Great Britain	622	1967
Heron	Breast muscle	5.2-12.0 (E)	Dieldrin 11.0-15.0	Great Britain	622	1967
Heron	Breast muscle	0.2 (T)		Great Britain	622	1967
Heron	Breast muscle	1.0-3.6 (D)		Great Britain	622	1967
Heron	Liver	6.1-30.0 (E)		Great Britain	622	1967
Heron	Liver	2.0-2.8 (D)	Dieldrin 13.8-20.3	Great Britain	622	1967

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a] DDT[b]		Other	Geog. loc.[c]	Ref.	Date[d]
Heron	Kidney	21.0	(E)	Dieldrin 10.5	Great Britain	622	1967
Heron	Kidney	4.0	(D)		Great Britain	622	1967
Cassin's auklet	Whole	1.0-15.4			Farallon Islands, Cal.	531	1967
Cassin's auklet	Breast muscle	2.0			Farallon Islands, Cal.	531	1967
Cassin's auklet	Brain	0.7			Farallon Islands, Cal.	531	1967
Cassin's auklet	Liver	1.0			Farallon Islands, Cal.	531	1967
Cassin's auklet	Subcu-taneous fat	56.0			Farallon Islands, Cal.	531	1967
Cassin's auklet	Egg	10.8			Farallon Islands, Cal.	531	1967
Western gull	Breast muscle	9.2			Farallon Islands, Cal.	531	1967
Western gull	Brain	1.8			Farallon Islands, Cal.	531	1967
Western gull	Subcu-taneous fat	211			Farallon Islands, Cal.	531	1967
Western gull	Egg	6.5			Farallon Islands, Cal.	531	1967
Pelagic cormorant	Breast muscle	0.8			Tomales Bay, Calif.	531	1967
	Liver	0.7			Tomales Bay, Calif.	531	1967
Brandt's cormorant	Breast muscle	4.4			Tomales Bay, Calif.	531	1967
Brandt's cormorant	Liver	3.3			Tomales Bay, Calif.	531	1967
Brandt's cormorant	Brain	1.2			Tomales Bay, Calif.	531	1967
Brown pelican	Breast muscle	84.5			Monterey Bay, Calif.	531	1967

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a] DDT[b]	Residues[a] Other	Geog. loc.[c]	Ref.	Date[d]
Common murre	Whole	7.3		Monterey Bay, Calif.	531	1967
Ancient murrelet	Whole	0.75		Farallon Islands, Cal.	531	1967
Red phalarope	Whole	1.0		Farallon Islands, Cal.	531	1967
Rhinoceros auklet	Whole	2.7		Farallon Islands, Cal.	531	1967
Fulmer	Whole	1.9		Farallon Islands, Cal.	531	1967
Sooty shearwater	Whole	8.4		Farallon Islands, Cal.	531	1967
Slender billed shearwater	Whole	32		Farallon Islands, Cal.	531	1967
Cassin's auklet	Whole	5.8		Farallon Islands, Cal.	526	1969
Ancient murrelet	Whole	0.75		Monterey Bay, Calif.	526	1969
Fulmar	Whole	0.41-3.4		Monterey Bay, Calif.	526	1969
Red phalarope	Whole	0.78		Monterey Bay, Calif.	526	1969
Rhinoceros auklet	Whole	2.7		Monterey Bay, Calif.	526	1969
Slender billed shearwater	Whole	32.0		Monterey Bay, Calif.	526	1969
Sooty shearwater	Whole	10.3-12.3		Monterey Bay, Calif.	526	1969
Peregrine falcon immature	breast muscle	13		Monterey Bay, Calif.	526	1969
Peregrine falcon adult	breast muscle	104-112		Monterey Bay, Calif.	526	1969
Peregrine falcon	Egg	- - 33.0(E)	BHC 0-2.8	Great Britain	512	1970
Peregrine falcon	Egg	0-0.9 (D)	Heptachlor epoxide 0-4.3	Great Britain	512	1970

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a] DDT[b]		Other	Geog. loc.[c]	Ref.	Date[d]
Peregrine falcon	Egg	0-1.2	(T)	HEOD 0-2.6	Great Britain	512	1970
Guillemot	Egg	0.5-6.5	(E)	BHC 0-0.1	Great Britain	512	1970
Guillemot	Egg	0-0.8	(D)	Heptachlor epoxide -	Great Britain	512	1970
Guillemot	Egg	0-0.4	(T)	HEOD 0-1.4	Great Britain	512	1970
Razorbill	Egg	0.9-5.4	(E)	BHC 0-0.2	Great Britain	512	1970
Razorbill	Egg	0-0.1	(D)	Heptachlor epoxide -	Great Britain	512	1970
Razor bill	Egg	0-1.0	(T)	HEOD 0.2-3.0	Great Britain	512	1970
Kittiwake	Egg	- - 2.2	(E)	BHC -	Great Britain	512	1970
Kittiwake	Egg	0-0.5	(D)	Heptachlor epoxide 0-0.1	Great Britain	512	1970
Kittiwake	Egg	0-0.5	(F)	HEOD 0-0.6	Great Britain	512	1970
Black-headed gull	Egg	0.5-2.8	(E)	Heptachlor epoxide 0-0.10	Great Britain	512	1970
Black-headed gull	Egg	-	(D)	BHC -	Great Britain	512	1970
Black-headed gull	Egg	0-0.10	(T)	HEOD 0-1.2	Great Britain	512	1970
Shag	Egg	0.3-7.9	(E)	BHC -	Great Britain	512	1970
Shag	Egg	0-0.6	(D)	Heptachlor epoxide -	Great Britain	512	1970
Shag	Egg	0-1.3	(T)	HEOD 0.1-4.2	Great Britain	512	1970
Golden plover	Egg	0.43-2.62	(E)	BHC 0.5	Great Britain	512	1970
Golden plover	Egg	-	(D)	Heptachlor epoxide 0.01-0.03	Great Britain	512	1970
Golden plover	Egg	0.05-0.23	(T)	HEOD 0.09-0.11	Great Britain	512	1970
Gannet	Liver	<2-520	(E)EF	Dieldrin <2-126	Great Britain	486	1973
Snowy egret	Egg	20.9	(E)OD		West Coast, Florida	386	1973
Black skimmer	Egg	4.50	(E)OD		West Coast, Florida	386	1973
Least tern	Egg	3.17	(E)OD		West Coast, Florida	386	1973
Brown pelican	Egg	2.46	(E)OD		West Coast, Florida	386	1973

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a]			Geog. loc.[c]	Ref.	Date[d]
		DDT[b]		Other			
Laughing gull	Egg	11.70	(E)OD		West Coast, Florida	386	1973
White ibis	Egg	8.74	(E)OD		West Coast, Florida	386	1973
Great egret	Egg	10.36	(E)OD		West Coast, Florida	386	1973
Great blue heron	Egg	20.0	(E)OD		West Coast, Florida	386	1973
Peregrine falcon	Egg	131	(A)OD		Alaska	384	1970
Peregrine falcon	Brain	58.2	(A)OD		Alaska	384	1970
Peregrine falcon	Fat	752	(A)OD		Alaska	384	1970
Peregrine falcon	Muscle	114	(A)OD		Alaska	384	1970
Peregrine falcon	Liver	398	(A)OD		Alaska	384	1970
Peregrine falcon	Ovary	1,117	(E)OD		Alaska	384	1970
Sandwich tern adult	Liver	1.9-3.6	(E)	Dieldrin 4.7-7.2 Telodrin 0.77-1.6	Netherlands	366	1970
Sandwich tern adult	Liver			Endrin 0.50-0.80	Netherlands	366	1970
Sandwich tern adult	Brain	1.1-2.1	(E)	Dieldrin 2.8-3.4 Telodrin 0.50-1.1	Netherlands	366	1970
Sandwich tern adult	Brain			Endrin 0.30-0.60	Netherlands	366	1970
Sandwich tern adult	Whole	1.9-3.7	(E)	Dieldrin 4.3-7.5 Telodrin 0.80-1.1	Netherlands	366	1970
Sandwich tern adult	Whole			Endrin 0.37-0.70	Netherlands	366	1970
Sandwich tern juvenile	Liver	0.90-3.4	(E)	Dieldrin 1.9-6.6 Telodrin 0.60-1.70	Netherlands	366	1970

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a] DDT[b]	Residues[a] Other	Geog. loc.[c]	Ref.	Date[d]
Sandwich tern juvenile	Liver		Endrin 0.10-1.2	Netherlands	366	1970
Sandwich tern juvenile	Whole	0.45-2.0 (E)	Dieldrin 1.4-4.1 Telodrin 0.19-0.50	Netherlands	366	1970
Sandwich tern juvenile	Whole		Endrin 0.10-0.70	Netherlands	366	1970
Sandwich tern chicks	Liver	2.0-12 (E)	Dieldrin 2.4-12 Telodrin 0.63-3.8	Netherlands	366	1970
Sandwich tern chicks	Liver		Endrin 0.19-1.3	Netherlands	366	1970
Eider (female)	Liver	1.2 (E)	Dieldrin 9.5 Telodrin 0.65	Netherlands	366	1970
Eider (female)	Liver		Endrin 0.47	Netherlands	366	1970
Eider (female)	Brain	0.62 (E)	Dieldrin 2.9 Telodrin 0.33	Netherlands	366	1970
Eider (female)	Brain		Endrin -	Netherlands	366	1970
Eider (female)	Breast muscle	0.13 (E)	Dieldrin 2.3 Telodrin 0.30	Netherlands	366	1970
Eider (female)	Breast muscle		Endrin 1.3	Netherlands	366	1970
Eider (female)	Fat mesenteral	10.0 (E)	Dieldrin 35.0 Telodrin 3.0	Netherlands	366	1970
Eider (female)	Fat mesenteral		Endrin 1.3	Netherlands	366	1970
Eider (dead)	Liver	1.1-5.2 (E)		Netherlands	366	1970
Eider (live)	Liver	0.10-0.24 (E)		Netherlands	366	1970
Peregrine falcon	Egg	107 (E)		Yukon Territory, Can.	349	1972

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a]		Geog. loc.[c]	Ref.	Date[d]
		DDT[b]	Other			
Peregrine falcon		99	(E)	Baja Calif., Mexico	349	1972
Peregrine falcon		47	(E)	Northwest Territory, Canada	349	1972
Peregrine falcon		27	(E)	Alaska	349	1972
Peregrine falcon		21	(E)	Alberta, Canada	349	1972
Peregrine falcon		20	(E)	Northwest Territory, Canada	349	1972
Peregrine falcon		17	(E)	British Columbia, Canada	349	1972
Peregrine falcon		13	(E)	Yukon Territory and Alaska	349	1972
Peregrine falcon		13	(E)	Quebec, Canada	349	1972
Peregrine falcon		12	(E)	Northwest Territory, Canada	349	1972
Peregrine falcon		11	(E)	Quebec, Canada	349	1972
Great blue heron	Egg	13	(E)	Interior plains, USA	349	1972
Gull	Egg	5-71	(E)	USA and Canada	349	1972
Tern	Egg	0.4-11	(E)	Interior plains, USA	349	1972
Double crested cormorant	Egg	5-30	(E)	USA and Canada	349	1972
Brown pelican		0.3-75	(E)	Pacific Coast, USA	349	1972
White pelican	Egg	2	(E)	Interior plains, USA	349	1972
Bald eagle	Egg	2-34	(E)	USA and Canada	349	1972
Gannet	Egg	15	(E)	Atlantic Coast, USA	349	1972

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a]		Geog. loc.[c]	Ref.	Date[d]
		DDT[b]	Other			
Brown booby	Egg	0.3	(E)	Pacific Coast, USA	349	1972
Spotted sandpiper	Whole	2.0	(E)	Alaska	349	1972
Lesser yellowleg	Whole	2.0	(E)	Alaska	349	1972
Common snipe	Whole	1.6	(E)est	British Columbia	349	1972
Dunlin	Whole	1.1	(E)est	British Columbia	349	1972
Killdeer	Whole	0.64	(E)est	British Columbia	349	1972
Long-billed curlew	Whole	3.2	(E)est	California	349	1972
Willet	Whole	2.5	(E)est	California	349	1972
Black belly plover	Whole	1.4	(E)est	California	349	1972
Marbled godwit	Whole	1.3	(E)est	California	349	1972
American avocet	Whole	1.0	(E)est	California	349	1972
Red phalarope	Whole	0.62	(E)est	California	349	1972
Dunlin	Whole	0.52	(E)est	California	349	1972
Short-billed dowitcher	Whole	0.25	(E)est	California	349	1972
Least sandpiper	Whole	0.24	(E)est	California	349	1972
Willet	Whole	3.9	(E)est	Manitoba, Canada	349	1972
Northern phalarope	Whole	0.56-3.7	(E)est	Manitoba, Canada	349	1972
Lesser yellowlegs	Whole	0.27-2.0	(E)est	Manitoba, Canada	349	1972
Semipalmated plover	Whole	0.30	(E)est	Manitoba, Canada	349	1972

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a]		Geog. loc.[c]	Ref.	Date[d]
		DDT[b]	Other			
American woodcock	Whole	0.6-13.0	(E)est	New Brunswick, Canada	349	1972
Common snipe	Whole	0.04	(E)est	New Foundland, Canada	349	1972
Long billed curlew	Whole	14.0	(E)est	Alberta, Canada	349	1972
Willet	Whole	0.72	(E)	Alberta, Canada	349	1972
Marbled Godwit	Whole	0.33	(E)	Alberta, Canada	349	1972
Common murre	Egg	14.6	(E)est	Pacific Coast, USA	349	1972
Cassin's auklet	Egg	10.5	est	Pacific Coast, USA	349	1972
Common murre	Whole	6.8		Pacific Coast, USA	349	1972
Cassin's auklet	Fat	5.2	est	Pacific Coast, USA	349	1972
Cassin's auklet	Whole	4.8		Pacific Coast, USA	349	1972
Craveri murrelet	Egg	1.1-3.1	est	Far off USA Coast, Pacific	349	1972
Craveri murrelet	Whole	2.6		Pacific Coast, USA	349	1972
Rhiniceros auklet	Fat	1.2	est	Far off USA Coast, Pacific	349	1972
Ancient murrelet	Egg	0.92		Far off USA Coast, Pacific	349	1972
Ancient murrelet	Fat	0.67	est	Pacific Coast, USA	349	1972
Ancient murrelet	Fat	0.53		Far off USA Coast, Pacific	349	1972
Marbled murrelet	Fat	0.19	est	Far off USA Coast, Pacific	349	1972

Table 7 (continued). PRESENCE OF ORGANOCHLORINE PESTICIDES IN FISH-EATING BIRDS

Species	Sample	Residues[a] DDT[b]	Residues[a] Other	Geog. loc.[c]	Ref.	Date[d]
Common murre	Egg	0.77		Far off USA Coast, Atlantic	349	1972
Common puffin	Egg	0.67		Far off USA Coast, Atlantic	349	1972
Dovekie	Fat	0.15	est	Far off USA Coast, Atlantic	349	1972

[a] Residue data are expressed as parts per million based on wet weight unless otherwise designated (i.e., est.=estimated wet weight derived by dividing residue concentration based on extractable fat by 10; EF=extractable fat basis; OD=oven dry weight basis).
[b] Total DDT residues unless otherwise designated (i.e., E=DDE; D=DDD(TDE); T=DDT).
[c] Geographic location within USA unless otherwise indicated.
[d] Refers to date of publication and not necessarily date of sample collection.
[e] Dash indicates below detectable limits. Space indicates residue was not sought (or not reported).
[f] Composite of heart, liver, kidney and muscle.
[g] Composite of heart, liver, kidney, muscle and brain.
[h] Peregrine falcon (Falco peregrinus) is included because it, in turn, preys on many species of fish-eating birds.

salmon from Michigan and striped bass from Maryland and Florida (329).

Industrial Toxicants

Polychlorinated biphenyls (PCB's) are as widely distributed as DDT. Because of similar molecular shape and composition, the physical and chemical properties of PCB's also confer the same lipophilic characteristic that allows biological accumulation and food chain magnification (see Section II).

Estuarine organisms like fiddler crabs and shrimp readily pick up PCB's from the sediments (465) and filter-feeding oysters accumulate these chemicals, like organochlorine pesticides, from the water (395).

Like the organochlorine pesticides, PCB's accumulate to high levels in organisms representing the tops of food chains (see Tables 8 and 9 for industrial toxicants found in estuarine/marine fishes and fish-eating birds, respectively). Fat from the eggs of California brown pelicans contained 200 ppm PCB's while similar samples from the Baltic white-tailed eagle contained 540 ppm (528). Although chemically-related, there is some solid analytical evidence that PCT's are not as widespread as PCB's in the environment (668). However, PCT's were found in the eggs and fat of herring gulls from Nova Scotia (205).

There is an ever increasing list of "industrial toxicants" that have been found in our waterways, many of which lead directly to the estuaries. Phthalate esters have been found in water collected from the Charles River in New England. Levels of 0.88 - 1.9 ppb were reported with higher levels associated with increasing distances upstream (296). Mayer, et al. (416) reported on PAE's in selected samples from North America. They found from 0.09 ppb DNBP (di-n-butyl phthalate) in Missouri River water to 200 ppb in Mississippi River channel catfish and 500 ppb in tadpoles. Similar values for another phthalate, DEHP (di-2-ethylhexyl phthalate), were 4.9, 400 and 300 ppb. These residue levels were roughly comparable to PCB levels in the same samples.

Although the above rivers drain directly into estuaries and one suspects that phthalates, like other adsorbed toxicants, would "salt out" upon reaching the saline environment, apparently very little published research on phthalates has been directed specifically towards that habitat. Phthalate esters were, however, detected in the eggs of gulls and double-crested cormorants, in the blubber of a common seal (Phoca vitulina) pup, in commercial fish food and in hatchery-reared juvenile Atlantic salmon (205).

Although the preliminary work of Bowes, et al. (63) was aimed at determining levels of chlorinated dibenzofurans and dibenzodioxins in wildlife populations exhibiting embryonic mortality, it did not reveal either of these two compounds. However, they reported hexachloronaphthalene in gull eggs but no chlorinated compounds of interest in sea lion samples.

Table 8. PRESENCE OF INDUSTRIAL TOXICANTS IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a,b]	Geog. loc.[c]	Ref.	Date[d]
Northern anchovy	Whole	1.0	Los Angeles, California	526	1969
Shiner perch	Whole	0.4-1.2	Northern California	526	1969
English perch	Whole	0.04-0.11	Northern California	526	1969
Jack mackeral	Whole	0.02	Southern California	526	1969
Hake	Whole	0.12	Puget Sound	526	1969
Bluefin tuna	Muscle	0.04	Southern California	526	1969
Bluefin tuna	Liver	0.04	Southern California	526	1969
Yellowfin tuna	Liver	-[e]	Galapagos Archipelago	526	1969
Yellowfin tuna	Liver	0.04	Southern California	526	1969
Skipjack tuna	Liver	0.01	Hawaii	526	1969
Skipjack tuna	Muscle	-	Hawaii	526	1969
Skipjack tuna	Liver	-	Galapagos Archipelago	526	1969
Sardine	Whole	0.68-4.72	Mediterranean Coast, Spain	35	1973
Sardine	Whole	0.34-0.40	Atlantic Coast, Spain	35	1973
Herring	Whole	0.03-2.28	Atlantic Coast, Canada	667	1974
Perch	Whole	0.04-0.29	Atlantic Coast, Canada	667	1974
Herring	Muscle	0.1-0.9	Scottish Coast	301	1971
Sprats	Muscle	0.1-0.6	Scottish Coast	301	1971
Striped mullet	*Whole*	*1.39 (estimate)*	*Mississippi River, Louisiana*	293	1971
Rainbow trout	Whole	0.55 "	Snake River, Idaho	293	1971
Herring	Whole	0.19-0.44	Northern Baltic Sea	325	1971
Herring	Whole (EF)	5.4-37	Northern Baltic Sea	325	1971
Herring	Whole	0.16-0.87	Middle Baltic Sea	325	1971
Herring	Whole (EF)	7.1-24	Middle Baltic Sea	325	1971
Herring	Whole	0.24-0.61	Southern Baltic Sea	325	1971

Table 8 (continued). PRESENCE OF INDUSTRIAL TOXICANTS IN ESTUARINE/MARINE FISHES

Species	Sample		Residues[a,b]	Geog. loc.[c]	Ref.	Date[d]
Herring	Whole	(EF)	25-62	Southern Baltic Sea	325	1971
Cod	Muscle	(EF)	7.9	Northern Baltic Sea	325	1971
Cod	Muscle		0.052	Northern Baltic Sea	325	1971
Cod	Liver	(EF)	21	Northern Baltic Sea	325	1971
Cod	Liver		5.9	Northern Baltic Sea	325	1971
Cod	Muscle	(EF)	8-13	Middle Baltic Sea	325	1971
Cod	Muscle		0.056-0.078	Middle Baltic Sea	325	1971
Cod	Liver	(EF)	18-32	Middle Baltic Sea	325	1971
Cod	Liver		4.5-9.5	Middle Baltic Sea	325	1971
Cod	Muscle	(EF)	5.5-9.9	Southern Baltic Sea	325	1971
Cod	Muscle		0.046-0.056	Southern Baltic Sea	325	1971
Cod	Liver	(EF)	11-20	Southern Baltic Sea	325	1971
Cod	Liver		4.1-6.1	Southern Baltic Sea	325	1971
Cod	Muscle	(EF)	5.5-13	Mouth of Baltic Sea	325	1971
Cod	Muscle		0.033-0.096	Mouth of Baltic Sea	325	1971
Cod	Liver	(EF)	4.1-18	Mouth of Baltic Sea	325	1971
Cod	Liver		1.4-4.9	Mouth of Baltic Sea	325	1971
Salmon	Whole	(EF)	10-60	Baltic Sea	325	1971
Salmon	Whole		0.39-2.7	Baltic Sea	325	1971
Whitefish	Whole	(EF)	0.84-17	Baltic Sea	325	1971
Whitefish	Whole		0.015-0.35	Baltic Sea	325	1971
Vendace	Whole	(EF)	0.59-20	Baltic Sea	325	1971
Vendace	Whole		0.059-0.36	Baltic Sea	325	1971
Sprat	Whole	(EF)	0.93-57	Baltic Sea	325	1971
Sprat	Whole		0.061-1.5	Baltic Sea	325	1971
Flounder	Whole	(EF)	1.3-87	Baltic Sea	325	1971
Flounder	Whole		0.026-0.98	Baltic Sea	325	1971

Table 8 (continued). PRESENCE OF INDUSTRIAL TOXICANTS IN ESTUARINE/MARINE FISHES

Species	Sample	Residues[a,b]	Geog. loc.[c]	Ref.	Date[d]
Plaice	Whole (EF)	0.9-14	Baltic Sea	325	1971
Plaice	Whole	0.015-0.14	Baltic Sea	325	1971
Eel	Whole (EF)	0.4	Zuider Zee, Netherlands	217	1973
Eel	Whole (EF)	4.7	Zuider Zee, Netherlands	217	1973
Lake trout	Belly tissue	2.7-13.8	Lake Superior, Michigan	480	1973
Lake trout	Belly tissue	Aroclor 1242 1.4-3.4 (30.8% of total)	Lake Superior, Michigan	480	1973
Lake trout	Belly tissue	Aroclor 1248 1.1-4.1 (26.4% of total)	Lake Superior, Michigan	480	1973
Lake trout	Belly tissue	Aroclor 1254 1.8-7.7 (37.2% of total)	Lake Superior, Michigan	480	1973
Lake trout	Belly tissue	Aroclor 1262 0.9-1.8 (5.6% of total)	Lake Superior, Michigan	480	1973

[a] Unless otherwise designated, data are for the polychlorinated biphenyl (PCB) Aroclor 1254 or particular Aroclor not indicated by author(s).
[b] Residues (parts per million) expressed on the basis of sample wet weight unless otherwise denoted (i.e., EF=extractable fat weight basis; OD=oven dry weight basis).
[c] Geographic location within USA unless otherwise indicated.
[d] Refers to date of publication, not date of sample collection.
[e] Dash indicates below limits of detection.

Table 9. PRESENCE OF POLYCHLORINATED BIPHENYLS IN FISH-EATING BIRDS

Species	Sample	Residues[a]		Geog. loc.[b]	Ref.	Date[c]
Sandwich tern	Egg	1-10		Great Britain	507	1970
Razorbill	Egg	7		Great Britain	507	1970
Little auk	Liver	1		Great Britain	507	1970
Guillemot	Liver	8		Great Britain	507	1970
Guillemot	Egg	5		Great Britain	507	1970
Kingfisher	Liver	0-40		Great Britain	507	1970
Gannet	Liver	<4-9.57	EF	Great Britain	486	1973
Snowy egret	Egg	161	OD	West Coast, Florida	386	1973
Black skimmer	Egg	2.10	OD	West Coast, Florida	386	1973
Least tern	Egg	11.6	OD	West Coast, Florida	386	1973
Brown pelican	Egg	1.30	OD	West Coast, Florida	386	1973
Laughing gull	Egg	17.2	OD	West Coast, Florida	386	1973
White ibis	Egg	9.8	OD	West Coast, Florida	386	1973
Great egret	Egg	7.5	OD	West Coast, Florida	386	1973
Great blue heron	Egg	7.5	OD	West Coast, Florida	386	1973
Sooty tern	Fat	0-39.7	EF	Ascensian Island, S. Atlantic	332	1973
Fairy tern	Fat	-[d]	EF	Ascensian Island, S. Atlantic	332	1973
Red phalarope	Whole	1.03		California	349	1972
Long-billed curlew	Whole	0.48		Alberta, Canada	349	1972
Willet	Whole	2.28		Alberta, Canada	349	1972
Marbled godwit	Whole	0.87		Alberta, Canada	349	1972
Peregrine falcon[e]	Eggs	0-6.0		Great Britain	512	1969
Guillemot	Eggs	2.0-8.0		Great Britain	512	1969
Razorbill	Eggs	6.0-7.0		Great Britain	512	1969

Table 9 (continued). PRESENCE OF POLYCHLORINATED BIPHENYLS IN FISH-EATING BIRDS

Species	Sample	Residues[a]	Geog. loc.[b]	Ref.	Date[c]
Kittiwake	Eggs	3.0-8.0	Great Britain	512	1969
Shag	Eggs	3.0-5.0	Great Britain	512	1969
Great crested grebe	Liver	30-40	Great Britain	507	1970
Great crested grebe	Egg	40	Great Britain	507	1970
Bittern	Liver	<1-10	Great Britain	507	1970
Osprey	Liver	5-20	Great Britain	507	1970
Osprey	Egg	0-2	Great Britain	507	1970
Peregrine falcon	Egg	0-6	Great Britain	507	1970
Water rail	Liver	1	Great Britain	507	1970
Moorhen	Egg	0-15	Great Britain	507	1970
Oystercatcher	Egg	0-1	Great Britain	507	1970
Lapwing	Egg	0	Great Britain	507	1970
Ringed plover	Egg	<1-1	Great Britain	507	1970
Great skua	Egg	25	Great Britain	507	1970
Herring gull	Egg	1	Great Britain	507	1970
Black-headed gull	Liver	0	Great Britain	507	1970
Kittiwake	Egg	5	Great Britain	507	1970
Common tern	Liver	0	Great Britain	507	1970
Common tern	Egg	1-2	Great Britain	507	1970
Pink-footed shearwater	Whole	0.4	Chile	528	1972
Sooty shearwater	Whole	1.1	New Zealand	528	1972
Slender billed shearwater	Whole	2.1	Australia	528	1972
Wilson's petrel	Whole	2.1	Hallet Station, Antarctica	528	1972

Table 9 (continued). PRESENCE OF POLYCHLORINATED BIPHENYLS IN FISH-EATING BIRDS

Species	Sample	Residues[a]	Geog. loc.[b]	Ref.	Date[c]
Wilson's petrel	Whole	33	Palmer Station, Antarctica	528	1972
Giant petrel	Whole	0.2	Palmer Station, Antarctica	528	1972
Snow petrel	Whole	0.08	Hallet Station, Antarctica	528	1972
Cassin's auklet	Whole	0.16	Farallon Islands,California	526	1969
Ancient murrelet	Whole	0.15	Monterey Bay, California	526	1969
Fulmar	Whole	0.08-0.34	Monterey Bay, California	526	1969
Red phalarope	Whole	0.10	Monterey Bay, California	526	1969
Rhinocerous auklet	Whole	0.36	Monterey Bay, California	526	1969
Slender billed shearwater	Whole	0.21	Monterey Bay, California	526	1969
Sooty shearwater	Whole	0.9-1.2	Monterey Bay, California	526	1969
Peregrine falcon mature	Breast muscle	22-109	California and Arctic	526	1969
Peregrine falcon immature	Breast muscle	10.5	California	526	1969
Black duck	Egg	0.25	Nova Scotia, Canada	387	1973
Black duck	Egg	0.50	New Brunswick, Canada	387	1973
Black duck	Egg	1.0-2.0	Quebec, Canada	387	1973
Black duck	Egg	0.5-3.00	Ontario, Canada	387	1973
Black duck	Egg	0.50	Maine, Canada	387	1973
Black duck	Egg	0.50	New Hampshire, Canada	387	1973
Black duck	Egg	1.0-2.0	Vermont, Canada	387	1973
Black duck	Egg	1.00-6.90	Massachusetts, Canada	387	1973
Black duck	Egg	--	Connecticut, Canada	387	1973
Black duck	Egg	0.50-2.00	New York, Canada	387	1973
Black duck	Egg	1.00-4.0	New Jersey, Canada	387	1973
Black duck	Egg	1.00-2.00	Delaware, Canada	387	1973

Table 9 (continued). PRESENCE OF POLYCHLORINATED BIPHENYLS IN FISH-EATING BIRDS

Species	Sample	Residues[a]	Geog. loc.[b]	Ref.	Date[c]
Black duck	Egg	0.50	Maryland, Canada	387	1973
Great crested grebe	Egg	38.0	Kent, England	506	1969
Great crested grebe	Egg	28.0-40.0	England	506	1969
Common murre	Egg	5.13	Farallon Islands, California	250	1971
Double-crested cormorant	Egg	17.2-43.5	Atlantic Coast, Canada	665	1972
Herring gull	Egg	5.54-12.6	Atlantic Coast, Canada	665	1972
Black duck	Egg	9.10	Atlantic Coast, Canada	665	1972
Ashy petrel	Whole	21.0	Pacific Coast, California	528	1972
Leach's petrel	Egg	130-370 EF	Baja, California	528	1972
Ashy petrel	Egg	37	Pacific Coast, California	528	1972
Black petrel	Whole	1.0	Gulf of California	528	1972
Least petrel	Whole	0.4	Gulf of California	528	1972
Fulmar	Whole	2.3	Alaska	528	1972

[a] Residue values are in parts per million based on wet weight of sample unless denoted otherwise (i.e., OD=oven dry weight basis; EF=extractable fat weight basis).
[b] Location within USA unless otherwise stated.
[c] Refers to date of publication and not necessarily to date of sample collection.
[d] Dash indicates below limits of detection.
[e] Peregrine falcon (Falco peregrinus) is included because it, in turn, preys on many species of fish-eating birds.

GEOGRAPHIC DISTRIBUTION

Scientists (and/or their sponsors) seem to consider this aspect of synthetic organic compounds (SOC's) fairly important. The fact that approximately 10 percent of publications included in the bibliography (Section VIII) deal with some aspect of 'geographic distribution' attests this interest.

By far, the majority of research on the distribution of SOC's has dealt with organochlorines, some of which included PCB's. However, some work has specifically addressed the distribution of PCB's at the oceanic or global (211, 240, 276, 278, 533), national (39, 59, 211, 325), state or provincial (18, 549), and local level (463).

A variety of media and taxa have been used to characterize geographic distribution of SOC's. However, the most popular estuarine or marine animals seem to be the least suitable from the standpoint of mobility. Migratory birds and, to a lesser extent, mammals have been repeatedly analyzed in this regard (18, 39, 42, 48, 59, 120, 286, 287, 299, 300, 348, 367, 369, 370, 446, 519, 613, 648), while fish (46, 133, 325, 329, 463, 570, 662, 667), shellfish and crabs (68, 72, 105, 110, 208, 463, 570), water and/or plankton (136, 158, 276, 278, 533, 646), and sediments (143, 274, 463, 540) have also been used.

Efforts to reveal the distribution of SOC's in the various large bodies of marine water include the Gulf of Mexico and Caribbean (68, 230), the Atlantic (68, 276, 278, 533), the Baltic (211, 325), and the polar Arctic and Antarctic (338).

The geographic distribution of pesticides has also been discussed in the light of several source considerations, including the location of their manufacture (373), agricultural runoff (540), sewage outfall (549), and atmospheric distribution (1, 579). Review or conference level attention has addressed this topic such as the 1971 work of Zitko and Choi (664), who produced a fairly comprehensive list of PCB's and other halogenated hydrocarbons found in marine fishes, mammals and fish-eating birds. With the exception of Escambia Bay (Florida, U.S.A.), where a PCB-spill had occurred, PCB levels in whole fish or muscle tissue ranged usually between 0.5 and 1 ppm (wet weight basis). Within coastal United States, the higher levels seemed to come from New Jersey and California while salmon from Lake Michigan were reported to contain almost 15 ppm PCB's. Seals (blubber) from the Archipelago of Stockholm contained 6 ppm PCB while those from the Sable Islands contained 20 ppm. By far, the tissues and eggs from fish-eating birds contained the highest levels and widest range of PCB levels. This is partially because of the number of samples analyzed and the global nature of the samples but of great significance is the variety of tissues sampled. For instance, muscle from a shot herring gull (Larus argentatus) contained only 5 ppm PCB but subcutaneous fat contained 75 ppm.

Similarly, Risebrough and de Lappe (528), in 1972, summarized PCB's in fish from representative areas of the oceans and found that the highest levels were in fish from Tokyo Bay and Long Island Sound (New York, U.S.A.). Their compilation indicated that PCB levels in fish were comparable for the Nova Scotia banks and Swedish waters. As a monitor of coastal waters, the eggs of the brown pelican (Pelicanus occidentalis) seem to be useful. Data presented by Risebrough and de Lappe are quite interesting in this respect. The lowest levels were found in pelican eggs (i.e., lipids) from Panama and Venezuela (4 and 5 ppm, respectively); intermediate levels were found in samples from Western Baja (California) and Florida (39-72 and 71 ppm, respectively) while the highest PCB levels (266 and 210 ppm) were found in eggs from pelican colonies in Los Coronados and Anacapa Island (California), respectively.

The NSF/IDOE Pollution Transfer Program (157) is making significant inroads into the distribution of SOC's but was still in the early stages of attaining its original objectives at the time of their writing.

Goldberg et al. (453) in the 1971 SCEP publication estimated that, on a global basis (but excluding estuaries), plankton contained 3×10^7g and fish 6×10^8g of DDT residues. They pointed out that both figures were insignificant fractions of the total annual input to the environment (i.e., 10^{11}g), but did not address the high levels frequently found in estuarine environments.

Estuarine shellfish have been used as biological integrators of SOC's in many scientific endeavors. Nimmo et al. (463) used a variety of estuarine organisms plus sediment cores to elucidate the distribution of the PCB, Aroclor 1254, in Escambia Bay. Check and Canario (105) used the quahog or hard-shell clam (Mercenaria mercenaria) in Rhode Island's coastal waters to characterize the distribution of common organochlorine pesticides. Another shellfish, Crassostrea commercialis, was the animal of choice for Clegg in a similar effort in Moreton Bay (Queensland, Australia).

The most extensive effort along these lines, however, has been that of the National Pesticide Monitoring Program in the United States as described by Butler (676). It (as reported in 1973) involved the analysis of over 8,000 samples for 15 organochlorines in 15 coastal states. DDT residues were ubiquitous, occurring in 63 percent of all samples analyzed. Dieldrin was the next most commonly found organochlorine (15 percent incidence). California, Alabama and Florida were noteworthy as having the greatest incidence of DDT residues in the 101-1,000 ppb (parts per billion) range. Of those states having greater than 5 ppb dieldrin, New York showed the greatest incidence, followed (in descending order) by California, Georgia, Texas and Virginia.

In a later report, Butler (86) reported on the spacial and temporal trends of the above monitoring program. He stated that, "The lowest average incidence of DDT positive samples were found, in order, in

Washington, Georgia and Maine. Highest incidence rates were observed in New Jersey, Alabama, North Carolina and California. However, the largest residues of DDT and its metabolites were found in samples collected from the estuaries of Florida, California and Texas."

Because of the rate at which filter-feeding oysters and clams purge themselves of residues and because many fishes have wide and unknown ranges of movement, some investigators have sought other estuarine organisms as biological integrators. One particularly productive effort was that described by Burnett (72) who used the common surf zone sand crab (Emerita analoga) to delineate the distribution of DDT along the California coast. His findings revealed that animals near the Los Angeles County sewer outfall contained over 45 times as much DDT as did animals near major agricultural drainages. The effluent from a DDT manufacturing plant was the probable source.

SECTION IV

TOXICOLOGICAL EFFECTS OF SYNTHETIC ORGANIC COMPOUNDS ON ESTUARINE LIFE

PESTICIDES

General and Lethal Effects

The sensitivity of a particular taxonomic group to any particular toxicant will vary appreciably. Although toxic to crustaceans, the carbamate Sevin is fairly nontoxic to fish and mammals (394). In very general terms, Table 10 (reworked from Butler, 74) displays the relative toxicities of different pesticide groups to estuarine fauna.

In a toxicity test which included 12 insecticides and 7 species of estuarine fish, the descending order of toxicity was: endrin, DDT, dieldrin, aldrin, dioxathion, heptachlor, lindane, methoxychlor, Phosdrin, malathion, DDVP, and methyl parathion (180). For a more comprehensive listing of the toxic effects on estuarine life, by pesticide, the reader is encouraged to look at Table 3 of reference number 683.

Some organochlorines, like mirex, a chemical used to control the imported fire ant, Solenopsis saevissima, in the southeastern states, are particularly toxic to estuarine organisms. For example, juvenile shrimp and crabs died when exposed to one particle of mirex bait; and 1 ppb (part per billion) mirex in sea water killed 100 percent of the shrimp exposed (396). At the other end of the food chain, correlative evidence exists that the organochlorine, dieldrin, has been responsible for the death of some bald eagles in the United States (39) and peregrine falcons in Great Britain (54).

Hexachlorobenzene has been shown to be especially toxic to birds under laboratory conditions (685), but no tests on estuarine species have been reported to the authors' knowledge.

For a cross section of lethal and other toxic effects of organochlorines, organophosphates, carbamates and other pesticides on estuarine organisms, see Tables 11,12,13 & 14, respectively.

Table 10. RELATIVE SENSITIVITY OF TYPICAL ESTUARINE ORGANISMS TO THREE MAJOR GROUPS OF PESTICIDES. HIGHER NUMBERS REFLECT GREATER SENSITIVITY[a]

Organism	Pesticide type		
	Herbicides	Organophosphates	Organochlorines
Plankton	1	0.5	3
Shrimp	1	1,000	300
Crab	1	800	100
Oyster	1	1	100
Fish	1	2	500

[a] Reworked from Butler (74).

Table 11. EFFECTS OF ORGANOCHLORINE PESTICIDES ON ESTUARINE ORGANISMS

Treatment	Taxa	Observed effects	Ref.
Seven pesticides; .1 to 5 ppb; 5 year monitoring	Clams Oysters	Different species take up pesticides at specific rates. Sublethal long range effects more significant than acute toxicity	84
Endrin, aldrin, heptachlor	Oysters	Linear relation between concentration and shell growth	76
Dieldrin, kepone	Oysters	Sharp threshold of toxicity relative to shell growth	76
DDT, 1 ppb	Clam	No effects for 3 months; 30% mortality 4th month	76
DDT - toxaphene, parathion - together and separately, < 3.0 ppb	Oysters	10% less body weight; tissue changes, loss of resistance to parasite	397
Twelve pesticides ranging from lindane, 9.10 ppm to CoRal 0.11 ppm	Oyster eggs and larvae	50% of eggs develop normally at given concentrations	146
Twelve pesticides ranging from lindane and aldrin <10 ppm, to N3514, <1.0 ppm	Clam eggs	Same as above	146
DDT >1 ppm and <1 ppm	Oyster	Remain closed or show spasmodic shell movements at higher levels; decrease in shell deposition at lower levels	78
DDT in oil spray, .2-1.6 lb/A	Isopods Amphipods Prawns	High mortality High mortality High mortality	571
0.3 to 0.8 lb/A	Blue crab	10-100% mortality	571
repeated applications	Crabs Insects Marsh crabs Fish Molluscs	High mortality High mortality Resistant Some deaths Not affected	
Aldrin 0.2 lb/A	Insects Prawns Crabs Fish	More affected than by DDT Less affected than by DDT Less affected than by DDT Less affected than by DDT	571

Table 11 (continued). EFFECTS OF ORGANOCHLORINE PESTICIDES ON ESTUARINE ORGANISMS

Treatment	Taxa	Observed effects	Ref.
Gamma BHC 0.2 lb/A	Crabs	Most toxic insecticide tested	571
DDT - 2 ppm fed	Fish Shrimp	50% mortality; DDT in dead laboratory animals less than in seemingly healthy ones in field	77
DDT 1-500 ppb	Phytoplankton	Photosynthesis reduced	657
DDT 0.2 lb/A	Fish Crabs	Some mortality among animals that could not avoid pesticides	657
Strobane 0.3 lb/A	Three species crabs	Same as DDT	657
BHC 0.1 lb/A	Fiddler crabs	Lost ability to escape predators	228
Dieldrin 0.6-12 ppb	Sailfin molly	Killed by 72 hours; raised serum glutamic oxaloacetic transaminase to 1,500 to 1,700 units.	688
3.0 ppb		Survived to 120 hours; raised SGOT to 6,006 to 11,954 units	
Dieldrin 3-12 ppb	Sailfin molly	100% mortality 1st to 31st week	689
1.5 and 7.5 ppb		More than half survived to week 34; growth and reproduction adversely affected	
Aldrin and dieldrin	Fiddler crab and trout tissues	Selectively inhibited cholinesterase activity in homogenized tissues. Cholinesterase very sensitive to small amounts of pesticide	253
DDT 1.0 ppb/2 wks	Fish	Maximum concentration reached at 2 weeks	272
0.1 ppb/5 wks		38,000 x test water conc. Loss of 78-87% in 8 weeks	
DDT, endrin, 0.1 to .00001 ppm	Minnows	Avoided water containing pesticides; did not distinguish concentration differences	262
DDT 50 ppm	Eel intestine	Inhibition of water absorption; inhibition of (Na^+ and K^+) activated Mg 2^+ - dependent adenosine triphosphatase	318

Table 11 (continued). EFFECTS OF ORGANOCHLORINE PESTICIDES ON ESTUARINE ORGANISMS

Treatment	Taxa	Observed effects	Ref.
Mirex, 1-5 particles of bait in standing sea water or Mirex in flowing sea water 1.0 to 0.1 ppb	Juvenile shrimp	40 to 100% mortality	396
	Juvenile shrimp	Up to 100% mortality delayed until shrimp in Mirex free water	
	Juvenile blue crab	Up to 96% mortality, delayed	
	Fiddler crabs	Accumulated Mirex in bodies	
	Fish	Accumulated Mirex in bodies. Gill parasites reduced	
DDT 2-5 - ppm	Shrimp Crab Fish Pinfish	35-100% mortality Accumulated DDT in bodies	81
DDT <1 ppm	Oyster	Feeding and shell growth stopped. Erratic shell movements	80
DDT 2-4 ppm on food	Fish Shrimp	50% mortality in 2-4 weeks	80
DDT in flowing sea water 0.1 ppm	Shrimp	Lowered Na^+ and K^+ in hepatopancreas	460
0.05 ppm	Shrimp	Change in Na^+ and K^+ only after day 20	
DDT in flowing sea water 10 ppb	Juvenile shrimp	100% mortality	461
DDT 0.05 to 0.17 ppb	Shrimp	DDT concentrates in hepatopancreas. Flushed from hepatopancreas within 6 weeks	464
0.12 to 0.20 ppb		100% mortality 18 to 28 days	
Mirex .001, .1, 1.0 and 10 ppb	Crab larvae	Larval stages prolonged. Increased mortality	55
DDT 10 ppm on detritus	Fiddler crabs	100% lost coordination by day 5. Three-fold accumulation in claw muscles	471
Toxaphene	Fish Shrimp, crabs	Established 96 hour TL_{50} values; includes data on synergy and histopathology	132
DDE	Duck	Eggshell thinning complete after 4 days on 40 ppm diet; electron microscopy	690

Table 11 (continued). EFFECTS OF ORGANOCHLORINE PESTICIDES ON ESTUARINE ORGANISMS

Treatment	Taxa	Observed effects	Ref.
DDE, dieldrin	Duck	20 ppm DDT or 10 ppm dietary doses resulted in eggshell thinning	147
DDT group, dieldrin, heptachlor, toxaphene	Duck	Established effects on eggshell thinning	257
DDE	Duck	LC_{50} values varied with age of ducks (1200-1600 ppm)	221
Dieldrin .1-50 ppm	Fiddler crabs	Levels correlated with maladaptive behavior and mortality. Latent effects	357

Table 12. EFFECTS OF ORGANOPHOSPHATE PESTICIDES ON ESTUARINE ORGANISMS

Treatment	Taxa	Observed effects	Ref.
Parathion	Oysters	Sharp threshold of toxicity relative to shell growth	76
Four pesticides ranging from guthion .62 ppm to TEPP 10 ppm	Oyster eggs Clam eggs	50% of eggs develop normally	146
Malathion Dursban 10-.01 ppm	Minnows	Did not avoid malathion Did avoid dursban	262
Paraoxon, DDVP Parathion, methyl parathion	Fiddler crab and trout tissues	Selectively inhibited cholinesterase activity in homogenized tissues. Cholinesterase sensitive to small amounts of pesticide	253
Malathion, naled, guthion, parathion	Fishes and pink shrimp	Revealed comparative AChE inhibition	123
Parathion	Duck	Established effect on eggshell thinning	257

Table 13. EFFECTS OF CARBAMATE PESTICIDES ON ESTUARINE ORGANISMS

Treatment	Taxa	Observed effects	Ref.
Sevin 0.1 ppm	Juvenile fish	Survived normally, neural parasite may not be related to toxicant	394
Sevin 0.01-10 ppm	Minnows	Did not avoid Sevin	262
Sevin	Gastropod (oyster drill)	Swelling at 6-7 hours exposure	649
Matacil, mesurol, zectran, Baygon, Sevin	Fiddler crab and trout tissue	Selectively inhibited cholinesterase activity in homogenized tissues. Cholinesterase very sensitive to small amounts of pesticides	253

Table 14. EFFECTS OF HERBICIDES, BACTERIOCIDES, AND OTHER PESTICIDES ON ESTUARINE ORGANISMS

Treatment	Taxa	Observed effects	Ref.
12 herbicides ranging from amitrol 733 ppm to silvex 6 ppm and the nematocide, nemagon 10 ppm	Oyster eggs Clam eggs	50% developed normally	146
Nineteen bacteriocides, algicides, fungicides, from untinted sulmet 1000 ppm to phygon .014 ppm	Oyster eggs Clam eggs	50% developed normally	146
2,4-D acid	Duck	Established effect on eggshell thinning	257
Four herbicides in sea water	Six general algae	Carbohydrate concentration depressed. Varies with salinity	626
2,4-D, 0.01-10 ppm	Minnows	Avoidance of herbicide	267
Antimycin A 7 ppb	Thirty-eight species fish Other fish Oysters Plankton Crabs	Killed in three days No effect	686
Polystream (chlorinated benzenes)	Oyster drill	Under recommended dosage, 50% of animals killed by day seven	649

Sublethal Effects

Environmental contaminants in concentrations too low to cause death in estuarine organisms may, nevertheless, have profound effects on the continued existence of many species. Even minute quantities of toxicants may cause behavioral, biochemical, physiological, reproductive, developmental and other changes that interfere with a species' ability to survive in its habitat. The categorization of these changes is somewhat of an arbitrary one selected for simplicity by the authors. It is obvious, however, that there is great overlap of these categories. Changes in behavior, for example, affect the organism's ability to perform courtship rituals (357, 471, 573), thus affecting reproduction. Physiological and biochemical changes likewise affect growth, development, and reproduction.

Behavior--

In 1955, Jordan (683), trying to control the crab, Sesarma africanum, noted loss of coordination without mortality occurring with exposure to concentrations of crude BHC as low as 0.65 ppm. In 1957, George and his associates (679) observed that, in an estuary sprayed with DDT, BHC, and Strobane, fiddler crabs showed lack of coordination which rendered them unable to escape predators.

Although crustaceans are generally more sensitive to pesticides than fishes (270), little has been done to investigate their ability to detect and avoid pesticide contamination. Hansen et al. (270) found that adult grass shrimp (Palaemonetes pugio) avoided water containing the herbicide 2,4-D at concentrations of 1 and 10 ppm, and selected the lower concentration when given a choice. However they did not avoid water containing DDT, endrin, dursban, malathion, or Sevin. Adult sheepshead minnows (Cyprinodon variegatus) avoided water containing DDT, endrin, dursban, and 2,4-D, but did not avoid Sevin or malathion. When exposed to different concentrations of the same pesticide, fish avoided the highest concentration of 2,4-D, but preferred the higher concentration of DDT (i.e., 0.1 vs. 0.01 ppm). They did not distinguish between other concentrations of 2,4-D or between different concentrations of endrin or dursban (262).

Hansen and co-workers (266), experimenting with mosquitofish (Gambusia affinis), tested the fishes' ability to avoid two organochlorine pesticides, two organophosphates, and one carbamate. They avoided DDT, dursban, malathion, Sevin and 2,4-D in water, but did not avoid endrin. The fish were also able to distinguish between certain concentrations of dursban and 2,4-D, seeking the lower concentration when given a choice between 10 ppm and 1 ppm.

Many vital activities such as escape from predators, feeding, and reproduction are integrally dependent on coordination and control of the organism's movements. In an early study, Springer and Webster (573) noted that fiddler crabs showed sluggish, uncoordinated behavior following DDT spraying of a salt marsh, and became ready prey to carnivorous birds. Clam worms (Nereis sp.) similarly became sluggish following the spraying

of their habitat with DDT (573). Odum and his associates (471) observed that fiddler crabs (Uca pugnax) survived being fed detritus containing 10 ppm DDT residues, but showed loss of coordination, stumbling and rolling over, as well as general sluggishness. Klein and Lincer found that adults of another species of fiddler crab (U. pugilator) were unable or less able to run from threat or to right themselves when fed 0.1 or 1.0 ppm dieldrin. Fiddlers were also unable to convey food to their mouths when fed 10 or 50 ppm dieldrin. Degree of impairment was dose-related and persisted after feeding of dieldrin ended (357).

At least two review articles (581, 583) conclude that the maladaptive behavioral responses of vertebrates in contact with various pesticides include hyperactivity and hypersensitivity to stimuli, rather than loss of coordination. Several strictly freshwater and some euryhaline species of fish displayed the above symptoms and became hyperirritable. Salmon (Salmo salar) in rivers where DDT had been sprayed became oversensitive to cold. Ogilvie and Anderson also reported that Atlantic salmon showed preference for lower temperatures than that to which they had been acclimated after 24 hours exposure to 5 and 50 ppb DDT. In addition. they showed hyperactive avoidance response to temperature change in the water (473). Mosquitofish aborted their young when exposed to DDT, DDD, methoxychlor, aldrin, dieldrin, endrin, toxaphene, heptachlor, and lindane. These pathological behaviors in fish would undoubtedly interfere with the normal reactions in the field that are essential to escape predators, to feed, or to reproduce.

The Stickels', in their respective review articles (581, 583), suggest that hyperirritability and hyperactivity in birds would interfere with proper incubation of eggs. Gulls and eagles increased egg breakage in their nests, possibly due to increased restlessness or clumsiness. Herring gulls containing high DDT residues also became aggressive and restless; behavior which would contribute to egg breakage, especially if the eggshells had been thinned due to DDE or other stresses. Woodwell (653) states that DDT may inhibit reproduction in carnivorous birds; birds which, because of their trophic position, would be most likely to contain the highest residues.

Conditioned response learning has been used as a measure of toxic sublethal effect. Atlantic salmon (S. salar) parr and speckled trout (Salvelinus fontinalis) yearlings showed no effect in avoidance learning by exposure to DDT (316). However, Warner et al. (629) found that conditioning responses were species-specific and dose-dependent in more-strictly freshwater species. For instance, toxaphene and TEPP actually improved learning of avoidance conditioning and habituation rates in goldfish, possibly as a result of increased responsiveness to stimuli.

Lowe reported that although only 22 percent of the blue crabs (Callinectes sapidus) survived when exposed for 9 months to 0.5 and 0.25 ppb DDT, the survivors fed, molted, and grew normally (392). Lowe et al. stated that

juvenile blue crabs and juvenile pink shrimp showed no symptoms of poisoning during 96 hours' exposure to 0.1 ppm mirex, but became irritable and paralyzed, then died within 18 days after being placed in mirex-free water (396).

Salinity preferences and interorganism responses have an important bearing on a species' ability to select appropriate habitat in an estuary. Mosquitofish exposed to DDT selected water of increased salinity, but malathion had no such effect (263). Schooling of juvenile silversides (Menidia menidia) was disrupted by exposure to Sevin. It was pointed out that the group hydrodynamic effect of schooling may be important in conserving energy and survival of schooling fish (635).

As Butler emphasizes, the effects of various chemicals on animal behavior cannot be predicted by chemical classification or chemical relationships (84). Similarly, Warner and his associates point out that since responses are species-specific, and dependent on the background behavior of each species, generalizations cannot be made across species (629).

Growth and Development--

Abnormalities of growth and development of estuarine species in response to minute quantities of synthetic organic compounds, in even a minimally contaminated habitat, will affect the survival of such species. The effects of sublethal quantities of synthetic organic compounds in the estuarine environment are not, generally, immediately observable. Studies on the growth and development of various organisms indicate that effects on these processes, while subtle, may be consistent enough to provide even a bioassay method (82).

There have been attempts to quantify the effects of pesticides on growth and photosynthesis of marine phytoplankton. Menzel et al. (424) exposed four species of phytoplankton to DDT, dieldrin, and endrin. Effects varied both with species and with concentration of toxicants used and effects on cell division ranged from none to total inhibition. Carbon uptake, as a measure of photosynthesis, varied considerably across species. Artificial seawater, supplemented with vitamins and trace elements, was used by Walsh (624) as a growth medium for several species of unicellular algae. The 30 herbicides tested inhibited oxygen evolution and autotrophic growth. The urea and triazine herbicides were the strongest inhibitors of oxygen evolution and the most toxic to growth. Their effects were immediate on all genera, indicating that herbicides are absorbed quickly by the marine algae. Results varied among the algal species tested, but consistently less herbicide was required for inhibition of growth than for inhibition of oxygen evolution. Powers et al. (716) found that DDE, at concentrations as low as 0.1 parts per thousand million (109), significantly inhibited growth of the marine dinoflagellate, Exuviella baltica.

Sears and Yentsch (552) used oxygen production as a measure of photosynthesis by three species of macroscopic algae. There was no consistent

effect of DDT on this function. However, the rate of photosynthesis varied greatly with the amount of stirring of the seawater, a factor which may have affected the observations.

Studies of the effects of pesticides on mollusks are of special interest since their sessile life style makes them suitable for bioassay use (82). Using the oyster (Crassostrea virginica) as a test organism for 200 different pesticides, Butler (82) found that chronic exposure to insecticides and fungicides slowed or halted shell deposition. However, in some instances, growth rate change was not evident until several months after the start of chronic exposure to pesticides. Some pesticides had no significant effect throughout the 5 months of exposure. Butler (84) concluded that many pesticides have a threshold of toxicity below which no ill effects occur despite prolonged exposure. He exposed one group of juvenile (2.5 cm.) oysters to sublethal concentrations of aldrin, malathion, and toxaphene from March through August and another group to dyrene, endrin, naled, and DMPA from July to December. Concentrations used were one-tenth the amount needed to cause a 50 percent decrease in shell growth in 96 hour bioassay test. No difference in growth from controls occurred. Lowe et al. (397) reared juvenile (2.7 cm.) oysters to maturity in seawater containing DDT, toxaphene, and parathion, separately and as a mixture. The individual pesticides caused no significant effects on growth during 9 months exposure to 1 ppb. However a combination of the 3 pesticides, each at 1 ppb, slowed growth rate 10 percent.

The effects of 52 compounds, including insecticides, herbicides, fungicides, etc., were noted on eggs and larvae of hard clams (Mercenaria mercenaria) and oysters. Most compounds were more toxic to larvae than to embryos, although the reverse was occasionally true. Also TLm values were shown useful only for rough comparisons of toxicity since some compounds reduce rates of growth of larvae at concentrations too low to cause appreciable mortality. Conversely, others may kill embryos at concentrations too low to affect growth of larvae. For example both endrin and dieldrin had 14-day TLm's (for oyster larvae) greater than 10 ppm, but at concentrations of only 1 ppm they reduced the rate of growth of these larvae drastically. Other compounds, such as Nemagon, aldrin and toxaphene, permitted development of embryos at higher concentrations than those at which the larvae could survive. Conversely, some compounds (e.g. Griseofulvin and Endothal) almost completely stopped embryonic development at concentrations too low to affect survival and growth of larvae (146).

Seven developmental stages of eggs of bay mussels (Mytilus edulis) were exposed for one hour to various concentrations of Sevin and its first hydrolytic product, 1-naphthol. At all stages, abnormalities of development, such as disjunction of blastomeres, reduction of growth rate, asynchronous and unaligned cleavages, were observed. Sensitivity decreased as age of embryo increased, but eggs from a single female also varied in sensitivity (28). Seed mussels reduced byssal attachment in high concentrations of 5 organochlorines, Carbaryl and Trichlorphon.

Endosulfan was the most toxic of the chemicals tested; 0.045 ppm causing 50 percent reduction in byssal attachment in mussels exposed for 24 hours. Trichlorphon was least toxic, having no effect on byssal attachment in concentrations less than 30 ppm. Queen scallops (Chlamys operculàris) showed greater sensitivity to Endosulfan than mussels (536).

Liu and Lee (705) reported that although adult mussels were able to tolerate a saturated solution (0.20 ppm) of Treflan and embryo shell development was not affected by Treflan at a concentration half that of its solubility in seawater, larval growth and metamorphosis were reduced. Solutions of methoxychlor as high as 0.092 ppm were not lethal to adult mussels, and eggs developed normally. However larval growth was depressed and metamorphosis inhibited. Malathion and 2,4-D adversely affected all life stages of the mussel at concentrations of 20 percent and 25 percent, respectively, of the estimated solubility in seawater of these toxicants.

Lowe (392) found no adverse effects on the growth of juvenile blue crabs (Callinectes sapidus) when exposed to 0.25 ppb DDT. Concentrations near the threshold of tolerance, (i.e. 0.5 ppb) caused pesticide poisoning symptoms, but those crabs surviving this exposure molted normally.

Epifanio (193) fed dieldrin-contaminated Artemia nauplii to the larvae of the crab Leptodius floridanus throughout their development to the megalopa stage. Sublethal effects were noted when food organisms contained 5.49 ppm dieldrin, and 100 percent of the larvae failed to complete development when food contained 33 ppm dieldrin. Because of the pumping rate of larvae, Epifanio suggested that the same effects would occur with considerably lower concentrations of dieldrin in seawater rather than in the food.

In another study, Epifanio (192) placed larvae of the Xanthid crabs, L. floridanus and Panopeus herbstii, in various concentrations of dieldrin in seawater. At 10 ppb dieldrin, larvae of L. floridanus failed to complete development, and only a small percent survived to magalopa in 5 ppm dieldrin. At 1 ppb, this species exhibited high mortality in the first zoeal stage, and time to reach megalopa was significantly increased, but at 0.5 ppb, survival was not affected. Survival of P. herbstii in 1 ppb dieldrin was not affected to the first crab stage. However, the rate of molt was significantly slowed. Toxicity depended more on stage of development than on length of time of exposure, but salinity was also a source of stress.

Results of studies on the effects of pesticides on growth and development of vertebrates, primarily fish, have not been as consistent as those on mollusks. The growth of juvenile spot (Leiostomus xanthurus) exposed to 0.1 ppm Sevin for 5 months in flowing seawater was not affected (394). In another study involving juvenile spot, Lowe (391) observed no difference in growth rate from controls in fish exposed for 5 months to 0.1 and 0.01 ppb toxaphene.

Grunion (Leuresthes tenuis) fry exposed to p,p'-DDT in flowing seawater at concentrations ranging from 0.001 to 500 ppb, developed significant asymmetry of their pectoral fin rays (612). The threshold for this effect was 0.01 ppb and degree of asymmetry was dose-related.

Chadwick and Shumway (104) exposed steelhead trout (Salmo gairdneri) embryos, alevins, and fry to dieldrin at concentrations ranging from 0.012 ppb to 52 ppb for as long as 130 days. Survival and number of days to hatching of embryos were not affected by any concentrations used. However, alevins from eggs exposed to the higher concentrations were smaller than controls at hatching and alevins were more sensitive to dieldrin than were the embryos. At an exposure of 0.39 ppb, there was delay in reaching fry stage and increased mortality, regardless of whether or not the embryo had been exposed to dieldrin earlier. Fry were the most susceptible to dieldrin regardless of previous exposure. When eggs were exposed to a range of 0 to 1.2 ppb dieldrin from time of fertilization for 60 days, no adverse effects on growth occurred in concentrations of 0.12 ppb or less. At 0.39 ppb, growth and survival was greatly reduced. Of this group exposed to dieldrin for 130 days, only 3 percent survived, but the survivors appeared to grow at the same rate as controls.

Weis and Weis (636) exposed early embryos of killifish (Fundulus heteroclitus) to DDT or malathion in concentrations up to 10 ppm. These pesticides had no significant gross developmental effects, but exposure to 10 ppm carbaryl or parathion arrested development prior to initiation of heartbeat. Embryos exposed to 10 ppm carbaryl for 3 days recovered after removal to clean water, however, a 4 1/2 day exposure caused cardiac abnormalities and hatching failure in embryos. Similar severe abnormalities were observed following exposure of embryos to 10 ppm parathion for 3 days and even one day at 1 ppm affected development. Interestingly, malathion, though closely related chemically to parathion, did not have any effect on development at concentrations up to 10 ppm.

Adult winter flounder (Pseudopleuronectes americanus) were exposed to sublethal amounts of DDT and dieldrin in combination. Their eggs showed abnormal gastrulation, and at hatching, 39 percent of the young showed vertebral abnormalities. The percent of eggs fertilized varied with the various combinations of the two pesticides, and exposure of adults to 1.74 ppm dieldrin alone prevented fertilization of eggs. There were dose-related effects on amount of DDT concentrated in the eggs and mortality associated with abnormal egg development. When the amount of DDT in the eggs was 2.39 ppm or greater, embryos that hatched showed vertebral abnormalities (565).

Piavis and Howell (714) observed that 10 ppm of the lampricide, TFM, resulted in an increased incidence of abnormalities in larval sea lamprey. This concentration also drastically reduced the number of viable embryos at one particular stage, and sometimes delayed hatching.

Chromosomal alterations in juvenile mallard ducks, on long-term doses of dieldrin, were only evident at unrealistically high levels (i.e., 100 ppm).

Burch and Low (71) concluded that the levels of dieldrin commonly found in water birds would probably be too low to elicit chromosomal aberrations.

Heath and Spann (289) found that both hen and drake mallards, that were on 1 and 10 ppm mirex for 5 months, ate less and gained significantly less weight than their control counterparts.

Cytology and Histology--

Oysters (Crassostrea virginica) exposed to 1 ppb DDT, toxaphene and parathion (in combination) exhibited abnormal leukocytic infiltration of the gonads and hyperplasia of germinal epithelium. Treated oysters exhibited slight edema beneath the gut, sometimes accompanied by leukocytic infiltration, and a dilation of the digestive tubule epithelium. In this study, Lowe et al. (397) reported that all of the oysters contaminated with the mixture were found to be parasitized by an unidentified mycelial fungus. Since none of the control oysters was parasitized it was assumed that the mixture had caused a breakdown in the oysters' defense mechanism.

Couch (128) found that mirex was capable of increasing the incidence of viral infection in pink shrimp (Penaeus duorarum). Although 6.6 percent of the controls were infected with the virus (given the name Baculovirus penaei), 40 percent of the shrimp exposed to 0.01 - 0.23 ppb mirex for 30 days exhibited infection and associated cytopathology.

Parrish and co-workers (481) observed tissue alterations in spot (Leiostomus xanthurus) after 4 days exposure to 1.35 ppb dieldrin in water. Gill lamellae exhibited subepithelial edema while damage to the visceral tissue included severe lysis and sloughing of the small intestine epithelium and apparent inflammation of the underlying lamina propria. Fish examined after 35 days of exposure to .135, .075, and .0135 ppb dieldrin and at the end of depuration (for the .135 and .075 ppb groups) showed no significant differences from control fish. The same species was exposed by Lowe (391) to 0.1 and 0.01 ppb toxaphene for 5 months. Fish that had been exposed to the 0.1 ppb exhibited a distinct thickening of the gill lamellae and some evidence of liver degeneration; but the latter was inconclusive.

Spot exposed to 0.1 ppm of the carbamate Sevin exhibited central nervous system lesions. These could, however, not be definitely attributed directly to Sevin, as there was evidence that it might have been caused by a parasite (394).

Eller (187) intermittently exposed cutthroat trout (Salmo clarki) to short-term treatments of endrin over a period of 42 weeks. The organochlorine was administered to some animals via the water and to others through contaminated food. Pathological conditions were found in the gills, liver, pancreas, brain and gonads. Hemorrhage, edema and possibly intracapillary congestion characterized the gill damage. Multiple cysts

containing unidentified protozoans infected the gill filaments. Hepatic lesions in young trout were similar to those described as preceding the development of hepatomas in nutritionally deficient fish. The severity of hepatic degeneration suggested nutritional deficiency accentuated by exposure to endrin. Marked hyperplasia of pancreatic islets and irregular, atypical oocytes were also observed.

Reproduction--

Successful reproduction is of obvious value to any species and behavioral changes as well as lack of coordination subtly affect this vital function. If an organism is unable adequately to perform species specific courtship behavior patterns it will be unable to mate: for example, lack of coordination in the fiddler crab (357, 471, 573) might well interfere with the elaborate ritual waving that precedes mating in these crustaceans.

Behavior may not be a factor in reproduction of phytoplankton or sessile invertebrates. On the other hand, Wurster (657) equated reduction of photosynthesis in phytoplankton to reduction of reproduction by cell division. He noted that DDT reduced photosynthesis of 4 species of marine algae; a diatom, a coccolithophore, a green alga, and a dinoflagellate. Concentrations of 10, 100, and 500 ppb yielded sigmoid growth curves typical of a dose-response relationship.

Diuron at concentrations of 1, 5, and 10 ppb, Neburon at 7, 5, 15, and 30 ppb, Monuron at 10, 50, and 100 ppb, and Fenuron at 100, 500, and 1,000 ppb, inhibited cell division of unicellular marine algae. Inhibition ranged from 10 percent at the lowest dosage to 75 percent at the highest (626).

The cell volume of an estuarine dinoflagellate decreased in various concentrations of the insect chemosterilant, Aziridine (Apholate). Prazer and Mahoney (504) speculated that this reflected an effect at the chromosomal level.

Butler (84), reporting on a 5 year bioassay of 240 different pesticides and other chemicals, found no effects on reproduction, but some beneficial effects of growth of oysters (Crassostrea virginica) exposed to sublethal levels of toxicants. However, in another study (74) he observed that oyster gonads stored approximately twice as much DDT as did the digestive organs. There were significant amounts in the gametes themselves: after 12 days exposure to 10 ppb DDT, eggs contained 25 ppm and sperm 9 ppm, while whole unsexed oysters contained an average of 10 ppm DDT. The effect(s) of this concentration in the gametes was not known (77). However, oysters exposed to a mixture of 1 ppb each of DDT, toxaphene, and parathion exhibited retarded gonadal maturation and more numerous immature ova (397).

Although the change was not marked, adult oysters exposed to 1 or 10 ppm abate or 1 or 10 ppm dibrom matured more slowly than controls and spawning was inhibited. Tripp suggested that poor conditions in the laboratory

were a factor since spawning was also somewhat delayed in control animals. A subsequent observation is of interest: when controls were returned to a natural habitat they spawned normally while pesticide treated oysters did not (596).

Exposure of eggs and larvae of oysters and clams (Venus mercenaria) to lindane and guthion led to hatch failure (145). In a report at an FAO Technical Conference (212), it was stated that lindane and guthion prevented hatching of eggs of oysters and mussels (Mytilis edulus), while DDT inhibited the metamorphosis of barnacle (Balanus balanoides) larvae.

Roberts (534) exposed the mussel, Mytilus edulis, to Endosulfan at concentrations of 0.1, 0.5, and 1 ppm. At 0.5 ppm the spawning period was prolonged. At 1 ppm onset of spawning was delayed, possibly due to interference with gamonic action. Developmental stages of the mussel were adversely affected by sevin (28).

Eisler (182) exposed snails (Nassa obsoleta) to DDVP, dioxathion, parathion, and phosdrin. The number of egg cases deposited per survivor did not differ from controls but exposure to lindane at 10 ppm or to dieldrin and endrin at 0.1 ppm reduced the number of egg cases deposited. Eisler also found that low concentrations of the organochlorine pesticides tested (0.01 ppm endrin, dieldrin, lindane, and DDT, or 0.1 and 1 ppm lindane, or 1 ppm DDT) were associated with an increase in number of egg cases deposited.

In a Texas estuary, Butler (82) found an absence of juvenile seatrout (Cynoscion nebulosus). DDT was present in the gonads of adult fish in amounts as high as 8 ppm. The pesticide concentration in the ovaries reached this peak just prior to spawning. Mature fish continued to spawn, but there was a dramatic decline in the juvenile population (88).

A review article by Davis (145) noted a relationship between fry mortality and DDT in hatchery raised lake trout. Highest mortality was among sac fry, especially just after the yolk sac was absorbed.

An FAO Technical Conference (435) reported several instances of pesticide effects on reproduction. Egg viability of trout directly correlated with DDT residue level. Apholate had a negative effect on ovarian development of Fundulus majolis.

The reviews of the early seventies (145, 453, 489) pointed out the initial correlations between DDE and eggshell-thinning in fish-eating birds. Since then, high DDE levels in eggs have been associated with eggshell-thinning and/or reproductive failure in: wild brown pelicans (52); cormorants (Phalacrocorax auritus) and white pelicans (Pelecanus erythrorhynchos) (18); common terns (Sterno hirundo) (215, 284); roseate terns (S. dougalli) (284); herring gulls (Larus argentatus) (234 , 346); common egrets (Casmerodius albus) (200); bald eagles (Haliaetus leucocephalus) (369), and; peregrine falcons (Falco peregrinus) (91).

Not only has reproductive failure in brown pelicans been associated with high DDE residues (52) but, conversely, population stability and reproductive success of at least two California colonies has significantly improved with decreased DDE contamination of anchovies, their major food source (670).

Wiemeyer et al. (731), in a unique egg exchange field experiment with the fish-eating osprey (Pandion haliaetus), found that the poor reproduction of this species is probably related to the contamination of the adults and/or eggs by a series of toxicants, which include dieldrin and DDE.

However, this DDE correlation does not always exist. Switzer and his colleagues (590, 591) found that eggshell-thinning in an isolated colony of common terns was not at all highly correlated to DDE content. Similarly, Potts (503) did not find a correlation between egg DDE content of wild shags (Phalacrocorax aristotelis) and their reproductive failure.

The correlative relationship between dietary DDE and eggshell-thinning in many wild aquatic waterfowl has been characterized by scientists under more controlled aviary and laboratory conditions.

Davison and Sell (147) found that either technical DDT or p,p'-DDT, at 20 ppm, caused significant eggshell-thinning and reduction in calcium. In this experiment, neither compound affected egg weight or egg production by the experimental mallard ducks.

Heath, Spann and Kreitzer (291) reported similar eggshell-thinning for dietary DDE at 10 and 40 ppm, plus shell cracking and embryonic mortality. DDD and DDT also impaired the reproduction of the captive mallards but less so than the DDE.

Another species of duck, the black duck (Anas rubripes) showed the same response to DDE. Longcore and co-workers (388, 390) found that 10 ppm dietary DDE brought about a 22 percent eggshell-thinning (at the equator) and experimentals produced 1/5 as many young as controls.

Dieldrin has been suspected in the reproductive failure of wild brown pelicans (Pelecanus occidentalis) (52), and bald eagles (369), and Potts (503) reported a significant correlation ('threshold type') between dieldrin and clutch/brood failure in shags.

Dieldrin residues in the eggs of gallinules (Porphyrula martinica and Gallinula chloropas), which fed on aldrin-treated rice fields, were not found to be related to clutch size or hatchability. However, the effect upon the survival of the young was not known (103). Davison and Sell (147) reported that 10 ppm dietary dieldrin caused significant eggshell-thinning and calcium reduction but had no effect on egg weight nor egg production of captive mallard ducks.

Other widespread compounds, such as mirex, have not been shown to similarly affect reproduction in birds. Heath and Spann (289) reported

that 1 or 10 ppm dietary mirex induced no perceptible reproductive effects in mallards as measured by eggshell thickness, egg production, shell cracking, embryonation, embryo survival, hatching and survival to 14 days. Hyde et al. (313) also found no significant reproductive effects of up to 100 ppm dietary mirex on the same species.

Research on the reproductive effects of pesticides on marine mammals is quite limited. However, sea lions (Zalophus californianus) on breeding islands off the coast of California exhibited an incidence of premature births substantially higher than normal. Tissue analysis of postpartum females giving birth prematurely showed high levels of DDT (63). DeLong et al. (148) found a mean of 824 ppm DDT in females giving birth prematurely as compared to 103 ppm DDT in females carrying young full term.

Biochemistry and Physiology-

By far, the majority in these areas of endeavor are, again, with organochlorine pesticides. Starting with the microorganism end of the spectrum, Keil et al. (344) exposed E. coli to 0.01 ppm DDT but were not able to find any significant effect on nucleic acid content. However, an increased uridine uptake was noted after 5 hours incubation.

Hollister, Walsh and Forester (698) elucidated the effect of 0.2 ppb mirex on 2 species of marine algae (Chlorococcum sp. and Chlamydomonas sp.). Exposure for 7 days resulted in no significant difference in rates of oxygen evolution between control and treated cultures.

Eisler and Weinstein (186) reported that quahaug clams (Mercenaria mercenaria) exposed to 1.1 ppm methoxychlor exhibited consistent changes in tissue levels of several metals, especially calcium and zinc. They suggested that the metal shifts might provide an early warning system to detect undesirable environmental conditions.

Nimmo and Blackman (460) found that 0.1 ppm DDT in flowing seawater lowered Na^+ and K^+ in shrimp hepatopancreas but a similar change in response to 0.05 ppm occurred only after day 20.

DDT was found to affect the amount of P (from labelled ATP) incorporated into proteins derived from lobster peripheral nerves in a number of ways. The exact effects, however, varied depending on the relative concentrations of ATP, Na^+, K^+, Mg^{++}, and Ca^{++} (154).

Homogenized tissues of fiddler crabs and trout exposed to aldrin and dieldrin were characterized by selective inhibition of cholinesterase (253).

Fish have been a popular taxa for physiological and biochemical studies. This is partially because of the extensive use of salmonids under freshwater test conditions. Eight organochlorine analogs were tested for their ability to change levels of brain ATPases in salmon (Salmo salar)

and trout (Salvelinus fontinalis). Interestingly, the in vitro efficacy of the various compounds did not correspond to their in vivo toxicities. DDA (10^{-1}M) and Kelthane (10^{-5}M) caused complete inhibition of ATPase. DDT, DDE, DDD and chloro-DDT were of similar impact but caused less inhibition than DDA and Kethane. Methoxychlor gave very low levels of inhibition and dieldrin - none.

Mehrle et al. (423) found that rainbow trout (Salmo gairdneri) fed either 1 ppm DDT or dieldrin exhibited significant increases in serum amino acids while Grant and Mehrle (246) showed a variety of physiological alterations (in the same species) in response to 4 to 145 ppm dietary endrin. Parameters affected included: serum electrolytes, osmolality, protein, cholesterol, cortisol, lactate, glucose and liver glycogen. Dietary dieldrin at doses beginning at 14 ppb were also shown to have a marked effect on phenylalanine metabolism and could induce the biochemical manifestations of phenylketonuria in rainbow trout. The authors suggested that their results indicated subtle effects that might alter the survivability of the species (422).

DDT at 50 ppm affected the water absorption ability of eel intestine. In this study, Janicki and Kinter (318) also reported an inhibition of (Na^+ and K^+) activated Mg^{++} - dependent ATP.

Endrin was not shown by Eisler and Edmunds (184) to significantly alter many parameters of blood and tissue chemistry of the northern puffer (Sphaeroides maculatus). Exposure to sublethal levels of endrin did, however, impair liver function, which was reflected by the transfer of major cations from hepatic tissue to the serum and by elevated serum cholesterol.

Lane and Scura (704) exposed sailfin mollies to a range of dieldrin levels. Those exposed to 0.6 to 12 ppb, which died by 72 hours had serum glutamic oxaloacetic transaminase (SGOT) levels from 1,500 to 1,700 units over controls. Mollies which survived 3.0 ppb to 120 hours had SGOT levels which were raised approximately 6,000 to 12,000 units.

Most toxicological research on birds has addressed lethality and, of that research concerning physiology or biochemistry, the duck has been the bird of choice for studies which might be interpreted as estuarine. DDE was shown to interfer with extra-renal salt excretion in the mallard duck (218), but if the glands were stimulated by pre-exposure to salt solutions, no such effect was noted. Loncore et al. (389) found that dietary DDE brought about changes in the mineral composition of eggshells produced by black ducks and mallards. Ten ppm resulted in significant changes in magnesium, barium and strontium in black duck eggshells while 5 ppm resulted in significant changes in magnesium and aluminum in the mallard eggshells. Ten ppm DDE also resulted in significant increases in the calcium of mallard eggshells.

A daily dietary combination dose of DDT (20 mg), DDD (15 mg) and DDE (15 mg) resulted in many physiological changes in the white pelican

(Pelecanus erythrorhynchos) as reported by Greichus et al. (694). Liver weight decreased and liver vitamin A levels increased but serum potassium and protein values were lowered.

Work on organophosphates is somewhat limited. Quahaug clams exposed to 37 ppm malathion were characterized by significant changes in tissue calcium, zinc and other metals (186).

Guilbault et al. (253) reported a selective inhibition of cholinesterase activity in fiddler crab and trout tissues in response to paraoxon, DDVP, parathion and methyl parathion. A similar effort by Coppage and Matthews (123) with fish and shrimp revealed the comparative AChE inhibition by malathion, naled, guthion and parathion.

Puffers, when exposed to 20 ppm methyl parathion or a combination of 10 ppm methyl parathion plus 15 ppb methoxychlor, refused to eat and when compared with controls, had less hemoglobin, erythrocytes, and a lower hematocrit. Eisler (177) also reported that the experimentals exhibited serum esterase inhibition, less liver magnesium and less zinc in their livers and gills.

Coppage (122) looked at the ability of a dozen organophosphates to inhibit brain acetylcholinesterase in the sheepshead minnow (Cyprinodon variegatus). Results indicated that effect is a function of pesticide concentration and exposure period. His data indicated that brain AChE levels are dependable indicators of exposure and death. In another similar experiment, Coppage et al. (681) found that acetylcholinesterase inhibition in the brain of the pinfish (Lagodon rhomboides) was correlated so well with malathion exposure that the technique appears to have diagnostic value for field situations. Exposure of the same species to the organophosphate, naled, resulted in parallel findings (680).

Guilbault et al. (253) also showed the selectivity of the carbamates matacil, mesurol, zectran, Baygon and Sevin on cholinesterase inhibition in fiddler crab and trout tissue.

Piavis and Howell (714) reported that the lampricide, TFM, at 10 ppm, resulted in either retarded hemoglobin or no hemoglobin production at all in embryonic sea lamprey (Petromyzon marinus). It was also shown to have other physiological effects on the lamprey, as reflected by changes in electrocardiograms. Lamprey, which died from 3 ppm TFM (perfused), were found to have red blood, indicating the absence of methemoglobin (6).

Physiological experimentation with herbicides has logically focused in on photosynthetic unicellular algae. Walsh and Grow (626) documented the depression of carbohydrate concentration in 6 genera of algae by 4 phenylurea herbicides in seawater, at several salinity levels. Chlorococcum was the most susceptible, with a depression of 49 percent at 5 ppt salinity. Hollister and Walsh (307) addressed the physiological response of 18 species of marine unicellular algae to the substituted ureas, neburon and diuron, and the triazines, atrazine and ametryne.

Using oxygen evolution as the parameter, atrazine was the least toxic, while the other 3 herbicides had approximately the same effect. Wide variations in response occurred among the species within several families.

INDUSTRIAL TOXICANTS

General and Lethal Effects

A great deal of research has been carried out on the effects of PCB's on estuarine life (Table 15). Perhaps most of it has been done at the E.P.A. Gulf Breeze Laboratory. Because of similarities in molecular formation, PCB's share many of the characteristics of organochlorine pesticides (see Section III). Like the pesticide mirex, PCB's accummulate and have been shown to be extremely toxic to some estuarine organisms. For instance, Duke et al. (164) showed that crabs concentrated the PCB Aroclor 1254 and 72 percent of the shrimp exposed to 5 ppb died after day 10. Nimmo and colleagues (461) found 1 ppb PCB to be lethal to shrimp. Bioassays by Hansen et al. (268) with Aroclor 1254 indicated that 5 ppb caused mortality to estuarine fish and the effect was delayed. In response to the change in emphasis of PCB production and subsequent increase in Aroclor 1016 manufacture, Hansen and co-workers (267) established the acute 96-hour LC_{50}'s for estuarine shrimp, fish and oyster.

Although no work has apparently been done on the effects of PCB's on strictly estuarine fish-eating birds, some data are available on ducks. Heath et al. (233), testing a series of PCB's revealed that toxicity was positively correlated with degree of chlorination and Haegele and Tucker (257) established the effect of 1254 on eggshell thinning.

Very little toxicological work has been done with dioxins, dibenzofurans, or phthalates and nothing to date, which has been directed strictly at the estuarine habitat, has been published to the authors' knowledge. Miller et al. (691) reported on the effects of tetrachloro-dibenzo-dioxin (TCDD) on various aquatic organisms. Approximately 50 percent of the young coho salmon exposed to .131 ppb died by day 20.

Zitko and his colleagues (692) reported on the acute and chronic oral toxicity of chlorinated dibenzofurans to immature brook trout. They concluded that 2,8-dichlorodibenzofuran has a low acute toxicity to that species since even a high level of 122 ppm produced no mortality.

Work on phthalate esters seems to be limited to freshwater or anadromous organisms. In an effort to establish LC_{50} values for freshwater organisms, Mayer and Sanders (415) reported DNBP to be less toxic to rainbow trout (96-hour LC_{50} = 6.5 ppm) than to the other fish tested. Stalling et al. (574) noted that phthalate esters are metabolized by freshwater fishes and both DEHP and DNBP are apparently not especially (acutely) toxic to freshwater invertebrates. Sanders et al. (545) reported that although freshwater invertebrates rapidly accumulate these compounds, their 96-hour TL_{50} (2.1 - >32/1) is appreciably greater than DDT, by

Table 15. SOME TYPICAL EFFECTS OF INDUSTRIAL TOXICANTS ON ESTUARINE/MARINE ORGANISMS

Treatment	Taxa	Observed effects	Ref.
Aroclor 1242 and Aroclor 1254 + radiocarbon	Phytoplankton	Radiocarbon uptake reduced at as low as 1-2 ppb	441
Aroclor 1242 in water .01 to .1 ppm	Marine diatom	Inhibited growth, RNA synthesis and chlorophyll index	343
Aroclor 1254 .94-100 ppb	Juvenile shrimp	51 to 100% mortality	461
Aroclor 1254 2.5-3.5 ppb	Adult shrimp	50% mortality, accumulated in hepatopancreas 23% died after return to sea water	
Aroclor 1254 in Corexit 7664 colloidal solution	Gammarus	Lethal threshold 0.001 to 0.01 ppm	642
emulsions	Gammarus	Lethal threshold .01 to .1 ppm	
Aroclor 1254 100 ppb 48 hours	Shrimp Oysters Pinfish	100% mortality Shell growth inhibited Concentrated PCB	164
5 ppb 20 days	Shrimp Crabs	72% mortality after day 10 Concentrated PCB	
Aroclor 1254 in sediment 61.0 ppm (dry wt.) to 1.4 ppm for 30 days	Shrimp Crabs	Amount of PCB residue in animal varies with amount in substrate	465
Aroclor 1254	Shrimp	60% died at 9.1 ppb (7 day exposure); no significant mortality at 0.62 ppb	462
Aroclor 1254 0.001-10 ppm	Shrimp Fishes	Demonstrated that some animals could avoid Aroclor 1254 under laboratory conditions	271
Aroclor 1016	Oyster Shrimp Fish	Established acute 96 hour LC_{50}'s	267

Table 15 (continued). SOME TYPICAL EFFECTS OF INDUSTRIAL TOXICANTS ON ESTUARINE/MARINE ORGANISMS

Treatment	Taxa	Observed effects	Ref.
Aroclor 1254	Oyster	5 ppb for 24 weeks reduced growth and produced tissue atrophy and degeneration	395
Aroclor 1254 1 ppb to 56 days 5 ppb 14-45 days	Fish	No apparent effect at 1 ppb Mortality occurred, though delayed at 5 ppb	268
Phthalate ester	Rainbow trout	LC_{50} (96 hour) = 6.47 ppm	415
Aroclor 1221 7.5 - 75 ppm	Killifish	Decreased ability to osmoregulate	355
Dioxin (TCDD) in water and food	Salmonids	Marked decrease in growth; latent effect	709
Dibenzofurans	Salmonids	Dietary doses up to 122 mg/kg resulted in no mortality	732
Aroclor 1254	Duck	Showed PCB influence on susceptibility of birds to virus	219
Aroclors 1232, 1242, 1248, 1254, 1260, 1262	Duck	Toxicity positively correlated with percent chlorine	233
Aroclor 1254	Duck	Established effect on eggshell thinning	257

comparison. However, the TL_{50} values for aquatic organisms are 700 to 11,000 times that which inhibited reproduction in one of the invertebrates tested (water fleas).

Sublethal Effects

Behavior--

Like the organochlorine pesticides, PCB's and other industrial toxicants have been shown to affect the behavior of microorganisms, fish and wildlife.

During the past decade, many authors have reported behavioral effects of chemical pollutants that interfere with several aspects of survival in the estuarine habitat. The ability to detect and avoid water, in which potentially toxic chemicals are present, is essential to survival in many instances. In this light, marine bacteria showed negative chemotaxis to toxic substances, including toluene, chloroform, ethanol and benzene. (661). Hansen et al. (271) demonstrated that estuarine fishes and shrimp displayed varying degrees of avoidance to the PCB, Aroclor 1254, at levels 0.001 to 10 ppm. Adult pink shrimp (Penaeus duorarum) failed to avoid various concentrations of the same PCB but grass shrimp were able to detect and avoid this chemical (271).

There is a considerable amount of information available on the ability of marine/estuarine vertebrates, primarily fish, to avoid water containing various toxic materials. Juvenile rainbow trout (Salmo gairdneri) showed agitated behavior when unable to avoid toxic terphenyl compounds and 'high-boilers' (tar-like decomposition products resulting from exposure of terphenyl compounds to a high radiation field) occurring as by-products of cooling nuclear reactors (255). As one might expect, all fish are not equally discriminating. Hansen, Schimmel and Matthews (271) reported that pinfish (Lagodon rhomboides) and mosquitofish (Gambusia affinis) avoided water containing Aroclor 1254, but sheepshead minnows were not able to do this.

Several other behaviors are altered by industrial pollutants. Larval barnacles (Balanus balanoides) exhibited reduced swimming activity, and adult barnacles showed reduced cirri beating rate when exposed to a by-product of vinylchloride production (EDC-tar). Effects were dose-related, and larvae were more sensitive than adults (538). The bivalve, Tellina tenuis, which normally responds to sand and water by burrowing, ceased burrowing when exposed to phenol (585). Similarly, juvenile pinfish and spot exposed to Aroclor 1254 ceased feeding (680).

The ability of anadromous fishes to select water of the right characteristics is obviously important to the survival of the species. Along these lines, Miller and Ogilvie (707) demonstrated a consistent downward shift in temperature selection by young brook trout (Salvelinus fontinalis), which had been exposed to 0.75 to 10 ppm phenol. A similar experiment with 25 to 100 ppm Aroclor 1254 did not yield a similar shift.

Growth and Development--

The PCB, Aroclor 1254, at 5 ppb in flowing seawater, significantly reduced growth rate in oysters over 24 weeks exposure, but 1 ppb for 30 weeks had no significant effect (483). Lowe et al. (395), using juvenile (3.1 cm) oysters, observed that both height and in-water weight increases were adversely affected by exposure to 5 ppb PCB's for 24 weeks. As in the study by Parrish et al. (483), Lowe found no effect of 1 ppb PCB over 30 weeks exposure. Another estuarine invertebrate shown to be adversely affected by PCB's is the mussel. Seed mussels reacted to fairly high concentrations of Aroclors 1254 and 1242 in the form of reduced byssal attachment (536).

Little has been done on the developmental effects of industrial toxicants on estuarine fishes but juvenile rainbow trout exposed to terphenyl compounds experienced swimming and postural difficulties (355) and Miller et al. (709) reported marked growth inhibition on both salmon and rainbow trout by TCDD.

Some subtle effects of industrial contaminants, like abnormal chick development in wild birds, may easily escape observation. Along these lines, Hays and Risebrough (284) brought attention to the possible involvement of dietary chlorinated dibenzo-p-dioxins and PCB's. PCB residues ranged from 10 to 180 ppm (lipid weight basis) for prey fish and abnormalities, similar to those produced in chickens by the above compounds, included feather loss in juvenile terns and eye, bill and foot deformities in newly-hatched birds.

Cytology and Histology--

Interestingly enough, many more publications have appeared on the cytological and histological effects of industrial toxicants than of pesticides over the last 5 to 10 years.

After exposure of the isopod, Gammarus oceanicus, to .01 - .1 ppm Aroclor 1254 (plus the emulsifier Corexit 7664) for more than 150 hours, Wildish (642) reported that some of the isopods which died had severely necrosed branchiae. A sublethal branchial edema was also found, and attributed to the Corexit 7664. No mortality, however, was attributed to Corexit 7664, but the possibility of synergism between emulsifier and PCB was mentioned (642).

Couch and Nimmo (129) exposed shrimp (Penaeus duorarum) to 3-5 ppb Aroclor 1254 for 20-30 days, at which time, 50 percent of the experimentals died. An examination of the hepatopancreases of shrimp surviving 30 days' exposure revealed that 30 to 50 percent of the cells had increased numbers of endoplasmic reticula and free and attached ribosomes. Nuclear degeneration and myelin bodies enclosing lipid droplets were also observed. The former were characterized by the appearance of vesicles in the nucleoplasm.

As a result of observing an unidentified virus in previous bioassays with shrimp (129), Couch (128) investigated the relationship between Aroclor 1254 and the occurrence of virus-induced cytopathology. Sixty percent of the shrimp exposed to 3 ppb of this PCB for 30 days were infected with virus but none of the controls were.

Oysters (Crassostrea virginica) exposed to 5 ppb Aroclor 1254 for 24 weeks were characterized by a number of histopathological changes (395, 483). Parrish et al. (483) reported degeneration of vesicular connective tissues concomitant with leukocytic infiltration. Tissue recovery was excellent after 12 weeks depuration.

Nimmo et al. (463) investigated the histopathological effects of Aroclor 1254 on a variety of estuarine organisms. Oysters (C. virginica) developed abnormal infiltration of leukocytes in the connective tissue (parenchyma) and digestive gland tubules of exposed oysters suffered atrophied (i.e., thinned) tubule epithelium and enlarged, abnormal lumen of tubule. Spot, an estuarine fish, showed fatty changes in the liver, characterized by the presence of vacuoles within the hepatocytes. Shrimp (P. duorarum) showed dramatic tissue changes associated with chronic PCB exposures, developing crystalloids in hepatopancreatic nuclei.

Pinfish (Lagodon rhomboides) exposed to 10 ppb Aroclor 1016 for 42 days showed no pathological alterations in the tissues. However, those exposed to 32 ppb had several liver and pancreatic alterations that distinguished them from the control fish. Hepatocytes appeared slightly enlarged and more basophilic, while less lipid material existed in the livers of the exposed fish than in those of the controls. The most noticeable alteration in the pinfish occurred in the form of severe vacuolation in the pancreatic exocrine tissue surrounding the portal veins. Hansen, Parrish and Forester (267) stated that Aroclor 1016 is similar to other PCB's in its toxicity to, and uptake and retention by estuarine animals.

Reproduction--

Industrial toxicants as well as pesticides have been shown to affect the reproductive success of several estuarine/marine species. Some evidence only leads to suspicion. For example, starfish (Acanthaster planci) concentrated PCB's in their gonads (418).

Although very little research appears to have been carried out on the reproductive effects of industrial toxicants on estuarine species, work with freshwater and tropical species indicate the need for extending the studies into the estuaries. For instance, Daphnia magna, continuously exposed to phthalate esters in water for 21 days (their full life cycle) showed reproductive impairment: 60 percent reduction of reproduction in 3 ppb phthalate, 70 percent reduction in 10 ppb, and 83 percent reduction in 30 ppb (545). Mayer et al. (416) exposed zebra fish (Brachydanio rerio) and guppies (Poecilia reticulata) to phthalate esters in their food. The zebra fish received 50 and 100 ppm in food, the guppies 100 ppm. The zebra fish showed increased fry mortality and guppies

experienced an 8 percent incidence of abortions in the experimental group.

One study that did deal with an estuarine species involved the sheepshead minnow (Cyprinodon variegatus). There was no change in fertilization success in eggs of the adult minnows exposed to the PCB Aroclor 1254 in concentrations as high as 201 ppm. However, survival of embryos and fry was reduced by exposure of adults to a range of 0.1 to 10 ppb PCB's. At concentrations of 7.0 ppb or greater, fry mortality increased dramatically (269).

In field research dealing with reproduction in fish-eating birds, PCB residues were not correlated to failure in at least one species, the brown pelicans (52), where the relationship was specifically investigated. Although commonly found in populations exhibiting reproductive problems, PCB's are usually highly correlated with the presence of DDE. In a lengthy review on the eggshell-thinning effects of environmental pollutants, Cooke (116) concluded that DDE is the major offender, but PCB's may play a minor role.

Heath et al. (292) found that low dietary doses (i.e., 25 and 50 ppm) of Aroclor 1254 produced no measurable effects on the reproduction of mallards. Haegele and Tucker (257) established the effect of at least one PCB (Aroclor 1254) on eggshell-thinning in ducks. This PCB, when fed to adult mallards did thin the eggshells produced but the doses were fairly massive (i.e., 1,000 ppm).

In the observations on sea lions mentioned earlier, Bowes et al. (63) noted higher levels of PCB's in tissues of females giving birth prematurely compared to those giving birth at full term. DeLong et al. (148) found a mean of 112 ppm PCB's in the premature partum females compared to a mean of 17 ppm in full term partum females.

Biochemistry and Physiology--

Zitko and Choi's review (664) of the physiological effects of industrial toxicants in 1971 included some estuarine species. However, relatively very little has been published since then.

At the microorganism level, E. coli growth was consistently stimulated by 0.01 ppm Aroclor 1242. Keil et al. (341, 344) reported that although this exposure level did not alter the uridine content after 24 hours incubation, increased uridine uptake was noted in all PCB cultures after 5 hours incubation. Keil et al. (343) also tested the effects of Aroclor 1242 (0.01 - 0.1 ppm) on marine diatoms and found that it inhibited growth, RNA synthesis and chlorophyll production.

Aroclor 1221 has been shown capable of impairing osmoregulation in the killifish at relatively high levels (7.5 - 75 ppm) by Kinter et al., (355) but spot developed fatty changes in the liver in response to a 2 week exposure to 5 ppb Aroclor 1254 (463).

Greichus and her co-workers (694) examined the physiological effects of Aroclor 1254 on white pelicans (P. erythrorhynchos). They found that 100 mg (daily dietary load) resulted in: an increase in spleen and liver weight; a lowering of serum potassium, protein, and calcium and a reduction in albumin fraction. Jefferies and Parslow (320) looked at the effect of the same chemical on the lesser black-backed gull (Larus fuscus). They found that 50 ppm (dietary)significantly increased the weight of thyroids in experimental gulls. Sections of thyroids from PCB-dosed birds revealed enlarged follicles filled with colloid.

SYNERGISTIC AND MODIFYING EFFECTS

No report on the effects of SOC's on estuarine life would be complete without including the area of synergistic effects and modifying factors. The term "synergism", unfortunately, has many definitions (706). For our present needs, we will consider it to mean more than the anticipated additive effects.

Several reviews have addressed this topic (34, 315, 544, 621), and many examples of synergistic or modifying effects involving pesticides (73, 95, 132, 146, 179, 180, 220, 222, 223, 306, 394, 397, 417, 436, 449, 544, 621, 647), herbicides (362, 626) and industrial toxicants (127, 204, 219, 268, 459, 621) exist in the literature.

Pesticide toxicity can affect disease resistance in shellfish (84) and ducks (220, 222), while organophosphates and general stress have been shown to interact on oyster toxicity (596). With respect to organochlorine poisoning, sensitive life stages have been reported for many species including trout (104), crabs (192), and other decapods and fishes (132).

A wide range of sensitivity to sublethal herbicidal effects on algae exists, depending on which particular species is used (626).

In the field, modifying factors are the rule rather than the exception. For instance, mirex leached from fire ant bait at a greater rate in the summer than in the spring. During this toxicological experiment on a variety of estuarine organisms, the greatest mortality occurred in the summer and to younger individuals (723).

Eisler (180) reported on the modifying factors affecting the toxicity of organochlorines and organophosphates to the mummichog, an estuarine fish. The toxicity of organophosphates increased with increasing temperature and salinity and decreasing pH. The toxicity of organochlorines was greatest at intermediate temperatures (20 - 25 C) and least at an intermediate pH (7 - 8). Salinity had little effect on organochlorine toxicity. Toxaphene, however, has been shown to interact with salinity, temperature and D.O. to produce varying toxicity in a number of estuarine organisms (132).

Since estuarine fish and wildlife are rarely exposed to only one chemical at a time in the 'real' world, it is of interest that prolonged pre-exposure of spot (Leiostomus xanthurus) to sublethal levels of toxaphene rendered the fish more sensitive to subsequent toxaphene toxicity (391). Lowe et al. (397) also reported that oysters exposed to a mixture of 1 ppm each of DDT, toxaphene and parathion showed reduced growth and histopathological effects. However, when these mollusks were exposed to the individual pesticides, similar results were not observed.

Like their chemical counterparts, PCB's have been shown to affect the resistance of both fish (268) and ducks (219) to disease. In addition, there is an indication of a similar effect on estuarine shrimp (127).

The interaction between normal variations in estuarine environmental parameters and industrial toxicants has been elucidated by Nimmo (459). He reported that sublethal levels of the PCB, Aroclor 1254, became lethal to estuarine penaeid shrimp when the test organisms were stressed by reduced salinity. Since this species is migratory and experiences a wide variation in salinity, this finding is particularly significant.

EFFECTS AT THE COMMUNITY AND ECOSYSTEM LEVELS

A variable amount of effort has gone into testing the effects of particular toxicants under "field conditions" (Table 16). Although a number of taxa are considered in this approach, this is still not attacking the problem on an interactive level.

Only recently has any serious experimental attention been given to the effects of synthetic organics at the community level. A variety of community parameters have been suggested as reflectors of a community's health. Margalef's "species richness" and Peilou's "species diversity and eveness" are but a few (111). Researchers are only now finding out that many of these parameters are not the panaceas they thought they were. The main problem lies with trying to use these techniques out of the context for which they were originally intended.

Aroclor 1254 was shown by Hansen (264) to affect the composition of estuarine animal communities. Control communities and those in 0.1 ppb PCB were dominated by arthropods, however, communities exposed to 1 and 10 ppb PCB were dominated by tunicates and other chordates. Numbers of phyla, species and individuals were decreased by the industrial toxicant but this was not reflected by the classical Shannon-Weaver index of species diversity.

At the community level, PCB's have been shown to alter species composition in mixed cultures of algae (445). Experiments by Moore and Hariss (441) suggest that the effects of PCB's are also more acute at the phytoplankton community level than at the single species culture level.

Table 16. AN OVERVIEW OF THE FIELD-TESTING OF PESTICIDES IN THE ESTUARINE ECOSYSTEM

Ecosystem	Pesticide	Observed parameters	Taxa	Ref.
Tidal marsh	DDT	Mortality Gross behavior Growth (snails)	Fish Crabs Shrimp Insects Mollusks Amphipods Worms Mites Birds	573
Tidal marsh	Strobane DDT HCB	Mortality Gross behavior on fiddlers	Fish Crabs Birds Mammals	228
Tidal marsh Ditch	Dieldrin	Mortality	Fish Crabs	273
Tidal marsh	DDT Aldrin Dieldrin BHC	Mortality	Fish Prawns Arthropods Iso- and amphipods Crabs Worms Mollusks Birds	571
Tidal marsh Ditch	DDT	Mortality and population; Residue monitoring	Fish Crabs	141
Estuaries	2,4-D	Mortality	Fish Crab Oysters Clam	514
Salt Marsh	Malathion	Mortality Cholinesterase	Fish Crab Shrimp Mollusks	593

In South Vietnam, where over 49 million kilograms of herbicides were sprayed on 2 million hectares of forest lands, including mangroves, many changes were noted by Westing (639). He found that sprayed areas were characterized by: alteration and simplification of plant and animal communities; a loss of mineral nutrients, and; a reduction in ecosystem productivity.

Woodwell, in his overview of the changing chemistry of the oceans (654), notes that changes in the structure of the community can be expected. The most conspicuous change observed in response to DDT was the loss of highly specialized top carnivores, like the fish-eating birds.

Pimentel and Goodman (498), in their review, addressed the environmental impact of pesticides at the ecosystem level. Some other attempts to address SOC's at the ecosystemic level have centered around their presence and movement on a global scale (112, 140, 528). One particularly agressive effort was that of Crammer (140) who approached the circulation of DDT on earth.

Model ecosystems (i.e., microcosms) have received increased attention over the last 5 years. Metcalf and his associates looked at the distribution and fate of pesticides and industrial toxicants using model ecosystems (431). The experimental herbicide, bentazon (56) and the organophosphate, malathion (61) received similar attention.

Attempts that combine a microcosm and a computor approach are likely to be the most successful, for computor programs alone are twice-removed from reality. Along these lines, Cox (137) found that the actual DDT content of a euphausiid shrimp was quite different from that predicted by a theoretical food assimilation model. On the other hand, models often indicate what types of information are lacking and also have the advantage that the effects of even extreme manipulations can be tested through many generations or seasonal cycles without any damage to the real world.

If it is possible to consistently and accurately describe some ecosystem parameter, then it ought to be theoretically possible to quantitate a change in that parameter. The absence of this kind of effort in the estuarine habitat is probably a reflection of our current inability to describe such changes, not evidence of their non-existence.

SECTION V

ARE THERE ALLOWABLE LEVELS?

California seems to have taken the lead in 1963 in describing the presence and effects of pesticides relative to water quality criteria (419). This precipitated many studies and many questions. Perhaps the most important question a decision-making politician or coastal-zone administrator ought to ask with reference to SOC's is "How much should be allowed in our waters and what chemicals should not be applied at all near the estuaries?" More recent attempts have been made to answer these and similar questions. The National Technical Advisory Committee to the Secretary of the Interior (609) zoned in on this topic and recommended that the following organochlorines not be applied near the marine habitat because of their extreme toxicity:

Aldrin	DDT
BHC	Dieldrin
Chlordane	Endosulfan
Endrin	Methoxychlor
Heptachlor	Perthane
Lindane	TDE
	Toxaphene

Mirex has been shown to be exceptionally toxic to estuarine invertebrates like shrimp and should be considered in this category. Hexachlorobenzene is particularly toxic to birds (729) and deserves special attention around rookeries.

A similar list (609) for organophosphates included:

Coumophos	Naled
Dursban	Parathion
Fenthion	Ronnel

In general terms, the above organochlorines and organophosphates are acutely toxic at concentrations of 5 ppm or less and should not be permitted to exceed 50 nanograms/l. The next group they discussed is generally not quite as toxic but should not be allowed to exceed 10 ppm in estuarine waters. This group included:

Arsenicals	2,4,5-T compounds
Botanicals	Phthalic acid compounds
Carbamates	Triazine compounds
2,4,-D compounds	Substituted urea compounds

This kind of information and guidance as to allowable levels of these and most other common toxicants, including radionuclides, heavy metals, PCB's, etc. is presently being updated by the Environmental Protection Agency (454).

SECTION VI

RECOMMENDED RESEARCH

Whenever a scientist is asked to make research recommendations, there is always the temptation to exaggerate areas of his own interests. This can, obviously, be quite counterproductive and we shall try to ignore the temptation. Many groups have made concerted efforts to recommend research needs for the subject considered herein (74, 111, 112, 157, 175, 242, 243, 455) and, to some extent, our own thoughts become a duplication of effort but, hopefully, more of an updating of needs.

MAINTAIN PRODUCTIVE ONGOING PROGRAMS

It goes without saying that there are existing programs that have to continue. One such program is the National Pesticide Monitoring Program. It is also imperative that we have established centers, like the E.P.A. Laboratories at Gulf Breeze, Florida and Corvallis, Oregon where SOC's which come under public scrutiny, can be quickly tested under estuarine conditions.

A continuing search for suspected contaminants should be the rule rather than the exception. Compounds of interest include, but are not restricted to: polar metabolites of widespread SOC's, like PCB's and PCT's, PVC by-products, halogenated aromatic and aliphatic hydrocarbons (like dry cleaning solvents), phthalates and other plasticizers, flame retardents and contaminants like the chlorinated dibenzofurans and dioxins.

There is a constant need for standardization of residue analysis techniques and perhaps, even more important, a standardization of reporting residues. PCB's and other multiple peak SOC's should be high on the list. Along these lines, it is also necessary to establish the analytical ability to detect the more polar metabolites of widespread pesticides and industrial toxicants. Especially important may be the presence of the organophosphate and carbamate metabolites which, although not easily detected are often toxic. Their absence from the literature may only reflect our inability to identify them.

SIGNIFICANCE OF RESIDUES

As Edwards (175) said in 1971, before one can define research needs, one must define objectives. He went on to say that the principal objective must be to provide information which will allow us to ultimately determine the environmental consequences of man's waste products.

Working backwards then, emphasis in future research should also be given to determining the significance of the residues being reported. This can be accomplished by stressing the diagnostic aspects of experimentation during the planning stage and encouraging toxicological studies that have direct relevance to the real world. We must orient our research designs more at elucidating the meaning of residues found in fish and wildlife. Conversely, microcosm work seems to have great potential and every effort must be made to improve our ability to extrapolate from this system to the natural environment.

ESTABLISH COOPERATIVE EFFORTS

Cole (679) adroitly pointed out that we can no longer afford the financial luxury of uncoordinated research efforts. A closer coordination of the individual research attempts is necessary and we must bring together the practical industrial scientist, the non-industrial scientists and responsible governmental decision-makers.

Within a particular subject area, every attempt must be made to coordinate the efforts of scientists working in the laboratory, those in the field and those that have to use the data to establish legal tolerance limits or effect necessary change. Multidisciplinary efforts should be made wherever the hypothesis to be tested justifies it. Unfortunately, inadequate funding has been responsible for too many projects falling short of what could be accomplished with an interdisciplinary approach.

On the global scene, interdisciplinary efforts should be made to more thoroughly characterize the kinetics, marketing patterns and use of the classical synthetic organic compounds plus the newcomers, like phthalate esters, other plasticizers, chlorinated dibenzofurans and dioxins and HCB. Estuarine monitoring programs along the coasts of less developed continents should be established. Trends must be recognized and characterized if they do exist.

We should determine the production and use figures, as well as input of SOC's, by country, if we are going to address the problem on a truly global scale. To do this, every effort must be made to encourage international cooperation.

SPECIFIC COMPOUNDS

As to specific chemicals that need experimental attention, PAE's, HCB, dioxins and dibenzofurans are high on the list.

PAE's are widespread in freshwater fish with higher residues appearing to be associated with industrial areas. They have been shown to be more toxic to aquatic organisms than warm-blooded animals. These esters also disturb reproduction and growth in freshwater invertebrates and fish, yet little is known about their effects on estuarine species.

Although not widely reported in the literature, HCB has been found in environmental samples. In view of the possible analytical confusion

with benzene hexachloride (BHC), HBC may be even more widespread than presently thought. With this potential and the documented toxicity of this compound to birds in mind, the effects of HCB on fish-eating birds should be of immediate concern.

The initial work with the dioxin, TCDD, indicates important effects on the growth and reproduction of anadromous and freshwater species. Again, nothing is known about the effects on estuarine species.

Dibenzofurans were not particularly lethal to the trout they were tested on, however, nothing is known about their sublethal effects. In addition, because of different osmoregulatory mechanisms, the effect(s) on euryhaline species may be considerably different.

Needless to say, new industrial compounds, like the wastes from vinyl chloride production, because of the great volume of production and projected increases in use, should be watched carefully.

AREAS OF RESEARCH

We must characterize the seasonal and geographic distribution of SOC's in estuaries. This was recommended by Butler (74) as early as 1964. Yet we make slow progress and cannot continue to look at the estuary in a vacuum. The juxtaposed terrestrial and marine habitats must be considered in this area of research.

The role that sediment plays in a variety of estuary types is of great importance. Are SOC's sequestered and/or deactivated by this part of the system? Are they released at expected times of the year? What are the biological, physical and chemical factors involved?

Interdisciplinary research on the fate and degradation of SOC's in the environment under field conditions is also of prime importance. We need to know more about their environmental kinetics, especially their metabolism in soil and water. The role of microorganisms in the breakdown of SOC's in the estuarine ecosystem is not well understood. Especially productive, may be research aimed at elucidating the synergistic impact of the whole microbial system, including bacteria, yeasts and fungi (60).

Increased attention must be given to the sublethal effects of SOC's on estuarine species. Lethality alone is not a sufficient criterion for reflecting damage to estuarine resources. One challenging area is that of establishing biological responses that can reflect certain types of contamination after they occur. It is commonly only this kind of ex post facto diagnostic evidence that remains after a spill or industrial breakdown.

We should extend research on behavioral and other sublethal effects that has been done on organochlorine pesticides to PCB's, dioxins, dibenzofurans and other industrial toxicants as they become obvious.

Sublethal effects should be looked at in species representing upper trophic levels like fish-eating birds and large, long-lived predatory fishes, that are likely to accumulate high residues. With respect to the shark, is there a relationship between shark attacks and toxicant-induced aggressive behavior?

Modifying factors and synergism have just now come into the limelight but much more needs to be done. The increased sensitivities of one sex over the other and one life stage over others stand out as important considerations. The toxicological consequences of varying environmental conditions in the estuary is largely unknown. What about the importance of the pesticide formulation at time of application? Does it make a significant difference to non-target organisms whether the application is an emulsion, in a granular form, an oil solution or some other solvent? How does the standard acetone-pesticide combination of the laboratory exposure compare with the variety of application approaches, which are aimed at maximum toxicity? What is the ecological significance of resistance acquired through natural selection? What of the effects of early life exposure on subsequent stages? Estuarine organisms are rarely exposed to one chemical. Multiple exposure with solid statistical design is indicated. With respect to this, research aimed at elucidating the modifying effects of sewerage and storm runoff is long overdue.

An impact that is becoming more obvious is that of 'latent' effect. It has been seen in the effect of a variety of organochlorines on estuarine invertebrates, fishes and birds (82, 182, 188, 357, 396, 463, 515, 584). Most observations along these lines were unexpected. There is a need to carry out experiments which are specifically designed to investigate latency in effect.

Perhaps the most challenging area of research is that which addresses impact at the ecosystem level. This is the final biological-physical-chemical integration that will reflect individual perturbations at any sublevel if, in fact, they are significant.

Cope, in the early part of this decade, noted that pesticide-wildlife studies had sought understanding at the cell, tissue, organ, organism and population levels of organization. He continued that it was long overdue that we consider the 'total' picture and attack the effects question at the ecosystem level. However, as L. Eugene Cronin (editorial in BioScience, April, 1970) commented, we did not yet know enough about estuaries to broadly describe or quantify even the flow of energy. He went on to state that we could not yet adequately assist the planner, engineer or public agency by providing useful predictions of the biological effects of specific environmental changes.

To predict the ecosystem's response to insult 'X' requires either an empirical and/or a theoretical basis. Since the science of computer modeling of systems is still in the embryonic stages and, we as scientists, do not yet seem capable of agreeing on just what is a healthy system, how can we characterize an unhealthy one? We need much more basic information

on the control mechanisms of populations, communities and ecosystems. This is necessary if we are going to have the predictive ability like that needed in other applied fields, such as wildlife management.

The area of field-testing toxicants has progressed in a manner that reflects individual idiosyncrasies and the idiomatic characteristic(s) of the respective funding and/or research organizations. Efforts should be made to, at least, roughly standardize field testing techniques with a keen awareness of the possible modifying and synergistic effects that one will encounter in the estuary. Interpretation of field exposures will require coordinated efforts in the laboratory under less real, but more controlled, conditions. More extensive utilization of microcosms and semi-field experiments, which bring us one step closer to the 'real' world are indicated. It is there that statistically and logistically complicated designs can reach fruition and elucidate specific modes of action, synergy, latent effects, food-chain magnification, etc.

PHILOSOPHY AND BUREAUCRACY

Because some questions aimed at the ecosystem level, can only be answered by using the intact system, it becomes of paramount importance that large estuarine preserves at various latitudes be established immediately for experimentation and to provide an example of how an unperturbed system operates.

An interface science ('Econology'?) should be established where the ecological impact of SOC's can be equated in economic terms. Trade-offs have and will continue to be made but how can the administrator make an intelligent trade-off until both sides of the equation have been given realistic values? This is an area where the marine scientist could learn from past experiences of his freshwater counterpart (175).

It is sad to see the lack of support for basic research on estuarine (and other) species. For it is only by proceeding along a broad front, supported by both basic and applied science, that we can expect to keep ahead of the multitude of potential problems associated with SOC's.

Research on the long-term chronic and sublethal effects, that may only manifest themselves after several generations, necessitates long-term support commitments. Recently, these kinds of commitments have, in general, only been made to governmental organizations, and even those are showing signs of waning. These organizations are to be commended for their work, but if private and academic research, under less structured conditions becomes a thing of the past, this short-sighted approach will result in only partial victory over environmental contamination.

SECTION VII

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SECTION VIII

APPENDIX

ADDENDUM TO BIBLIOGRAPHY

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SUBJECT PROFILE INDEX

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16. ABSTRACT

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This review and indexed bibliography concerns the presence and effects of pesticides (i.e., insecticides, herbicides, fungicides, etc.) and industrial toxicants in the estuarine ecosystem. The industrial toxicants refer, primarily, to polychlorinated biphenyls, but phthalate esters, polychlorinated terphenyls, chlorinated dibenzo-dioxins and dibenzofurans are also discussed. The review covers literature of the last decade, with emphasis on the most recent 5 years. However, the 700-plus references in the bibliography span a much wider range. A permuted keyword retrieval system (SPINDEX) is provided to allow practical use of the bibliography by scientists, academicians, and societal decision makers.

17. KEY WORDS AND DOCUMENT ANALYSIS

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