American National Standard

# guidelines for handling and disposal of capacitor- and transformer-grade askarels containing polychlorinated biphenyls

C107.1-1974

Attachment 1-5

ANSI
C107.1-1974

# American National Standard Guidelines for Handling and Disposal of Capacitor- and Transformer-Grade Askarels Containing Polychlorinated Biphenyls

Secretariat
National Electrical Manufacturers Association

Approved January 9, 1974
American National Standards Institute, Inc

NONS 219076

**American National Standard**

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# Foreword

(This Foreword is not a part of American National Standard Guidelines for Handling and Disposal of Capacitor- and Transformer-Grade Askarels Containing Polychlorinated Biphenyls, C107.1-1974.)

Recognizing that the polychlorinated biphenyls present in askarels were being identified as environmental pollutants and the urgent need for procedures and guides for their safe use and disposal, the Board of Directors of the Power Equipment Division of the National Electrical Manufacturers Association (NEMA) in June 1970 appointed a committee to review the general problem and propose recommendations. In September 1970, the NEMA board approved, as recommended by the committee, the sponsorship of an American National Standards Committee for the use and disposal of askarels in electrical equipment.

American National Standards Committee C107 was established in April 1971. NEMA was designated as the secretariat. An organizational meeting was held on September 14, 1971. Mr. W. P. Papageorge, Monsanto Company, was appointed chairman. Committee members included representatives of capacitor and transformer manufacturers, utilities, governmental departments and agencies, and maintenance and disposal service companies. A steering committee and two subcommittees – one to develop guidelines for the capacitor industry, the other for the transformer industry – were established.

The scope and primary objective of the committee was designated as the development of Procedures and guides for the safe use, maintenance, and disposal of askarel and askarel-soaked materials used in electrical equipment.

Additional objectives proposed were as follows:

(1) To serve as a source for technical information and advice for federal, state, and local authorities and for the information of all others concerned.

(2) To encourage the development of suitable disposal facilities and maintain a list of their capabilities and locations, for the information of all concerned.

(3) To serve as the advisory group for U.S. participation in the International Commission on Rules for the Approval of Electrical Equipment (CEE), the International Electrotechnical Commission (IEC), the Pan American Standards Commission (COPANT), the International Conference on Large High Tension Electric Systems (CIGRE), and other international organizations.

The final draft of the guidelines was completed in late November 1972. NEMA issued the guidelines as an Official Standards Proposal in January 1973. It was submitted to the American National Standards Institute for approval as an American National Standard and was approved by the Standards Institute on January 9, 1974.

Suggestions for improvement of this standard will be welcome. They should be sent to the American National Standards Institute, 1430 Broadway, New York, N.Y. 10018.

This standard was processed and approved for submittal to ANSI by American National Standards Committee on Use and Disposal of Askarel and Askarel-Soaked Materials in Electrical Equipment, C107. Committee approval of the standard does not necessarily imply that all committee members voted for its approval. At the time it approved this standard, the C107 Committee had the following members:

W. P. Papageorge, Chairman
A. M. Salazar, Secretary

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| Doble Engineering Company | A. L. Rickley |
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# American National Standard Guidelines for Handling and Disposal of Capacitor- and Transformer-Grade Askarels Containing Polychlorinated Biphenyls

## 1. Scope

This standard establishes guidelines for the safe use, maintenance, and disposal of askarel and askarel soaked material used in capacitors and transformers.

## 2. General Information

**2.1 General** The term "askarel" generally describes a broad class of nonflammable synthetic chlorinated hydrocarbon insulating liquids widely used in capacitors, transformers, reactors, and accessory equipment operated at power frequencies.

Askarels consisting of or containing polychlorinated biphenyls (PCBs) have been used in many applications for more than 40 years, but only recently was evidence discovered that PCBs are widely dispersed in the environment. Systematic investigations of the biological effects of PCBs have been undertaken within the past few years to establish the effects of specific formulations upon specific species. Some studies have shown that PCBs may be an environmental contaminant. Simultaneously, significant steps have been taken by U.S. industry to limit further releases of PCBs to the environment.

PCBs have been used in three broad types of applications for the past 40 years, as follows.

(1) "Open-ended" applications; for example, in paints, specialty inks, paper coatings, plastics, etc.

(2) "Nominally closed" applications, for example, as the working fluid in hydraulic or heat transfer systems.

(3) "Closed electrical system" applications, specifically as the insulating fluid in certain kinds of transformers and capacitors[1].

[1] The Monsanto Company is the sole U.S. producer of PCBs. It has discontinued supplying the material for all applications given in 2.1(1) and 2.1(2).

Evaluations of the benefits, risks, and alternatives involved in the continued use of PCBs in closed electrical systems are summarized in 2.2 through 2.4.

**2.2 Benefits.** Askarel-filled transformers do not burn or sustain fire under conditions of internal electrical arcing.

Askarel-filled power and industrial capacitors are significantly smaller, more reliable, more durable, and safer than oil-filled capacitors. As a result, askarels have supplanted mineral oils in more than 90% of the power and industrial capacitors made today. Over the past few decades most of the equipment that incorporates such capacitors has been designed to take particular advantage of the size, safety, and reliability benefits of askarel capacitors (for example, many types are today less than 14% of the size of equivalent oil capacitors and have a life expectancy of 10 to more than 20 years).

Various federal, state, and local codes, therefore, require their continued use in or adjacent to public, commercial, and industrial buildings, which locations present the greatest potential danger to life and property.

**2.3 Risks.** In the United States, medical records over a nearly 40 year period show that the only adverse health effects experienced by U.S. workers exposed to askarels, either during the manufacture of these liquids or of electrical equipment containing these liquids, have been limited to occasional cases of nonchronic chloracne or other temporary skin lesions or irritations.

Askarel-filled transformers and capacitors are delivered to customers as sealed units from which there is no escape of askarel under normal operation. Although certain types of equipment failures can permit loss of some askarel to the environment, transformer failures are limited to approximately 0.02% of the units in service per year. With respect to capacitors, such losses are limited to approximately 0.02% of the askarel put into service per year. In addition, limited amounts of PCBs can get into the environment during the manufacture, delivery, improper use, maintenance, repair, and disposal of transformers and capacitors.



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Specific control measures have been instituted by individual manufacturers and are supplemented and strengthened by national standards and procedures such as this standard, which provides information to prevent the inadvertent loss of PCBs to the environment at all stages from initial askarel manufacture through ultimate disposal.

2.4 Alternatives. For technical and local and national code reasons, it would be impossible to replace most askarel filled transformers now in service with oil filled units of equivalent ratings without major construction changes that would be required to compensate for the fire resistance of the askarel filled units. For certain applications and locations, dry type transformers may replace askarel-filled transformers.

For new installations, although many of the foregoing limitations would still apply, building and installation design provisions could be made to accommodate the use of oil-filled, open dry type, or sealed dry-type transformers, provided that necessary technical, code, physical size, and cost considerations are properly evaluated.

The principal alternative to askarels for capacitors is mineral oil, but such replacement would return capacitor technology to its pre 1932 level and would necessitate the redesign and replacement of such widely used equipment as fluorescent light fixtures and racks for power and induction-heating capacitors, which could not now accommodate the increased size of oil capacitors while maintaining their present ratings.

The cost of askarel liquids is about five to ten times more than mineral oil. Thus, long before there were any environmental concerns about PCBs, there was a strong economic incentive to find other, less expensive insulating liquids with the desirable characteristics of askarels. Since the 1930s at least ten major chemical or electrical companies have invested large amounts of time and money in this search, all with no success. Although potential substitutes that are more costly than askarels (such as fluorinated liquids) have also received some consideration, little is known about either their electrical performance or possible ill effects upon the environment. There are today no fluids that can be used as a direct replacement for askarels.

2.5 Interdepartmental Task Force on PCBs. An in-depth study of PCBs has recently been completed by five Executive Branch Departments of the federal government. This Interdepartmental Task Force on PCBs issued their report, entitled *PCBs in the United States Industrial Use and the Environment*, in May 1972.[2] The following conclusion is quoted from page 4 of this report:

"The use of PCBs should not be banned entirely. Their continued use for transformers and capacitors in the near future is considered necessary because of the significantly increased risk of fire and explosion and the disruption of electrical service which would result from a ban on PCB use. Also, continued use of PCBs in transformers and capacitors presents a minimal risk of environmental contamination. The Monsanto Company, the sole domestic producer, has reported voluntarily eliminating its distribution of PCBs to all except manufacturers of electrical transformers and capacitors."

Reference should be made to the Interdepartmental Task Force Report for additional information and conclusions.

## 3. Capacitor Guidelines

3.1 General. The environmental effects of askarels are under in-depth study by governmental and other agencies. Askarels have been considered relatively harmless to humans based on about 40 years of safe industrial usage. There has been no known instance of human injury when they were used under the normally accepted precautions and conditions of handling in both manufacturing and use applications.

Traces of askarels are being found in the environment and in fish and bird life. The long-term genetic and ecological effects are not yet completely understood. For these reasons, care should be taken to contain askarels and minimize their entry into the environment.

There are two general classes of askarels used by the electrical industry. The higher chlorinated grades are the more persistent in nature. Because of their high degree of nonflammability they are used in transformers where personnel safety is of paramount importance.

Capacitor grade askarel has a lower degree of chlorination (composed primarily of the 3-chlorine isomers of biphenyl) and a higher degree of biodegradability. Generally, it has not been found in animal life. It is used in capacitors, where the extreme degree of nonflammability required in transformers is of less importance.

Although capacitor and transformer grade askarels both contain members of the PCB family, they do differ in composition, degree of biodegradability, persistence in nature, electrical stability, chemical stability, and degree of nonflammability. (Both are recognized as nonflammable.) It is for these reasons that Section 3 of this standard is intended to apply to capacitor-grade askarel.

[2] Available from the National Technical Information Service, U.S. Department of Commerce, Springfield, Va. 22151.

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Table 1
Typical Properties of Aroclor 1016

| Property | Test Method (See Note) | Typical Values |
|---|---|---|
| Color | APHA* | 40, max |
| Condition | – | Clear |
| Specific Gravity at 25/15.5°C | ANSI C59.68-1963 (R1973) [2] | 1.362–1.372 |
| Acidity (mg KOH/g) | ANSI Z11.131-1964 (R1974) [3]<br>ANSI Z11.59-1958 (R1971) [4] | 0.010, max |
| Moisture | – | 35 ppm, max |
| Refractive index at 25°C | ASTM D 1817-66 (1972) [5] | 1.6215–1.6235 |
| Inorganic (free) chlorides | ANSI C59.55-1963 (R1973) [6] | 0.05 ppm, max |
| Pour point | ANSI Z11.5-1966 (R1972) [7] | −14°C or lower |
| Dielectric constant (1000 Hz at 100°C) | ANSI C59.22-1967 (R1973) [8] | 4.70–4.90 |
| Resistivity (500 V dc at 100°C, 0.1-inch gap) | ANSI C59.51-1965 (R1973) [9] | 500 × 10⁹ Ω - cm, min |
| Hydrolysis stability test (as chlorides) | ANSI C59.106-1970 [10] | 0.5 ppm, max |
| Thermal stability test (as chlorides) | ANSI C59.111-1970 [11] | 0.4 ppm, max |
| Distillation range (corrected)<br>10% distilled by weight<br>90% distilled by weight | ANSI A37.9-1974 [12] | <br>323°C, min<br>356°C, max |
| Higher Boiling Homologues | ASTM D 3303-74 [1] | 0.4%, max |
| Sulfates | ANSI C59.2-1974 [13] | None |
| Dielectric strength at 25°C | ANSI C59.19-1968 (R1973) [14] | 35 kV, min |
| Flash point, Cleveland open cup | ANSI Z11.6-1973 [15] | 338°F, min |
| Fire point | – | None to boiling point |
| Corrosion test (6 hours at 210°C with bright aluminum foil), change in weight of aluminum | – | 0.0% |
| Viscosity at 100°F (SUS)† | ANSI Z11.2.1956 (R1971) [16] | 71–81 |
| Specific heat at 25°C | – | 0.30 |
| Coefficient of expansion | ANSI C59.57-1963 (R1973) [17] | 0.00068 cm³/cm³/°C |
| Fixed chlorine | Carius | 41.3 ± 0.5% |
| Power factor at 100°C, 60 Hz | Munch [18] | |
| Bulk<br>Drums | | 1%, max<br>4%, max |

NOTE: Numbers in brackets refer to correspondingly numbered test methods in 5.1, References to the Text.

*American Public Health Association.

†Saybolt Universal seconds.

**3.2 Capacitor-Grade Askarel.** In September 1971 a new grade of capacitor impregnant, Aroclor 1016, was made available to the industry. This new grade contains a typical concentration of 0.4% by weight of the higher-boiling homologues of the chlorinated biphenyls. (See ASTM D 3303-74 [1].)[1] Aroclor 1016 replaces Aroclor 1242, which previously was the major capacitor impregnant and contained around 7% of the higher boiling homologues (the more persistent in nature). Thus 0.4% level of the higher-boiling homologues should be the maximum concentration acceptable in any capacitor impregnant. Aroclor 1242 and 1254, previously used as impregnants, do not meet this requirement and should no longer be used in capacitors designed and manufactured for alternating-current applications.

Aroclor 1016 has the same Underwriters' Laboratories, Inc, nonflammability rating as Aroclor 1242. Its typical properties are given in Table 1.

**3.3 Plant Housekeeping and Employee Safety.** The procedures and limits given in 3.3 are intended to be minimum requirements to be met by manufacturers and users of capacitors containing askarel. Handling, control, and disposal procedures are given, together

[1] Numbers in brackets refer to corresponding numbers in 5.1, References to the Text.



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with exposure limits and indicated antidotes and clean-up procedures

3.3.1 Material. Askarel for use in capacitors should consist of homologues and isomers of chlorinated biphenyl with the concentration of the higher boiling homologues at about 0.4%. (See ASTM D 3303-74 [1].) Aroclor 1016 is considered to be the standard impregnant meeting these requirements.

Commonly used solvents include benzene, kerosene, acetone, trichloroethane, trichloroethylene, and perchloroethylene. Typical vapor pressure data for Aroclor 1016 are:

0°C = 0.001 mmHg
25°C = 0.006 mmHg
150°C = 4.3 mmHg
200°C = 29.0 mmHg

At 25°C and 760 mmHg pressure, saturated air contains approximately 0.09 mg/l.

NOTE
1 mg/l = 90.0 ppm (v/v)
1 ppm (v/v) = 0.011 mg/l

3.3.2 Bulk Fluid Shipment, Receiving, and Transfer. Shipment of askarel from point of manufacture to point of receiving should be done in closed containers such as rail tank cars, truck tanks, marine or barge tanks, or sealed drums. Containers should be labeled as to contents and carry a label cautioning against loss of fluid to the open environment. Containers used to transport askarel should not be used for storage or to transport other material without being completely cleaned of all traces of askarel. (Cleaning procedures must take cognizance of precautions against excessive exposure and of the need for proper disposal of contaminated cleansing solvents and materials as set forth in 3.5.) Transfer from shipping containers to processing systems should be through closed piping or tubing with appropriate valves, pumps, etc. Provision should be made for trapping and disposing of fluid lost by leakage or spills from the transfer system and from the storage containers.

Drums to be retired from use should be cleaned before crushing, delivery to scrap dealers, or other disposal. Contaminated cleaning fluids and materials should be disposed of as indicated in 3.5.

3.3.3 General Safety Precautions. Although it is generally accepted that exposure to capacitor-grade askarel is not hazardous provided that simple precautions are taken, exposure should still be avoided.

3.3.3.1 Vapors. The odor of askarel is noticeable well below the maximum air concentrations considered safe. Up to 1.0 milligram per cubic metre of air has been determined to be the maximum safe level of exposure during an 8-hour workday. (See reference [10].) The procedure for performing the necessary analyses is contained in Section B3 of Appendix B. This procedure or its equivalent should be used.

Breathing vapor or fumes from heated askarel should be avoided. Provisions should be made for adequate ventilation and regulation of manufacturing operations to avoid open exposure to askarel (especially at temperatures of 55°C or higher). The gases produced when askarel is decomposed by very high temperatures (such as that of an electric arc) in the presence of air or organic insulating materials contain a high percentage of hydrogen chloride, and small percentages of carbon dioxide, carbon monoxide, and oxygen. Minute concentrations of this combination of gases are very unpleasant and irritating, thus giving ample warning of their presence. If exposure to high concentrations of askarel is necessary under emergency conditions, an approved gas mask or self-contained breathing apparatus should be worn. Such exposure should be under the surveillance of other personnel capable of effecting rescue in case of an accident. If the odor of askarel is detected by the person wearing protective equipment, he should immediately go into fresh air. All gas masks, respirators, and replacement parts should have U.S. Bureau of Mines approval and be maintained on a regular schedule in accordance with the manufacturer's recommendation.

3.3.3.2 Liquid. In contrast to the situation in which mineral insulating oils are handled, there is virtually no fire hazard in handling askarel. A limited solvent action (similar to that for paint thinner) on the fats and oils of the skin with prolonged contact may lead to drying and chapping of the skin. As with insulating oil, some people are allergic to askarel, and continued exposure may result in skin irritation. Both the liquid and vapor are moderately irritating to eye tissue.

Operating procedures should be such as to minimize or eliminate contact with askarels. Use of eye protection is recommended. The use of porous gloves that can absorb and retain askarels is to be avoided. Barrier creams[4] or resistant gloves[5] should be used if contact is unavoidable. Use of enclosed transfer and handling equipment, processing equipment, and mechanical washers reduces direct contact.

Medicinal washes or mild detergents followed by the application of cold cream will reduce the irritation resulting from the contact of an open cut or abrasion with askarel.

Safety glasses with side shields or a face shield should be worn when handling askarels. If liquid aska-

[4] For example, PLY No. 9 Gel (Malburn Company, Detroit, Mich.) or Kerodex No. 71 (Arcst Laboratories, New York, N.Y.), or the equivalent.

[5] For example, Edmont Style 5-352 (Merrick of Bridgeport, Bridgeport, Conn.), or the equivalent.



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rel contacts the eyes, the eyes should be irrigated immediately with large quantities of running water for 15 minutes and then examined by a physician. (A drop of castor oil has been found to reduce irritation.)

Persons developing a skin irritation or respiratory tract irritation while working with askarels should be placed under the supervision of a physician.

Ingestion or swallowing of askarels is not generally regarded as a problem of the industry. Should accidental ingestion occur, a physician should be consulted. Hands should be washed with warm water and soap before eating, drinking, smoking, or using toilet facilities.

3.3.4 Manufacturing Housekeeping. Manufacturing equipment and operating procedures should safeguard against loss of askarels to the environment through proper containment and disposal procedures.

Enclosed systems of sealed piping, properly gasketed joints, valves, containers, and processing chambers should be used for any portion of the operation where askarel temperatures may exceed 55°C. Enclosure should preferably extend to all other portions of the system insofar as practicable.

Containment provisions should be established around all askarel processing areas to ensure against inadvertent loss to sewer systems by spillage, leakage, or other uncontrolled conditions or events.

Spills of askarel should be removed promptly by means of absorptive material, such as sawdust, or trapped and removed by pumping or other suitable means.

Waste fluids containing askarel not suitable for reconditioning or reuse should be collected (by means of traps, drip pans, trays, etc.) from the various parts of the manufacturing and processing area (including washers or other cleaning devices). Disposal should be made in accordance with 3.5.

Wiper rags, clothing, and other extraneous materials saturated with askarels should be collected within the containment area for properly controlled laundering or disposal (see 3.5).

3.3.5 **Disposal of Askarel Wastes.** Methods for disposal of liquids and saturated solids generated by the manufacturing operation should be in accordance with those outlined in 3.5, and should include (but not be limited to) the following wastes.

(1) Contaminated liquid askarel that is unsuitable for reclaiming as a dielectric fluid.

(2) Liquid askarel from solvent operations or water and detergent type washers.

(3) Saturated earth or other absorbent media from filtering operations.

(4) Saturated sawdust or other absorptive materials from spills.

(5) Saturated filters from vapor-control devices and other filters.

(6) Saturated wastes (paper, rags, etc.).

(7) Saturated, spent gasket materials.

(8) Askarel-contaminated vacuum pump oils.

(9) Askarel-contaminated steam jet vacuum system condensates.

3.3.6 Miscellaneous Procedures. Other safety considerations include the following:

(1) Spills by leakage from finished capacitors should be cleaned up promptly by means of absorbent media, which should then be moved to containers provided for that purpose within the containment area, and later disposed of properly.

(2) Askarel wastes should never be disposed of down effluent drains or sewers. The utmost care must be exercised to prevent accidental loss by these avenues to the environment.

(3) Capacitors failing tests or otherwise designated for disposal must be controlled and handled in accordance with the intent of the procedures given in 3.3.6(1) and 3.3.6(2), finally being disposed of by one of the means outlined in 3.5.

**3.4 Control of Water Effluents.** The industry goal is to eliminate askarel in plant water effluent streams. However, it is recognized that existing drain systems in capacitor manufacturing plants are probably contaminated as a result of past practices, and askarel traces may continue to show up in effluent streams for some time. However, the level should continue to decrease with the proper containment of askarel wastes and no further discharges into drain systems. Other sections of this standard provide that no askarel wastes of any kind be disposed of in any water effluent streams and that accidental spills be prevented from getting into such streams.

3.4.1 Concentration Limits. The 1972 Environmental Protection Agency proposals are to keep PCB levels in rivers and lakes below 0.01 part per billion. This is currently under review by the EPA and standards are expected to be promulgated in 1971. Plant effluent streams should be managed and controlled in a manner anticipating these government standards.

3.4.2 Monitoring Streams. On a regular basis consistent with plant situations, all effluent streams should be analyzed. The procedure for performing the necessary analyses is contained in Section B4 of Appendix I. This procedure or its equivalent should be used.

3.4.3 Methods for Minimizing Effluent Stream Contamination. The ideal approach is to isolate potential effluent streams that could be contaminated with askarels during manufacturing processes and prevent them from being discharged from the plant. Carbon adsorption, limestone beds, and solvent extraction are two



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niques that can be applied to reduce the askarel content of effluent streams. These techniques may be most useful in cleaning up water used in plant processing and to permit recycling.

3.5 Scrap-Disposal Procedures. The manufacture and use of capacitors involve processes that produce askarel-saturated solids and liquids containing or composed entirely of askarel which should be disposed of as wastes. Specific sources of these materials are described throughout this standard. They may be placed into three categories:

(1) Capacitor units impregnated with askarel, production and field rejects.

(2) Manufacturing process liquid wastes containing askarel.

(3) Solid waste purposely or accidently saturated with askarel.

Disposal should be done in a manner that is consistent with proper concern for the environment and minimizes any release of askarels to the environment.

3.5.1 Disposal of Capacitor Units. Scrap capacitor units can be generated during manufacturing processes or during field service.

Production rejects are those capacitors that are rejected after the impregnation process in the course of production by the capacitor manufacturer. They may be rejected for mechanical or electrical reasons, or because of obsolescence.

Field rejects are those units that are rejected or, for other reasons, are to be scrapped after shipment from the plant where they were manufactured.

3.5.1.1 Production Rejects. Rejected capacitors in capacitor manufacturing plants at present a concentration of askarel. It is important that their disposition be made in a manner consistent with a proper concern for the environment. Therefore, capacitors should be disposed of only in supervised dry landfill sites that meet all applicable state requirements.

Care should be exercised to ensure that no loss of liquid will occur during transportation to the disposal site.

Incineration of scrap capacitors in facilities designed to accept such solids should provide an alternative means of disposal as such services become available in the future.

3.5.1.2 Field Rejects. Small capacitors (defined as containing less than 2 pounds of askarel) are practically always used as components in other electrical or electromechanical equipment. Typical examples of large quantities in use of such capacitors are fluorescent lamp ballasts and residential air conditioning equipments. Failure of such capacitors may result in scrapping of the device of which it is a part (as in a fluorescent ballast) or replacement and scrapping of the individual capacitor (as in a room air conditioner). However, the majority of such capacitors do not fail in service, but are scrapped as a result of wearing out or obsolescence of the devices in which the capacitors are used as components. Thus, the matter of disposal is characterized by a low concentration of small quantities of askarel throughout the country and, indeed, throughout the world. Fortunately, the nature of the devices and equipment in which such capacitors are used is such that they are normally disposed of in dry landfills as a matter of convenience. Since it is impractical at present to exercise any meaningful control over the disposition of the bulk of such devices and equipment, it is imperative that askarel used for impregnating small capacitors be limited to the recently introduced type, which contains a typical concentration of 0.4% of the higher-boiling homologues. (See ASTM D 3303-74 [1].)

Large capacitors (those incorporating more than 2 pounds of askarel) should be disposed of according to the procedure for production rejects (see 3.5.1.1).

3.5.2 Disposal of Liquid Wastes. All waste askarel or liquid wastes containing askarel should be disposed of in accordance with one of the procedures given in 3.5.2.1 through 3.5.2.3.

3.5.2.1 Incineration. Present knowledge indicates that proper incineration must involve a suitable balance between dwell time and temperature in the incinerator, plus oxygen availability and, finally, suitable scrubbers to remove the HCl that will be formed, for example, 2-second dwell time at 2000°F and 3% excess oxygen in stack gas, or 1.5 second dwell time at 2700°F and 2% excess oxygen in stack gas.

These facilities should meet the applicable requirements of the state in which they are located, and should control effluents within the limits set forth in this standard.

3.5.2.2 Toxic and Hazardous Waste Disposal Sites. Certain landfill sites have been classified by state governments and the federal government as suitable for the disposal of toxic and hazardous liquids. Where such approved sites exist, they may be used for the disposal of liquid wastes described in this standard. (See Appendix A.)

3.5.2.3 Packaging and Shipment.

3.5.2.3.1 Transportation to the disposal facility should be in containers that will prevent leakage and accidental loss of askarel to the environment.

3.5.2.3.2 Containers should be labeled as to contents and precautions relative to loss to the environment.

3.5.2.3.3 Containers used for this purpose should not be used for any other purpose until they are removed from service and they are completely cleaned. Any

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solvents used in cleaning these containers will be contaminated with askarel and should be disposed of according to the same procedures described in 3.5.2.

3.5.3 Disposal of Solid Wastes. (See 3.5.1 for scrap capacitors.) All solid wastes that have been saturated with askarel should be disposed of by the following procedure:

(1) The saturated wastes should be placed into leakproof containers and transported to a supervised dry landfill site meeting state requirements. Alternatively, they can be disposed of by incineration in state-approved facilities.

(2) Solid absorbents used for spills can be disposed of uncontained in the supervised dry landfill site, transport to the site should be in closed containers. Alternatively, incineration can be used in accordance with 3.5.2. (See Appendix A for a listing of facilities.)

**3.6 Labeling.** Capacitor units vary greatly in size and in end use or application. Small capacitor units are frequently applied as a component of another piece of equipment, such as a fluorescent lighting ballast, a roadway or area lighting luminaire, a motor, etc. In such applications a label on the capacitor unit referencing approved disposal procedures would not normally be visible when the piece of equipment is disposed of.

For the foregoing reasons, the methods of providing disposal instructions for small capacitors and large capacitors are treated separately in 3.6.1 and 3.6.2, respectively.

**3.6.1 Small Units.** Small capacitors are defined as those that contain askarel in quantities up to about 2 pounds each and in which the free liquid does not exceed 0.4 pound. They are hermetically sealed in metallic cases. Such capacitors are applied as a component of a large piece of equipment. Attaching a label to such equipment referencing this standard or describing disposal procedures would be of limited practical value.

**3.6.2 Large Units.** Large capacitor units are defined as those that contain more than 2 pounds of askarel. The capacitor manufacturer should affix a label in a conspicuous place, referencing this standard or describing disposal procedures consistent with it. This label should contain, as a minimum, the following information:

CAUTION: This capacitor contains a polychlorinated biphenyl (PCB). To avoid possible environmental contamination, it should be disposed of only in supervised dry landfill areas meeting state requirements or in incineration facilities designed for disposal of PCBs. See American National Standard C107.1-1974 for further information. Copies are available from: American National Standards Institute, 1430 Broadway, New York, N.Y. 10018.

## 4. Transformer Guidelines

### 4.1 General

**4.1.1 Types of Transformer Askarels.**[6] Askarels of various compositional types are currently in use (for the general properties and types, see ASTM D 2283-73a [20].) Under arcing conditions, the gases produced, though predominantly consisting of noncombustible hydrogen chloride, can contain varying amounts of combustible gases depending upon the askarel type.

**4.1.2 Safety Precautions.** Based on about 40 years of safe industrial usage, askarels have been considered as relatively harmless materials to humans. There has been no known instance of human injury when askarels are used under the normally prescribed conditions of precaution and handling.

Although it has been generally thought that exposure to askarels is not hazardous provided that simple precautions are taken, exposure should still be avoided or minimized.

**4.1.2.1 Vapors.** The odor of askarel is noticeable well below the maximum safe air concentrations. Depending upon the composition of the askarel used, from 0.5 to 1.0 milligram per cubic metre of air has been determined to be the upper safe level of exposure during an 8-hour workday. (See reference [19].) The procedure for performing the necessary analyses is contained in Section B3 of Appendix B. This procedure or its equivalent should be used.

Breathing vapor or fumes from heated askarels should be avoided. High concentrations of vapors can cause irritation of the eyes, nose, throat and upper respiratory tract. Provisions shall be made for adequate ventilation and regulation of manufacturing operations to avoid open exposure of hot askarels (55°C or higher). The gases produced when askarel is decomposed by very high temperatures (such as that of an electric arc) in the presence of air or organic insulating materials contain a high percentage of hydrogen chloride, and small percentages of other gases. Minute concentrations of this combination of gases are very unpleasant and irritating, thus giving ample warning of their presence. If exposure to high concentrations of askarels or its arced products is necessary under emergency conditions, an approved gas mask of the organic canister type, or self-contained breathing apparatus, must be worn. Such exposure should be under the surveillance of other personnel capable of effecting rescue in case of accident. If the odor of askarel or its arced products is detected

[6] The following trademarks are among those employed by electrical manufacturers for identifying the askarels used in their products: Aroclor, Chlorextol, Inerteen, Nepolin, Pyranol, and Saf-T-Kuhl.



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by the person wearing protective equipment, he should immediately go into fresh air. All gas masks, respirators, and replacement parts should have U S Bureau of Mines approval and be maintained on a regular schedule in accordance with the manufacturer's recommendation.

4 1 2 2 Liquid. In contrast to the situation in which mineral insulating oils are handled, there is no fire hazard in handling askarels. A limited solvent action (similar to that for paint thinner) on the fats and oils of the skin with prolonged contact may lead to drying and chapping of the skin. As with insulating oil, some people are allergic to askarel and continued exposure may result in skin irritation. Both the liquid and vapor are moderately irritating to eye tissue.

Operating procedures should require avoidance of contact with any askarels. The use of porous gloves that can absorb and retain askarels is to be avoided. Resistant gloves and aprons of the neoprene, polyethylene, or fluoroelastomer[1] type should be used if contact is unavoidable. In case of spillage on the clothing, the clothing should be removed as soon as practical, the skin washed, and the clothing laundered.

Medicinal washes or mild detergents followed by the application of cold cream will reduce the irritation resulting from the contact of an open cut or abrasion with askarel.

Safety glasses with side shields or a face shield should be worn when handling askarels. Eyes that have been exposed to liquid askarel should be irrigated immediately with large quantities of running water for 15 minutes and then examined by a physician if the irritation persists. (A drop of castor oil has been found to reduce irritation.)

Persons developing a skin irritation or respiratory tract irritation while working with askarels should be placed under the supervision of a physician.

Ingestion or swallowing of askarels is not generally regarded as a problem of the industry. Should accidental ingestion occur, a physician should be consulted. Hands should be washed with warm water and soap before eating, drinking, smoking, or using toilet facilities.

4.1.3 **Transport Container Marking.** Any container, such as tank cars, tank trucks, drums, cans, etc, used to transport transformer askarels, new or used, should be labeled with the following:

CAUTION: This product contains polychlorinated biphenyls (PCBs). Care should be taken to prevent entry into the environment through spills, leakage, use, vaporization, or disposal of liquid or containers. Avoid prolonged breathing of vapors or mists. Avoid contact with eyes or prolonged contact with skin. If skin contact occurs, remove by washing with soap and water. Following eye contact, flush with water. In case of spillage onto clothing, the clothing should be removed as soon as practical, skin washed, and clothing laundered.

4.1.4 **Receiving, Handling, and Storage of Askarels.** Askarels are shipped in tank cars, tank trucks, steel drums, metal cans, and test sample containers. When received, all containers should be inspected for leaks.

4 1 4 1 **Storage Tanks.** Storage tanks should be erected so that inspection can be made for leaks or spills. Construction should be such that inadvertent leakage or spills are prevented from reaching streams and sanitary or storm sewers.

4.1 4 2 **Tank Cars and Tank Trucks.** All bulk shipment equipment should be inspected for leaks immediately upon receipt. Drain pans must be provided to prevent spillage from unloading hoses and connections. Askarel liquid collected in drain pans should be placed in drums labeled "SCRAP ASKAREL" for disposition.

4 1 4 3 **Steel Drums, Cans, and Test Sample Containers.** On delivery, all such shipments should be carefully inspected for leaks. The containers should be stored indoors in an area especially selected for this purpose. A curb should enclose the area to provide a basin for containing the askarel from one or more containers should the containers be damaged. The area must not have a drain that is connected to a sanitary or storm sewer.

If an indoor storage area is not possible, the containers should be stored under a lean-to.

4.1.5 **Control of Water Effluents.** The industry goal is to eliminate askarel in plant water effluent streams. However, it is recognized that existing drain systems from manufacturing plants, repair shops, and installation sites may be contaminated as a result of past practices. Other sections of this standard provide that no askarel wastes of any kind be disposed of in any water effluent streams and that accidental spills be prevented from getting into such streams.

4 1 5 1 **Concentration Limits.** The 1972 Environmental Protection Agency proposals are to keep PCB levels in rivers and lakes below 0.01 part per billion. This is currently under review by the EPA and standards are expected to be promulgated in 1974. Plant effluent streams should be monitored and controlled in a manner anticipating these government standards.

4 1 5 2 **Monitoring Streams.** All plant effluent streams should be monitored on a regular basis, consistent with plant situations. The procedure for performing the necessary analyses is contained in Section 11 of Appendix B. This procedure or its equivalent should be used.

[1] For example, Viton.



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4.1.5.3 Methods for Minimizing Effluent Stream Contamination. The ideal approach is to isolate totally all effluent streams that could be contaminated with askarels during manufacturing processes and prevent them from being discharged from the plant.

4.1.6 Disposal Procedures and Services.

4.1.6.1 Sources of Materials Requiring Special Handling and Disposal Procedures. Liquids containing PCBs and solids containing or contaminated with PCBs may be obtained from any of the following sources: transport containers, transformer manufacturing processes, in-test failures, liquids contaminated beyond reclamation, in-service transformer leaks and failures, askarel filled transformers scrapped for any reason, etc.

4.1.6.2 Classification for Disposal of Materials Containing PCBs. In general, there are three types of materials requiring disposal: liquids, burnable solid materials containing PCBs, and nonburnable solid materials contaminated with PCBs.

4.1.6.2.1 Liquids. Liquids containing PCBs requiring disposal by high temperature incineration may consist of the following:

(1) PCBs contaminated with mineral oil.

(2) Mineral oil contaminated with PCBs.

(3) Nonreclaimable contaminated transformer askarels, arced askarels, askarels from manufacturing spills, and sump accumulation, etc., rich in PCBs, and askarels from holding basins, drip and drain pans, washings, sample jars and containers, etc.

4.1.6.2.2 Burnable Solid Waste Materials Containing PCBs. These materials can be disposed of by high-temperature incineration and consist of cellulosic materials, rags, pressboard, wood, sawdust, fuller's earth in bulk or in cloth bags, blotter papers, nitrile or cork gaskets, etc.

4.1.6.2.3 Nonburnable Solid Waste Materials Containing, or Contaminated with, PCBs. These materials may consist of coil structures, steel, copper, aluminum filter units of the steel mesh construction type, askarel drums, cans, etc.

Materials of this nature should be allowed to drain with the liquid collected in drip pans, etc. Further removal of adhering PCBs can be accomplished by washing or solvent extraction with kerosene or other approved washing liquids such as perchloroethylene or trichloroethylene. Accumulated liquids can be disposed of as indicated in 4.1.6.2.1. Solid materials may be handled as normal scrap.

4.1.6.3 Shipment of Scrap Liquids for Disposal. All liquid scrap material should be placed in appropriate metal transport drums, properly labeled, for shipment to a company offering an acceptable disposal service.

4.1.6.4 Shipment of Burnable Solid Waste Material Containing PCBs for Disposal. Material of this type should be placed in open head drums with suitable closures and with the drum properly labeled for shipment to a company offering an acceptable disposal service.

4.1.6.5 Liquid and Solid Waste Disposal Service Organizations.

4.1.6.5.1 General. Disposal of askarels and askarel-soaked materials should be accomplished by means in which there is no significant release of askarel to the environment. At present, disposal is accomplished by carefully controlled incineration of liquids and soaked software, and by controlled landfill burial of apparatus and other hardware from which askarel has been previously drained and washed.

Present knowledge indicates that proper incineration must involve a suitable balance between dwell time and temperature in the incineration plus oxygen availability and, finally, suitable scrubbers to remove the HCl that will be formed, for example, 2 second dwell time at 2000°F and 3% excess oxygen in stack gas, or 1.5-second dwell time at 2700°F and 2% oxygen in stack gas.

These facilities should meet the applicable requirements of the state in which they are located and should control effluents within the limits set forth in this standard.

Controlled landfill or deep-well disposal can be used where permitted by federal, state, and local regulations.

4.1.6.5.2 Costs. In addition to the normal costs of collecting scrap liquids and solids for disposal, additional costs borne by the owner of such scrap include shipping containers, cost of transport to the disposal service organization, and a disposal fee usually based upon a per gallon or per pound charge.

4.1.6.5.3 Disposal Services. Organizations offering disposal services are listed in Appendix A, including their location, facilities available, types of material handled, and disposal procedures used. Specific shipping directions, disposal procedures, and costs should be obtained from the organization.

4.2 Specific Guidelines.

4.2.1 Plant Housekeeping. It is necessary to assume that in filling equipment with askarel, and during further handling of this equipment, an askarel spill may occur. Therefore, it is necessary to provide facilities and a procedure for cleanup to prevent contamination.

4.2.1.1 Askarel Filling Area.

4.2.1.1.1 The location of the askarel filling area should be adjacent to the test area and final shipping area to minimize the danger of damage of units during handling.

4.2.1.1.2 The main manufacturing area for filling equipment with askarel should be provided with impervious surface floors or suitable basins so con-



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structed that any inadvertent leakage or spills are prevented from reaching streams, sanitary sewers, or storm sewers. All askarel handling equipment, such as pumps, hoses, etc, shall be of the askarel resistant type.

4.2.1.3 Drip pans shall be provided for hose connections and filling valves.

4.2.1.2 Special Containers for Scrap Materials.

4.2.1.2.1 Drums labeled "SCRAP ASKAREL" should be available for handling all spilled and waste askarel from sumps, failed units, drip pans, sample jars, etc.

4.2.1.2.2 Open head drums with suitable closures and labeled "SCRAP BURNABLE ASKAREL WASTE" should be available for handling contaminated cellulose insulation, rags, paper pressboard, wood, gaskets, sawdust, etc.

4.2.1.2.3 Separate containers for handling steel, copper, and aluminum, each adequately marked, shall be provided for the components of contaminated core and coil assemblies. These containers are required for the various materials when repairing or scrapping assemblies.

4.2.1.2.4 Containers for supplies of material for absorbing small askarel spills or cleanup of larger spills should be provided.

4.2.1.3 Conditioning of Askarels.

4.2.1.3.1 Askarel Conditioning Equipment. The conditioning unit should be located either in the storage tank area or in the main transformer manufacturing area for filling with askarel.

4.2.1.3.2 Fuller's Earth. Conditioning of new askarel or recycled askarel requires fuller's earth treatment. The spent fuller's earth in cartridges or bags, when replaced, should be allowed to drain thoroughly over drip pans to remove as much liquid askarel as possible. The cartridge units of steel mesh construction should be placed in the "STEEL CONTAMINATED WITH ASKAREL" container for disposition. Cloth bags filled with fuller's earth should be placed in the "SCRAP BURNABLE ASKAREL WASTE" container for disposition.

4.2.1.4 Teardown of Units for Repair or Scrap.

4.2.1.4.1 Drain all askarel from the unit either into a holding tank for reuse or into the drum labeled "SCRAP ASKAREL" for disposition, and then allow sufficient time for all of the askarel to drain from the core and coils.

4.2.1.4.2 Remove the core and coil assembly from the transformer. Sufficient absorbent material should be placed on the floor to absorb any askarel fluid that still drips from the transformer.

4.2.1.4.3 Place all materials in the appropriate salvage containers during the dismantling for later disposition.

4.2.1.4.4 All used materials including rags, sawdust, tape, etc, regardless of quantity, shall be put into the appropriate containers for disposition.

4.2.2 Transformer Labeling.

4.2.2.1 New Transformers. All new transformers that contain PCBs shall have a label of adequate durability, permanently and prominently attached to the tank by the manufacturer, giving adequate warning and instructions. A suggested label includes the following:

CAUTION: The insulating liquid in this transformer contains polychlorinated biphenyls (PCBs). Care should be taken to prevent entry into the environment. In case of malfunction or leaks, consult the instruction manual or the manufacturer.

4.2.2.2 In Service Transformers. The transformer manufacturer should make available suitable labels with a similar warning as shown in 4.2.2.1 for use on existing transformers.

4.2.3 Information for Transformer Users.

4.2.3.1 General. Askarel filled transformers are delivered to customers as sealed units from which there is no escape of askarel under normal operation. Although certain types of equipment failures can permit loss of some askarel to the environment, such cases are extremely rare.

4.2.3.2 Transportation and Receiving. Immediately upon receipt of the equipment and following any transportation or handling accident that could affect the integrity of the tank, bushings, or radiators, the transportation vehicle, tank, and fittings should be examined for any leakage or spillage that may have occurred in shipping. If leakage is evident, the cause should be corrected and the spillage soaked up with absorbent material, such as sawdust, followed by a cleanup of the affected area with rags soaked with kerosene or other approved solvent such as perchloroethylene or trichloroethylene. All materials used should be collected for proper disposition as described in 4.1.6.

4.2.3.3 Installation and Periodic Inspection. Following installation, the unit should again be inspected for any damage or leakage. It is recommended that periodic in-service inspections be made for any leaks.

4.2.3.4 Filling, Filtering, or Drying Askarel. Most askarel units are shipped with the proper amount of askarel, but if it becomes necessary to top off a unit, the manufacturer's instructions should be followed.

If it is necessary to dry an askarel unit or to treat an askarel unit with fuller's earth, instructions should be followed. When filtering or conditioning askarel, all of the precautions previously described for drip pans, proper disposal of filter media, etc, apply.

4.2.3.5 Sampling. It is common practice to sample askarel from a transformer for periodic maintenance testing. As previously described, such samples should be



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taken in a manner to avoid any contamination of the environment. Washings should be collected for proper disposal. Field and laboratory test samples, washings, etc, should also be collected for proper disposal.

4.2.3.6 Transformer Disposal. The ultimate disposal of an askarel filled transformer may be accomplished in either of two ways:

(1) Complete drainage and dismantling with the proper disposal of the askarel and askarel soaked components as described in 4.1.6.

(2) Disposition of askarel transformers by means of junk or scrap dealers. This should be avoided unless a transformer is first drained, followed by soaking the interior with a suitable solvent. Accumulated liquids and washings are to be disposed of as described in 4.1.6.

## 5. References

### 5.1 References to the Text

[1] Standard Method of Test for Rapid Gas Chromatographic Estimation of Higher Boiling Homologues of Chlorinated Biphenyls for Capacitor Askarels, ASTM D 3303-74.

[2] American National Standard Method of Test for Specific Gravity of Askarels, C59.68-1965 (R1973) (ASTM D 1810-63 (1973)).

[3] American National Standard Method of Test for Neutralization Number by Color-Indicator Titration, Z11.131-1964 (R1974) (ASTM D 974-64 (1973), IP 139/65).

[4] American National Standard Method of Test for Neutralization Number by Potentiometric Titration, Z11.59-1958 (R1971) (ASTM D 664-58 (1968), IP 177/64).

[5] Standard Method of Test for Density of Rubber Chemicals, ASTM D 1817-66 (1972).

[6] American National Standard Method of Test for Inorganic Chlorides in Askarels, C59.55-1963 (R1973) (ASTM D 1821-63 (1973)).

[7] American National Standard Method of Test for Pour Point of Petroleum Oils, Z11.5-1966 (R1972) (ASTM D 97-66 (1971), IP 15/67).

[8] American National Standard Method of Test for Power Factor and Dielectric Constant of Electrical Insulating Liquids, C59.2-1967 (R1973) (ASTM D 924-65 (1973)).

[9] American National Standard Method of Test for Specific Resistance (Resistivity) of Electrical Insulating Liquids, C59.51-1965 (R1973) (ASTM D 1169-64 (1973)).

[10] American National Standard Method of Test for Hydrolyzable Chlorine Compounds in Chlorinated Aromatic Hydrocarbons (Askarels), C59.106-1970 (ASTM D 1820-65 (1971)).

[11] American National Standard Method of Test for Thermal Stability of Chlorinated Aromatic Hydrocarbons (Askarels), C59.111-1970 (ASTM D 1936-64 (1971)).

[12] American National Standard Method of Test for Distillation of Road Tars, A37.9-1974 (ASTM D20-72).

[13] American National Standard Methods of Testing Electrical Insulating Oils, C59.2-1974 (ASTM D 117-71 (1973)).

[14] American National Standard Method of Test for Dielectric Breakdown Voltage of Insulating Liquids Using Disk Electrodes, C59.19-1968 (R1973) (ASTM D 877-67 (1971)).

[15] American National Standard Method of Test for Flash and Fire Points by Cleveland Open Cup, Z11.6-1973 (ASTM D 92-72, IP 36/67).

[16] American National Standard Method of Test for Saybolt Viscosity, Z11.2-1956 (R1971) (ASTM D 88-56 (1973)).

[17] American National Standard Method of Test for Coefficient of Thermal Expansion of Electrical Insulating Liquids of Petroleum Origin, and Askarels, C59.57-1963 (R1973) (ASTM D 1903-63 (1973)).

[18] MUNCH, R. H. Measuring the dissipation factor, dielectric constant, and resistivity of liquids. *Insulation/Circuits*, vol 16, Mar 1970, pp 46–49.

[19] Chlorodiphenyls. *Hygienic Guide Series*. Westmont, N.J.: American Industrial Hygiene Association, Jan-Feb 1965.

[20] Standard Specification for Chlorinated Aromatic Hydrocarbons (Askarels) for Transformers, ASTM D 2283-73.

### 5.2 General References

*Aroclor*® *Polychlorinated Polyphenyls (Biphenyls)*. Technical Bulletin O-F/1R. St. Louis: Monsanto Industrial Chemicals Company, Nov 1971.

DRINKER, C. K. Further observations on the possible systemic toxicity of certain of the chlorinated hydrocarbons. *Journal of Industrial Hygiene and Toxicology*, vol 21, 1939, pp 155–159.



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DRINKER, C. K., WARREN, M. F., and BENNET, G. A. The problem of possible systemic effects from certain chlorinated hydrocarbons. *Journal of Industrial Hygiene and Toxicology*, vol 19, 1937, pp 283-311

ELKINS, H. B. *The Chemistry of Industrial Toxicology*. New York, John Wiley & Sons, Inc, 1959

GREENBURG, L., MAYERS, M. R., and SMITH, A. R. The systemic effects resulting from exposure to certain chlorinated hydrocarbons. *Journal of Industrial Hygiene and Toxicology*, vol 21, 1939, pp 29-38

*Threshold Limit Values for Chemical Substances and Physical Agents in the Workroom Environment*. Cincinnati: American Conference of Governmental Industrial Hygienists, 1973

TREON, J. F., CLEVELAND, F. P., CAPPEL, J., and ATCHLEY, R. W. The toxicity of the vapors of Aroclor 1242 and Aroclor 1254. *American Industrial Hygiene Association Quarterly*, vol 17, 1956, pp 204-213

## 6. Revision of American National Standards Referred to in This Document

When the American National Standards referred to in this document are superseded by a revision approved by the American National Standards Institute, Inc, the revision shall apply.



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# Appendixes

(These Appendixes are not a part of American National Standard Guidelines for Handling and Disposal of Capacitor and Transformer-Grade Askarels Containing Polychlorinated Biphenyls, C107.1-1974, but are included for information purposes only.)

## Appendix A
## Disposal Services

In addition to the supervised dry landfill sites that may be used for the disposal of askarel-containing scrap, the following additional known facilities and services have been established, and others may be available.*

Chem Trol Pollution Services, Inc
P.O. Box 200
1550 Balmer Road
Model City, N.Y. 14107
Phone: 716 754-8231

This organization has facilities and services capable of handling

(1) Liquids: Askarels alone or mixed with solvents or oils. Disposal by high-temperature incineration.

(2) Solids (software): Askarel-soaked compounds, rags, cartons, absorbing earths, etc. Disposal by incineration or scientific landfill.

(3) Solids (hardware): Capacitors, transformer tanks, cores, askarel soaked metals. Disposal by scientific landfill. Has solvent extraction capability.

Monsanto Company
800 North Lindbergh Boulevard
St. Louis, Mo 63166
Phone: 314 694-3352

This organization has facilities and services capable of handling askarel liquids alone or mixed with other oils or solvents by high-temperature incineration. Liquid is pumped through a gun with atomizing steam into incinerator. Temperatures are maintained at 2000°F-2500°F with auxiliary natural gas. Exit gases are quenched to 180°F by contact with water. Gas is then passed through a high-energy venturi scrubber for removal of particulates. Before exhausting to air (110°F), gases are passed through a packed column scrubber to remove HCl.

Nuclear Engineering Company
Eastern Division
P.O. Box 146
Morehead, Ky 40351
Phone: 606 784-6611

Nuclear Engineering Company
Disposal Division
Sheffield, Ill. 61361
Phone: 815 454-2624

This organization provides containerization, transportation, and disposal services of all liquids and solids (including hardware). Disposal is in conventional chemical and scientific landfill area. Licensed by Atomic Energy Commission for radioactive waste disposal. The organization also has two West Coast locations, in the states of California and Washington.

Rollins Environmental Services, Inc
P.O. Box 2349
Wilmington, Del 19899
Phone: 302 658-8451

This organization has facilities and services capable of handling

(1) Liquids: Askarel alone or mixed with other solvents or oils. Disposal is by high temperature incineration.

(2) Solids (software): Askarel soaked [illegible] rags, cartons, absorbing earths, etc. [illegible] incineration at combustion temperatures up to 2000°F. Incineration gases are scrubbed, and entrained askarel is removed before exhausting to air.

Rollins Environmental Services maintains disposal facilities in the following areas:

Philadelphia/Camden
Rollins Environmental Services, Inc.
Route 322
Logan Township
Bridgeport, NJ 08014

*The listing given herein is representative of some of the sources that provide this service, and is not presumed to be complete. Any other organizations that wish to be listed should notify the Standards Institute so that they may be included in the next edition of this standard.



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APPENDIX

Baton Rouge.
Rollins Environmental Services, Inc
Scenic Highway & West Cheatham Lane
Scotlandville
East Baton Rouge Parish, La 70807

Houston
Rollins Environmental Services, Inc
Tidal Road & Highway 134
Deer Park, Tex 77536

## Appendix B

## Analytical Procedures and Laboratory Service Organizations

### B1. General

Analytical procedures for determining PCBs in air, water, and sediments are given in Sections B3 and B4.

### B2. Laboratories

The following laboratories are representative of those offering services for PCB analyses [1]

Carus Chemical Corporation
1375 Eighth Street
La Salle, Ill 61301
Phone 815 223-1500

Limnetics, Inc (Subsidiary of Carus Corporation)
6132 West Fond du Lac Avenue
Milwaukee, Wis 53218
Phone. 414 461-9500

Gollob Analytical Service Corporation
47 Industrial Road
Berkeley Heights, N.J. 07922
Phone 201 464-3331

### B3. An Analytical Procedure for the Determination of Airborne PCBs

B3.1 Scope This procedure is based on techniques used by the Monsanto Industrial Chemical Company for the isolation and determination of PCBs in water, soil/sediment, and biological materials. Absolute confirmation of PCB structures is not obtained with this method Where needed, additional structure proof should be obtained using techniques such as mass spectrometry for further identification of gas chromatography fractions.

B3.2 Principle Airborne PCBs are absorbed in toluene by drawing the air through one or more fritted bubblers or impingers in cylinders filled with toluene After a suitable amount of air is sampled, the scrubbing solvent is diluted or concentrated, and interfering components, if present, are removed by chemical treatment and column absorption chromatography The amount and type of PCBs present are determined by electron capture gas chromatography (LC/GC)

B3.3 Reagents

(1) Hexane Pesticide grade

(2) Toluene Pesticide grade

(3) Sodium sulfate Anhydrous, granular, analytical reagent (AR) grade Heat at 400°C for 1 hour prior to use

(4) Alumina adsorption For chromatographic analysis, 80/200 mesh Heat at 400°C for a minimum period of 4 hours and deactivate with 5 % (w/w) distilled water

For alumina column preparation fill a chromatographic column with hexane up to the point where the reservoir joins the column, and push a glass wool plug to the bottom with a glass rod In a 50 ml beaker measure 35 ml (about 30 grams) of deactivated alumina, and pour this slowly into the column Tap or vibrate the column to settle the alumina and top the alumina with 2 to 3 cm of anhydrous sodium sulfate Wash the column with 50 to 100 ml of hexane prior to the addition of the sample

(5) Distilled water Extracted with hexane to re

[1] The listing given herein is representative of some of the sources that provide laboratory analysis services for PCBs compliance. Any others that should be listed should notify the Standards Institute so that they may be included in the next edition of the standard



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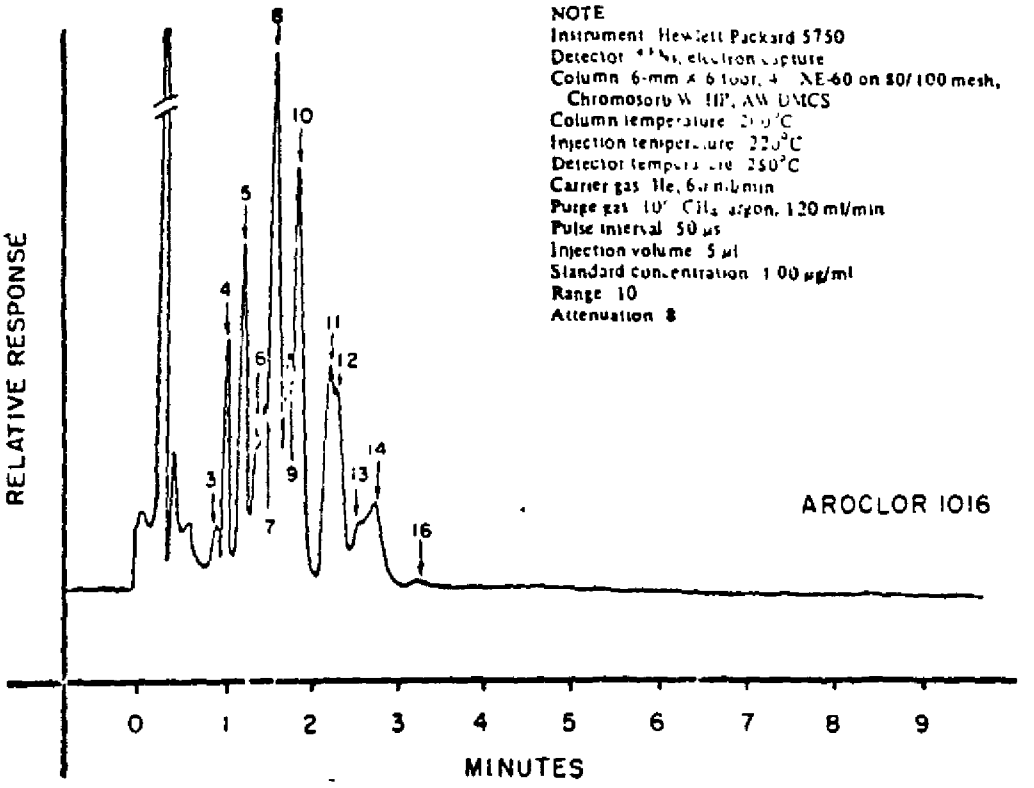


**Fig. B1**
**Aroclor 1016 Electron Capture Chromatogram**

move hexane soluble electron capturing impurities.

(6) Sulfuric acid AR grade, specific gravity = 1.84.

(7) Potassium hydroxide. AR grade.

(8) Ethanol Formula 2B.

(9) Alcoholic potassium hydroxide, 2.5% (w/v) Dissolve about 12.5 grams of AR grade KOH in 500 ml of ethanol

(10) Sulfuric acid-water, 1:1 (v/v) Carefully add 270 ml of AR grade sulfuric acid to 30 ml of distilled water in a 500 ml beaker

(11) PCB standards Aroclor 1016, 1221, 1242, 1248, 1254, and 1260 (See Fig B1 through B7)

B3.4 Apparatus

(1) Gas scrubbing bottles, high form, ground-glass joint, fritted coarse disks

(2) Separatory funnels equipped with ground glass stoppers and TFE fluorocarbon stopcocks, capacities of 125, 250, 500, 1000, and 2000 ml

(3) Kuderna-Danish evaporative concentrators, 500-ml capacity, equipped with three-ball Snyder columns and graduated 5-ml capacity vials.

(4) Chromatographic columns, glass, 10 inches × 20 mm (OD), with a 5 inch × 50 mm (OD) reservoir at the top, equipped with TFE fluorocarbon stopcocks

(5) Flat bottomed boiling flasks, 125 ml capacity

(6) Liebig condenser, 200 mm in length.

(7) Hot plate.

(8) Water bath

(9) 10-µl syringes.

(10) Dry-test meter or wet-test gas meter.

(11) Laboratory vacuum pump

(12) Rotating vacuum evaporator.

(13) Usual laboratory glassware.

B3.5 Sampling The air to be sampled for airborne PCBs is drawn through a gas scrubber (or scrubbers) and a dry-test meter using a laboratory vacuum pump See Fig B8. The sampling flow rate is controlled by bleeding air via a needle valve located between the



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APPENDIX

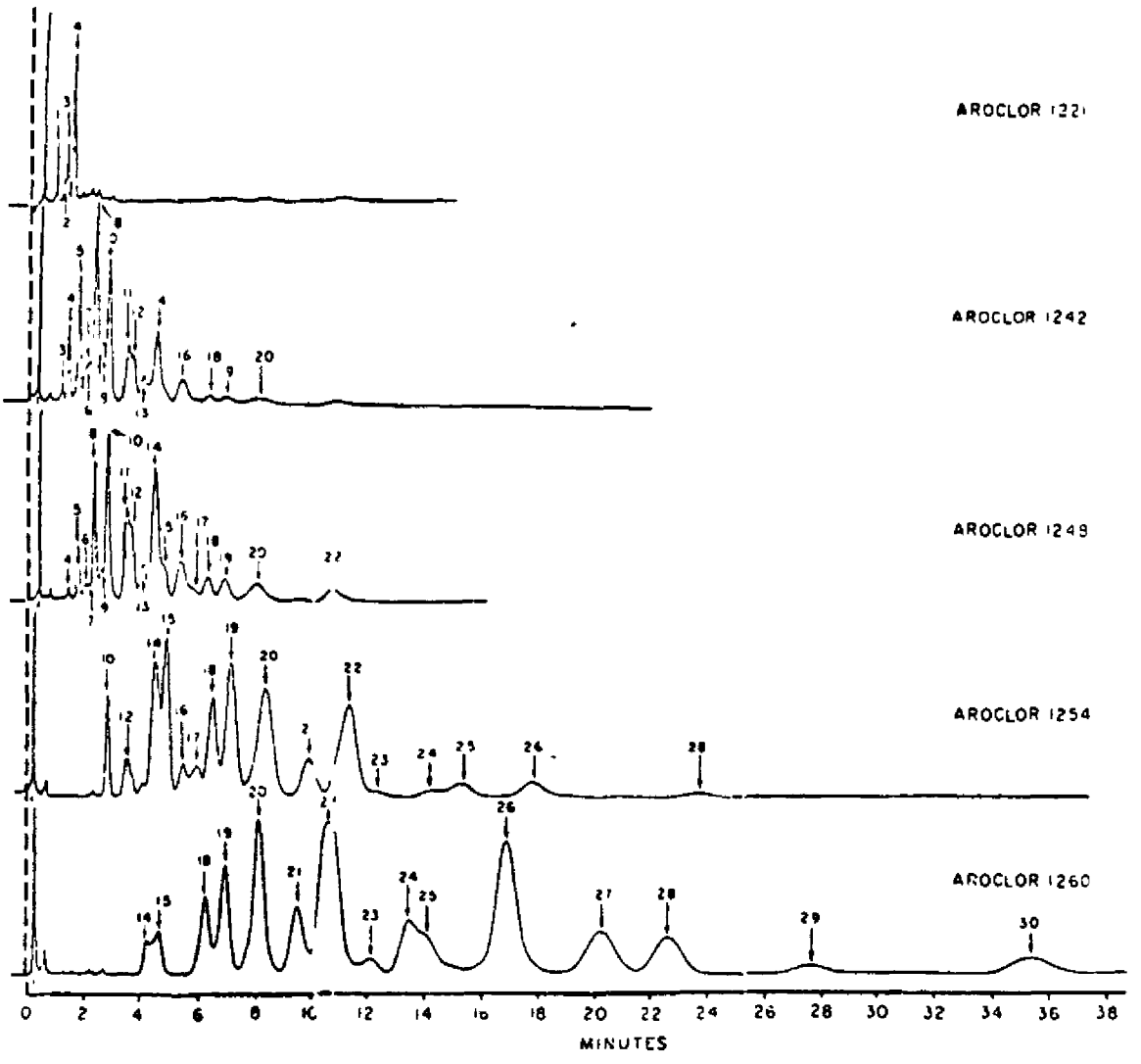


Fig. B2
Comparison of Electron Capture Chromatograms for
Aroclor 1221, 1242, 1248, 1254, and 1260

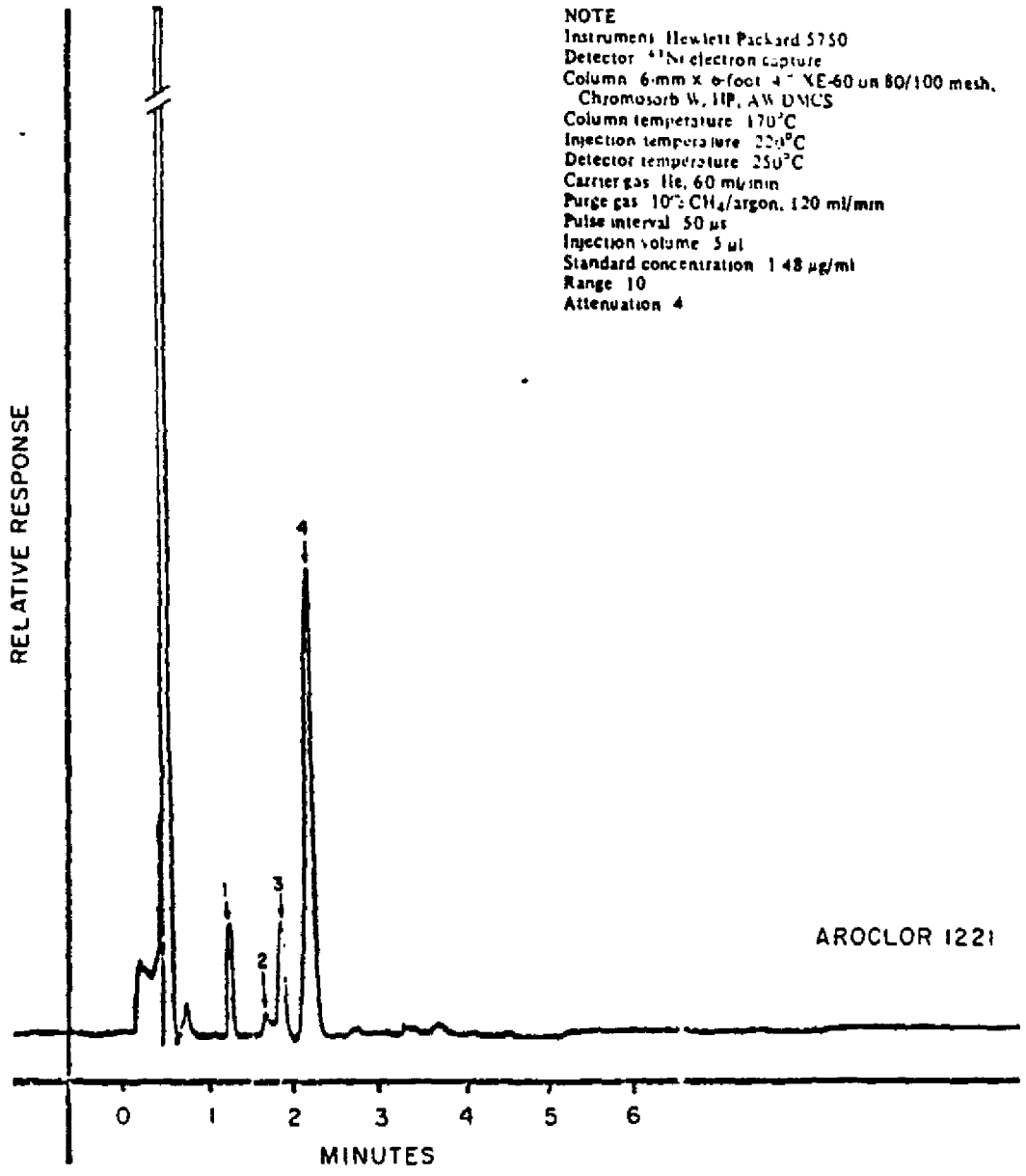


Fig B3
Aroclor 1221 Electron Capture Chromatogram

NOTE
Instrument Hewlett-Packard 5750
Detector $^{63}$Ni electron capture
Column 6 mm × 6 foot 4% XL-60 on 80/100 mesh, Chromosorb W, HP, AW-DMCS
Column temperature 190°C
Injection temperature 220°C
Detector temperature 250°C
Carrier gas He, 60 ml/min
Purge gas 10% $CH_4$/argon, 120 ml/min
Pulse interval 50 µs
Injection volume 5 µl
Standard concentration 1.03 µg/ml
Range 10
Attenuation 8

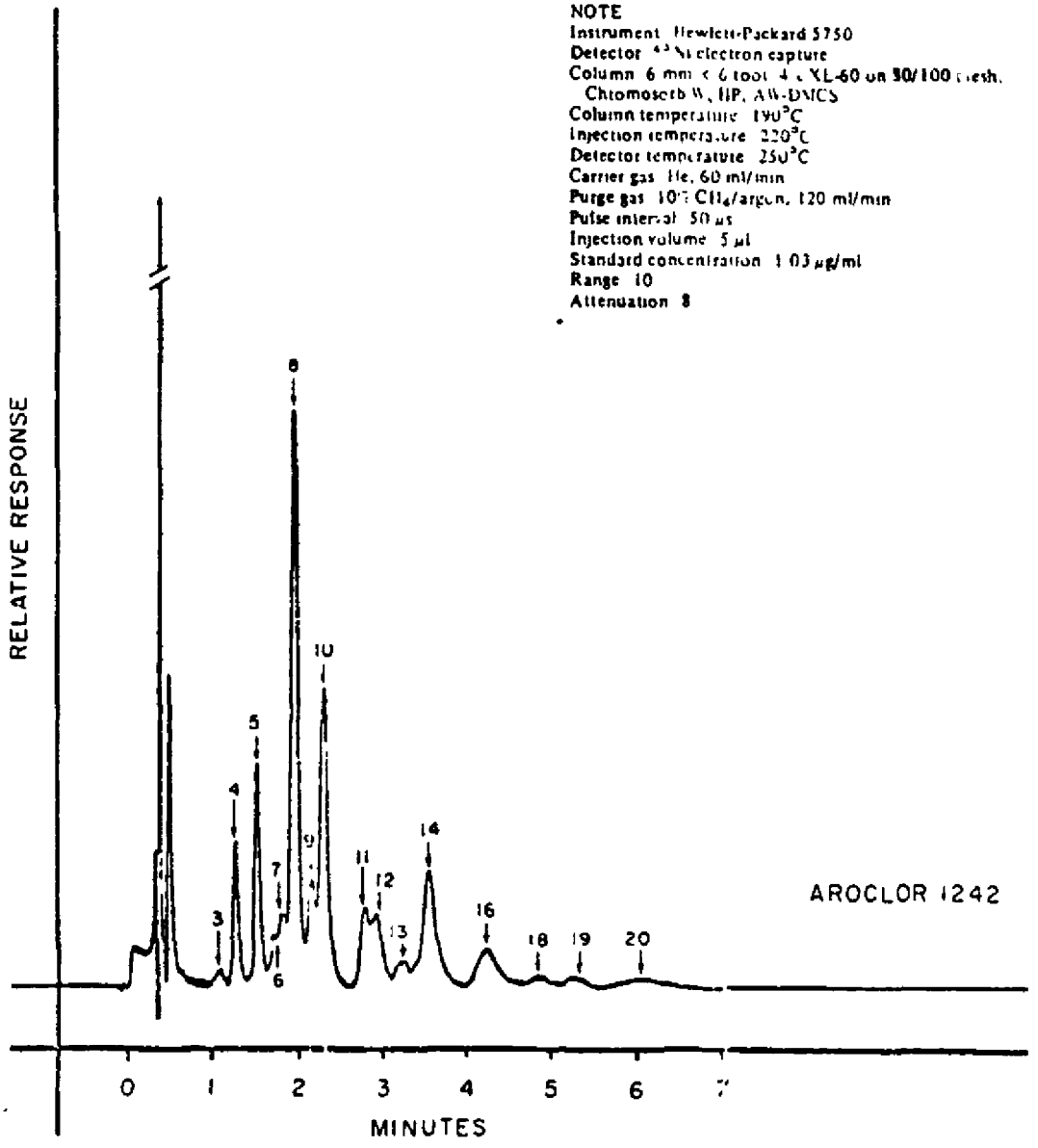


Fig B4
Aroclor 1242 Electron Capture Chromatogram

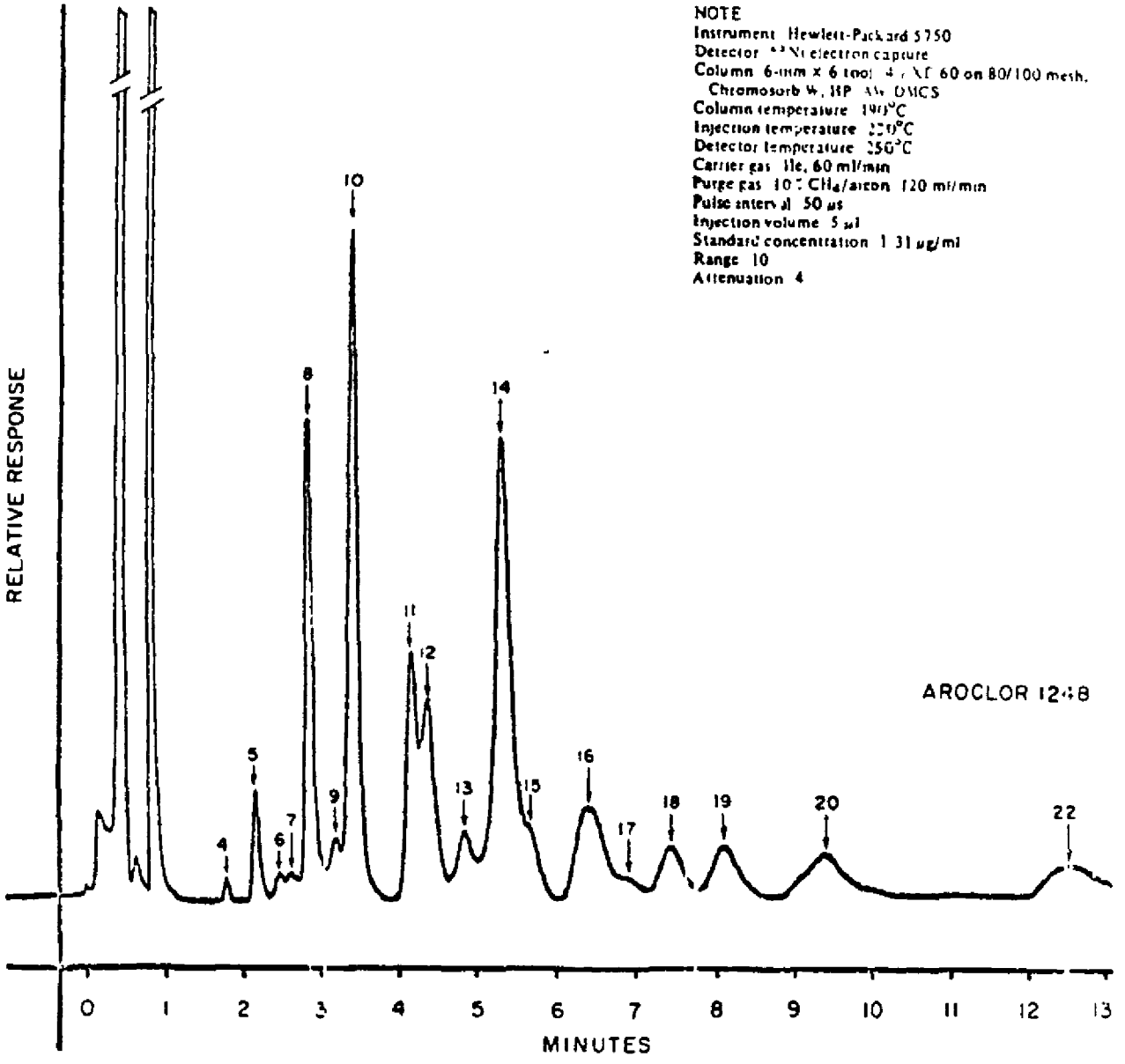


Fig. B5
Aroclor 1248 Electron Capture Chromatogram

APPENDIX

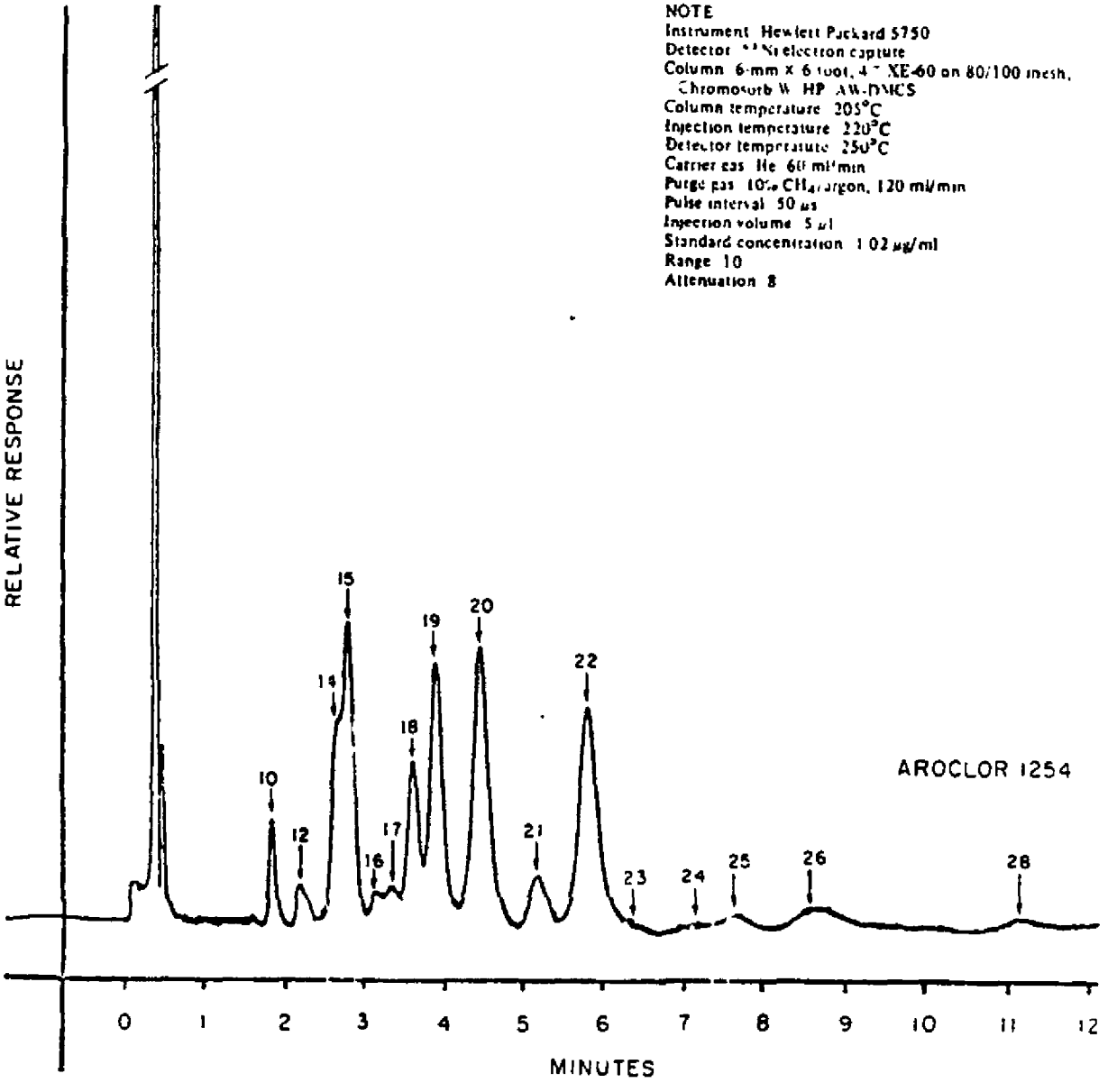


Fig B6
Aroclor 1254 Electron Capture Chromatogram

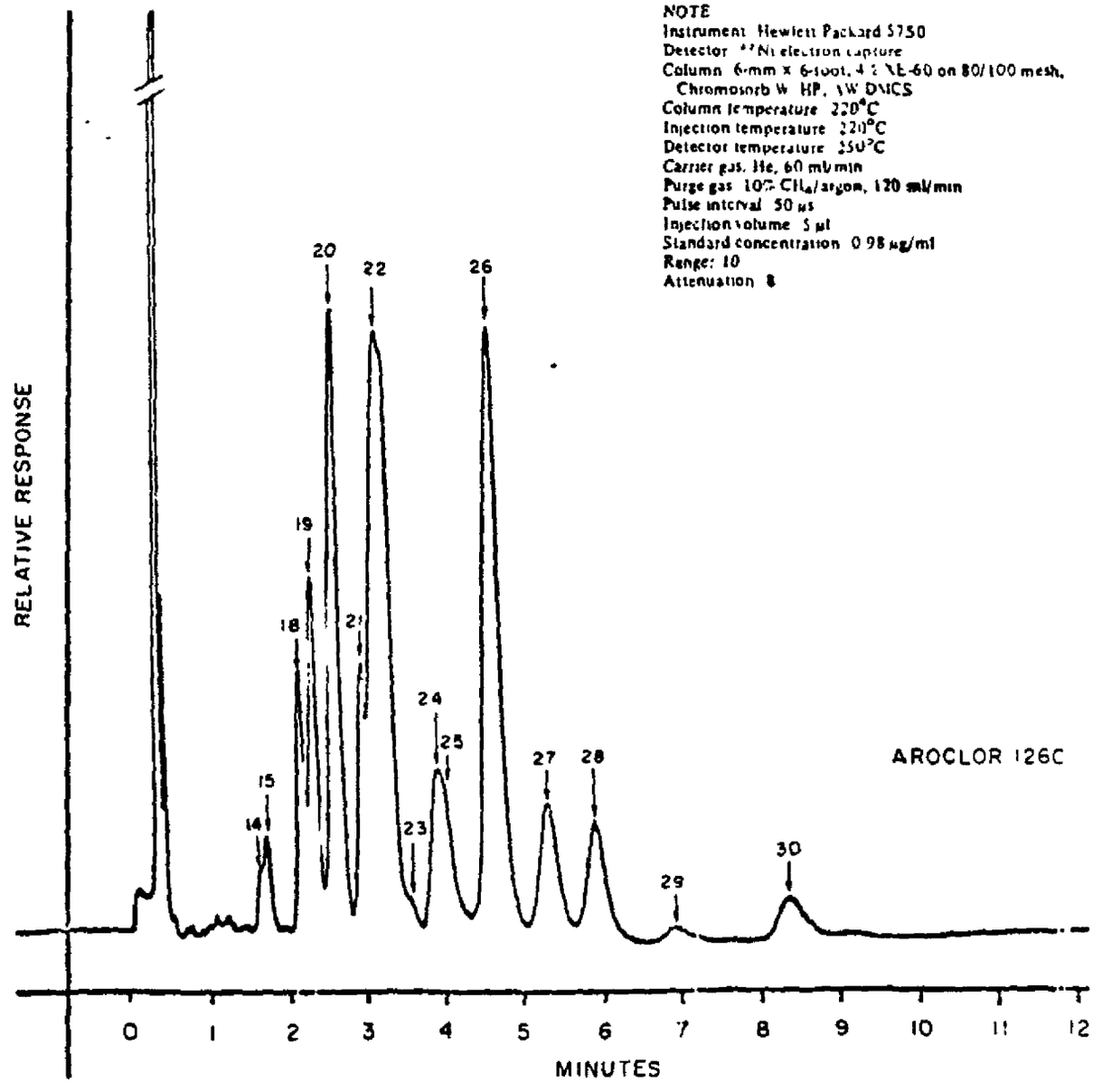


Fig B7
Aroclor 1260 Electron Capture Chromatogram

APPENDIX

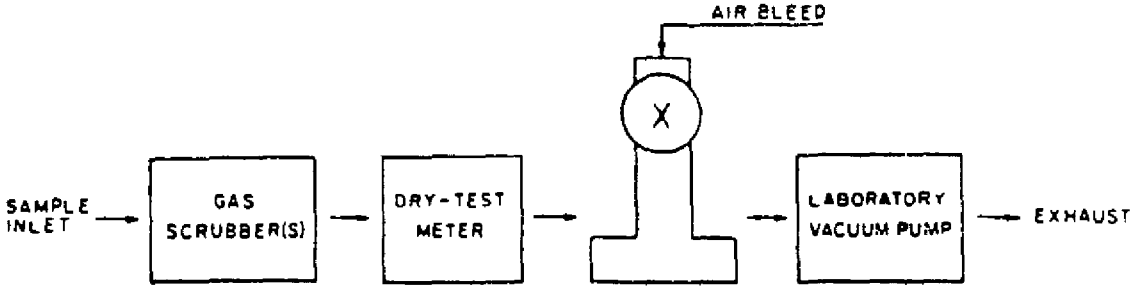


**Fig B8**
**Sampling Train**

pump and the meter At the end of the sampling period the metered gas volumes are corrected for temperature and pressure to cubic metres at 25°C and 760 mmHg

It is important to note that neither the capacity nor the efficiency of the gas scrubber(s) for removal of airborne PCBs have been experimentally evaluated For this reason, it is best to minimize the sampling flow rate and maximize the sampling time period to obtain measurable amounts of PBCs When high flow rates must be employed or a larger capacity may be needed, it is recommended that several gas scrubbers be used in tandem

It is cautioned that until the efficiency and capacity of the toluene gas scrubber have been experimentally established, this procedure should be used only to measure relative PCB levels sampled under equivalent conditions

B3.6 Procedure

(1) After scrubbing the desired amount of air, record the metered volume, pressure, and temperature

(2) Quantitatively transfer the scrubbing solvent to a round-bottomed flask and reduce the volume to approximately 2 ml by rotary vacuum evaporation

(3) Quantitatively transfer the concentrate to a 30 ml beaker with the aid of several small portions of toluene

(4) Inject a fraction of a microlitre of the concentrate into the gas chromatograph to check for interferences and determine the approximate level of PCBs present If no interferences are present dilute or concentrate the sample to a known volume, as determined by the electron capture chromatogram, and proceed with the gas chromatographic analysis

(5) If interferences are present, proceed with the chemical treatment and column chromatographic cleanup procedures

(6) Transfer the concentrate to a 125 ml extraction flask with the aid of several small portions of solvent

(7) Evaporate the concentrate just to dryness with a gentle stream of dry filtered air and add 25 ml of 2.5% alcoholic potassium hydroxide

(8) Add a boiling chip, put a water condenser in place, and allow the solution to reflux for 45 minutes

(9) After cooling, transfer the solution to a 250-ml separatory funnel with the aid of 25 ml of distilled water

(10) Rinse the extraction flask with 25 ml of hexane and add it to the separatory funnel

(11) Stopper the separatory funnel and shake vigorously for at least 1 minute Allow the layers to separate, and transfer the lower aqueous phase to a second separatory funnel

(12) Extract the saponification solution with a second 25-ml portion of hexane After the layers have separated, add the first hexane extract to the second separatory funnel, and transfer the aqueous alcohol layer to the original separatory funnel

(13) Repeat the extraction with a third 25-ml portion of hexane Discard the saponification solution and combine the hexane extracts

(14) Carefully add 25 ml of the sulfuric acid solution (9 1 concentrated sulfuric acid–water) to the hexane extracts

(15) Stopper the separatory funnel and shake vigorously for at least 1 minute Allow the layers to separate, and discard the lower aqueous acid layer Repeat this step until the acid layer is colorless

(16) Wash the hexane with a 25-ml portion of water Discard the water wash

(17) Filter the hexane extract through a 4 inch funnel, plug with glass wool that is covered with a layer of sodium sulfate, into a Kuderna Danish evaporative concentrator

(18) Add a small boiling chip put the Snyder column in place and reduce the hexane volume to less



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than 5 ml by heating the apparatus in a 80°C to 90°C water bath

(19) After cooling, remove the 5-ml graduated tube and transfer the hexane extract to an alumina adsorption column, washing it in with several 5 ml portions of hexane

(20) Carefully add 100 ml of hexane to the column reservoir, and collect the total eluent in either a 250 ml volumetric flask or a Kunderna Danish evaporative concentrator

(21) If the column eluent is collected in a volumetric flask, dilute to volume with hexane, and proceed with the gas chromatographic analysis

(22) If the column eluent is collected in a Kunderna-Danish evaporative concentrator, reduce solvent volume, cool, dilute to volume, and proceed with gas chromatographic analysis

**B3.7 Electron Capture Gas Chromatographic Procedure**

(1) Instrument Gas chromatograph (for example, Hewlett Packard Model 5750, or the equivalent)

(2) Detector High-temperature $^{63}Ni$, electron capture cell

(3) Column 6-mm X 6-foot glass column, 4% XE-60 on 80/100 mesh, Chromosorb W, HP, AW-DMCS

(4) Column temperature 180°C to 190°C

(5) Injection port temperature 195°C to 215°C

(6) Detector temperature. . 00°C

(7) Pulse interval 50 μs

(8) Flow rates Helium carrier, approximately 60 ml/min, argon-methane purge, approximately 120 ml/min

Using EC/GC as the determinative step, inject, in duplicate, 1 to 10 μl of each solution into the chromatograph By comparison with standard solutions injected, in duplicate, under the same operating conditions, determine the amount and type of Aroclor using the individual or total peak height and area methods.

**B3.8 Sample Concentration.** Concentration of sample extracts is necessary, prior to cleanup by chromatographic or chemical means to reduce sample size and increase sensitivity. The preferred method of concentrating allows minimum loss through volatilization or chemical decomposition and requires a minimum time The three methods of solvent volume reduction most commonly used are evaporation by exposure to a stream of air, evaporation employing a Kunderna-Danish evaporative concentrator equipped with a Snyder column, and evaporation under reduced pressure All three techniques have been used without encountering any significant losses from volatilization or chemical alteration However, the Kunderna Danish evaporative concentrator and the stream of air methods are easier to use

**B3.9 Column Adsorption Chromatography and Chemical Cleanup** Silica gel,[9] magnesia silica gel,[9] and alumina deactivated with 0%, 1.0%, 1.5%, 2.0%, and 5% water were investigated as adsorbants for the elimination of interferences Alumina (5% water) was found to be more effective and reproducible than either silica gel or a magnesia silica gel.[9] The activity of alumina varies with age and lot, therefore, 5% water was added to the alumina, after heating for a minimum of 4 hours at 400°C, to ensure a reproducible activity

Saponification, and subsequent extraction of the sample with sulfuric acid, is an effective way to remove a number of chlorinated hydrocarbon interferences as well as other matrix interferences PCBs are not affected

**B3.10 Column Performance** Column performance is the key to effective gas chromatographic analysis and, as such, the choice of column materials is particularly important Ideally, the support employed should be inert, mechanically strong, and of high surface area Chromosorb W, HP, AW-DMCS fulfills these requirements and is recommended for this work

A variety of polar and nonpolar liquid phases have been investigated The following columns have been found to provide adequate separation, etc, for use in PCB analysis by election capture 4% (w/w) DC 200, SF-96, OV-17, SE 30, SE 54, XE-60, Apiezon L, and 6% QF-1 DC 200 and XE 60 or QF-1 have been found to be the most suitable of these liquid phases.

Another important consideration when working with an extremely sensitive detector and, consequently, low levels of materials is column conditioning With polar phases such as XE 60 and QF-1, operating a new column overnight at a temperature 25°C to 50°C higher than that to be used during analysis results in a more stable column. A no-flow conditioning technique is employed to condition nonpolar columns The column is purged with carrier gas, heated for 30 minutes at an elevated temperature without carrier flow, and then cooled to room temperature At the end of this cycle, the carrier flow is resumed and the conditioning is completed as in the case of the polar liquid phase Two precautions should be observed during conditioning, the column should not be connected to the detector, and the maximum safe temperature of the liquid phase should not be exceeded

Since all liquid substrates bleed to one degree or another and columns eventually degrade, all new columns should be characterized with two column performance indicators the number of theoretical plates (N) and a tailing factor (T) p,p' DDT is employed to check these parameters because it is known to degrade on

[9] For example Chromosil



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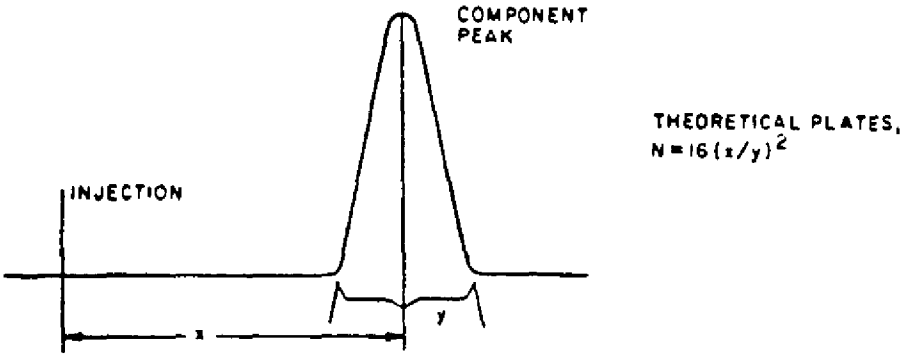


Fig. B9
Calculating Column Efficiency

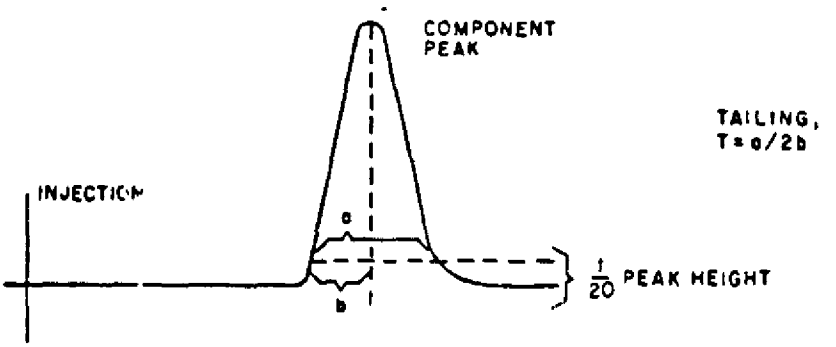


Fig. B10
Calculating the Tailing Factor

"poor" columns. In this manner one can determine whether the performance of a new column is satisfactory and when the column performance begins to fall off. A column is considered good if the number of theoretical plates per foot is of the order of 400 to 500, with tailing factors of 1.0 to 1.1. Calculation of these parameters is shown in Fig. B9 and B10. Additionally, there should be no significant extraneous peaks upon injection of a pure p,p'-DDT standard.

Other chromatographic conditions that can be adjusted are column temperature and flow rates. Although resolution of a mixture increases with decreasing temperature, a temperature should be chosen that allows the elution of all components within a convenient time period. The flow rates shown are optimum for a given instrument, column, and detector system. These should be adjusted if better results can be achieved.

Any system of instrument and column suitable for chlorinated pesticide analysis is satisfactory for PCB analysis. The use of the high-temperature $^{63}$Ni electron capture cell is highly recommended. The ability to operate at higher temperatures prevents maintenance problems due to contamination from high boiling components. Glass columns also should be employed.

B3.11 Detection and Measurement. Quantitative determinations employing the electron capture detector are nonstoichiometric measurements made by comparing peak heights or areas for known concentrations with those for unknown compositions. Three variations of the peak height or area quantification procedures have been employed.

Case I. If a chromatogram of PCB unknown, uncharacterized with respect to standard PCB with no evidence of interferences



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Case II EC gas chromatogram of PCB unknown, altered with respect to standard PCB with no evidence of interferences

Case III EC gas chromatogram of PCB unknown, unaltered with respect to standard PCB with evidence of interference

The amount of PCBs in Case I samples is determined by preparing a plot of the major peak height or area versus concentration. With Case III samples, a peak free from interference is used. When dominant interferences are present, one or more of the chemical cleanup procedures is employed.

In all cases, the response of the electron capture detector must be linear for quantitative analysis.

B3.12 Contamination. In determining PCBs in biological materials by electron capture gas chromatography, laboratory sources of contamination can be a major problem. The samples and extracts should never be allowed to come in contact with materials other than glass, TFE-fluorocarbon, or metal. Laboratory glassware should be thoroughly washed with hot, soapy water, and rinsed with distilled water, acetone, and then hexane. All equipment should also be rinsed again with hexane just prior to use, and blanks should be frequently carried through all steps of the procedures to ensure against the possibility of contamination.

B3.13 Sensitivity

(1) Concentration: 2 ppb (parts per billion)
(2) Absolute sensitivity: $0.5 \times 10^{-9}$ gram
(3) Volume injected: 5 µl
(4) Final volume of extract: 5 ml
(5) Sample size: 250 ml

## B4. Analysis of Water and Sediment for PCBs

B4.1 Scope. This methodology is used by the Monsanto Industrial Chemical Company for the determination of the amount and type of PCBs in water and sediment samples. Absolute confirmation of PCB structures is not obtained with this method. Structure proof can be obtained using additional techniques such as mass spectrometry to further identify the gas chromatography fractions.

B4.2 Principle. The PCBs in water and sediment samples are extracted into an organic solvent. Interfering components are then removed from the extracts by chemical treatment and column adsorption chromatography. The amount and type of PCBs present are determined by electron capture gas chromatography (ECGC).

B4.3 Reagents

(1) Hexane. Pesticide grade.
(2) Acetonitrile. Pesticide grade.
(3) Sodium sulfate. Anhydrous, granular, analytical reagent (AR) grade. Heat at 400°C for 1 hour prior to use.
(4) Alumina adsorption. For chromatographic analysis, 80/200 mesh. Heat at 400°C for a minimum of 4 hours and deactivate with 5% (w/w) distilled water.

For alumina column preparation fill a chromatographic column with hexane up to the point where the reservoir joins the column, and push a glass wool plug to the bottom with a glass rod. In a 50-ml beaker measure 35 ml (about 30 grams) of deactivated alumina, and pour this slowly into the column. Tap or vibrate the column to settle the alumina, and top the alumina with 2 to 3 cm of anhydrous sodium sulfate. Wash the column with 50 to 100 ml of hexane prior to the addition of the sample.

(5) Distilled water. Extracted with hexane to remove hexane-soluble electron capturing impurities.
(6) Sulfuric acid. AR grade, specific gravity = 1.84.
(7) Potassium hydroxide. AR grade.
(8) Ethanol. Formula 2B.
(9) Alcoholic potassium hydroxide, 2.5% (w/v). Dissolve about 2.5 grams of AR grade KOH in 500 ml of ethanol.
(10) Sulfuric acid-water, 9:1 (v/v). Carefully add 270 ml of AR grade sulfuric acid to 30 ml of distilled water in a 500 ml beaker.
(11) PCB standards. Aroclor 1242, 1248, 1254, and 1260. (See Fig. B2 and Fig. B4 through B7.)

B4.4 Apparatus.

(1) Separatory funnels equipped with ground-glass stoppers and TFE-fluorocarbon stopcocks, capacities of 125, 250, 500, 1000, and 2000 ml.
(2) Kuderna-Danish evaporative concentrators, 500-ml capacity, equipped with three-ball Snyder columns and graduated 5 ml capacity vials.
(3) Chromatographic columns, glass, 10 inches × 20 mm (OD) with 1.5 inch × 30 mm (OD) reservoir at the top, equipped with TFE fluorocarbon stopcocks.
(4) Sintered glass filter funnels, 600 ml capacity, 90-mm disk diameter, medium porosity.
(5) Flat bottomed boiling flasks, 125 ml capacity.
(6) Liebig condenser, 200 mm in length.
(7) Hot plates.
(8) Water bath.
(9) Reciprocating variable-speed shaker.
(10) 10 µ syringes.
(11) 32 oz. glass mortars and pestles.
(12) 8 inch × 12 inch × 2 inch (2 1/2 qt) heat-resistant glass baking dishes.



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(13) U.S. Standard sieve, No. 30
(14) Usual laboratory glassware.

B4.5 Sampling It is to be assumed that a rather wide variety of sampling techniques may be employed in collecting samples submitted for analysis. For this reason, water and sediment samples should be treated as given in B4.5.1 and B4.5.2.

B4.5.1 Water Where possible, the entire water sample, including the container in which it was collected, should be extracted with hexane. With larger samples, where this is not physically possible, the containers should be simply agitated and a 250-ml portion used for the analysis. (See B4.6.)

B4.5.2 Sediment Any excess water should be decanted, and the entire sediment transferred to a glass baking dish to air dry at room temperature. The dried material should be transferred from the dish into a mortar and ground. The ground sediment should be sieved, remixed, and a 250 gram portion taken for analysis. (See B4.7.)

B4.6 Extraction of Water Samples

(1) After agitating, transfer the entire aqueous sample or a 250-ml aliquot into a graduated glass cylinder. Record the volume of the sample and quantitatively transfer it to a separatory funnel with distilled water.

(2) Rinse the graduated cylinder with two 50-ml portions of hexane and add each to the separatory funnel.

(3) Stopper the separatory funnel and shake vigorously for at least 1 minute. Allow the layers to separate, and transfer the lower aqueous phase to a second separatory funnel.

(4) Extract the water sample a second time with a 50 ml portion of hexane. After the layers have separated, add the first hexane extract to the second separatory funnel, and transfer the aqueous layer to the original separatory funnel.

(5) Repeat the extraction with a third 50-ml portion of hexane. Discard the aqueous layer and combine the hexane extracts.

(6) Filter the combined extracts through a 4-inch funnel, plugged with glass wool that is covered with sodium sulfate. Collect the filtrate in a Kuderna-Danish evaporative concentrator, add a small boiling chip, put the Snyder column in place, and reduce the hexane volume to less than 5 ml by heating the apparatus in an 80°C to 90°C water bath. (CAUTION: Solvent vapors must be vented into a hood.)

(7) After cooling, remove the 5 ml graduated tube and transfer the hexane extract to an alumina adsorption column, washing it out with several 5 ml portions of hexane.

(8) Carefully add 100 ml of hexane to the column reservoir, and collect the total eluent in either a 250 ml volumetric flask or a Kuderna-Danish evaporative concentrator.

(9) If the column eluent is collected in a volumetric flask, dilute to volume with hexane, and proceed with the gas chromatographic analysis.

(10) If the column eluent is collected in a Kuderna-Danish evaporative concentrator, reduce solvent volume, cool, dilute to volume, and proceed with the gas chromatographic analysis.

B4.7 Extraction of Sediment and Soil Samples

(1) Decant off any excess water and transfer the entire sediment sample to a glass baking dish. Air dry at ambient temperature (heat should be applied).

(2) When dry, transfer the soil/sediment to a mortar and grind. Sieve the ground material through a No. 30 mesh sieve and weigh 250 grams (to the nearest 0.01 gram) into a 16-ounce narrow-neck screw-cap (aluminum-foil liner) glass bottle.

(3) Moisten the soil with about 10 ml of water and add 150 ml of acetonitrile. Cap the bottle tightly, and mechanically shake it for a minimum period of 1 hour.

(4) Quantitatively transfer the acetonitrile extract into a 600-ml sintered glass filter funnel containing a 1/4-inch layer of anhydrous sodium sulfate. Collect the filtrate in a 600 ml beaker (vacuum filtration may be necessary).

(5) After the acetonitrile has completely drained into the beaker, wash the bottle twice with 50 ml portions of acetonitrile, adding each wash to the funnel after the previous wash has completely percolated through the sediment.

(6) Quantitatively transfer the extract to a Kuderna-Danish evaporative concentrator, add a small boiling chip, put the Snyder column in place, and reduce the solvent volume to less than 5 ml by heating the apparatus in an 80°C to 90°C water bath. (CAUTION: Solvent vapors must be vented into a hood.)

(7) After cooling, remove the 5 ml graduated tube and transfer the concentrate of extracts to a 125 ml extraction flask with the aid of several small portions of solvent.

(8) Evaporate the extract just to dryness with a gentle stream of dry, filtered nitrogen and add 25 ml of 2.5% alcoholic potassium hydroxide.

(9) Add a boiling chip, put a water condenser in place, and allow the solution to reflux for 45 minutes.

(10) After cooling, transfer the solution to a 250 ml separatory funnel with the aid of 25 ml of distilled water.

(11) Rinse the extraction flask with 25 ml of hexane and add it to the separatory funnel.

(12) Stopper the separatory funnel and shake it



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vigorously for at least 1 minute. Allow the layers to separate, and transfer the lower aqueous phase to a second separatory funnel

(13) Extract the saponification solution with a second 25-ml portion of hexane After the layers have separated, add the first hexane extract to the second separatory funnel and transfer the aqueous alcohol layer to the original separatory funnel

(14) Repeat the extraction with a third 25-ml portion of hexane Discard the saponification solution and combine the hexane extracts.

(15) Carefully add 25 ml of the sulfuric acid solution (9 1 concentrated sulfuric acid–water) to the hexane extracts.

(16) Stopper the separatory funnel and shake vigorously for at least 1 minute Allow the layers to separate, and discard the lower aqueous acid layer. Repeat this step until the acid layer is colorless.

(17) Wash the hexane with a 25-ml portion of water. Discard the water wash.

(18) Filter the hexane extract through a 4-inch funnel, plugged with glass wool that is covered with a layer of sodium sulfate, into a Kuderna-Danish evaporative concentrator.

(19) Add a small boiling chip, put the Snyder column in place, and reduce the hexane volume to less than 5 ml by heating the apparatus in an 80°C to 90°C water bath

(20) After cooling, remove he 5-ml graduated tube and transfer the hexane extract to an alumina adsorption column, washing it in with several 5-ml portions of hexane.

(21) Carefully add 100 ml of hexane to the column reservoir, and collect the total eluent in either a 250-ml volumetric flask or a Kuderna Danish evaporative concentrator.

(22) If the column eluent is collected in a volumetric flask, dilute to volume with hexane, and proceed with the gas chromatographic analysis.

(23) If the column eluent is collected in a Kuderna-Danish evaporative concentrator, reduce solvent volume, cool, dilute to volume, and proceed with gas chromatographic analysis.

**B4.8 Electron Capture Gas Chromatographic Procedure**

(1) Instrument Gas chromatograph (for example, Hewlett-Packard Model 5750, or the equivalent)

(2) Detector: High-temperature $^{63}Ni$, electron capture cell

(3) Column 6 mm X 6-foot glass column, 4% XE-60 on 80/100 mesh, Chromosorb W, HP, AW-DMCS

(4) Column temperature 180°C

(5) Injection port temperature 195°C

(6) Detector temperature 300°C

(7) Pulse interval 50 μs

(8) Flow rates Helium carrier, approximately 60 ml/min, argon-methane purge, approximately 120 ml/min

Using EC/GC as the determinative step, inject, in duplicate, 1 to 10 μl of each solution into the chromatograph By comparison with standard solutions injected, in duplicate, under the same operating conditions, determine the amount and type of Aroclor using the individual or total peak height and area methods

The electron capture detector should also be used to guide the isolation procedures. Water and sediment extracts can be checked for the presence of PCBs or interferences, or both, by injection of microlitre portions of the extracts at various points in the extraction and concentration schemes. In this manner it can be determined whether the sample needs to be concentrated or diluted and whether the cleanup procedures should be employed.

B4 9 Extraction. The extraction of PCBs from water, employing hexane as the extractant, has been found to be quantitative and sufficiently simple and rapid for use as a routine procedure.

The evaluation of this method was based on spiking water samples with standard acetone solutions of PCBs The spiking method consisted of adding the PCB in 25 to 50 μl of acetone to 500 ml of tap water in a 32-ounce narrow neck screw-cap jar After the sample was thoroughly mixed, duplicate 225- to 250-ml aliquots were taken and subjected to the proposed sample preparation and worked up as outlined. The results were quantified by preparing a calibration curve using standard hexane solutions of the PCBs used to spike the water samples The major isomer peak height was used to construct the calibration plot.

The average recovery and deviation achieved substantiated the applicability of the method for the quantitative recovery and analysis of PCBs from water at the ppb-ppm level

No PCB recovery experiments from spiked sediment and soil samples have been performed Instead, several of the residual solids representative of some of the types of sediment or soil analyzed were reextracted with hexane/acetone (40 60) in a soxhlet extractor to test for the efficiency of the acetonitrile extraction step. The hexane, after isolation by dilution with distilled water, was then carried through the purification steps. Recoveries by soxhlet extraction have indicated that the acetonitrile extraction of PCBs was essentially quantitative in the cases checked.

B4 10 Sample Concentration. Concentration of sample extracts is necessary, prior to cleanup by chromato-



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graphic or chemical means, to reduce sample size and increase sensitivity. The preferred method of concentrating allows minimum loss through volatilization or chemical decomposition and requires a minimum time. The three methods of solvent volume reduction most commonly used are evaporation by exposure to a stream of air, evaporation employing a Kuderna-Danish evaporative concentrator equipped with a Snyder column, and evaporation under reduced pressure. All three techniques have been used, and no significant losses from volatilization or chemical alternation have been encountered. However, the Kuderna Danish evaporative concentrator and the stream-of-air methods are easier to use.

B4.11 Column Adsorption Chromatography and Chemical Cleanup. Silica gel, a magnesia silica gel,[9] and alumina deactivated with 0%, 1.0%, 1.5%, 2.0%, and 5% water were investigated as adsorbants for the elimination of interferences. Alumina (5% water) was found to be more effective and reproducible than either silica gel or a magnesia silica gel.[9] The activity of alumina varies with age and lot; therefore 5% water is added to the alumina, after heating for a minimum of 4 hours at 400°C, to ensure a reproducible activity.

Saponification, and subsequent extraction of the sample with sulfuric acid, is an effective way to remove a number of chlorinated hydrocarbon interferences as well as other matrix interferences. PCBs are not affected.

B4.12 Column Performance. Column performance is the key to effective gas chromatographic analysis and, as such, the choice of column materials is particularly important. Ideally, the support employed should be inert, mechanically strong, and of high surface area. For these reasons, Chromosorb W, H-P, AW-DMCS is recommended for this work.

A variety of polar and nonpolar liquid phases have been investigated. The following columns have been found to provide adequate separation, etc. for use in PCB analysis by electron capture: 4% (w/w) DC-200, SF-96, OV-17, SE-30, SE-54, XE 60, Apiezon L, and 6% QF-1. DC-200 and XE-60 or QF-1 are the most suitable of these liquid phases.

Another important consideration when working with an extremely sensitive detector and, consequently, low levels of materials is column conditioning. With polar phases such as XE-60 and QF-1, operating a new column overnight at a temperature 25°C to 50°C higher than that to be used during analysis results in a more stable column. A no-flow conditioning technique is employed to condition nonpolar columns. The column is purged with carrier gas, heated for 30 minutes at an elevated temperature without carrier flow, and then cooled to room temperature. At the end of this cycle, the carrier flow is resumed and the conditioning is completed as in the case of the polar liquid phase. Two precautions should be observed during conditioning, the column should not be connected to the detector, and the maximum safe temperature of the liquid phase should not be exceeded.

Since all liquid substrates bleed to one degree or another and columns eventually degrade, all new columns should be characterized with two column performance indicators: the number of theoretical plates ($N$) and a tailing factor ($T$). p,p' DDT is employed to check these parameters because it is known to degrade on "poor" columns. In this manner, one can determine whether the performance of a new column is satisfactory and when the column performance begins to fall off. A column is considered good if the number of theoretical plates per foot is of the order of 400 to 500, with tailing factors of 1.0 to 1.3. Calculation of these parameters is shown in Fig. B9 and B10. Additionally, there should be no significant extraneous peaks upon injection of a pure p,p'-DDT standard.

Other chromatographic conditions that can be adjusted are column temperature and flow rates. Although resolution of a mixture increases with decreasing temperature, a temperature should be chosen that allows the elution of all components within a convenient time period. The temperatures given are optimum for 42% chlorinated biphenyl; temperatures are increased when specifically analyzing for the higher chlorinated biphenyls; that is, 54%, 60%, etc. The flow rates shown are optimum for a given instrument, column, and detector system. These should be adjusted if better results can be achieved.

Any system of instrument and column suitable for chlorinated pesticide analysis is satisfactory for PCB analysis. The use of the high-temperature $^{63}$Ni electron capture cell is highly recommended. The ability to operate at higher temperatures prevents maintenance problems due to contamination from high boiling components. Glass columns also should be employed.

B4.13 Detection and Measurement. Quantitative determinations employing the electron capture detector are nonstoichiometric measurements made by comparing peak heights or areas for known concentrations with those for unknown compositions. Except for sharp peaks, peak area measurements are usually more reproducible than peak height measurements but are extremely time consuming, unless a recording integrator is employed. However, peak height measurements are as accurate as disk integration or triangulation and, if the

[9] For example, Florisil

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peak shape represents a Gaussian curve, the height may be considered independent of the base. Consequently, peak height measurements are generally used Three variations of the peak height or area quantification procedures have been employed.

Case I EC gas chromatogram of PCB unknown, unchanged with respect to standard PCB with no evidence of interferences

Case II EC gas chromatogram of PCB unknown, altered with respect to standard PCB with no evidence of interferences

Case III. EC gas chromatogram of PCB unknown, unchanged with respect to standard PCB with evidence of interference

The amount of PCBs in Case I samples is determined by preparing a plot of the major peak height or area versus concentration For Case II, a plot is prepared of the total sum of all major peaks versus concentration With Case III samples, a peak free from interference is used. When dominant interferences are present, one or more of the chemical cleanup procedures is employed

In all cases, the response of the electron capture detector must be linear for quantitative analysis

B4.14 Contamination In determining PCBs in water, soil, and sediment by electron capture gas chromatography, laboratory sources of contamination can be a major problem. The samples and extracts should never be allowed to come in contact with materials other than glass, TFE-fluorocarbon, or metal Laboratory glassware should be thoroughly washed with hot, soapy water and rinsed with distilled water, acetone, and then hexane. All equipment should also be rinsed again with hexane just prior to use, and blanks should be frequently carried through all steps of the procedures to ensure against the possibility of contamination.

B4.15 Sensitivity

(1) Concentration. 2 ppb
(2) Absolute sensitivity. $0.5 \times 10^{-9}$ gram
(3) Volume injected: 5 µl
(4) Final volume of extract: 5 ml
(5) Sample size: 250 ml



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The standard in this booklet is one of nearly 5,600 standards approved to date by the American National Standards Institute, formerly the USA Standards Institute.

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MDNS 219110

# Polychlorinated Biphenyls

*A Report Prepared by the*

Committee on the Assessment of
Polychlorinated Biphenyls in the Environment

Environmental Studies Board
Commission on Natural Resources
National Research Council

NATIONAL ACADEMY OF SCIENCES
Washington D.C. 1979

Attachment 1-6

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FICF: The project that is the subject of this report was approved by the Governing Board of the National Research Council, whose members are drawn from the Councils of the National Academy of Sciences, the National Academy of Engineering and the Institute of Medicine. The members of the Committee responsible for the report were chosen for their special competences and with regard for appropriate balance.

This report has been reviewed by a group other than the authors according to procedures approved by a Report Review Committee consisting of members of the National Academy of Sciences, the National Academy of Engineering, and the Institute of Medicine.

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# PREFACE

Recognizing the hazards posed to human health and the environment by polychlorinated biphenyls (PCBs), the Office of Research and Development of the U.S. Environmental Protection Agency (EPA) requested that an assessment of the problem be conducted by the National Research Council. The Environmental Studies Board, given responsibility for the study, appointed the Committee on the Assessment of Polychlorinated Biphenyls in the Environment early in 1970. The charge to the Committee was to provide EPA with a detailed assessment of the nature of the problem and of alternatives for reducing the current substantial level of environmental contamination from PCBs.

Instead of contributing yet another extensive review of the fate of PCBs and the risks to health associated with them, the Committee set itself three objectives: (1) to develop a model of PCB distribution throughout the environment with estimates of current reservoir burdens, (2) to analyze the economic impact of various control options, and (3) using PCB data as the test case, to assess the effectiveness of the proposed EPA testing guidelines for evaluating toxic substances.

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The Committee benefited greatly from discussions with individuals from academia, industry, and government agencies. We are particularly grateful to Frank Whitmore of Versar, Inc., for developing the model of PCB distribution. We also appreciate the assistance of several other persons at various stages of the preparation of the report: David Downer, Arthur Stern, Norbet Page (Office of Toxic Substances), Phill Taylor, Sam Conger (EPA Water Quality Analysis Branch-STORET), Wade Fowler (EPA Scientific Advisory Board), Steven Malkenson (EPA Office of Planning and Management), Nathan Karch (Council on Environmental Quality), Herman Feltz (U.S. Geological Survey), and Fred Kutz (EPA Office of Pesticides Program).

On behalf of the Committee, I would like to thank Suellen Pirages and Elizabeth Panos of the National Research Council for their contributions in preparing this report. Other staff members providing assistance included Judith Cummings, Nancy Fairfax, Raphael Kasper, Christina Olson, Connie Reges, William Robertson, and Robert Rooney.

Finally, I wish to express my gratitude to each member of the Committee for his efforts towards the successful completion of the study.

Alfred M. Beeton
Chairman, Committee on
the Assessment of Polychlorinated
Biphenyls in the Environment

# INTRODUCTION

Thirty-six years after the introduction of polychlorinated biphenyls (PCBs) into the world market, scientists discovered that the compound was accumulating in tissues of fish taken from the Baltic Sea. Monitoring efforts within several countries subsequently disclosed extensive global environmental contamination from this group of organics, which comprises theoretically more than 200 individual isomers of chlorinated biphenyls. The problem of environmental contamination by PCBs has now been well documented. Comprehensive reviews have reported the existence of significant quantities of PCBs in atmosphere, soil, water, sediment, fish, wildlife, and even in samples of human blood and tissue (NIEHS 1972, Hutzinger et al. 1974, Kimbrough 1974, Durfee et al. 1976, Kornreich et al. 1976, Nisbet 1976, U.S. EPA 1976, WHO 1976, Roberts et al. 1978).

The term polychlorinated biphenyls refers to a class of organics produced by chlorination of a biphenyl and composed of 10 possible forms. These forms arise from a specified number of chlorine substitutions on the biphenyl molecule, and correspond to the chemical structures monochlorobiphenyl, dichlorobiphenyl,

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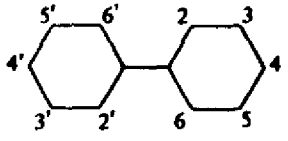
trichlorobiphenyl, etc. Several isomers for each PCB molecule are possible, the number depending on available substitution sites on each biphenyl portion (2 6, 2'-6'). However, not all possible isomers are likely to be formed during the manufacturing processes. In general, the most common ones are those that have either an equal number of chlorine atoms on both rings or a difference of only one chlorine atom between rings. More complete information on chemical identity and the manufacturing process is presented in Appendix D.




Commercial PCB mixtures are manufactured under a variety of trade names (Appendix D). The chlorine content of any product may vary from 18 to 79 percent and depends on the extent of chlorination during the manufacturing process or on the amount of isomeric mixing engaged in by individual producers. Each company has a specific system for identifying the chlorine content of its product. For example, Aroclor®1248, 1254 and 1260 indicate 48 percent, 54 percent and 60 percent chlorine, respectively; Clophen A60, Phenochlor DP6 and Kaneclor 600 designate that these products contain mixtures of hexachlorobiphenyls.

The only important U.S. producer of PCBs was Monsanto Industrial Chemicals Co., which had plants at Anniston, Alabama, where production of PCBs ended in 1970, and Sauget, Illinois, where production ceased in 1977. Sold under Monsanto's registered trademark of Aroclors, mixtures of PCBs had been used originally as a coolant/dielectric for transformers and capacitors, as heat transfer fluids, and as protective coatings for woods when low flammability was essential or desirable. Producers and users alike, apparently unaware of any potential hazards from exposure to PCBs, initially operated in accordance with earlier results of toxicity tests that indicated no effects (Penning 1930). The expansion of open-ended applications between 1930 and 1960, incorporating PCBs into such commodities as paints, inks, dedusting agents, and pesticides, led to the widespread dissemination of which we are now aware. By 1937, toxic effects were noted in occupationally



exposed workers and threshold limit values were imposed at manufacturing sites.

The general pattern of release of PCBs to the environment changed significantly during the early 1970s. Until then, essentially no restrictions were imposed either on the use or on the disposal of PCBs. After evidence became available in 1969 and 1970 that chronic exposure could result in hazards to human health and the environment, Monsanto voluntarily banned sales of PCBs for "open-ended" and "nominally closed" applications (see Chapter 1, Tables 1.1 and 1.2), and the loss rate from industrial use was reduced through stringent control measures. However, significant reservoirs of mobile PCBs (those available for transport among environmental media and biota) still exist along with even larger, currently immobile reservoirs. The latter include those materials containing PCBs that are still in service and those deposited in landfills and dumps. The major factor affecting future release of PCBs from these sources will be regulatory action governing storage and disposal of the chemical.

The new Toxic Substances Control Act (TSCA), Public Law 94-469, specifically prohibits production of PCBs within the United States, regulates disposal of materials contaminated by PCBs, and restricts the use of any such materials already in service. The effect of these measures should be to eliminate further releases into the environment and, eventually, to reduce quantities existing in the environment. However, because of the extreme stability of PCBs, environmental levels will not be reduced substantially for many years, and the problem of dealing with existing sources of mobile PCBs will remain.

Cleanup efforts have been initiated in those environments that are heavily contaminated, but complete removal may not be either technically or economically feasible. For this reason, the U.S. Environmental Protection Agency (EPA) commissioned the National Research Council to review the current dimensions of the PCB problem and to recommend future action.

The intent of this report is to provide EPA with an assessment of (1) the current state of the PCB problem, (2) the cost-effectiveness of selected options for controlling continued environmental pollution, and (3) the appropriateness of proposed guidelines for [illegible] the potential hazard and persistence of [illegible] the type of product [illegible] "PCB-like"

problems. To accomplish its purpose this study has three primary components.

- Pertinent information on environmental aspects of PCBs was reviewed and summarized with emphasis on transport processes and distribution among environmental compartments. A model of the distribution of PCBs in the environment was constructed.
- An economic analysis was performed to determine, to the extent possible, the economic costs and environmental and health benefits that might be derived from implementing various options for controlling further PCB release and for reducing the present level of contamination. It should be noted that this analysis used the 1978 EPA published regulations for use and disposal of PCBs (U.S. EPA 1978a,b). New regulations are to be published in 1979 that may reflect different disposal requirements.
- Using EPA draft guidelines for testing potential toxicity of new chemicals, an assessment of PCB toxicity was made. An approach that might be used for identifying potential hazards prior to extensive manufacture and use of new chemicals was developed.

The study has a generic aspect in that the model of PCB distribution, the approach to quantifying persistence and toxicity, and the analysis of control options are applicable to studies of other persistent environmental pollutants.

REFERENCES

Durfee, R.L., G. Contos, F.C. Whitmore, J.D. Barden, E.E. Hackman III, and R.A. Westin (1976) PCBs in the United States--Industrial Use and Environmental Distribution. Report No. EPA 56016-16-005. Springfield, Va.: Versar, Inc.

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

Kimbrough, R.D. (1974) The toxicity of polychlorinated, polycyclic compounds and related chemicals. Pages 445-498, CRC Critical Reviews in Toxicology. Cleveland, Ohio: CRC Press.

Kornreich, M., B. Fuller, J. Dorigan, P. Walker, and L. Thomas (1976) Environmental Impact of Polychlorinated Biphenyls. McLean, Va.: The Mitre Corporation.



National Institute of Environmental Health Sciences (1972) Conference on PCBs. Environ. Health Perspect. 1:1-85.

Nisbet, I.C.T. (1976) Criteria Document for PCBs. Report No. PB-225 397. Washington, D.C.: U.S. Department of Commerce.

Penning, C.H. (1930) Physical characteristics and commercial possibility of chlorinated diphenyl. Ind. Eng. Chem. 22:1180-1183.

Roberts, J.R., D.W. Rodgers, J.R. Bailey, and M.A. Rorke (1978) Polychlorinated Biphenyls: Biological Criteria for an Assessment of Their Effects on Environmental Quality. Associate Committee on Scientific Criteria for Environmental Quality. Ottawa, Canada: National Research Council of Canada.

U.S. Environmental Protection Agency (1976) Proceedings of National Conference on Polychlorinated Biphenyls, November 19-21, 1975, Chicago, Illinois. Report No. EPA-560/6-75-004. Washington, D.C.: U.S. Environmental Protection Agency.

U.S. Environmental Protection Agency (1978a) Polychlorinated biphenyls (PCBs): Disposal and marking. Federal Register 43(34):7149-7164.

U.S. Environmental Protection Agency (1978b) Polychlorinated biphenyls (PCBs): Manufacturing, processing, distribution in commerce, and use bans. Federal Register 43(110):24802-24818.

World Health Organization (1976) Polychlorinated Biphenyls and Terphenyls. Environmental Health Criteria 2. Geneva: World Health Organization.

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## FINDINGS AND RECOMMENDATIONS

### Chapter 1. PCB Transport Throughout the Environment

1. A model of the distribution of PCBs throughout the U.S. environment estimates the total cumulative PCB use from domestic sources through 1975 aa 6.1 x $10^8$ kg, with 38 percent representing mono-, di-, and trichlorobiphenyl; 23 percent, tetrachlorobiphenyl; and 39 percent, penta- through decachlorobiphenyl.

2. From production and use data, loaaes of PCBa from production-uae functions and the amounts entering the mobile environmental reservoir total 8.2 x $10^7$ kg, distributed as follows:

| | Amounta (x $10^7$ kg) |
|---|---|
| mono-, di-, trichlorobiphenyl: | 3.1 |
| tetrachlorobiphenyl: | 1.8 |
| penta- through decachlorobiphenyl: | 3.3 |

3. The North Atlantic Ocean appeara to be the dominant sink for PCBs, accounting for 50 to 80 percent of PCBa in the environment, and freshwater sediment is a major continental reservoir.



4. Environmental load estimates for the United States and the Atlantic Ocean are presented for the following compartments:

| | Amounts (x $10^6$ kg) |
|---|---|
| Atmosphere | 0.018 |
| Hydrosphere | |
| Fresh water | 0.012 - 0.035 |
| Freshwater sediment | 1.400 - 7.100 |
| Freshwater biota | 0.030 |
| Marine sediment | 0.660 - 2.700 |
| Marine water | 6.000 - 66.00 |
| Marine biota | 0.030 |
| Lithosphere | 0.140 - 2.800 |

5. Biodegradation has been a minor factor in PCB destruction for those isomers of tetra- through decachlorobiphenyls.

### Recommendations

1. Because the North Atlantic Ocean appears to be the ultimate sink, further studies are needed to firmly establish PCB concentrations and distribution in the marine environment.

2. Although modeling waa useful in assessing large-acale distribution and fate of toxic substances, more adequate data bases are needed to develop transport modela of environmental pollutanta. For example, further research is needed to determine the processes of tranafer between the environmental compartments.

3. EPA should continue to develop better analytical techniques, particularly those enabling assessment of the extent of PCB fractionation. Because of continuing analytical problems it is difficult to determine the extent of ecosystem contamination. For aquatic systems, the most effective indicator of both the degree of PCB pollution and the potential hazards to humans and wildlife is the PCB content of fish. To use this indicator effectively, factors such as age, species, and nutritional habits of the fish must be taken into account. Detailed information on individual isomers should also be reported together with the specific chromatography.

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4. Freshwater sediment is an important continental sink for PCBs, and special attention should be given to monitoring programs and to studies of PCB cycling from sediment through human food chains.

## Chapter 2. Policy Assessment

1. Substantial differences exist in the costs of various options for the control of PCBs. Among the alternatives analyzed--which included EPA proposed regulations on disposal and use, options for dredging and disposal of contaminated river sediment, and U.S. Food and Drug Administration (FDA) regulations limiting permissible residues in fish destined for human consumption--average costs varied by as much as five orders of magnitude.

2. Existing EPA regulations on disposal and use are projected to result in the control of 260 x $10^6$ kg of PCBs with average costs of up to $33/kg of PCBs controlled. Proposed EPA regulations would increase the total quantity of PCBs controlled by about 0.1 percent (beyond those controlled by existing regulations) with average costs becoming as high as $133,000/kg.

3. A case study of the Hudson River indicated that alternatives for dredging sediment could remove a total of about 178 x $10^3$ kg of PCBs from potentially mobile river sediment at costs ranging from $110/kg to $3,150/kg.

4. It is estimated that up to 0.25 percent of the total PCBs released to the environment enter the human diet; at least 400 kg of PCBs would have to be removed from the mobile environmental reservoir in order to achieve a reduction of 1 kg of PCBs in food. This figure (400 kg) is derived using data from the Great Lakes, a closed system, and may be too low for more open systems. FDA limitations on PCB concentrations in commercially distributed fish could reduce PCB input to the American diet by about 64 kg at a cost of less than $125,000/kg. (A reduction of an additional 34 kg would cost less than $293,000/kg.)

5. Based on the figures used in the analysis in this report, FDA regulation of PCB limits in fish destined for human consumption appears a more cost-effective method of controlling dietary intake and human health effects than some of the most costly EPA proposed



[illegible] on use and disposal. Some dredging options may be preferable to either EPA or FDA regulations.

6. It is difficult to assess the benefits of controlling exposure to PCBs with currently available information. The analysis in this report conservatively estimates the benefits as $30/kg to $300/kg of PCBs removed from the environment or withheld from discharge, but notes that even these figures could be high or low by a factor of 100 or more.

### Recommendations

1. Policymakers should use available data to calculate the incremental or average costs of PCB control options in conducting economic analyses of proposed regulations.

2. As greater control of PCBs is sought, regulatory efforts should be concentrated on those options which are most cost effective, that is those with the lowest average or incremental costs per comparable effect or benefit.

3. Economic analyses should include cost data on all options--not just those under a single agency's regulatory authority. These analyses could then show--as does the analysis presented in this report--that some options beyond a particular agency's authority are substantially more cost effective than some that lie within the agency's jurisdiction. A comprehensive, coordinated government policy toward control of pollutants should reflect such findings.

## Chapter 3. Guidelines for Evaluating the Potential Toxicity of Chemicals

1. Based on the original uses of PCBs, this chemical might have passed toxic substance evaluation as proposed under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), thus allowing its use on a large scale. Acute toxicity is minimal for all species except some aquatic invertebrates, and intended use patterns may not have warranted additional chronic tests.

2. Results from the proposed TSCA Reference Set tests would require continued testing with the Hazard

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Evaluation Studies leading to detection of some PCB toxic effects.

3. Most health hazards resulting from PCB exposure other than in occupational settings are subtle (e.g., reduced reproductive capabilities, changes in behavioral patterns) and would not be detected with either the FIFRA or TSCA scheme.

4. Common laboratory animals do not always respond to PCB exposure in a manner similar to humans. No one test species exhibits "typical" human symptoms of PCB poisoning. Thus, choice of test species becomes an important factor in evaluating the potential toxicity of a substance.

## Recommendations

1. Whenever chemical, physical, and environmental chemistry data indicate that a substance is both persistent and lipophilic, its development for large-scale use should be reassessed and alternatives considered.

2. Testing schemes should include models of anticipated release and movement through the environment identifying potential reservoirs, and tests should be performed within these specified environmental ecosystems.

3. Test guidelines should stress assessment of subtle health hazards, such as reproductive alterations and behavioral-learning impairment.



# PCB TRANSPORT THROUGHOUT THE ENVIRONMENT

In considering PCB sources, mechanisms of transport through the environment, and mobile or immobile reservoirs, it is often useful to consider global cycles (Nisbet and Sarofim 1972). Models of these cycles can provide insights into the general behavior of compounds and can establish a framework for subsequent research. This chapter presents a review of PCB distribution in the major environmental compartments for the United States (i.e., atmosphere, hydrosphere, lithosphere) and the means of transport among and within them.

In any attempt to design a distribution model based on results from a number of different studies, several possible sources of error must be recognized. Lack of uniformity in sampling and analytical techniques is the major problem encountered. Sampling difficulties occur for measurements in air, water, and sediment. In air, the amount of PCBs in the vapor phase relative to that attached to particles still is uncertain, mainly because of problems in sampling techniques. Similar difficulties are encountered in efforts to distinguish between dissolved vs. particle-bound PCBs in water. Measurement of PCBs in sediment also presents problems. Many studies have not used existing, proven techniques

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for collecting sediment cores, but instead have analyzed "grab" samples. Grab sample analyses can result in overestimates or underestimates of PCB levels depending on the depth of sample taken, and may not reflect accurately the concentration of the sediment column.

The comparisons of measurements of PCBs in environmental samples have also been complicated by a lack of uniformity in analytical techniques. Because each commercial mixture contains a number of PCB isomers and thus exhibits a wide range of vapor pressures and solubility characteristics, the most common approach to quantifying PCB concentrations is to compare gas chromatographic records of the sample with records of Aroclor, Clophen, Phenoclor or Kanechlor standards. But this approach is not entirely appropriate, unless both the number of peaks recorded and the interpeak height ratios are identical for the sample and standard being compared (Webb and McCall 1973, Murphy and Rzeszutko 1977). Analytical fractionation techniques make it difficult to identify adequately all peaks on the gas chromatographic record, and it is quite possible that underestimates of the quantities of mono- and dichlorobiphenyls have resulted. Because of these problems in identifying the chemical components of a given mixture, PCB concentrations are often reported as "most closely resembling" or calculated as "similar to" a certain known Aroclor standard. Thus, it becomes difficult to discern whether or to what extent individual component fractionations actually occur within the environment.

The ability to understand and assess PCB movement throughout the environment can only remain speculative until the sampling and analytical problems are resolved; the following interpretations of data were made with these limitations in mind.

## SOURCES OF CONTAMINATION

### Estimates of General PCB Distribution Within the United States

During the period from 1930 to 1975, U.S. commercial sales of PCBs totalled nearly 570 x $10^6$ kg from domestic sources and about 1.4 x $10^6$ kg from imports (Durfee et al. 1976). As of 1975, more than half of this amount, or 340 x $10^6$ kg, was estimated to be in service.



[illegible]tribution of the remainder was estimated using 1957 [illegible] 1974 production and sales figures supplied by Monsanto, information supplied by other industrial concerns, and analysis of PCB use patterns with respect to losses, waste disposal, and other aspects of environmental entry. The amount and location of PCBs are presented in Table 1.1.

### Routes of Entry into the Environment

There is no substantial evidence suggesting that PCBs are produced in the environment, either from natural sources or from chemical transformation of other compounds. Therefore, all environmental contamination by PCBs is inferred to have resulted from the production and use of materials and equipment containing PCBs. It also can be inferred, with some certainty, that PCBs entering the United States through commercial imports or via transnational atmospheric transport have been inconsequential when compared to the amounts entering the U.S. environment via domestic routes.

Major domestic uses of PCBs are indicated in Table 1.2. The voluntary limitation of PCB production imposed by Monsanto in 1971 restricted sales to applications in closed electrical systems. The 1973 estimated imports of PCBs to the United States totalled only 1.0 percent of Monsanto's production for that year, suggesting further voluntary elimination of uses other than for closed systems. The major applications of imported PCBS at that time were for pattern waxes (decachlorobiphenyl) and replacement coolant fluids for mining machines. Further evidence of the decrease in PCB use lies in the

TABLE 1.1 Environmental Distribution of PCBs Not in Service as of 1975

| | Amount (x $10^6$ kg) |
|---|---|
| Mobile PCBs in the Environment | 68.2 |
| Degraded or Incinerated | 25 |
| In Landfills or Equipment Dumps | 130 |

[a] Modified from Durfee et al. (1976)

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TABLE 1.2 Domestic Uses of PCBs

| Category | Type of Product | Percent of New Total Use |
|---|---|---|
| Closed Electrical Systems | Transformer, capacitors, other (minor) electrical insulating/cooling applications | 61 until 1971; 100 after 1971 |
| Nominally Closed Systems | Hydraulic fluids, heat transfer fluids, lubricants | 13 until 1971; 0 after 1971 |
| Open-End Applications | Plasticizers, surface coatings, ink and dye carriers, adhesives, pesticide extenders, carbonless copy paper, dyes | 26 until 1971; 0 after 1971 |

fact that although Monsanto ceased all PCB production in 1977, large-scale increases in imports of these compounds did not occur.

Two important routes of entry into the environment have been losses during the process of manufacture and leakage from electrical equipment and other products containing PCBs. Losses from open-end and nominally closed systems in service undoubtedly have been large and have contributed to contamination of soil, water, and atmosphere.

Mobilization of PCBs located in landfills and freshwater sediment offers significant additional potential for contamination, but few data are available for estimating current or future rates of dispersion. In studies of New York landfills some volatilization of PCBs is noted, but movement of PCBs from these sources into groundwater reservoirs is negligible (Lels and Metry 1978, Hetling et al. 1978). Disposal of materials containing PCBs--as for example, through municipal incineration--appears to be one of many minor routes of entry, and application of PCB-contaminated oils as dedusting agents may mobilize relatively large quantities of PCBs formerly in service.

A summary estimate of the quantity of PCBs in landfills as of 1975 is shown in Table 1.3. The estimated rate of PCB entry into on-land disposal sites from all sources in 1975 was 5.5 x $10^4$ kg/yr (Durfee et al. 1976). Assuming that this estimate is valid for the



[illegible] 1973 through 1978, the total accumulation in landfills by 1978 can be estimated as 140 x $10^4$ kg. However, this total may be an overestimate as disposal on land may have declined with the passage of TSCA and the attention given this aspect of the PCB problem. Some obsolete equipment has been stored instead of dumped in landfills, but the major reduction of PCBs deposited in land sites has resulted through a decrease in production wastes due to diminished domestic manufacture and use of PCBs.

In the future, the primary source of PCB entry to on-land disposal sites is likely to be the obsolescence of electrical equipment, because no regulations are anticipated that could change thia situation. Thus, it is not unreasonsble to suggest a maximum entry rate of about 5.5 x $10^4$ kg/yr to landfills for at least 10 years and perhaps as many as 20, although by the year 2000 the rate should have decreased somewhat.

PCBs newly deposited in landfills should be much less mobile than those in old landfill sites, because of the Resource Conservation and Recovery Act (RCRA) requirements for better designed landfills and tighter controls on monitoring site deterioration and chemical leakage. Much of the new deposits will be contained sufficiently to prevent leakage or volatilization and new sites must be designed to reduce mobilization of toxic materials by leaching.

## Sources of PCBs

**Industrial and Municipal Effluents** Recent estimates of industrial and municipal PCB discharges into aquatic systems have been determined through monitoring programs

TABLE 1.3 Reservoir of PCBs in Landfills as of 1975

| Source | Estimated Amount (x $10^6$ kg) |
|---|---|
| Production Wastes | 50 |
| Obsolete Equipment | 36 |
| Other | 36 |
| Total | 122 |

[illegible]PCF Modified from Durfee et al. (1976).

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initiated by EPA. The results indicate that the average PCB content of industrial effluents (five categories) is 0.68 x $10^3$ kg/yr, and the PCB content of discharges from publicly owned wastewater treatment works (POTW) is 110 x $10^3$ kg/yr (Versar, Inc. 1978).

The sampling efforts producing the industrial discharge data, on which these estimates were based, have been far from comprehensive, but they may provide results representative of the five industrial segments covered (i.e., timber processing, nonferrous metals, ore mining, pulp and paper manufacturing, and machinery and mechanical products). The estimate of PCB discharge from POTW is high by perhaps a factor of three and reflects a nonrandom sampling of 66 treatment plants; some of these were in the vicinity of known PCB operations (Versar, Inc. 1978).

Even if the estimated POTW discharge rate is high, the municipal sector apparently releases a significant amount of PCBs into the aquatic environment. The major source is believed to be the mobile environmental reservoir, because the municipal wastes are presumed to have passed from a use cycle to the environment before entering a wastewater stream. Leakage from equipment in use or being serviced accounts for the remainder of PCBs in POTW discharges.

If it is assumed that approximately 40 x $10^3$ kg/yr of total PCBs are discharged from municipal sewage-treatment plants, the release rate of PCBs in sewage sludge that is released to the environment should total about 240 x $10^3$ kg/yr based on an approximate 6:1 ratio for partitioning of PCBs between sludge and water effluent (Sargent et al. 1977). Approximately 85 to 90 percent of PCBs in the portion of sludge that is incinerated (on the order of 15 percent of total sludge) may be destroyed via combustion (Whitmore 1975); the remaining PCBs in sludge represent a potential threat to the environment as well as a substantial but perhaps temporary sink for PCBs that were previously environmentally mobile.

Paper Recycling The use of PCBs in carbonless copy paper from 1957 through 1971 has resulted in the contamination of paper products and foods by recycling practices and entry of contaminated effluents into the environment via paper-recycling facilities. Projections based on available data have indicated that both average concentrations and occurrence of "hot spots" are now



[illegible], rapidly and will continue to decrease (Carr [illegible], 1977). Some fraction of the PCBs in carbonless [illegible] paper and recycled paper is also released to the [illegible]sphere through incineration under conditions that do not destroy PCBs completely.

Release of PCBs to Soil In light of the diminishing use of PCBs in industry, the major routes of future entry into soil apparently will be from disposal sites, in-service equipment, and atmospheric transport. Disposal sites represent a sizable reservoir of PCBs, but little is known about the lifetime of these compounds in a site (for example, a landfill) or their effective rate of release. A recent study of the Hudson River area indicates that negligible groundwater contamination by PCBs has occurred in areas surrounding old landfills (Leis and Metry 1978, Hetling et al. 1978), but extrapolation from the local to the national scale is not possible.

Accidental and deliberate releases pose a potential threat of excessive contamination, but only locally. Diminished industrial use should result in a reduction in the number of spills. It is also hoped that application of PCBs or PCB-contaminated oils for dust control will be reduced through public awareness and regulatory actions; but if it is not, such applications will continue to have local impact on concentrations of PCBs in soil.

Incineration The PCB regulations proposed by EPA suggest that ultimately incineration will be the required method of disposal, but at this time it is not clear whether the necessary facilities will be available, or what conditions of burning time and temperature are needed to incinerate large quantities of contaminated wastes (U.S. EPA 1978). Research is in progress on the incineration conditions required to destroy PCBs, but the investigation is not yet complete. One problem needing immediate resolution concerns the effect of applying "combustion efficiency" criteria based strictly on combustion conditions ($CO/CO_2$ in exhaust) rather than on destruction of PCBs by the incineration process.

[illegible]nical Landfills The situation is also unclear [illegible]ding availability and adequacy of chemical waste [illegible]fills for disposing of PCBs. At present, four such

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sites exist in the United States and more are planned, but little information is available about the potential for loss of PCBs from landfills through long periods of residence. Another problem, affecting both incineration and disposal in landfills, is how to assign responsibility for spills or other accidental losses that may occur during transport of contaminated waste across state lines.

## ATMOSPHERIC COMPARTMENT

Recently available data indicate that atmospheric transport is an important factor in PCB distribution on both local and global scales (Harvey and Steinhauer 1976a, Risebrough et al. 1976, Fuller et al. 1977, and Bidleman et al. 1978). Quantities of the compound detected in a variety of samples collected at remote areas (e.g., in water, snow, and soil) provide strong evidence for this conclusion, even though the details of atmospheric transport are not well understood. Current data do not permit accurate characterization of such factors as vapor/particulate partitioning, individual component fractionation, spatial and temporal variations in deposition, volatilization or degradation rates, or identification of ultimate sinks. Accurate measurement has been hampered, as stated earlier, mainly by technical problems in sampling and analytical procedures. But despite the problems, it has been possible to reach several important conclusions concerning atmospheric transport of PCBs.

### Physical and Chemical Forms of Atmospheric PCBs

Current information about PCB vapor pressures suggests that PCBs should exist both in the vapor form and attached to particles in the atmosphere. Determination of PCB levels in air has indicated that more than 99 percent of the total atmospheric burden of PCBs can pass through a glass-fiber filter and thus must be collected on various adsorbent traps (e.g., silicone oil coated ceramic chips, polyurethane foam, or XAD-2 resin [Bidleman et al. 1978]). However, it cannot be determined whether PCBs are associated with particles too small to be captured by the glass-fiber filter or whether 99 percent of the total burden of PCBs actually



[illegible] as a vapor. It is thought that this collection [illegible] may be further hampered by vaporization of PCBs [illegible] ciated with particulates during the collection process. However, Harvey and Steinhauer (1974) state:

> ...the observation that only 1 percent or less of the total chlorinated hydrocarbon concentration in the sampled air is on the filter argues strongly against significant particle transport.

Murphy and Rzeszutko (1977) tried to resolve this phase-partitioning question by measuring individual Aroclors in both air and precipitation (Tables 1.4 and 1.5). Two assumptions were made. The first was that all Aroclor 1260 detected in air is associated with particulates. It is likely that this is a valid assumption because the vapor pressure range for Aroclor 1260 compounds is on the order of $10^{-4}$ mm Hg to $10^{-5}$ mm Hg. The second assumption, based upon the magnitude of an effective Henry's Law Constant for Aroclor 1260, contended that all PCBs in rain result from particulate scavenging. This assumption might be in error because of the uncertainties involved in calculating and measuring Henry's Law Constant. The value of the partition coefficient calculated from Henry's Law Constant indicates that the concentrations of scavenged vapor in precipitation should be negligible (Mackay and Wolkoff 1973). Using their measured Aroclor partitioning and the above assumptions, Murphy and

**TABLE 1.4 PCBs in Precipitation**

| Location (Number of Samples) | Average Concentration (Range) (ng/l) | Percent Dissolved |
|---|---|---|
| Norway, Rain (5)[1] | 4.7 (1.2 - 7.6) | 44 |
| Norway, Snow (11) | 4.2 (1.5 - 7.0) | 56 |
| Lake Michigan, Rain[2] | 120[a] (190 ± 140)[b] | 66 |
| Lake Michigan, Snow (4) | 210 ± 97 | 36 |

[a] Weighted mean concentration.
[b] Arithmetic mean.

[illegible] from Lunde et al. (1977).
[illegible] from Murphy and Rzeszutko (1977).

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TABLE 1.5 PCB Concentrations in Continental Air ($10^{-9}$ g/m$^3$)

| Location and Date | No. Samples | Range | Comment |
|---|---|---|---|
| Kingston, R.I., 1973-75[1] | 6 | 1 - 15 | Calculated as Aroclor 1254 |
| Organ Pipe National Park, 1974[1] | 6 | 0.02 - 0.41 | No reference to Aroclor |
| Hays, Kansas, 1974[1] | 3 | 0.03 | No reference to Aroclor |
| Northwest Territories, Canada, 1974[1] | 3 | 0.002 - 0.07 | No reference to Aroclor |
| La Jolla, Calif., 1974[1] | 6 | 0.5 - 14 | No information on Aroclor |
| Vineyard Sound, Mass., 1973[1] | 2 | 4 - 5 | Calculated as Aroclor 1254 |
| Univ. R.I., 1973[2] | | 2.1 - 5.8 | Calculated as Aroclor 1254 |
| Providence, R.I., 1973[2] | | 9.4 | Calculated as Aroclor 1254 |
| Chicago, Ill., 1975-76[3] | 4 | 3.6 - 11.0 | 4% as 1242; 97% as "vapor" |
| Jacksonville, Fla., 1976[4] | | 3 - 36 | |
| Lake Michigan, 1976-78[5] | 6 | 0.57 - 1.6 | 74% as 1242; 88% as "vapor" |
| Milwaukee, Wis., 1978[5] | 2 | 2.7 | 59% as 1254; 27% as 1260; 84% as "vapor" |

SOURCE:
[1]Modified from Bidleman et al. (1978).
[2]Modified from Bidleman and Olney (1974).
[3]Modified from Murphy and Rzeszutko (1977).
[4]Modified from Fuller et al. (1977).
[5]Modified from Doskey (1978).



... also calculated the amount of PCBs on particulates ... be 27 percent of the total atmospheric burden.

Calculations based on another method for estimating vapor/particulate partitioning indicate that less than 5 percent of PCBs sampled in oceanic air could be associated with particulates (Junge 1977). Similar calculations for continental urban air indicate that 10 to 45 percent of Aroclor 1254 and 1260 and less than 5 percent of Aroclor 1242 could be associated with particulate matter. These estimates remain speculative because adsorption depends not only on the vapor pressure of individual mixtures, but also on the nature and amount of available surface area of the adsorbent, factors not known for most air masses.

Vapor-pressure data support the expectation that the more volatile isomers should preferentially accumulate in the atmosphere. PCB vapors with chlorine contents above a standard Aroclor 1016 mixture (greater than 41 percent chlorine) have been shown to contain more of the less chlorinated and thus more volatile isomers, for example the mono-, di- and trichlorobiphenyls (Mieure et al. 1975). This finding suggests that an atmospheric sample may bear only a slight resemblance to the composition of the original PCB product composition (Mieure et al. 1975).

## Distribution of PCBs in Air

The current state of knowledge about the atmospheric cycle of PCBs is reflected in the summary of data on continental and marine air levels presented in Tables 1.5, 1.6, and 1.7. Relatively few measurements have been made of large-scale, geographically representative areas, comparing changes through time.

Atmospheric PCB concentrations for oceanic and rural continental areas range from 0.002 ng/m³ to 1.6 ng/m³, with more than 99 percent reported as existing in the vapor phase. Data obtained from atmospheric samples taken in the North Atlantic indicate an order of magnitude decrease in PCB levels between 1973 and 1977 (Bidleman, Department of Chemistry, University of South Carolina, personal communication, 1978). But measurements designed to investigate long-term trends in these locations do not exist, and thus it is impossible to ... whether these data reflect a global trend. ... concentration of 0.05 ng/m³ is used for calculating

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TABLE 1.6 PCB Concentrations in Marine Air ($10^{-9}$ g/m$^3$)

| Location and Date | No. Samples | Range | Comments |
|---|---|---|---|
| Bermuda, 1973 | 4 | 0.15 - 0.5 | Calculated as Aroclor 1254 |
| Bermuda, 1973 | 7 | 0.19 - 0.66 | Calculated as Aroclor 1254 |
| Bermuda, 1974 | 25 | 0.08 - 0.48 | Matched Aroclor 1248 most closely |
| Bermuda-U.S. Cruises 1972-74 | 17 | <0.05 - 1.6 | Matched Aroclor 1248 most closely |
| Chesapeake Bay, 1973 | 3 | 1.0 - 2.0 | Calculated as Aroclor 1254 |
| Grand Banks, 1973 | 5 | 0.05 - 0.16 | Calculated as Aroclor 1254 |
| Georges Bank, 1973 | 5 | 0.58 - 1.6 | Calculated as Aroclor 1254 |

SOURCE: Modified from Bidleman et al. (1978).



...Trends in the Concentration of PCBs in ...tic-Bermuda Air, Seasonal and Long-Term ...r of Samples)

| ...ling Period | PCB - 1248 ($10^{-9}$ g/m$^3$) |
|---|---|
| April 1973 - Sept. 1973 | 0.76 ± 0.55 (5), 4 were < 0.33 |
| Oct. 1973 - March 1974 | 0.51 ± 0.53 (7), 5 were < 0.24 |
| April 1974 - Sept. 1974 | 0.20 ± 0.09 (19) |
| April 1975 - Sept. 1975 | 0.076 ± 0.063 (6) |
| April 1976 - Sept. 1976 | 0.077 ± 0.035 (11)[a] |
| Oct. 1976 - March 1977 | 0.074 ± 0.099 (11)[a] |

[a]These samples matched Monsantos Aroclor 1254 mixture more closely.

SOURCE: Compiled from Bidleman, Department of Chemistry, University of South Carolina, personal communication (1978).

the atmospheric PCB burden for oceanic and rural areas because the data converge around this value.

Measurements from urban/metropolitan areas also are quite variable, but data indicate concentrations ranging from 0.5 ng/m³ to 36 ng/m³. The average is about 4 ng/m³ or 5 ng/m³. Data on long-term trends do not currently exist for these areas.

Even less information exists about PCB levels in rain and snow, and it is difficult at present to predict levels for various geographically representative areas. The published data are presented in Table 1.4. Two data sets indicate that large variations can exist, both temporally and spatially. Whether the Lake Michigan measurements represent typical urban/metropolitan levels and whether the values measured at Norwegian sites represent those of remote regions is not certain.

Based on the above discussion the PCB burden in the atmosphere is conservatively estimated to be 18 x 10⁹ kg using the following data:

| | |
|---|---|
| Area of U.S. Atmosphere: | 2,500 km x 6,500 km |
| Height of Assumed Affected Air Column: | 2 km |
| Rural and Oceanic Area: | 90 percent of affected area; 0.05 ng/m³ concentration |
| Urban/Suburban Area: | 10 percent of affected area; 5.0 ng/m³ concentration |

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Removal_of_PCBs_from_the_Atmosphere

PCBs are removed from the atmosphere via wet and dry deposition. The relative importance of each has not been resolved, mainly because of difficulties in measuring dry deposition on various surfaces. The most extensive work on wet deposition has been conducted in lake Michigan (Murphy and Rzeszutko 1977). The applicability of these findings to other geographically representative regions has yet to be confirmed, but several important observations can be made at this time. In Lake Michigan, calculated yearly PCB deposition was found to be relatively uniform, although a large degree of variability was observed in rain and snow samples. The weighted mean concentration of PCBs in rain was 120 ng/l, which indicates a gross flux to the lake of 0.83 x $10^{-4}$ g/m²-yr (4,800 kg/yr).

The relationship between PCB concentrations in air and rain was studied using the concept of washout coefficients (Slinn et al. 1978). The washout coefficient for vapor depends upon the partitioning of PCBs between vapor and water, which is essentially Henry's Law Constant (H), whereas that for a particulate depends upon the collision efficiency of raindrop and particle, which is a function of the processes of diffusion, interception, and impaction. Because the precipitation-capture processes for the two phases are different, it is important to know the vapor/particulate partitioning (see discussion above).

The following relationship has been used to calculate vapor scavenging by rain: $W = \alpha p_e \bar{C}$, where W = wet flux of PCBs, $\alpha$ = washout coefficient (concentration in rain divided by concentration in air, no density correction), $p_e$ = annual amount of rainfall, and $\bar{C}$ = average PCB concentration in air (Murphy and Rzeszutko 1977). Inspection of the Lake Michigan results revealed $\alpha$ to be 6.2 x $10^3$ and 2.5 x $10^4$ for Aroclors 1242 and 1254, respectively. A value of 7.8 x $10^4$ as $\alpha$, for total PCBs, has been obtained using data for dieldrin and DDT (Bidleman et al. 1978).

Washout coefficients also can be derived from $\alpha = RT/H$, where R = the universal gas constant (8.2 x $10^{-5}$ atm-m³/mol-deg°K), T = absolute temperature, and H = Henry's Law Constant (Slinn et al. 1978). This would imply that H represents the effective Henry's Law Constant (because PCBs are mixtures) with values of 4 x



[illegible] mol for Aroclor 1242 and 1 x $10^{-4}$ atm-m³/mol for 1254.

[illegible] results are two to three orders of [illegible] lower than those obtained if the effective [illegible]'s Law Constant for various Aroclors is calculated [illegible] both solubility and vapor-pressure data (Mackay and Leinonen 1975). In order to determine H, however, the vapor pressure and aqueous solubility must refer to material under similar physical conditions (Mackay, Department of Chemical Engineering, University of Toronto, Canada, personal communication, 1978). Most individual chlorinated biphenyl isomers are solid at room temperature whereas Aroclors, which are complex isomeric mixtures, are viscous fluids. These mixtures will be partitioned into individual components when present in air or water, and therefore, must be treated accordingly (Doskey 1978). Thus published Aroclor H values may be in error by two or three orders of magnitude (Mackay, Department of Chemical Engineering, University of Toronoto, Canada, personal communication, 1978). The calculated $\alpha$ coefficients from rain data (Murphy and Rzeszutko 1977) and volatilization rate experiments (Doskey 1978) seem to bear this out, but more extensive data are required. Nevertheless, it appears that the equation $W = \alpha p_e \bar{C}$ gives a reasonable approximation for the wet deposition of PCBs (Murphy and Rzeszutko 1977).

Data on dry deposition fluxes, either to water or land surfaces, are not currently available. Measurements of PCB fallout have been made using glass plates covered with mineral oil and nylon or silicon screens coated with oil (Table 1.8). The data obtained using these techniques should not be considered representative of fallout rates, because the collection media do not reflect soil, vegetation, or water surfaces. However, such measurements can be useful in determining distributional patterns for various geographical locations.

HYDROSPHERIC COMPARTMENT

PCB contamination in the aquatic environment originates [illegible] several sources including precipitation, land [illegible]ff, industrial and municipal waste discharges, and [illegible]rial spills. Atmospheric transportation may be a [illegible]ant factor in the global distribution of PCBs to

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TABLE 1.8 Comparative Monthly Fallout of PCBs

| Location and Date | Range ($ng/m^2$-month) | Comment |
|---|---|---|
| Sweden, 1969-71[1] | 550 - 10500 | As Clophen A50 |
| Sweden, 1972-73[2] | 320 - 2900 | As Clophen A50 |
| Iceland, 1971-73[3] | ND - 1050 | Limit of Detection = 11 |
| Czechoslovakia, 1972-73[6] | 120 - 3400 | As Clophen A50 |
| Tsavo East, Kenya, 1972-73[6] | ND - 20 | As Clophen A50 |
| Nakuru, Kenya, 1972-73[6] | ND - 15 | |
| Gambia[6] | ND | |
| Sri Lanka[6] | ND | |
| La Jolla, Calif., 1972[4] | 9000 | As Aroclor 1254 |
| Southern Calif., Bight, 1973-74[5] | 3000 | As Aroclor 1254 |

ND indicates not detected.

SOURCE
[1]Modified from Sodergren (1973).
[2]Modified from Sodergren (1972).
[3]Modified from Bengtsson and Sodergren (1974).
[4]Modified from McClure and Lagrange (1977).
[5]Modified from Young et al. (1975).
[6]Modified from Sodergren (1975).

oceans and lakes, but coastal zones and estuaries near urban areas probably receive the major portion of PCB contamination from sources other than air (Young et al. 1977). PCBs initially deposited on land go through successive cycles of evaporation, deposition, and adsorption, until eventually these chemicals are carried via water runoff to lake and ocean sediments, the ultimate sinks.

When PCBs are introduced into the aquatic environment, they are dissolved in the water, adsorbed onto solids suspended in the water, or accumulated in different trophic levels (Sodergren 1973, Haque et al. 1974, Sayler and Colwell 1976, and Halter and Johnson 1977). PCBs have low water solubility and high octanol/water partition coefficients; thus, these compounds are adsorbed readily onto suspended solids, especially those high in organic carbon (Hamelink et al. 1971, Haque and Schmedding 1975, and Dexter and Pavlou 1978). Adsorption of PCB residues on the organic



[illegible] of suspended solids and the subsequent [illegible]ration of these particles into the sediment in [illegible] systems is considered a major means for [illegible]mobilizing PCBs (Oloffs et al. 1973, Haque et al. 1974, Moein 1976, Moein et al. 1976, and Hetling et al. 1978).

Mechanical and biological processes in aquatic systems may result in some recycling of PCBs. Degradation of material in the sediment may change its organic composition and release PCBs to the water column. Redistribution may also occur through erosion (Olsson et al. 1978). PCBs may be transported to the water column and surface film by the movement of sediment-generated bubbles, for example, hydrogen sulfide bubbles (Sodergren and Larssom 1979). It has been observed that a substantial portion (greater than 50 percent) of PCBs in pelagic carnivorous fishes apparently has been recycled through bottom sediment at least once (Weiniger 1978). Marine polychaetes are also able to accumulate PCBs from contaminated sediments (Courtney and Langstone 1978).

Despite major decreases in the influx of PCBs to coastal marine sites, only minor decreases in PCB concentrations can be observed in bottom sediment and flatfish (Young et al. 1977); this observation suggests continued recycling. During periods when contamination of the sediment becomes sufficiently high, PCBs may be released to the water column (Veith and Comstock 1975). If these releases were significantly greater than the rate of deposition, long-term pollution problems may result with the sediment acting as a source of PCB residues rather than a sink.

Data on deposition rates are scarce, and the basis for making estimates comes either from samples of layered sediment or from actual measurements of PCBs on settling particles. More studies are required because the available information is too scattered and incomplete to allow accurate determination of deposition rates.

Analysis of sandy, aerated marine sediment indicates that the less chlorinated isomers of a particular PCB mixture were not present, although this class of isomers could be found in anaerobic sediment samples (Eder [illegible]76). Since it is unlikely that the less chlorinated [illegible] would be retained differently in sediments, the [illegible] is suggest preferential decomposition.
[illegible]tion, in some Wisconsin rivers indicated that

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certain isomers of less chlorinated Aroclors disappeared downstream from the point source and that the proportion of highly chlorinated PCB compounds increased at the source (Veith 1972). This observation may also indicate preferential volatilization of the lighter Aroclor fractions.

PCBs can be degraded in activated sludge and it has been shown that certain bacteria can metabolize chlorobiphenyls (Ahmed and Focht 1973, Tucker et al. 1975, Tulp et al. 1978, and Furukawa et al. 1978). However, the highly chlorinated isomers are degraded more slowly than the less chlorinated forms. Thus, degradation of chlorobiphenyl isomers below penta- may occur in bottom sediment primarily as a result of microbial metabolism. Verification of this hypothesis is difficult, because field data are fragmentary and most laboratory studies have been conducted under conditions widely different from those of the natural aquatic environment. Available results do suggest, however, that degradation of most pentachlorobiphenyls is extremely slow and that of hexa- and more substituted chlorobiphenyls is practically negligible. The conclusion is that degradation is an insignificant factor in eliminating the highly chlorinated biphenyls from sediment.

## PCBs in Fresh Water

The concentration of PCBs in fresh water (including suspended particles) is generally low. Even if the sediment contains appreciable quantities, the concentration in the overlying water may be below limits of detection using available analytical procedures. Median residue levels in the major drainage basins (major rivers) of the United States range from 0.1 µg/l to 3.0 µg/l (measured between 1971 and 1974: see Dennis 1976). Freshwater concentrations of PCBs in large lake systems have been observed to be much lower, probably due to their greater distances from point sources of PCBs. For the Great Lakes, the highest value reported in open waters has been for Lake Michigan: 31 ng/l (Table 1.9). The low values reported for Lake Superior and Lake Ontario were used as conservative estimates for freshwater levels of PCBs because the Great Lakes represent 98 percent of the total freshwater volume in the United States. The low and high estimates for

1.9) PCBs in the waters of the Great

| | ng/l |
|---|---|
| Lake Superior (as Aroclor 1254) | |
| 1972-73 | 0.8[1] |
| 1976 | 5.0[2] |
| Lake Michigan | 31.0[3] |
| | 9.0[4] |
| Lake Ontario | 1.0-3.0[5] |
| Lake Huron | 5.0-7.0[6] |
| Lake Erie | 27.0[7] |

SOURCE:
[1]Veith et al. (1977).
[2]Swain (in press).
[3]Haile (1977)
[4]Edie, NOAA-Great Lakes Environmental Research Laboratory, Ann Arbor, Michigan (unpublished data)
[5]de Lappe et al. (in press)
[6]Swain, Large Lakes Research Station, Grosse Ile, Michigan (unpublished data).
[7]Glooschenko et al. (1976).

freshwater levels in the four geographical areas used in this report are 1 ng/l and 3 ng/l (Table 1.10).

## PCBs in Freshwater Sediment

Results obtained from monitoring sediment concentrations in European and African lakes and rivers are presented in Table 1.11. All PCB levels are reported on a dry weight basis. The lakes are divided into two groups: those subjected only to airborne contamination, and those contaminated with industrial and municipal wastes as well as fallout. The sediment concentration ranged from 4 µg/kg to 320 µg/kg for lakes receiving PCBs from all sources and 2 µg/kg to 20 µg/kg for those contaminated by atmospheric pollution only.

Sediment from the major drainage basins of the United States has been surveyed extensively for chemical contamination. From 1971 through 1974, median residue levels of PCBs in bottom sediment of 17 major rivers were as follows: 1.2 µg/kg to 64.5 µg/kg (1971), 2.0 µg/kg to 60.0 µg/kg (1972), 5.0 µg/kg to 50.0 µg/kg (1973), and 2.0 µg/kg to 160.0 µg/kg (1974). The

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TABLE 1.10 Estimated PCBs in Fresh Water from Four Geographical Areas Across the U.S.

| Area | $H_2O$ Volume ($km^3$) | PCB Concentration (ng/l) low | high | Amount of PCB (kg) low | high |
|---|---|---|---|---|---|
| Pacific-Continental Divide | 27 | 1 | 3 | 27 | 81 |
| Appalachian Mountains-Atlantic Coast | 100 | 1 | 3 | 100 | 300 |
| St Lawrence Drainage | 11,320 | 1 | 3 | 11,320 | 33,960 |
| Gulf of Mexico Drainage | 129 | 1 | 3 | 129 | 387 |
| | | | Total | 11,556 | 34,728 |

highest residue levels were detected in the eastern portion of the United States (Table 1.12).

Bottom sediment from 26 streams discharging into the San Francisco Bay had been collected and analyzed (Law and Goerlitz 1974). PCB residues were found in all the streams surveyed with values ranging from 1 µg/kg to 1,400 µg/kg. The values found in two streams not located near industrial or commercial developments were greater than would be expected: 180 µg/kg and 610 µg/kg. No significant differences were found between the average residues of streams discharging into the northern or southern sections of the Bay.

From 1950 to 1977, the Hudson River in New York State received wastewater from two plants that used PCBs in producing transformers and capacitors. The plants appear to have been major contributors to contamination of the river; PCB concentrations in the sediment near the outfall from these plants ranged from 6.6 mg/kg to 67,000 mg/kg (Nadeau and Davies 1974, Hetling et al. 1978). The amount of PCBs stored in the sediment of the Hudson River is estimated to be approximately 300 x $10^3$ kg (Toffelmire and Quinn, New York State Department of Conservation, personal communication, 1978).

As Figures 1.1 and 1.2 indicate, distribution of PCBs within the Hudson River sediment is not uniform. Maximum concentrations occur at depths of 5 cm to 30 cm rather than at the plane where water and sediment meet. Because the samples were collected from areas of high sediment accumulation, the observed depth distribution may reflect the reduction (beginning in 1976) and eventual cessation (in 1977) of PCB discharges from the



1.11 PCB Residues in Sediment of Lakes and ... in Europe and Africa (No. Samples)

| ...spheric Sources | | µg/kg dry weight |
|---|---|---|
| Lake Havgardssjon (10) | Sweden | 10[3] |
| Lake Lyngsjo (3) | Sweden | 8[3] |
| Lake Notsjon (3) | Sweden | 20[3] |
| Lake Kinariochy (3) | Scotland | 2[6] |
| **All Sources** | | |
| Lake Vattern (20) | Sweden | 170[1] |
| Lake Malaren (15) | Sweden | 68[2] |
| River Viskan (10) | Sweden | 45[2] |
| Lake Trummen (15) | Sweden | 4[1] |
| Lake Oresjo (2) | Sweden | 117[3] |
| River Kavlinge (5) | Sweden | 30[1] |
| Lake Mcilwaine (10) | Rhodesia | 120[5] |
| Hartbeespoort Dam (10) | South Africa | 320[4] |
| Voelvlei Dam (10) | South Africa | 70[4] |
| Lake Nakuru (10) | Kenya | 20[7] |

SOURCE:
[1]Modified from Hakansson (1977).
[2]Modified from Oden and Ekstedt (1976).
[3]Modified from Soderqren (unpublished).
[4]Modified from Greichus et al. (1977).
[5]Modified from Greichus et al. (1978b).
[6]Modified from Wells, Department of Agriculture and Fisheries of Scotland, personal communication.
[7]Modified from Greichus et al. (1978a).

General Electric plants at Hudson Falls and Fort Edward. If so, now, less contaminated sediment must have ... cumulated over older deposits. The values recorded ... undoubtedly affected by erosion during and by the ... nt of backfilling that occurs after floods. Other ... s in PCB deposition may be migration from sediment ... ring water columns and redistribution by ...

... rition of PCBs has occurred where channels ... and the rate of redeposition has been

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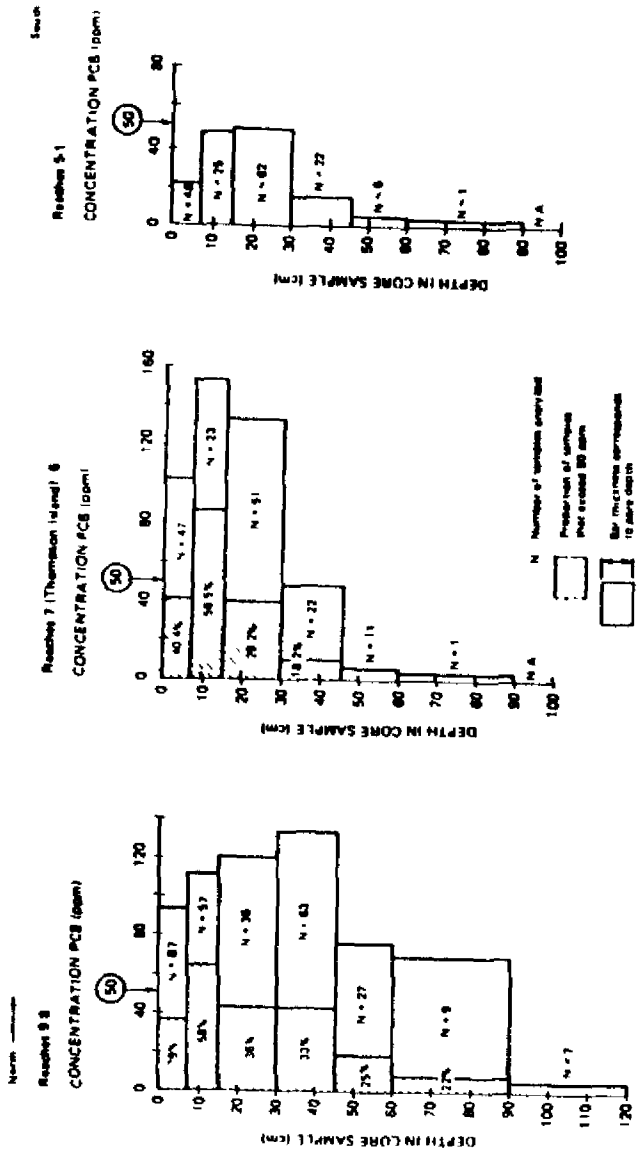
TABLE 1.12 PCBs in Bottom Sediments in the Major Drainage Basins of the United States and Puerto Rico (Sample Size)

| Drainage Basin | 1971 Median of Positive Detections in ug/kg | 1971 Range of Positive Detections in ug/kg | 1972 Median of Positive Detections in ug/kg | 1972 Range of Positive Detections in ug/kg | 1973 Median of Positive Detections in ug/kg | 1973 Range of Positive Detections in ug/kg | 1974 Median of Positive Detections in ug/kg | 1974 Range of Positive Detections in ug/kg |
|---|---|---|---|---|---|---|---|---|
| North Atlantic Slope | 50 (3) | 10 - 100 | 10 (56) | 5 - 800 | 10 (58) | 4 - 4000 | 11 (99) | 2 - 800 |
| South Atlantic Slope and Eastern Gulf of Mexico | 64 (19) | 10 - 200 | 30 (101) | 5 - 500 | 10 (123) | 5 - 600 | 11 (171) | 3 - 530 |
| Ohio River | 9 (1) | 9 | - | - | 20 (4) | 20 | 15 (6) | 9 - 61 |
| St Lawrence River | 10 (1) | 10 | 20 (10) | 3 - 800 | 20 (24) | 5 - 11000 | 20 (40) | 3 - 700 |
| ison Bay and Upper ississippi River | - | - | ND (3) | ND | ND (14) | ND | ND (26) | ND |
| - i ri. River | 1 (2) | 0.3 - 2 | 2 (22) | 2 | ND (17) | ND | ND (5) | ND |
| | 10 (4) | 10 - 80 | 3 (25) | 2 - 2400 | 20 (57) | 20 | 10 5 (81) | 1 - 1 |
| | (3) | 3 - 290 | 4 (66) | 2 - 250 | 50 (188) | 14 - 190 | 27 (15 ) | |
| | - | - | 2 (7) | 2 | ND (6) | ND | ND (3) | |
| | - | - | 2 (9) | 2 | ND (7) | ND | ND (4) | ND |
| | - | - | 20 (6) | 2 - 190 | ND (14) | ND | 2 (26) | 2 - 1 |
| | - | - | 2 (9) | 2 | ND (12) | ND | ND (5) | ND |
| | - | - | ND (11 | ND | ND (1) | ND | ND (4) | ND |
| | - | - | 2 (1) | 2 | - | - | ND (1) | D |
| | - | - | - | - | ND (2) | ND | - | - |
| | - | - | ND (11 | ND | ND (1) | ND | - | - |
| | - | - | - | - | 5 (6) | 5 | 160 (18) | 10 - 440 |

- Dots no samples.
- Indicates not detected.

Modified from Dennis (1976).




SOURCE Sanders (1979)

FIGURE 1.1 Upper river detailed core network, 1975-1978. Summary of 672 PCB analyses of sediment samples (312 cores) from the upper Hudson River. Core samples were taken from river banks and shallows, not from deep-channel areas. Large pieces of woody debris were abundant in Reaches 5 through 1 and 6, but not common in Reaches 5 through 1.



- rid as 30 cm/yr (Figure 1.2, Albany Basin, -ntown, and New York Harbor). Short core samples in and marginal shallow water areas from Kingston to Foundry Cove reflect sediment accumulation in areas not dredged.

During the period 1971 to 1972, the Chester River in the Upper Chesapeake Bay was monitored. PCB residues were found with sediment concentrations ranging from nondetectable to 300 µg/kg. The level detected at any one time depended on river flow rates (Clark and Murdock 1972). During 1974, bottom sediment was sampled from various lakes and rivers in Connecticut. The data indicated widespread contamination, particularly in highly developed areas. Levels up to 3,500 µg/kg were recorded (U.S. EPA 1976).

Table 1.13 summarizes the widespread contamination of freshwater bottom sediment throughout the United States (Crump-Wiesner et al. 1973). Samples were taken from lakes and streams remote from industrial centers to represent regional conditions. Twenty percent of the sediment samples taken across the nation contained PCBs with levels ranging from 5 µg/kg to 2,400 µg/kg.

The occurrence and distribution of PCBs in sediment from the Great Lakes have been studied extensively (Veith 1975, Glooschenko et al. 1976, Haile 1977, Veith et al. 1977, PLUARG 1978, Swain 1979, Eisenreich et al. 1979). Table 1.14 illustrates the concentrations detected in specific lake systems. These data seem consistent with sediment data from other sections of the country.

An evaluation of the degree of PCB contamination throughout the United States indicates regional variation. In Table 1.15 residue data from the United States Geological Survey (USGS) are summarized for 1972 through 1974. These data show an increase in the geographical dispersion of PCBs. In 1972, 5 states measured PCBs in bottom sediment; in 1973, 8 states, and in 1974, 13 states reported contamination with these compounds (U.S. EPA 1976). Those states reporting PCB contamination during this period continue to detect levels of the chemical today. Data from U.S. monitoring -ams and from reports in the literature through 1977 - ate that the highest level of PCBs in freshwater - found in heavily industrialized, populous particularly in the eastern part of the country. - ites are found in rural areas.

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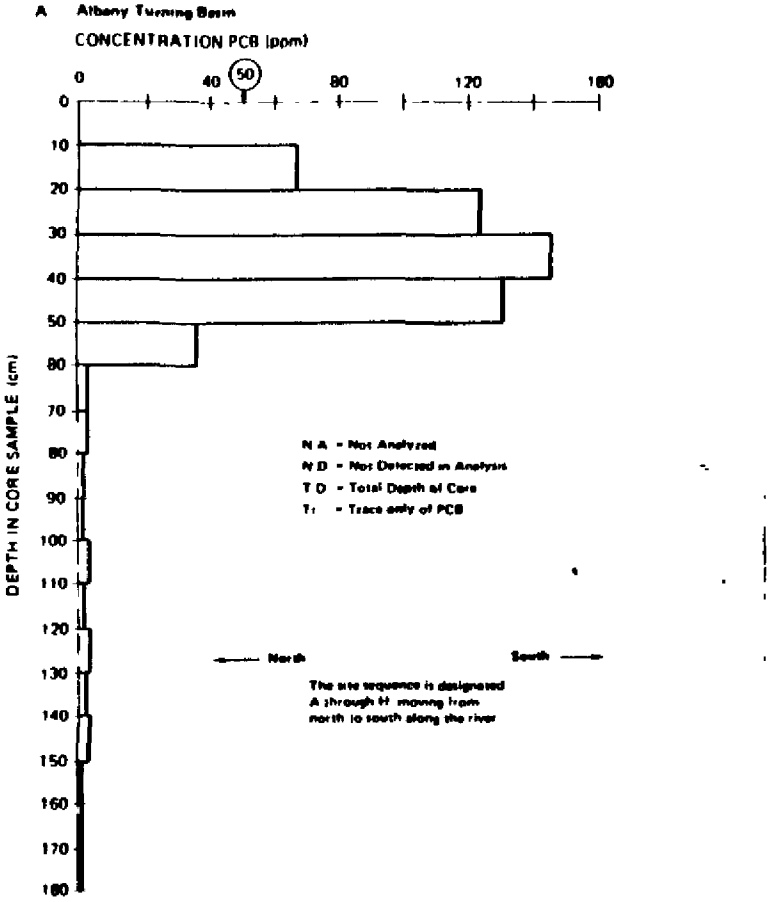


SOURCE Sanders (1979)

FIGURE 1 2 Single core reconnaissance of the Hudson River Summary of PCB analyses from eight single-core samples from a reconnaissance traverse of the lower river regions from Albany to New York Harbor Each bar represents two analyses, one for each of two different PCB isomers (Aroclor 1242 and 1254)



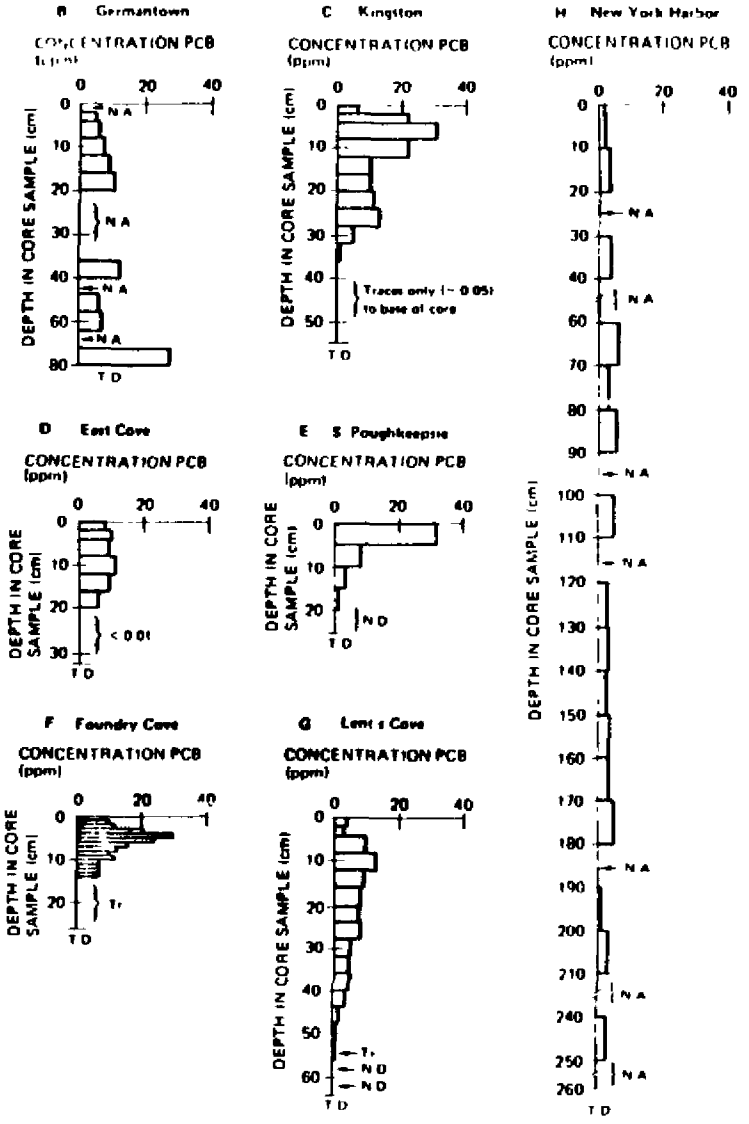


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TABLE 1.13 Summary of PCB Residue Data for Bottom Sediments January 1971 - June 1972 (Number of Samples)

| State | Range (µg/kg) | Median Concentration (µg/kg) |
|---|---|---|
| Alaska (3) | --- | --- |
| Arkansas (23) | 20 - 2,400 | 60 |
| California (13) | 20 - 190 | 85 |
| Connecticut (1) | 40 | --- |
| Hawaii (4) | --- | --- |
| Georgia (12) | 10 - 1,300 | 300 |
| Maryland (11) | 10 - 1,200 | 30 |
| Mississippi (8) | 50; 170 | --- |
| New Jersey (12) | 8 - 250 | 20 |
| Oregon (4) | 15; 140 | --- |
| Pennsylvania (16) | 10 - 50 | 20 |
| South Carolina (11) | 30 - 200 | 50 |
| Texas (293) | 7.9 - 290 | 80 |
| Virginia (10) | 5 - 80 | 40 |
| Washington (10) | --- | --- |
| West Virginia (2) | 10 | --- |

SOURCE: Modified from Crump-Wiesner et al (1973)

Table 1.16 presents estimates of total PCB burdens in sediment for particular regions throughout the United States: the Pacific Coast to the Continental Divide, the Appalachian Mountain Range to the Atlantic Coast, the St. Lawrence drainage system including the north-central and Great Lakes region, and the Gulf of Mexico drainage system including the south-central region of the United States. When calculating a total PCB burden for sediment, it is necessary to consider possible mixing activity of bottom-living organisms. This activity may carry PCB residues deeper into the sediment than would be otherwise anticipated. For that reason, a depth of 15 cm has been used for potential PCB



TABLE 1.14 PCBs in Sediment of the Great Lakes (No. Samples)

| | | µg/kg |
|---|---|---|
| Lake Superior | | 30.0[1] |
| | (4) | 7.0±0.5[2] |
| | (17) | tr - 20.0[3] |
| Lake Michigan | | 38.0[1] |
| | (7) | 12.0-25.0[4] |
| Lake Huron | | 9.0-33.0[1] |
| | (9) | tr - 20.0[3] |
| Lake Erie | | 74.0-250.0[1] |
| Lake Ontario | | 77.0-89.0[1] |
| | (7) | 43.0-240.0[4] |

tr indicates trace.

SOURCE:
[1]PLUARG (1978).
[2]Veith et al. (1977).
[3]Glooschenko et al (1976)
[4]Haile (1977)

contamination. With an average sedimentation rate of 3.3 mm/yr (a conservative estimate based on values obtained in the Great Lakes), 15 cm represents the average amount of sediment accumulated during the last 50 years (Edgington and Robbins 1976).

The area from the Pacific Coast to the Continental Divide seems to have the lowest PCB level in sediment. Most recorded values are below 10 µg/kg; therefore 2 µg/kg to 20 µg/kg are chosen as low and high estimates of PCB levels in sediment from this region. Thus, the calculated PCB burden in sediment for the region from the Pacific Coast to the Continental Divide is from 5 x $10^9$ kg to 65 x $10^9$ kg (Table 1.16).

The Appalachian Mountain-Atlantic Coast region and the St. Lawrence drainage basin represent areas where consistently high levels of PCBs have been recorded in various components of the aquatic ecosystem. The Hudson River is the most contaminated water system in the Atlantic Coast area: approximately 300 x $10^9$ kg of PCBs have been deposited in the river sediment (Toffelmire [illegible], New York State Department of Conservation, personal communication, 1978). Low and high estimates [illegible] freshwater sediment for the Appalachian-

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TABLE 1.15 Summary of PCB Residue Data for Bottom Sediments (Means and Ranges in ug/kg)

| | 1972 | | 1973 | | 1974 | |
|---|---|---|---|---|---|---|
| | Mean | No. of Stations | Mean | No. of Stations | Mean | No. of Stations |
| Arkansas | 0 | 6 | 0 | 6 | 12.8 (<2-29) | 15 |
| California | 0 | 1 | 0 | 15 | 33.5 (<2-65) | 18 |
| Connecticut | 127.5 (<5-E800) | 16 | 14.1 (<5-40) | 16 | 71.5 (<2-350) | 40 |
| Florida | 93.6 (<5-400) | 27 | 35.8 (<2-600) | 64 | 43.3 (<2-530) | 68 |
| Georgia | --- | --- | 0 | 1 | 13.6 (<1-51) | 9 |
| Illinois | --- | --- | 20 | 1 | --- | --- |
| Indiana | --- | --- | --- | --- | 35 | 4 |
| Michigan | 2 | 1 | --- | --- | 17.6 (<4-60) | 13 |
| [illegible] | --- | --- | 0 | 1 | 16.4 (<3-56) | [illegible] |
| New Jersey | --- (<5,<10) | 2 | 230.2* (<3-4000) | 20 | 172.2 (<1-800) | 15 |
| New York | --- | --- | 519* (<2-1300) | 17 | 95.8 (<2-450) | 17 |
| Pennsylvania | 50.7 (<5-E100) | 9 | 11.7 (<5-20) | 5 | 105.6 (<6-700) | 23 |
| Texas | 61.7 (<4-120) | 41 | 17.6 (<3-45) | 58 | 45.5 (<1-153) | 58 |
| Wisconsin | --- | --- | 10 | 9 | 0 | 21 |
| Puerto Rico | --- | --- | 0 | 6 | 242 (<10-640) | 16 |

--- indicates no samples.
E indicates estimated.
* indicates mean figure does not include anomalous values.

SOURCE: Modified from Nisbet (1976).



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TABLE 1.16 Estimated PCB Levels in Sediment from Four Geographical Areas Across the U.S.

| Area | Sediment Volume ($km^3$) | PCB Concentration (µg/kg) low | high | Amount of PCBs ($\times 10^3$ kg) low | high |
|---|---|---|---|---|---|
| Pacific Coast-Cont Div. | 3.2 | 2 | 20 | 5 | 65 |
| Appal. Mount.-Atlantic | 5.8 | 100 | 500 | 580 | 2900 |
| St. Lawrence Drainage | 23.7 | 30 | 150 | 700 | 3500 |
| Gulf of Mexico Drainage | 10.5 | 10 | 50 | 100 | 500 |

NOTE: The following assumptions are made - the sediment is mainly organogenic with a water content of approximately 90 percent. The PCB residues are stored within the upper 15 cm of the sediment layer.

Atlantic Coast area are 100 µg/kg and 500 µg/kg, respectively with the associated PCB burden ranging from 580 x $10^3$ kg to 2,900 x $10^3$ kg (Table 1.16).

The Great Lakes are not only the dominant water areas of the St. Lawrence drainage system, but they also constitute approximately 98 percent of the total freshwater volume of the United States. Based on the results presented in Table 1.14, 30 µg/kg to 150 µg/kg are chosen as low and high estimates for PCBs in sediment. The estimated range of PCB burden in the sediment of the St. Lawrence drainage area amounts to approximately 700 x $10^3$ kg to 3,500 x $10^3$ kg, thus representing the greatest PCB reservoir in the continental United States.

The drainage area of the Gulf of Mexico contains sediment with concentrations that range from non-detectable to 530 µg/kg, excluding samples taken in the vicinity of heavy industrial contamination. Samples below 30 µg/kg were in the majority; thus 10 µg/kg and 50 µg/kg are considered reasonable low and high estimates, indicating a PCB load of 100 x $10^3$ kg to 500 x $10^3$ kg stored in the bottom sediments.

### Freshwater Biota

[illegible]ution of PCBs in freshwater fish has been of [illegible]al concern because fish are a major source of PCB [illegible]mination in the diet of humans. Concentrations in [illegible]ertain fish species high in fat content have been in [illegible]xcess of the FDA tolerance level of 5 mg/kg.

The occurrence of PCBs in fish differs among species, as would be expected, due to differences in food gathering habits, behavior patterns, and in content of body fat. Even when fish samples are taken from the same environment, PCB concentrations in species with large amounts of fat tissue, such as carp, catfish, lake trout, or coho and chinook salmon, are greater than those in low-fat species, such as yellow perch, walleye, or northern pike. In a 1975 study using fish samples from Green Bay, Lake Michigan, PCB concentrations in the edible tissues ranged from 10.0 mg/kg to 20.0 mg/kg in lake trout, 5.1 mg/kg to 51.6 mg/kg in carp, and 3.2 mg/kg to 5.6 mg/kg in yellow perch (Kleinert 1976). The lake trout and carp had a fat content in excess of 10 percent of body weight, whereas the yellow perch fat content was less than 4 percent.

Large differences in concentrations of PCBs have been reported in fish taken from the Hudson River. For example, yellow perch had PCB levels of 16.0 mg/kg to 38.0 mg/kg and the content in goldfish ranged from 45 mg/kg to 900 mg/kg (New York State Department of Environmental Conservation 1978). Elsewhere in various fresh waters of New York State, the PCB content of fish ranged generally from 0.2 mg/kg to 2.1 mg/kg, although a few species had as much as 14 mg/kg.

PCB concentrations within species can vary geographically. For example, coho salmon samples taken from the Wisconsin portion of Lake Michigan had PCB levels of 0.02 mg/kg to 26.0 mg/kg (Kleinert 1976); those from the New York waters of Lake Erie had 0.9 [illegible]/kg to 5.7 mg/kg of PCBs; 1.7 mg/kg to 7.3 mg/kg of [illegible]Bs were detected in samples taken from New York waters [illegible] Lake Ontario (New York State Department of [illegible]ironmental Conservation 1978).

[illegible]ata from various sources indicate that the PCB [illegible]ntrations for species in certain environment[illegible], such [illegible] Michigan, the upper region of the Mississippi [illegible] Fox River in Wisconsin, and the Hudson River [illegible] [illegible]e, exceeded the FDA tolerance level of 5 [illegible] waters are known to have received large

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amounts of PCBs from industrial wastes in the [illegible]. PCB content in fish in the upper Mississippi River declined, but no such trend is evident in fish from the Great Lakes (Kleinert 1976).

Based on a maximum standing fish crop in the Great Lakes of 5 x $10^8$ kg and a mean PCB level of 3 mg/kg, the PCB content of freshwater fish in the United States is estimated to be 15 x $10^3$ kg (Veith 1975, Graham 1976, and Veith et al. 1977). It seems reasonable that the remainder of the Great Lakes biota would carry an amount approximately equal to this, giving a high estimate of 30 x $10^3$ kg. Other U.S. freshwater biota and associated PCB levels are negligible in comparison.

PCBs in the Oceans

Although some data exist on samples taken from the Pacific Ocean and the Gulf of Mexico, most measurements of PCBs in marine waters have used samples collected in the North Atlantic Ocean. A summary of most of the published data is presented in Table 1.17.

Clear-cut explanations of the observed spatial and temporal trends remains elusive. It is especially difficult to explain measurements which suggest that PCB concentrations in North Atlantic surface waters dropped by a factor of more than 40 between the summers of 1972 and 1973 (Harvey et al. 1974). The data have generated lively debates, but a satisfactory explanation of them has yet to emerge (Duce and Duursma 1977). Consequently, it is fair to say that we have a poor understanding of historical trends in marine levels of PCBs. This limitation is unfortunate because it prevents accurate predictions of residence times and ultimate fates of PCBs in an important environmental pool.

A review of the data presented in Table 1.17, however, does indicate generally higher levels of PCBs in the North Atlantic than in the Pacific Ocean and the Gulf of Mexico. Airborne PCBs may enter the North Atlantic Ocean mainly between latitudes 40° and 45°N as a consequence of the southwesterly winds prevailing from the United States (Harvey and Steinhauer 1976b). Speculations have been made on the relative importance of the several ways to distribute PCBs within the oceans (Harvey and Steinhauer 1976a). These include: (1) dilutions of PCBs by vertical and horizontal advection



(2) association of PCBs with descending [illegible], (3) evaporative codistillation of surface waters in low- and mid-latitudes, and [illegible] chemical degradation of PCBs in the mixed [illegible]. Currently, however, no quantitative information [illegible] on the relative importance of each process. [illegible] removal is probably the most important pathway [illegible] moving PCBs from the mixed layer to bottom sediment, [illegible] few measurements of PCB accumulation rates have been published (Harvey and Steinhauer 1976a).

For purposes of calculating the oceanic PCB load, an average concentration of 2.6 ng/l was used for estimating the amount of PCBs in the upper 200 m of the North Atlantic Ocean (see Table 1.17: surface values are 2.6 ng/l and mixed layer values are reported as 3.6 ng/l and 1.8 ng/l). A conservative estimate of 1 ng/l was used as the amount of PCBs in water depths below 200 m (Table 1.17 indicates reported values of 2 ng/l and 1.3 ng/l). The area of the North American Basin was taken as 10 x $10^{12}$ $m^2$ with a depth of 4,200 m (Harvey and Steinhauer 1976a). Calculations using these data suggest that North American Basin waters contain an upper limit of 6.6 x $10^7$ kg of PCBs. Calculations by other investigators suggest a burden of 4.2 x $10^7$ kg (Harvey and Steinhauer 1976a). However, it has been suggested that all measurements of PCBs in oceans have been biased upwards by sampling contamination, and that reported measurements may be too high by at least one order of magnitude (Risebrough et al. 1976). Accordingly, we have adopted as a lower limit of PCB content the value of 6 x $10^6$ kg.

PCBs in Marine Sediment

PCBs are transported to ocean waters adsorbed on suspended organic particles in river water. Sewage sludge dumped along coastal areas also contributes to the concentration as does deposition of polluted dredged spoils. It is estimated that approximately 4 x $10^8$ kg of sewage sludge and 13 x $10^8$ kg of dredged spoils are dumped off the Atlantic and Gulf Coast annually (Council on Environmental Quality 1970). If an average PCB concentration of 400 µg/kg wet weight is assumed (Berg [illegible] Peoples 1977), approximately 7 x $10^3$ kg of PCB [illegible] are transferred to the coastal waters. Annual

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TABLE 1.17 PCB Concentration of Oceanic Water (ng/l)

| Date | Location and Type | Range | Mean | Referance |
|---|---|---|---|---|
| 3/73 | Saragasso Sea (surface) | <.9 - 3.6 | <1.5 | Bidleman and Olney (1974) |
| 9/73 | Saragasso Sea - N.Y. (surface) | | .8 | Harvey et al. (1974) |
| 9/73 | Azores-Barbados (surface) | | 2 | Harvey et al. (1974) |
| 1973/74 | Gulf of Mexico (surface) | | 1.8 | Giam at al. (1976) |
| 1974 | British Coastal waters (surface) | <.15 - .52 | 0.23 | Dawson and Riley (1977) |
| 9/75 | Iceland-New England (surface) | 1-7 | 4 | Harvey and Steinhauer (1976a&b) |
| 9/75 | South Atlantic (11° S - 35° S) (surface) | 0.3 - 3.7 | 0.9 | Harvey and Steinhauer (1976a&b) |
| 1972 | North Central Pacific Gyre (surface) | | 5 | Williams and Robertson (1975) |
| 1973 | Calif. current (surface) | 0.3 - .5 | | Young (1975) |
| 1974 | New England Continental Shelf (surface) | | 0.8 | Harvey et al. (1974) |
| | Atlantic Northeast Trades (surface) | 3.5 - 5 | | Harvey et al. (1976a&b) |
| 9/75 | North Atlantic (surface) | 0.3 - 8.0 | 2.6 | Harvey and Steinhauer (1976a&b) |
| 1975 | Pacific offshore waters (surface) | ND[a] | | Risebrough et al (1976) |
| 3/73 | Western boundary of Southern Calif. bight (surface) | 0.27 - 0.49 | 0.4 | Risebrough et al. (1976) |
| 5/73 | Calif. Coastal waters (surface) | 1.1 - 5.9 | 2.1 | Risebrough et al. (1976) |
| 2/75 | Atlantic Northeast Trades (mixed layer)[b] | 0.9 - 8.7 | 3.6 | Harvey and Steinhauer (1976a&b) |
| 9/75 | North Atlantic (mixed layer) | 1.7 - 1.8 | 1.8 | Harvey and Steinhauer (1976a&b) |
| 2/75 | Atlantic Northeast Trades (deep water)[c] | 0.8 - 4.2 | 2.0 | Harvey and Steinhauer (1976a&b) |
| 9/75 | North Atlantic (deep water) | 0.7 - 2.4 | 1.3 | Harvey and Steinhauer (1976a&b) |

[a]Limit of detection ranged from 0.03 - 0.3 ng/l.
[b]Mixed layer is taken as the upper 200 meters.
[c]Deep water is taken as below a depth of 200 meters.




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PCB transport from U.S. rivers to the oceans is estimated at 2 x $10^3$ kg (Nisbet and Sarofim 1972).

Data on PCB levels in marine sediment are available primarily for coastal areas, estuaries, and harbors. Measurements in sediment deposited on the continental shelves seem to be lacking. Whereas most of the PCBs delivered to the open ocean remain in solution or are adsorbed onto suspended solids, it is likely that most of the PCBs delivered to the coastal regions are deposited in the sediment. The greater abundance of suspended organic solids in coastal waters (100 times the amount in the open ocean) plus a more rapid rate of deposition contribute to the removal of a larger proportion of PCBs from the water column in these areas. Therefore, it is inferred but not yet proven that most of the PCBs deposited in marine sediment will be found within the continental terraces.

The Gulf of Mexico Because the Gulf of Mexico receives sediment runoff from about two-thirds of the U.S. land area and one-half the area of Mexico, the content of PCBs in sediment of the Gulf is representative of the runoff situation for a large area of the continent (Table 1.18). PCB levels from 0.2 µg/kg to 39 µg/kg were found in sediment sampled in coastal areas of Texas and Louisiana (U.S. EPA 1976). In Escambia Bay, Florida, where sediment has been sampled since 1969, it has been established that an industrial outfall is responsible for part of the PCB contamination in that area. Sediment sampled near the outfall reached a PCB level of 486 mg/kg in 1969, but the contamination appears to have decreased since then (Table 1.19 and Figure 1.3).

The North Atlantic Ocean In a preliminary survey in 1973 in St. Margaret's Bay, Nova Scotia, investigators were unable to detect any PCB residues (limit of detection 1 µg/kg) in sediment samples taken from water depths of 5 m to 20 m and 60 m to 75 m (Hargrave and Phillips 1975). In surface sediment off Long Island and New Jersey, PCB levels were found at 10 µg/kg and 40 µg/kg, respectively (West and Hatcher, Atlantic Oceanographic Research Laboratory, NOAA, personal communication, 1978). In a dump site in the Christiensen Basin, at the head of Hudson Shelf Valley (73°45'W; 40°27'N, ESE of harbor light entrance to the New York Harbor), one core sample revealed 2,000 µg/kg

[illegible] of PCBs in Selected [illegible] the Gulf of Mexico (1971-1975)

| | Year of Sampling | Number of Stations | ng/g |
|---|---|---|---|
| [illegible]ta | 1971-72 | 44 | 56.0 |
| | 1973-74 | 24 | 26.0 |
| Sediment | 1973-75 | 36 | 0.3 |
| Water | 1973-74 | 34 | 1.8 |
| Air | 1973-75 | 8 | 0.4 |

SOURCE: Taken from Giam et al. (1976).

TABLE 1.19 Residues of Aroclor 1254 in Sediment Samples from Escambia Bay and River (mg/kg) (See Figure 1.1 for Station Locations)

| Location | Depth | 1970 | 1971 | 1972 |
|---|---|---|---|---|
| 1 | | | | |
| 2 | | | | |
| 6 | | | | |
| 8 | | ND | | |
| 12 | | 1.9 | | |
| 13 | | 1.8 | | |
| 14 | | 30.0 | | |
| 15 | | 4.2 | | |
| 16 | | 3.3 | | |
| 17 | | 4.9 | | |
| 18 | | 0.6 | | |
| 19 | | 2.5 | | |
| 20 | | 1.4 | | |
| 21 | 0 - 5 cm | 78.0 | 8.1 | 0.97 |
| | 5 - 10 cm | 30.0 | 0.12 | 5.8 |
| | 10 - 15 cm | 6.1 | ND | --- |
| | 15 - 20 cm | 0.4 | ND | --- |
| 22 | 0 - 5 cm | 10.0 | 0.91 | 0.14 |
| | 5 - 10 cm | 11.0 | ND | ND |
| | 10 - 15 cm | 15.0 | ND | ND |
| | 15 - 20 cm | 20.0 | ND | --- |
| | 20 - 25 cm | 18.0 | ND | --- |
| | 25 - 30 cm | 1.2 | ND | --- |
| 23 | 0 - 5 cm | 0.19 | 0.19 | --- |
| | 5 - 10 cm | 0.08 | ND | --- |
| | 10 - 15 cm | 0.02 | ND | --- |
| | 15 - 20 cm | ND | ND | --- |

[illegible] not detected.

[illegible]d from U.S. EPA (1976).

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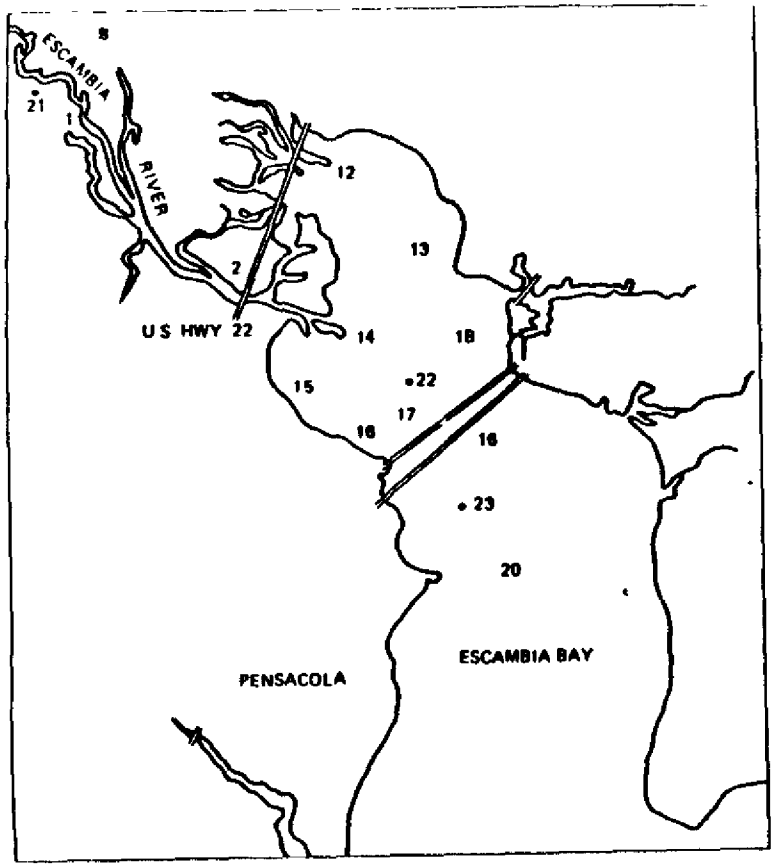


SOURCE Modified from U S EPA (1976)

FIGURE 1 3 Escambia Bay sampling stations

in the upper 4 cm and 400 µg/kg in a zone 4 cm to 23 cm below the sediment surface. In coastal sediments off Nova Scotia, New Brunswick and in the Northumberland Strait, PCBs were near or below the detection limit of 5 µg/kg (Leonard 1977). However, in a 1976 survey of four major harbors in the Atlantic Provinces of Canada, levels of PCBs ranged from 10 µg/kg to 2,900 µg/kg

[illegible] 1977). In deep [illegible] of [illegible] Canada, no PCBs were found [illegible] of [illegible] for Baffin Bay at water depths of 750 m to [illegible] m, Lancaster Sound at depths of 520 m to 579 m, and Smith Sound at 534 m (Pocklington, Bedford Institute, Halifax, Nova Scotia, personal communication, 1978).

Pacific Ocean The coastal waters of southern California have been extensively sampled for evidence of PCB residues. Dated sediment from the Santa Barbara Basin indicate that deposition of PCBs began about 1945 (Hom et al. 1974). Through 1967, the levels of PCBs detected were increasing continuously (Table 1.20). Under anaerobic conditions burrowing organism activity is negligible so that PCBs can remain undisturbed in sediment for at least 30 years.

Despite major decreases in PCB emissions over a 3-year period by a dominant discharger to a coastal marine site in Los Angeles, California, only minor decreases in PCB levels were observed in the bottom sediment (Young et al. 1977). In areas near the outfall, levels varied from 7 mg/kg in the top 5 cm in 1972 to about 4 mg/kg in 1975. In areas 12 km from the outfall, the corresponding values were 0.5 mg/kg in both 1972 and 1975.

Sediment samples taken off the Peruvian Coast at depths of 510 m and 915 m did not have detectable PCB residues; the limit of detection was 5 µg/kg (Pocklington, Bedford Institute, Halifax, Nova Scotia, personal communication, 1978).

That PCBs can be transported to considerable water depths is indicated by their presence in the

TABLE 1.20 PCB Levels in Dated Sediments from the Santa Barbara Basin (µg/kg, dry weight)

| | |
|---|---|
| 1940 - 1945 | 31 |
| 1947 - 1952 | 49 |
| 1955 - 1960 | 66 |
| [illegible] - 1967 | 100 |

Reprinted [illegible] with modifications] with permission from [illegible], Hom, W., R.W. Risebrough, A. Soutar, [illegible] Deposition of DDT and PCB in dated sediments [illegible] Basin, California (1974). American [illegible]

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bathydemersal fish, Antimora rostrata, a permanent resident of the lower continental slope in Pacific waters (Risebrough et al. 1976). These fish swim slowly over the bottom, crop benthic epifauna, and root in the sediment for infauna. The PCB residues found in tissue samples taken from these fish are believed to originate from the sediment.

The North Sea Sediment from the central North Sea and the Norwegian Depression has been sampled (Eder 1976) at water depths of about 50 m and 450 m, respectively. In the upper 20 cm of the sediment, PCB levels averaged 28 µg/kg in the North Sea; in the anaerobic sediment of the Norwegian Depression, levels of 56 µg/kg were noted (Table 1.21). The sampling sites in the North Sea are sufficiently remote from any point sources of direct pollution suggesting that atmospheric transport of PCB residues must have been the source. It also was observed that the percentage of less chlorinated PCBs was higher in the anaerobic sediment of the Norwegian Depression than in the well-aerated sand sites in the North Sea. If the ratios of high to low chlorine isomers were identical for both areas at the time of deposition, this observation would suggest that biological degradation may have occurred in the sand

TABLE 1.21 PCB Concentrations in the Central North Sea and Norwegian Depression

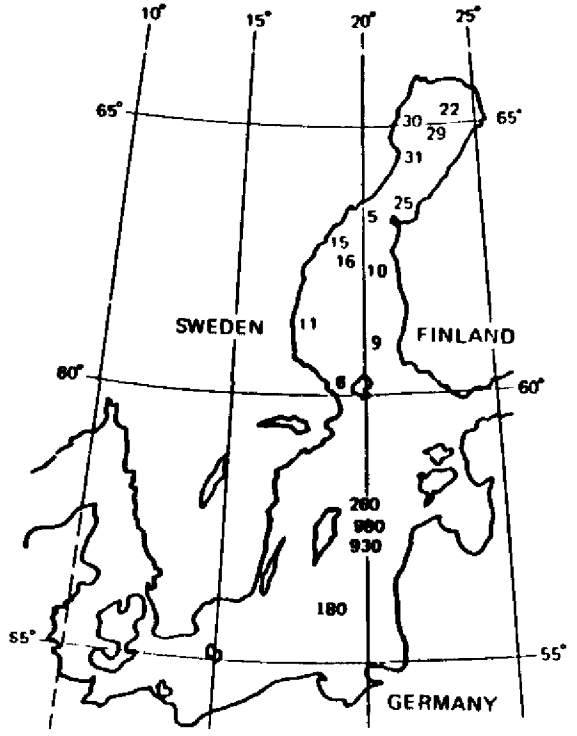
| Site Number | I | II | III | IV |
|---|---|---|---|---|
| Latitude | 55°30'N | 55°30'N | 57°40'N | 57°49'N |
| Longitude | 06°30'E | 04°50'E | 07°10'E | 08°29'E |
| Depth below sea level | 50 m | 45 m | 420 m | 505 m |
| Sediment Type | Homogeneous silty fine sand | Fine to medium sand | Clay and silt | Viscous clay and silt |
| PCB (Clophen) (weighted average, µg/kg dry sediment) | | | | |
| Upper 10 cm | 20.4±3.6 | 14.0±2.8 | 28.0±33.9 | 23.7±6.3 |
| Below 10 cm | 11.5±2.8 | 10.5±12.3 | 40.5±30.9 | 19.3±6.7 |

SOURCE: Reprinted (with modifications) with permission from [Chemosphere 2:101-106, Eder, G., PCBs and compounds of the DDT group in sediments of the central North Sea and the Norwegian Depression], Copyright (1976), Pergamon Press, Ltd.



physical factors related to solubility and ... may be important also and could result in PCB ...t from sediment.

... Baltic Sea From 1972 through 1973, sediment taken ...m various areas and depths in the Baltic Sea was ...mpled and analyzed for PCB residues (Oden and Ekstedt 1976). Only surface sediments were studied (0 to 2 cm). In the northern part of the Baltic Sea, the PCB content was fairly low (about 20 µg/kg) and without large geographical variation (Figure 1.4). Similarly low




SOURCE: Modified from Oden and Ekstedt (1976)

FIGURE 1.4 PCB concentrations in sediment samples taken in the Baltic Sea ... (ppb)

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it els had been observed in earlier studies (Linko et al. 1974). The high contamination (about 980 µg/kg) found in the central and southern portions was most probably the result of waste disposal practices.

The marine sediment data presented in the above subsections were used to estimate the quantity of PCBs stored in coastal areas of the United States. We have inferred that most PCBs are accumulated in sediment within 30 km from the coast (within 2 km on the Pacific Coast). The PCB burdens indicated in Table 1.22 were calculated for the upper 5 cm of the sediment layer.

PCBs in Marine Biota

The mass of marine biota over the eastern continental shelf of North America is taken to be 10 percent of the global marine biota, or about $1.5 \times 10^{11}$ kg (National Research Council 1977). Using a concentration of 0.2 mg/kg (Risebrough et al. 1972, Graham 1976), a plausible upperbound estimate of PCBs in North Atlantic marine biota is $3 \times 10^{4}$ kg.

Removal from the Hydrosphere

In the course of determining the mass distribution of PCBs in the Lake Michigan ecosystem, transport

TABLE 1.22 Estimated PCB Levels in U.S. Coastal Sediments

| | Sediment Volume ($km^3$) | PCB Concentration (ug/kg) low | high | Amount of PCBs ($\times 10^3$ kg) low | high |
|---|---|---|---|---|---|
| Pacific Coast | 10 | 10 | 50 | 24 | 120 |
| Atlantic Coast | 30 | 10 | 40 | 590 | 2400 |
| Gulf of Mexico | 4 | 5 | 25 | 20 | 98 |
| | | | Total | 634 | 2618 |

NOTE: The following assumptions are made - the sediment has a mass density of 2. The PCBs are stored in the upper 5 cm of the sediment layer within a distance of 30 km from the shoreline for Atlantic and Gulf of Mexico and 2 km for the Pacific Coast.



[illegible] removing hydrophobic contaminants were [illegible]ile 1977). A fractionation of the PCB [illegible] in the water indicated that nearly 60 percent [illegible] associated with heavy particles. The mean settling [illegible] was estimated to be $0.25 \times 10^{4}$ g/m²-yr. Dated sediment samples from the Santa Barbara Basin in California, showed that deposition of PCBs began about 1945 (Hom et al. 1974). The rate of deposition in this study was calculated to be $1.2 \times 10^{4}$ g/m²-yr. PCB deposition rates in sediment from the northwestern Mediterranean region had been estimated also (Elder and Fowler 1977). These calculations were based on PCB contents in a series of cores taken at a water depth of about 400 m. The values ranged from $0.8 \times 10^{4}$ g/m²-yr to $1.2 \times 10^{4}$ g/m²-yr over a period of 15 to 20 years. Fecal pellets from Eupausiids were found to contain PCB residues in the range of 1,000 µg/kg to 30,000 µg/kg dry weight, and it was suggested that sinking zooplankton fecal pellets could have contributed to deposition of PCB compounds in deep ocean sediment.

The results of these studies are in remarkably close agreement, considering the different techniques employed. The findings suggest that PCB deposition rates are similar in both lakes and marine coastal areas. Based on these data, a mean deposition rate of $1.0 \times 10^{4}$ g/m²-yr is proposed.

Rate of Release of PCBs Several studies indicate that PCBs may be released from the site of deposition in the sediment by biotic or abiotic processes. After such release, PCBs can be incorporated in the food web or reintroduced into the water column (Veith and Comstock 1975, Young et al. 1977, Courtney and Langstone 1978, Olsson et al. 1978, Sodergren and Larssom 1979, and Weiniger 1978). However, the results reported are only qualitative and do not provide estimates of the quantity being released.

LITHOSPHERIC COMPARTMENT

The estimated distribution and amount of PCBs in the lithosphere are summarized in Table 1.23. The data used to derive the estimates are discussed in the following subsections. The uncertainty involved in attempting to estimate the distribution and amount of PCBs in the

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TABLE 1.23 Estimated PCB Levels in the Lithosphere

| Compartment | Biomass (kg) | PCB Concentration (mg/kg) low | high | Amount of PCBs (kg) low | high |
|---|---|---|---|---|---|
| Soil (1 cm) | $1.35 \times 10^{14}$ | $2 \times 10^{-7}$ | $2 \times 10^{-3}$ | $2.7 \times 10$ | $2.7 \times 10^{5}$ |
| Plants | $6.35 \times 10^{11}$ | $2 \times 10^{-3}$ | $4 \times 10^{-2}$ | $1.3 \times 10^{5}$ | $2.5 \times 10^{6}$ |
| Wildlife | $6.35 \times 10^{8}$ | $2 \times 10^{-2}$ | $4 \times 10^{-1}$ | $1.3 \times 10^{1}$ | $2.6 \times 10^{2}$ |
| Livestock | $6.35 \times 10^{10}$ | $2 \times 10^{-3}$ | $10^{-1}$ | $1.3 \times 10^{2}$ | $6.3 \times 10^{3}$ |
| Man | $1.6 \times 10^{10}$ | $3.5 \times 10^{-1}$ | | $4.9 \times 10^{3}$ | $4.9 \times 10^{3}$ |
| | | | Total | $1.4 \times 10^{5}$ | $2.8 \times 10^{6}$ |

biota must be noted, particularly in the two largest compartments, soil and plants.

PCBs in Soil

Little information is available on the concentration of PCBs in soil. These compounds generally are not detectable in agricultural soil. Although many "hot spots" with high PCB levels exist in some industrialized locations, the estimated average soil concentration for metropolitan areas in the United States is about 0.002 mg/kg (Carey and Gowen 1976). In Japan, the average concentration is 0.08 mg/kg (Fujiwara 1975).

Whereas the pattern of PCB pollution in lakes appears to be controlled by atmospheric fallout, results of a nationwide network of sampling stations indicate that PCB concentrations in stream sediment are the result of accumulation downstream from known point sources of PCB discharge or from industrialized urban areas (STORET-U.S. EPA Water Quality Analysis Branch, Monitoring and Data Support Division). The contrast in patterns suggests that PCBs in rainfall make negligible contributions to sediment in streams. A possible explanation for this is that the primary source of water in stream flow is water that seeps out of the ground (defined as the base flow of streams). Only rural contributions to streams come via rain that falls on the water in the channel. (Appendix A, Table A.1, shows that the aggregate surface area of streams is only one quarter that of lakes and large reservoirs.) The inferred lack of PCBs from groundwater discharge is



consistent with data from New York State (private) negligible losses from landfill to ground water (Leis and Metry 1978, Metling et al. 1978). But if PCBs are widely dispersed in rainfall yet virtually absent from ground water, one must conclude that PCBs are accumulating in the soil and near-surface sediment underlying the soil. As already noted, data are scarce, therefore it appears to be important to monitor PCB levels in soil to find out how much enters from rainfall and what their fate is after entering.

The compartmental biomass and estimated PCB concentrations are presented in Table 1.23. Using these values plus a land area of $9 \times 10^{8}$ km$^2$, soil depth of 1 cm and soil density of 1.5 g/cm$^3$, the range of PCB burden was calculated as $2.70 \times 10$ kg to $2.7 \times 10^{5}$ kg.

Plants

An average PCB concentration in vegetation has been reported only for remote areas of Germany (0.02 mg/kg, from Klein and Weisgerber 1976). The high and low estimates in Table 1.23 were based on ranges given in the German study and represent mg/kg fresh weight. The biomass for plants was estimated in relation to livestock biomass ($6.35 \times 10^{11}$ kg, from Robinson 1975), and the PCB burden was calculated as $1.3 \times 10^{5}$ kg to $2.5 \times 10^{6}$ kg.

Wildlife

Most of the available data on PCB concentrations found in birds deal with predators. The level in tissue samples generally depends on the type of diet for each species studied. Avian species feeding mainly on small birds and mammals have PCB concentrations in liver samples that range from nondetectable to 50 mg/kg fresh weight; those feeding on fish have concentrations ranging from nondetectable to 900 mg/kg fresh weight (Prestt et al. 1970). The data indicate that liver concentrations are approximately 1.5 to 2 times the concentrations found in whole body samples. Residues in birds that feed on worms and insects are considerably lower compared to the above values. PCB tissue levels of 0.55 mg/kg fresh weight have been reported for starlings (Sodergren and Ulfstrand 1972), and 0.1 mg/kg

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to 0.65 mg/kg fresh weight reported for robins (White 1976). PCB levels in tissue of herbivorous birds average 0.2 mg/kg fresh weight (calculated for mourning doves, Kreitzer 1974).

Little information is available on PCB levels in wild mammals. Mice in Iceland contained tissue levels of 0.06 mg/kg fresh weight (Bengtsson and Sodergren 1974). Lipid samples from foxes and polar bears in Greenland contained 2.8 mg/kg and 21 mg/kg of PCBs, respectively (Clausen and Berg 1975).

PCB levels in wildlife were investigated during an extensive study of the Hudson River. Muscle and liver samples from snapping turtles in the Hudson Estuary had concentrations ranging from 0.05 µg/kg to 174 µg/kg (Metling et al. 1978). Samples of liver tissue of selected animals indicated PCB ranges of nondetectable to 0.26 µg/kg in raccoons, 0.18 µg/kg to 0.90 µg/kg in mink and 0.44 µg/kg to 1.70 µg/kg in otters (New York State Department of Environmental Conservation 1978).

It is difficult to derive an average PCB concentration for wildlife. For the purpose of Table 1 23, low and high PCB concentrations are estimated as 0.02 mg/kg and 0.4 mg/kg fresh weight, and the biomass value was estimated in relation to livestock biomass ($6.35 \times 10^8$ kg, Robinson 1975). The range for PCB burden in wildlife is calculated to be $1.3 \times 10^1$ kg to $2\ 6 \times 10^2$ kg.

## Livestock

Levels of PCBs in livestock may be derived from data on PCB concentrations in meat. In Japan, average concentrations in wet weight are 0.02 mg/kg in beef, 0 05 mg/kg in pork, 0.004 mg/kg in mutton, and 0.06 mg/kg in chicken (Fujiwara 1975). Because similar values have been reported from other countries, a concentration range of 0.002 mg/kg to 0.1 mg/kg was used for Table 1.23. The biomass was estimated using USDA Agricultural Statistics 1972 and 1974 for the number and average weight of livestock on U.S. farms. The estimated burden for this compartment ranges from $1.3 \times 10^2$ kg to $6.3 \times 10^3$ kg.



## PCBs in Humans

Levels of PCBs in human adipose tissue have been well documented. A summary of the data is presented in Table 1.24. In those cases in which the original data were not presented in mg/kg of lipid, the values were recalculated assuming that adipose tissue contains 76 percent lipid.

The concentrations detected in adipose tissue can be converted into whole body levels, assuming that lipid constitutes 30 percent of body weight. Accordingly, a PCB concentration of 0.35 mg/kg body weight has been suggested. A population size of $2 \times 10^8$ with an average weight of 70 kg per person was used for calculating biomass ($1.4 \times 10^{10}$ kg) in Table 1.23. The PCB burden associated with humans is $4.9 \times 10^3$ kg.

## Transport Between Lithosphere and Atmosphere

Volatilization rates for PCBs appear to depend on experimental conditions. A loss of 40 to 50 percent was observed for Aroclor 1254 in either dry or wet sand, at 26°C over a period of 4 weeks. In contrast, loss of Aroclor 1254 from a soil containing 3.1 percent organic

TABLE 1.24 PCBs in Human Adipose Tissue

| Country | Mean Concentration (mg/kg) |
|---|---|
| Denmark[1] | 5.0 |
| Japan[2] | 4.7 |
| West Germany[3] | 7.9 |
| Austria[3] | 4.6 |
| U.K.[3] | 1.3 |
| Norway[3] | 0.9 |
| Canada[4] | 1.2 |
| East Germany[5] | 6.4 |
| Israel[5] | 3.6 |
| New Zealand[6] | 0.9 |
| U S.A [7] | 1.2 |

SOURCE:
[1]Modified from Kraul and Karlog (1976).
[2]Modified from Fujiwara (1975).
[3]Modified from Acker and Schulte (1974).
[4]Modified from Mes et al. (1977)
[5]Modified from Wassermann et al. (1974)
[6]Modified from Solly and Shanks (1974).
[7]Modified from Kutz and Strassman (1976).

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...ater was regarded as negligible. PCBs in other soils ...th lower organic content may undergo some ...olatilization. When allowance is made for differences in experimental conditions, the rate of vaporization of Aroclor 1254 from sand appears to be of the same order of magnitude as the volatilization loss of Aroclor 1254 from itself which is 2.0 x $10^{4}$ g/$cm^2$-day at 26°C (Haque et al. 1974). If a linear relationship is assumed between PCB concentration in the top 1 cm of soil and the rate of vaporization, the expected vaporization rate from a typical sandy soil concentration of 2 µg/kg (Carey and Gowen 1976) would be 4 µg/$m^2$-day, or on the same order as the atmospheric deposition rate in the Great Lakes area (Murphy and Rzeszutko 1977).

The results indicate that PCBs deposited by atmospheric fallout onto sand, and presumably other inorganic materials (such as rock or concrete), will not accumulate but will reenter the atmosphere via volatilization at a rapid rate (2.0 µg/$cm^2$-day). PCBs deposited onto high-organic soils will reenter the atmosphere at a slower rate. However, the low concentrations observed in soils (Carey and Gowen 1976) combined with the estimated rate of deposition (Table 1.4) indicate that the residence time would be on the order of one month or less.

High-organic soils in the United States are typically covered--at least partially--with vegetation. Foliage and grasses may be efficient collectors of those PCBs associated with either particulate matter or precipitation fallout, and the vegetation probably would not easily release PCBs via volatilization; this assumption is supported by data from a study in New York. PCB levels were measured in or on foliage samples taken at control sites, upwind of facilities that formerly used PCBs. Ranges of 0.2 µg/kg to 0.8 µg/kg were reported (Leis and Metry 1978). PCBs would be introduced to the upper soil layer through leaf litter decomposition processes, but the average concentration in soil would be diluted by the additional presence of noncontaminated plant materials.

PCBs deposited on agricultural lands may never accumulate in the top layer of soil if the food crop is removed totally. Lands supporting row crops, grains, and other similar plants are typically turned under yearly, so that the PCBs are spread throughout a greater depth than the 1 cm mentioned above. Crop foliage and



...ture growth typically are removed and used as food or bedding for livestock.

Until more experiments are conducted studying processes of transfer between the atmosphere and the lithosphere, many questions regarding the net flux between these major environmental compartments cannot be answered.

## DISTRIBUTION MODEL FOR PCBs FROM DOMESTIC SOURCES

This section of the report uses estimates of domestic PCB usage from 1930 to 1977 to derive a Mobile Environmental Reservoir (MER). The MER is defined as that portion of PCBs existing in a form that can be accumulated in the environmental media and biota, including amounts detected in air, water, sediment, upper levels of soil, and biota trophic levels. Material in landfills is specifically excluded as PCBs within these sites are considered generally to be immobile.

The composition of the MER derived from data on domestic production and use, is categorized by chlorine content of the biphenyl molecule. These categories are described below in descending order of biodegradability:

| | |
|---|---|
| Category I: | PCBs containing one to three chlorine atoms per molecule (mono-, di-, tri-isomers). |
| Category II: | PCBs containing four chlorine atoms per molecule (tetra-isomers). |
| Category III: | PCBs containing five or more chlorine atoms per molecule (penta- through deca-isomers). |

### Magnitude and Composition of the MER

Data on PCB production and sales, including information on domestic sales by category of use and Aroclor designation, were obtained for the period 1957 to 1974 (Durfee et al. 1976). Using these data and known distributions of chlorine atoms per molecule for each Aroclor (Hutzinger et al. 1974) the yearly domestic sales can be calculated for each of three categories for the specified period

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The available data were extended to cover the entire period through 1977 by deriving best-fit empirical functions from Categories I, II and III of PCBs used in both electrical equipment (closed systems) and nonelectrical applications (open-end and nominally closed systems). The resulting equations are:

Electrical Applications (units are kg/yr)

$$\text{Category I Usage}^* = 2.4 \times 10^5 e^{0.097t} \quad (t = 0 \text{ at } 1930)$$
$$\text{Category II Usage} = 1.16 \times 10^5 e^{0.098t} \quad (t = 0 \text{ at } 1930)$$
$$\text{Category III Usage} = 2.24 \times 10^5 e^{0.102t} \quad (t = 0 \text{ at } 1930)$$

*In order to establish a zero base (i.e. start of production), subtract $3 \times 10^6$ kg.

Nonelectrical Applications (kg/yr)

$$\text{Category I Usage} = 2.43 \times 10^5 e^{0.177t} \quad (t = 0 \text{ at } 1950)$$
$$\text{Category II Usage} = 2.21 \times 10^5 e^{0.155t} \quad (t = 0 \text{ at } 1959)$$
$$\text{Category III Usage} = 7.35 \times 10^4 e^{0.105t} \quad (t = 0 \text{ at } 1950)$$

In these equations, time (t) is measured in years from the initial dates of production (t = 0) as specified for each equation. Nonelectrical applications became significant around 1950 and correlates approximately with commercialization of Aroclor 1248. After 1971, sales for nonelectrical applications effectively became zero.

The model described above results in a total cumulative use of PCBs from domestic sources through 1975 of $6.1 \times 10^8$ kg, 7 percent higher than the value of $5.7 \times 10^8$ kg previously estimated (Durfee et al. 1976). The production percentages, extended through 1977 for each of the three PCB categories, are:

Category I: 38 percent
Category II: 23 percent
Category III: 39 percent

The assumed loss factors for production and use of PCBs summarized in Table 1.25 are based on engineering judgment applied to the available information (Durfee et al. 1976). The factors are assumed to apply to all categories, as noted, through the cessation of domestic production in 1977.

Table 1.26 summarizes the cumulative losses to the MER calculated by applying loss factors from Table 1.25 to the production-use model above. For Category I cumulative losses in electrical use, the basic equation



TABLE 1.25 Assumed Loss Factors for Production and Usage of PCBs (1930-1977)

| Function | Percent of Overall Loss Factor | Percent of Fraction Entering the MER |
|---|---|---|
| Primary Production | 2[a] | 50 |
| Transport/Handling | 0.5[a] | 100 |
| First Tier Usage | 2[b] | 50 |
| In-Service Electrical Equipment | 0.1 of material in service per year[c] | 100 |
| Material in Nonelectrical Applications | 10 of material in service per year[d] | 100 |

[a]Applicable to total domestic production.
[b]Applicable to material used in each of two general applications (electrical and nonelectrical)
[c]Useful service life of 40 years.
[d]Useful service life of 6 years.

without zero correction was used, and the result corrected by subtracting $0.2 \times 10^7$ kg. A similar zero correction, by subtracting $0.05 \times 10^7$ kg, is required for Category II losses from electrical service. These corrections are required because of the extrapolations made for the period from 1930 to 1957; however, the factors are small when compared to the total cumulative amounts in the MER for each category.

The results in Table 1.26 indicate that, as of 1977, a total of about $8.25 \times 10^7$ kg of PCBs had entered the MER. The amounts by PCB category were:

| | Amounts ($\times 10^7$ kg) |
|---|---|
| Category I: | 3.12 |
| Category II: | 1.83 |
| Category III: | 3.30 |

The estimate of $8.25 \times 10^7$ kg is directly comparable with the 1972 estimate of $8.2 \times 10^7$ kg (Nisbet and Sarofim 1972), as well as the value derived in 1975 for all PCBs available to the biota (ignoring degradation processes) of $7.8 \times 10^7$ kg (Durfee et al. 1976). The MER model suggests that the cumulative amounts of PCBs

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TABLE 1.26 Estimated Cumulative Losses of PCBs to the Mobile Environmental Reservoir as of 1977 by PCB Category (Chlorine Content) and Production/Use Function

| | PCB Losses to the MER as of 1977 (kg x $10^7$) | | | |
|---|---|---|---|---|
| Production/Use Function | Category I | Category II | Category III | Total |
| Primary Production | 0.30 | 0.18 | 0.31 | 0.79 |
| Transportation/Handling | 0.15 | 0.09 | 0.16 | 0.40 |
| First Tier Usage | 0.37 | 0.23 | 0.38 | 0.98 |
| In-Service Elec. Equip. | 0.14* | 0.13* | 0.19 | 0.46 |
| Non-Electrical Applications | 2.16 | 1.20 | 2.26 | 5.62 |
| | 3.12 | 1.83 | 3.30 | 8.25 |

*Zero-corrected (see text).



in the mobile environment as of 1972 and 1975 were significantly less than previously estimated.

Removal of PCBs from the MER

The major identified process for removal of PCBs from the MER (as defined above) is land disposal of sewage sludge. Using the estimate of 2.4 x $10^3$ kg/yr of PCBs in sewage sludge (see the subsection on "Industrial and Municipal Effluents" above), and applying this value over a 20-year period, it is estimated that about 4.8 x $10^4$ kg of PCBs have been removed from the MER via sewage treatment and subsequent sludge disposal.

## SUMMARY COMPARISON OF INVENTORY

The environmental inventory is summarized in Table 1.27. The total estimated range of domestically derived PCBs in or removed from the environment is between 1.3 x $10^7$ kg and 8.4 x $10^7$ kg. These values compare favorably to the estimated MER of 8.25 x $10^7$ kg.

The major portion of the environmentally available PCBs is contained in marine waters, accounting for 50 to 80 percent of the total inventory. The total amount of PCBs in the North American Basin is estimated to be 0.6 x $10^7$ kg to 6.6 x $10^7$ kg. In comparison to these values, water concentrations from the Pacific Ocean and Gulf of Mexico attributable to domestically produced PCBs can be considered negligible. It should be noted that the values in Table 1.27 are based on measurements referring mainly to PCBs in Category III and that the high estimate is greater than the estimate of environmental releases of Category III materials. It is possible that some of the oceanic PCBs originate from European as well as North American sources. This compartment appears to be a dominant reservoir for PCBs even if the oceanic inventory is high by a factor of two. The rate at which PCBs will eventually reach (and perhaps become bound to) deep ocean sediment, however, cannot be estimated on the basis of current knowledge.

The close agreement between environmental measurements and the results of the model obtained in this analysis should not be interpreted as an indication that biodegradation and other processes of destruction of PCBs do not occur in the environment. Biodegradation

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TABLE 1.27 Summary of Environmental Inventory of U.S.-Derived PCBs

| Compartment | Estimate* (kg) Low | | High |
|---|---|---|---|
| Air | | $1.8 \times 10^4$ | |
| Lithosphere | $1.4 \times 10^5$ | | $2.8 \times 10^6$ |
| Fresh Water Sediments | $1.4 \times 10^6$ | | $7.1 \times 10^6$ |
| Fresh Waters | $1.2 \times 10^4$ | | $3.5 \times 10^4$ |
| Fresh Water Biota | | $3.0 \times 10^4$ | |
| Oceanic Sediments | $6.6 \times 10^5$ | | $2.7 \times 10^6$ |
| Oceanic Waters | $6.0 \times 10^6$ | | $6.6 \times 10^{7**}$ |
| Oceanic Biota | | $3.0 \times 10^4$ | |
| Sewage Sludge | | $4.8 \times 10^6$ | |
| Total | $1.3 \times 10^7$ | | $8.4 \times 10^7$ |
| Percentage of Total in Oceanic Waters | 50% | | 79% |

*Where only one estimate was made it is placed between the low and high columns; only two significant digits are used for the estimates and totals.

**Some PCBs in this estimate may be derived from European sources.

may be the ultimate fate of the PCBs now in the MER. The results, however, do indicate that biodegradation (or other processes resulting in chemical alteration of the molecules) has not destroyed a major fraction of environmental PCBs at this time.

## RECOMMENDATIONS

1. Because the North Atlantic Ocean appears to be the ultimate sink, further studies are needed to determine PCB concentrations and distribution in the marine environment.

2. Although modeling was useful in assessing large-scale distribution and fate of toxic substances, more adequate data bases are needed to develop transport models of environmental pollutants. For example,



further research is needed to determine the processes of transfer between the environmental compartments.

3. EPA should continue to develop better analytical techniques, particularly those enabling assessment of the extent of PCB fractionation. Because of continuing analytical problems it is difficult to determine the extent of ecosystem contamination. For aquatic systems, the most effective indicator of both the degree of PCB pollution and the potential hazards to humans and wildlife is the PCB content of fish. To effectively use this indicator, factors such as age, species, and nutritional habits of fish must be considered. Detailed information on the individual PCB isomers should also be reported together with the specific Aroclor designation.

4. Freshwater sediments are important continental sinks for PCBs and special attention should be given to monitoring programs and studies of PCB cycling from sediment through human food chains.

## REFERENCES


Acker, L. and E. Schulte (1974) Chlorkohlenwasserstoffe im Menschlichen fett. Naturwissenschaften 61 32.

Ahmed, M. and D.D. Focht (1973) Degradation of PCBs by two species of Achromobacter. Can. J. Microbiol. 19:47-52.

Bengtsson, S.A. and A. Sodergren (1974) DDT and PCB in airborne fallout and animals in Iceland. Ambio 3(2):84-86.

Berg, A.K. and R.S. Peoples (1977) Distribution of PCBs in a municipal wastewater treatment plant and environs. Sci. Total Environ. 8:197-204.

Bidleman, T.F. and C.E. Olney (1974) Chlorinated hydrocarbons in the Saragasso Sea atmosphere and surface water. Science 183:516-518.

Bidleman, T.F., C.P. Rice, and C.E. Olney (1978) High molecular weight chlorinated hydrocarbons in the air and sea: Rates and mechanisms of air/sea transfer. Pages 321-351, Marine Pollutant Transfer. Edited by H.L. Windom and R.A. Duce. Lexington, Mass.: D.C. Heath and Company.

Carey, A.E. and J.A. Gowen (1976) PCBs in agricultural and urban soil. Pages 195-198, Proceedings of National Conference on Polychlorinated Biphenyls, November 19-21, 1975, Chicago, Illinois. Report No.


MONS 219151

EPA-560/6-75-004. Washington, D.C.: U.S. Environmental Protection Agency.

Carr, R.A., G.Y. Contos, R.L. Durfee, V.C. Fong, and E.G. McKay II (1977) Involvement of PCBs in the Pulp and Paper Industry. Office of Toxic Substances. Final Task Report No. EPA 560/6-77-005, Contract No. 68-01-3259. Washington, D.C.: U.S. Environmental Protection Agency.

Clark, W.D. and L.C. Murdock (1972) Chester River Study: A Joint Investigation by the State of Maryland Department of Natural Resources and Westinghouse Electric Corporation, Vol. I. Annapolis, Md.: Westinghouse Electric Oceanic Division.

Clausen, J. and O. Berg (1975) The content of polychlorinated hydrocarbons in Artic ecosystems. Pure Appl. Chem. 42:223.

Council on Environmental Quality (1970) Environmental Quality. The First Annual Report. Washington, D.C.: U.S. Government Printing Office.

Courtney, W.A.M. and W.J. Langstone (1978) Uptake of PCB (Aroclor 1254) from sediment and from sea water in two intertidal polychaetes. Environ. Pollut. 15:303-309.

Crump-Wiesner, H.J., H.R. Feltz, and M.L. Yates (1973) A study of the distribution of polychlorinated biphenyls in the aquatic environment. J. Res. USGS 1(5):603-607.

Dawson, R. and J.P. Riley (1977) Chlorine-containing pesticides and polychlorinated biphenyls in British coastal waters. Estuarine Coastal Mar. Sci. 5:55-69.

de Lappe, B.W., R.W. Risebrough, A.M. Springer, T.T. Schmidt, J.C. Sharopshire, E.F. Letterman, and J.R. Payne (In press) Sampling and measurement of hydrocarbons in natural waters. Proceedings of International Symposium on Analysis of Hydrocarbons. Edited by B.F. Afghan and D. Mackay.

Dennis, D.S. (1976) Polychlorinated biphenyls in the surface waters and bottom sediments of the major dralnage basins of the United States. Pages 183-194, Proceedings of National Conference on Polychlorinated Biphenyls, November 19-21, 1975, Chicago, Illinois. Report No. EPA-560/6-75-004. Washington, D.C.: U.S. Environmental Protection Agency.

Dexter, R.N. and S.P. Pavlou (1978) Mass solubility and aqueous activity coefficients of stable organic



chemicals in the marine environment. PCBs. Mar. Chem. 6 41-53.

Doskey, P.V. (1978) Transport of Airborne PCBs to Lake Michigan. M.S. Thesis, Water Chemistry Program. Madison, Wis.: University of Wisconsin.

Duce, R.A. and E.K. Duursma (1977) Inputs of organic matter to the ocean. Mar. Chem. 5:319-339.

Durfee, R.L., G. Contos, F.C. Whitmore, J.D. Barden, E.E. Hackman III, and R.A. Westin (1976) PCBs in the United States--Industrial Use and Environmental Distribution. Report No. EPA 56016-16-005. Springfield, Va.: Versar, Inc.

Eder, G. (1976) PCBs and compounds of the DDT group in sediments of the central North Sea and the Norwegian Depression. Chemosphere 2 101-106.

Edgington, D.N. and J.A. Robbins (1976) Records of lead deposition in Lake Michigan sediments since 1800. Environ. Sci. Technol. 10 266-273.

Eisenreich, S.J., G.J. Hollod and T.C. Johnson (1979) Accumulation of PCBs in Surficial Lake Superior Sediments: Atmospheric Deposition. A report of the Limnological Research Center, University of Minnesota, Minneapolis, Minnesota.

Elder, D.L. and S.W. Fowler (1977) PCBs: Penetration into the deep ocean by zooplankton fecal pellets transport. Science 197:459-461.

Fujiwara, K. (1975) Environmental and food contamination with PCBs in Japan. Sci. Total Environ. 4:219.

Fuller, B., J. Gordon, and M. Kornreich (1977) Environmental Assessment of PCBs in the Atmosphere. EPA Report No. EPA-450/3-77-045. McLean, Va.: The Mitre Corporation.

Furukawa, K., K. Tonomura, and A. Kamibayashi (1978) Effect of chlorine substitution on the biodegradability of polychlorinated biphenyls. Appl. Environ. Microbiol. 35:223-227.

Giam, C.S., H.S. Chan, and G.S. Neff (1976) Concentrations and fluxes of phthalates, DDTs and PCBs to the Gulf of Mexico. In, Marine Pollutant Transfer. Edited by H. Windom and R. Duce. Lexington, Mass.: D.C. Heath and Company.

Glooschenko, W.A., W.M.J. Strachan, and R.C. Sampson (1976) Distribution of pesticides and PCBs in water, sediments, and seston of the Upper Great Lakes. Pestic. Monit. J. 10:61-67.

Graham, J.M. (1976) Levels of PCBs in Canadian commercial fish species. Pages 159-160, Proceedings

MONS 219152

of National Conference on Polychlorinated Biphenyls, November 19-21, 1975, Chicago, Illinois. Report No. EPA-560/6-75-004. Washington, D.C.: U.S. Environmental Protection Agency.

Greichus, Y.A., A. Greichus, B.A. Ammann, D.J. Call, D.C.D. Hamman, and R.M. Pott (1977) Insecticides, polychlorinated biphenyls and metals in African Lake ecosytems. I. Hartbeespoort Dam, Transvaal and Voelvlei Dam, Cape Province, Republic of South Africa. Arch. Environ. Contam. Toxicol. 6:371-383.

Greichus, Y.A., A. Greichus, B.D. Ammann, and J. Hopcraft (1978a) Insecticides, polychlorinated biphenyls and metals in African Lake ecosystems. III. Lake Nakuru, Kenya. Bull. Environ. Contam. Toxicol. 7:454-461.

Greichus, Y.A., A. Greichus, H.A. Draayer, and B. Marshall (1978b) Insecticides, PCBs and metals in African Lake ecosystems. II. Lake Mellwaine, Rhodesia. Bull. Environ. Contam. Toxicol. 7:444-453.

Haile, C.L. (1977) Chlorinated Hydrocarbons in the Lake Ontario and Lake Michigan Ecosystems. Thesis. Madison, Wis.: University of Wisconsin.

Hakansson, L. (1977) Sediments as indicators of contamination - investigations in the four largest Swedish Lakes. Report No. SNV PM 839. Environmental Protection Board of Sweden.

Halter, M.T. and H.E. Johnson (1977) A model system to study the desorption and biological availability of PCB in hydrosoils. Pages 178-195, Aquatic Toxicology and Hazard Evaluation, ASTM STP 634. Edited by F.L. Mayer and J.L. Hamelink. Philadelphia: American Society for Testing and Materials.

Hamelink, J.L., R.C. Waybrant, and R.C. Ball (1971) A proposal: exchange equilibria control the degree chlorinated hydrocarbons are biologically magnified in lentic environments. Trans. Am. Fish Soc. 100:207.

Haque, R., D.W. Schmedding, and V.H. Freed (1974) Aqueous solubility, adsorption, and vapor behavior of polychlorinated biphenyl Aroclor 1254. Environ. Sci. Technol. 8:139-142.

Haque, R. and D.W. Schmedding (1975) The method of measuring the water solubility of hydrophobic chemicals: solubility of five polychlorinated biphenyls. Bull. Environ. Contam. Toxicol. 14:13-18.

Hargrave, B.T. and G.A. Phillips (1975) The lack of accumulation of DDT residue by benthic invertebrates



and demersal fish in St. Margaret's Bay, Nova Scotia. Int. Counc. Expl. Sea, CM-E:32.

Harvey, G.R. and W.G. Steinhauer (1974) Atmospheric transport of polychlorobiphenyls to the North Atlantic. Atmos. Environ. 8:777-782.

Harvey, G.R. and W.G. Steinhauer (1976a) Biogeochemistry of PCB and DDT in the North Atlantic, Vol. I. Pages 203-221, Environmental Biogeochemistry. Edited by J.O. Nriagu. Ann Arbor, Mich.: Ann Arbor Science Publishers, Inc.

Harvey, G.R. and W.G. Steinhauer (1976b) Transport pathways of polychlorinated biphenyls in Atlantic water. J. Mar. Res. 34:561-575.

Harvey, G.R., W.G. Steinhauer, and H.P. Miklas (1974) Decline of PCB concentrations in North Atlantic surface water. Nature 252:387-388.

Hetling, L., E. Horn, and T.J. Tofflemire (1978) Summary of Hudson River PCB Study Results. Technical Report No. 51. Albany, N.Y.: New York State Department of Environmental Conservation.

Hom, W., R.W. Risebrough, A. Soutar, and D.R. Young (1974) Deposition of DDE and PCB in dated sediments of the Santa Barbara Basin. Science 184 1197-1199.

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

Junge, C.E. (1977) Basic considerations about trace constituents in the atmosphere as related to the fate of global pollutants. Pages 7-25, Part 1--The Mechanisms of Interaction between Environments and Mathematical Modeling and the Physical Fate of Pollutants. Edited by I.H. Suffet. New York: Wiley-Interscience.

Klein, W. and I. Weissgerber (1976) PCBs and environmental contamination. Environ. Qual. Saf. 5:237.

Kleinert, S.J. (1976) The PCB Problem in Wisconsin. Report for the Joint Hearing of the Assembly of Environmental Quality Committee and Senate and Assembly of Natural Resources Committees on HR 212 Administrative Rules, Madison, Wisconsin, September 21.

Kraul, I. and O. Karlog (1976) Persistent organochlorinated compounds in human organs collected in Denmark 1972-73. Acta Pharmacol. Toxicol. 38:38.

Kreitzer, J.F. (1974) Residues of organochlorine pesticides, mercury and PCBs in mourning doves from

MONS 219153

Eastern United States - 1970-71. Pestic. Monit. J. 7:195.

Kutz, F.W. and S.C. Strassman (1976) Residues of polychlorinated biphenyls in the general population of the United States. Page 139, Proceedings of National Conference on Polychlorinated Biphenyls, November 19-21, 1975, Chicago, Illinois. Report No. EPA 560/6-75-004. Washington, D.C.: U.S. Environmental Protection Agency.

Law, L.M. and D.F. Goerlitz (1974) Selected chlorinated hydrocarbons in bottom materials from streams tributary to San Francisco Bay. Pestic. Monit. J. 8:33-36.

Leis, W.M. and A. Metry (1978) Study of Migration of PCBs from Landfills and Dredge Spoil Sites and Related to the Hudson River. Final Report submitted by Weston Environmental Consultants-Designers, West Chester, Pennsylvania to New York State Department of Environmental Conservation, Albany, New York.

Leonard, J. (1977) Organohalogens in Coastal Sediments from the Maritime Provinces, Canada. Bedford Institute of Oceanography Report Series, June. Halifax, Nova Scotia: Bedford Institute of Oceanography.

Linko, R.R., P. Rantamaki, and K. Urpo (1974) PCB residues in plankton and sediments in the southwestern coast of Finland. Bull. Environ. Contam. Toxicol. 12:733-738.

Lunde, F., J. Gether, N. Gjos, and N.S. Lande (1977) Organic micro-pollutants in precipitation in Norway. Atmos. Environ. 11:1007-1014.

Mackay, D. and P.J. Leinonen (1975) Route of evaporation of low-solubility contaminants from water bodies to the atmosphere. Environ. Sci. Technol. 9:1178-1181.

Mackay, D. and A.W. Wolkoff (1973) Rate of evaporation of low-solubility contaminants from water bodies to atmosphere. Environ. Sci. Technol. 7:611-614.

McClure, E.V. and J. Lagrange (1977) Deposition of heavy chlorinated hydrocarbons from the atmosphere. Bull. Environ. Contam. Toxicol. 17(2):219-224.

Mes, J., D.S. Campbell, R.N. Robinson, and D.J.A. Davies (1977) Polychlorinated biphenyls and organochlorine pesticide residues in adipose tissue of Canadians. Bull. Environ. Contam. Toxicol. 17:196.

Mieure, J.P., O. Sicks, R.G. Kaley, B.W. Saeger (1975) Characterization of polychlorinated biphenyls. Pages 84-93, Proceedings of National Conference on



Polychlorinated Biphenyls, November 19-21, 1975, Chicago, Illinois. Report No. EPA 560/6-75-004. Washington, D.C.: U.S. Environmental Protection Agency.

Moein, G.J. (1976) Study of Distribution and Fate of PCBs and Benzenes After Spill of Transformer Fluid. Report No. EPA 904/9-76-014. Washington, D.C.: U.S. Environmental Protection Agency.

Moein, G.J., A.J. Smith, and P.L. Stewart (1976) Follow-up study of the distribution and fate of PCBs and benzenes in soil and ground water samples after an accidental spill of transformer fluid. Pages 368-372, Proceedings of 1976 National Conference on Control of Hazardous Materials Spills. Rockville, Md.: Information Transfer, Inc.

Murphy, T.J. and C.P. Rzeszutko (1977) Precipitation inputs of PCBs to Lake Michigan. J. Great Lakes Res. 3(3-4):305-312.

Nadeau, R.J. and R.P. Davies (1974) Investigation of PCB in the Hudson River. Washington, D.C.: U.S. Environmental Protection Agency.

National Research Council (1977) An Assessment of Mercury in the Environment. Committee for Scientific and Technical Assessments of Environmental Pollutants, Environmental Studies Board, Commission on Natural Resources. Washington, D.C.: National Academy of Sciences.

New York State Department of Environmental Conservation (1978) Toxic Substances Impacting on Fish and Wildlife. Division Fish and Wildlife Monthly Report 12, March 20. Albany, N.Y.: New York State Department of Environmental Conservation.

Nisbet, I.C.T. (1976) Criteria Document for PCBs. Report No. PB-225 397. Washington, D.C.: U.S. Department of Commerce.

Nisbet, I.C.T. and A.F. Sarofim (1972) Rates and routes of transport of PCBs in the environment. Environ. Health Perspect. 1:21-38.

Oden, S. and J. Ekstedt (1976) PCB and DDT in Baltic sediments. Ambio Special Report 4:125-129.

Oloffs, P.C., L.J. Albright, S.Y. Szeto, and J. Lau (1973) Factor affecting the behavior of five chlorinated hydrocarbons in two natural waters and their sediments. J. Fish. Res. Bd. Can. 30.1619-1621.

[illegible], M., S. Jensen, and L. Reutergard (1978) Seasonal variation of PCB levels in fish - An important

MONS 219154

factor in planning monitoring programs. Ambio 7:66-69.

PLUARG-International Reference Group on Great Lakes Pollution from Land Use Activities (1978) Environmental Management Strategy for the Great Lakes System. Final Report to the International Joint Commission. Windsor, Ontario, Canada.

Prestt, I., D.J. Jeffries, and N.W. Moore (1970) Polychlorinated biphenyls in wild birds in Britain and their avian toxicity. Environ. Pollut. 1:3.

Risebrough, R.W., B.W. de Lappe, and W. Walker II (1976) Transfer of higher molecular weight chlorinated hydrocarbons to the marine environment. Pages 261-321, Marine Poilutant Transfer. Edited by H.L. Windom and R.A. Duce. Lexington, Mass.: D.C. Heath and Company.

Risebrough, R.W., V. Freeland, G.R. Harvey, H.P. Miklas, and G.M. Carmigan (1972) PCB residues in Atlantic zooplankton. Bull. Environ. Contam. Toxicol. 8:345-355.

Robinson, J. (1975) Mathematical models in ecochemistry. Pure Appl. Chem. 42:139-153.

Sanders, J.E. (1979) Sedimentologic Aspects of the PCB Pollution of the Hudson River. (Manuscript in preparation)

Sargent, D.H., R.A. Carr, and G.A. Vogel (1977) PCBs Removal in Publicly-Owned Treatment Works. Final Report, Contract No. 68-01-3273, Task 13. Criteria and Standards Division. Washington, D.C.: U.S. Environmental Protection Agency.

Sayler, G.S. and R.R. Colwell (1976) Partitioning of mercury and PCB by oil, water, and suspended sediment. Environ. Sci. Technol. 10:1142-1145.

Slinn, W.G.N., L. Hasse, B.B. Hicks, A.W. Hogan, D. Lal, P.S. Liss, K.O. Munnich, G.A. Sehmel, and O. Vittori (1978) Some aspects of the transfer of atmospheric trace constituents past the air/sea interface. Atmos. Environ. 12(11):2055-2087.

Sodergren, A. (1972) Chlorinated hydrocarbons residues in airborne fallout. Nature 236:395-397.

Sodergren, A. (1973) Transport, distribution and degradation of organochlorine residues in a South Swedish lake ecosystem. Vatten 2:90-108.

Sodergren, A. (1975) Monitoring DDT and PCB in airborne fallout. Environ. Qual. Saf. 3:803-810.



Sodergren, A. and P. Larsson (1979) Transport of PCBs from Lake Sediments to the Atmosphere via the Aquatic Surface Microlayer. (To be published)

Sodergren, A. and P. Ulfstrand (1972) DDT and PCB relocate when caged robins use fat reserves. Ambio 1:36-40.

Solly, S.R.B. and V. Shanks (1974) Polychlorinated biphenyls and organochlorine pesticides in human fat in New Zealand. N.Z. J. Sci. 17:535.

Swain, W.R. (1979) Chlorinated organic residues in fish, water, and precipitation from the vicinity of Isle Royale, Lake Superior. J. Great Lakes Res. (In press)

Travers, I.C. and R.C.H. Wilson (1977) PCBs in the Atlantic Provinces. Surveillance Report No. EPS-5-AR-77-13. Halifax, Nova Scotia: Environmental Protection Service.

Tucker, E.S., V.W. Saeger, and O. Hicks (1975) Activated sludge primary biodegradation of PCB. Bull. Environ. Contam. Toxicol. 14:705-713.

Tulp, M.Th.M., R. Schmitz, and O. Hutzinger (1978) The bacterial metabolism of 4,4' dichlorobiphenyl and its suppression by alternative carbon sources. Chemosphere 7:103-108.

U.S. Environmental Protection Agency (1978) Polychlorinated biphenyls (PCBs); Disposal and marking. Federal Register 43(36):7149-7164.

U.S. Environmental Protection Agency (1976) Review of PCB Levels in the Environment. Report No. EPA-560/7-76-001. Washington, D.C.: U.S. Environmental Protection Agency.

Veith, G.D. (1972) Recent fluctuations of PCBs in the Green Bay Wisconsin region. Environ. Health Perspect. 1:51-52.

Veith, G.D. (1975) Baseline concentrations of polychlorinated biphenyls in Lake Michigan fish, 1971. Pestic. Monit. J. 9:21-29.

Veith, G.D. and V.M. Comstock (1975) Apparatus for continuously saturating water with hydrophobic organic chemicals. J. Fish. Res. Bd. Can. 32:1849-1851.

Veith, G.D., D.W. Kuehl, F.A. Puglisi, G.E. Glass, and J.G. Eaton (1977) Residues of PCBs and DDT in the western Lake Superior ecosystem. Arch. Environ. Contam. Toxicol. 5:487-499.

[illegible], Inc. (1978) Cross Annual Discharge to the Waters in 1976 - Polychlorinated Biphenyls Monitoring and

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Data Support Division, Revised Report No. 16, Task II, Contract No. 68-01-3852. Washington, D.C.: U.S. Environmental Protection Agency.

Wassermann, M., L. Tomatis, and D. Wassermann (1974) Storage maps of organochlorine compounds (OCC) in humans. In Epidemiological Deductions for Further Monitoring. CEC-EPA-WHO International Symposium-Environmental and Health, Paris 1974, Paper #115.

Webb, R.G. and A.L. McCall (1973) Quantitative PCB standards for electron capture gas chromatography. J. Chromatogr. Sci. 11:366-373.

Weiniger, D. (1978) Accumulation of PCBs by Lake Trout in Lake Michigan. Thesis. Madison, Wis.: University of Wisconsin.

White, D.H. (1976) Nationwide residues of organochlorine in starlings, 1974. Pestic. Monit. J. 10:10.

Whitmore, F.C. (1975) A Study of Pesticide Disposal in a Sewage Sludge Incinerator. Office of Solid Waste Management Programs, Final Report, Contract No. 68-01-1587, NTIS PB-253 485. Washington, D.C.: U.S. Environmental Protection Agency.

Williams, P.M. and K.J. Robertson (1975) Chlorinated hydrocarbons in sea-surface films and sub-surface waters at nearshore stations in the North Central Pacific Gyre. Fish. Res. Bull. 73:445-447.

Young, D.R. (1975) Inputs and Distributions of Chlorinated Hydrocarbons in Three Southern California Harbors. Report TM 214. El Segundo, Calif.: Southern California Coastal Water Research Project.

Young, D.R., D. McDermott-Erhlich, and T.C. Heesen (1975) PCB Limits to the Southern California Bight. El Segundo, Calif.: Southern California Coastal Water Research Project.

Young, D.R., D. McDermott-Ehrlich, and T.C. Heesen (1977) Sediments as source of DDT and PCB. Mar. Pollut. Bull. 8:254-257.

2

# POLICY ASSESSMENT

The data presented in Chapter 1 indicate the extensive environmental distribution of PCBs and the existence of several localized hot spots for which removal or cleanup action may be justified. State and federal officials have been coordinating efforts to reduce PCB levels at the hot spots and to prevent future contamination (U.S. EPA 1978a,b; Hetling et al. 1978). This chapter will attempt to determine the costs of alternative control and cleanup policies and the potential benefits to human health and the environment.

## CONCEPTS OF BENEFIT-COST ANALYSIS

Benefit-cost analysis evaluates beneficial and adverse effects of a proposed action in terms of a common monetary unit, like dollars. (For detailed discussions of benefit-cost analysis and its relationship to the underlying principles of welfare evaluation, see Prest and Turvey 1965, and Haveman and Weisbrod 1975.) Alternative actions are ranked according to net benefits, that is, benefits minus costs. Social good is [illegible] of the monetary value of net beneficial

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effects, without regard to whom the benefits accrue. The bulk of the practice of benefit-cost analysis involves the development and implementation of rules for measuring or inferring monetary values for beneficial and adverse effects where these are not directly observable.

## Economic Benefits

Monetary benefits are a measure of changes in utilities or welfare of individuals. It has been said that to economists, "Man is the measure of all (economic) things"; thus, human preferences provide the basis for defining and measuring economic benefits. Only those things that individuals use either directly or indirectly can be included in a measure of benefits. This means that some effects cannot be measured in economic terms, e.g., certain ecological effects, such as alterations in the size of species populations, or subtle changes that affect freshwater or marine trophic levels. However, where such ecological changes affect commercially exploitable resources, such as fisheries or the use of ecological systems for outdoor recreation, the effects can be expressed in economic measures of benefits.

The benefits of controlling chemicals such as PCBs can accrue to individuals and society either directly in the form of increases in the availability of goods and services (e.g., improved health or improved recreation opportunities), or indirectly through increases in production efficiency. Assigning monetary values to benefits resulting from chemical control policies involves three distinct stages, each of which must be understood if the benefits are to be measured.

Stage One Reduction in the quantities of chemicals released into the environment leads to reduction in the concentration found in various environmental reservoirs. Modeling the functional relationships between changes in chemical releases and changes in ambient chemical levels is a complex matter, yet these processes must be understood before the benefits of chemical regulation can be estimated.

Stage Two Changes in ambient chemical concentrations lead to changes in human use of the environment. [illegible] relationships between changes in [illegible] effects on human activities is an essential part of the estimation process and should be termed the measurement process to distinguish it from the process of valuation discussed in the next stage.

Stage Three The effects of changes in ambient quality have their counterpart in changes in an aggregate willingness to pay for uses of the environment. Here the economic concepts of demand and value are central to the analysis. However, the successful implementation of the valuation process of Stage 3 must often await the development of a better knowledge of the physical, ecological, and other processes governing Stages 1 and 2.

## Costs

The costs of a regulatory action are defined and measured by determining which utility or benefit opportunities are foreclosed by the action. The control of chemical contamination involves the use of scarce resources that might otherwise have been put into different activities. Costs associated with these "new" uses are called "opportunity costs" and can be measured in monetary terms by society's or the aggregate willingness to pay for those items that must be given up.

How can the opportunity cost of regulation be measured? When regulation requires producers to use real resources such as labor, capital, and materials for the installation of treatment and control equipment, special handling of materials, or development of special disposal methods, the market values of these resources can be taken as measures of opportunity cost. For example, under certain conditions wages are a measure of the marginal productivity of labor in the economy and therefore reflect the value of other goods foregone when labor is diverted from one sector of the economy to [illegible] to control chemicals.

[illegible]latory actions may also reduce the availability [illegible] services to consumers in a more direct [illegible] example, regulation may take the form of a [illegible] particular chemical or the

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production of particular goods using the chemical as an ingredient, or it may involve restrictions on allowable chemical tolerances in food (e.g., the FDA maximum PCB concentrations in fish). Such policies do not absorb resources directly, but they do impose costs in the form of losses in consumer welfare. In some cases, the cost of substituting alternative goods for those banned by regulation provides a basis for an estimate of true economic cost. A well developed economic methodology exists for making estimates of this sort.

## Other Economic Impacts

Other types of economic impact associated with regulatory decisions can be described and quantified in monetary terms, but they are not commensurate with the economic efficiency benefits and costs described earlier. The incommensurability arises because the measures of impact are usually based on changes in the dollar volume of a particular class of economic transaction rather than on changes in the quantities of goods and services available to consumers. The distinction is similar to the one often made in the benefit-cost literature between real and pecuniary effects (Haveman and Weisbrod 1975).

To illustrate the point, consider the case of a chemical plant that closes because of regulatory action, thus terminating an annual million-dollar payroll. This surely has economic impact on a region, but it is not necessarily a real cost in the sense of a reduction in available goods and services. The relationship between the measure of impact and the measure of cost depends on the mobility of the labor force and other resources and on how speedily adjustments can be made. These factors will vary widely from case to case. Where a firm or an industry reduces output because of governmental regulations, there is a cost in economic efficiency only to the extent that, for a time, resources remain unemployed or underemployed. Lost output is an appropriate measure of cost in this case. The cost of resources absorbed in relocation or other activities designed to speed the adjustment process (e.g., labor retraining programs) should be taken into account. If all workers laid off from a plant find alternative employment immediately and at the same wage, there is no real cost other than the expense of relocation. If [illegible]



of the labor force can find alternative employment, there is a cost in the form of lost output, and the lost payroll may be used as a measure of this cost.

## COST-EFFECTIVENESS ANALYSIS

A cost-effectiveness analysis can be used in circumstances where a desired level of beneficial effect has been established and where the policy has been set to achieve that level at minimum cost. A cost-effectiveness criterion ranks alternative policies in terms of the cost required to achieve the given level. Because all the policies being evaluated achieve the same level, the beneficial effect will be the same and the various options will differ only in terms of their costs. One advantage of cost-effectiveness analysis is that problems created by the noncommensurability of benefits and costs are avoided. It should be noted, however, that whereas this type of analysis can be useful in evaluating alternative options for achieving a specific goal, it cannot be used to determine the choice of the goal itself.

## EFFECTS ON HUMAN HEALTH AND THE ENVIRONMENT

As the data presented in Chapter 3 and Appendix D indicate, the major effects on human health and the environment resulting from exposure to PCBs take the form of subtle impairments rather than gross morphological or pathological changes. Acute exposure of either humans or wildlife rarely occurs, and those effects that have been observed are the result of cumulative contacts over long periods of time. The section entitled "Assessment of PCB Hazard" in Chapter 3 presents the scientific evidence relating to PCB toxicity and illustrates some adverse effects of unintentional release of the chemical to the environment.

### Effects on Human Health

Exposure of humans to PCBs can occur through a number of different routes--occupational exposure, ingestion of contaminated foods or fish, exposure to contaminated

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soil or water, and transmission from mother to child through breast feeding. The levels of contamination are generally low, except where occupational exposure or direct contact occurs. Because of the low concentrations and the subtle nature of the toxic effect, it is extremely difficult to assess precisely those modifications that may have arisen in the health of human and animal populations as a result of environmental exposure to PCBs (see Table 3.6, Chapter 3). Well-documented cases of PCB intoxication in the human population have been limited to industrial accidents involving workers directly responsible for PCB production, excessive exposure to manufactured products containing PCBs, and consumption of edible products contaminated with PCBs during processing (Matthews et al. 1978).

Some of the nonspecific effects on health that may be attributed to low-level exposure to PCBs are abnormal fatigue, abdominal pain, numbness of limbs, swelling of joints, chronic cough, menstrual irregularity, and headache. Abnormal tooth development, hyperpigmentation, and low weight in newborn children also may be complications resulting from PCB exposure. Abnormalities in blood lipids, anemia, lymphocytosis, and adrenocortical hypofunction have been recorded in a number of chronic diseases associated with PCB intoxication. In addition to dermatological abnormalities, such as acne and hyperpigmentation, there have been suggestions of increased incidence of cancer in some of the Japanese who were exposed to PCB through contamination of cooking oil (Urabe 1977, Umeda et al. 1978).

Other potential nonspecific abnormalities that may occur as a result of chronic exposure to PCBs have been suggested by controlled experiments using nonhuman primates. In addition to alterations in the menstrual cycle and births of abnormally small infants, nonhuman primates experienced a greater frequency of early abortions following low-level exposure to PCBs (see Table 3.6, Chapter 3). Infants born to mothers exposed to PCBs during gestation and lactation also show some loss of immunological competence as well as learning and behavioral deficiencies. These abnormalities persist indefinitely.

Even though a large percentage of the human population has been exposed to varying amounts of PCBs, primarily through the food chain (i.e., contaminated



fish, livestock, vegetation, and human milk), no well-documented cases of health problems associated with such environmental exposures exist. Thus, a detailed evaluation of the effects on human health in the general population is not currently available, although there are indications that even limited exposure to PCBs may produce injurious effects that will be difficult to detect even with close scrutiny.

## Effects on the Environment

Assessment of effects on the environment is even less clear-cut than analyses of problems associated with health. The data in Chapter 1 suggest that PCBs are found in many places. However, as the results in Chapter 3 indicate, the toxic effects are subtle and difficult to detect. Reduced reproductive capability is the most common result of exposure to PCBs among many species. But other morphological and functional changes are noted as well, particularly in the livers of test animals. PCBs accumulate in adipose tissue, and it has been suggested that severe effects could arise when the animal is under sufficient stress to mobilize the PCB-containing lipids. Because PCB concentrations are magnified as the compound moves through food chains (see Appendix D, Bioaccumulation), the most pronounced toxic effects are noted in the upper trophic levels. Excessive accumulations may lead to reduction in the size of populations and to consequent alterations in the composition of communities and, ultimately, of the ecosystem.

## ECONOMIC ANALYSIS OF PCB CONTROL STRATEGIES

In principle there are three categories of options for controlling or reducing potentially adverse effects of PCBs on health and the environment. They are:

- reducing the flow of additional PCBs into the environment (reducing or eliminating direct discharge of PCBs as an industrial waste product or controlling the use and disposal of materials currently in service that are contaminated with PCBs);
- controlling the exposure of populations [illegible] regulatory measures, such as limiting consumption of

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fish, discouraging the practice of breast feeding, or setting strict limits on permissible levels of PCBs in foods); and

- reducing stocks of PCBs in environmental reservoirs (dredging river sediment, cleaning or upgrading existing landfills previously used for disposal of products containing PCBs, or enhancing biodegradation rates).

It is possible to assess, if only approximately, the cost and effectiveness of specific control strategies drawn from one or all of these three categories.

## Cost Effectiveness of Control Strategies

The following specific control options have been considered for this report.

1. Governmental Regulations on Disposal of PCBs. EPA has published final regulations on the disposal of substances containing PCBs (U.S. EPA 1978a) and is currently considering revisions and additions to the regulations (U.S. EPA 1978b).

2. Dredging and Disposal of Contaminated River Sediment. The Department of Environmental Conservation in New York State has evaluated proposals to dredge sediment contaminated with PCBs from the upper Hudson River as a means of controlling further flow of PCBs into the estuary. Analyses are underway to determine the technological feasibility of effectively reducing PCB levels in various localized "hot spots" (Hetling et al. 1978).

3. Limitations of Permissible Residues in FDA Regulated Products. The FDA has implemented a temporary set of tolerances for PCB residues in several classes of food and currently has under consideration a proposal for further reductions (U.S. FDA 1977b). These limits apply to milk and dairy products (current limitation--1.5 mg/kg, proposed level--1.5 mg/kg fat basis), poultry (current limitation--5.0 mg/kg, proposed level--3.0 mg/kg fat basis), eggs (current limitation--0.5 mg/kg, proposed level--0.3 mg/kg), and fish and shellfish (current limitation--5.0 mg/kg, proposed level--2.0 mg/kg). The FDA reports that in all categories, except fish and shellfish, the observed contamination levels are well below the proposed limits. Therefore, the only



significant portion of the proposed regulation would be associated with commercial distribution of fish and shellfish.

For each option we assess effectiveness in terms of the quantities of PCBs removed from the environment and destroyed or immobilized, or as in the case of the FDA regulations, the quantity of PCBs prevented from entering the human diet. In each case, we assume perfect compliance with the regulations. We also provide estimates of the cost-per-kg affected for each option. These estimates do not include administrative costs or the costs of inspection, policing and other enforcement activities.

Controlling Use and Disposal Table 2.1 summarizes the available information on costs of existing and proposed regulations governing use and disposal of PCBs. The values were derived from two reports commissioned by EPA (Versar, Inc. 1977, Westin et al. 1978), and from additional information provided by the Agency's Office of Toxic Substances. The data upon which the costs are based are presented in Appendix B.

The first column of Table 2.1 presents the quantities of PCBs that would be prevented from reaching the environment, either through destruction or immobilization, for each policy option. The second column gives the total cost of each policy option. The third column provides the average cost of each option in dollars per kilogram of PCB controlled. If within each option there were no variation in the cost-per-kg removed under different circumstances, then this figure could be taken as representing the incremental or marginal cost of removal for that option. Table 2.1 does not address all the EPA regulations, because some data on costs are not available. Also the proposed EPA regulations would impose no special requirements on disposal of small appliance capacitors or high intensity and fluorescent lighting ballasts, although estimates indicate that nearly 81.8 x $10^6$ kg of PCB are contained in these items and are currently in service (Westin et al. 1978). The Committee does not at present have estimates of the cost of controls on these PCB materials.

The ranking of options in Table 2.1 is based on the assumption that PCB releases prevented by each regulation would have resulted in equivalent

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TABLE 2.1 Costs and Quantities Controlled for EPA Regulations on Use and Disposal

| Policy | Quantity Controlled (kg) | Control Costs[b] | Average Cost Per kg |
|---|---|---|---|
| A. Incinerate fluids from PCB transformers (required) | 123,000,000 | $ 75,000,000 | $ .61 |
| B Shred and incinerate high voltage capacitors (required) | 90,900,000 | $350,000,000 | $ 3.85 |
| C Shred and incinerate low voltage capacitors (require) | 31,800,000 | $143,000,000 | $ 4.49 |
| D. Flush and drain transformers after A. (required) | 12,300,000 | $105,000,000 | $ 8.50 |
| E. Place transformers in chemical waste landfill (required)[a] | 1,360,000 | $ 45,100,000 | $ 33.16 |
| F. Replace PCB-containing electromagnets and incinerate fluid (proposed) | 100,000-150,000 | $ 2,800,000-3,400,000 | $ 18.67-34.00 |
| G Replace PCB-containing [illegible] in mining equipment (proposed) | | | |
| (1) Loaders | 13,000 | $ 2,000,000 | $ 155.00 |
| (2) Continuous Miners | 1,136 | $ 640,000-2,240,000 | $ 562.00-1,[illegible] |
| H. Drain and incinerate mineral oil transformer fluids if contaminated by more than 50 ppm PCBs (proposed)[c] | 10,900 | $612,000,000-769,000,000 | $70,000 00-[illegible] |
| I. Flush and replace contaminated hydraulic fluids in die casting (proposed) | data incomplete[d] | data incomplete[d] | $133,000 |

[a]It is assumed that transformers will be disposed of in chemical waste landfills rather than by incineration.
[b]Options A through E will be undertaken as PCB-containing equipment is taken out of service. For example some PCB-containing transformers may remain in service for up to forty years. The control costs here will be incurred over the useful life of the equipment in question. These costs have not been discounted to the present. Discounting would reduce computed control cost, and would have the greatest effect on control options affecting transformers and transformer fluids. Discounting would have relatively little effect on options F through I, since these options would have to be undertaken shortly after the effective date of the proposed regulations.
[c]Fluids with greater than 500 ppm are covered by Policy A.
[d]Incremental costs are estimated from average costs per die-casting machine. Aggregate costs and quantity controlled are not known because of lack of data on the number of affected machines.

SOURCE. Modified from Versar (1977), Westin et al. (1978).

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environmental and health risks. However, some forms of release could be potentially more threatening than others, especially to health. For example, continued use of PCBs in mining equipment (option G) could subject operators to significant risk of severe contamination. Even with this qualification, the data of Table 2.1 indicate a classic pattern of increasing marginal costs of control, rising to extremely high levels for the last increments of material controlled. The options listed would, if implemented, control a total of 260 x $10^4$ kg of PCBs. Virtually all control is accomplished with options A through E at average costs of $33/kg or less. The remaining options listed (F through I) would contribute less than 0.1 percent to further control PCBs but would add significantly to the total cost; marginal costs of the last option rise to more than $100,000/kg of PCBs.

Dredging Hudson River Sediment The New York State Department of Environmental Conservation has examined several options for dredging contaminated sediment in the upper Hudson River and encapsulating it in special landfill sites. Information on these options is presented in Table 2.2. Maintenance dredging refers to routine maintenance of river channels. Although some PCBs would be removed by this dredging, the only portion of the cost that can be properly attributed to PCB control is that associated with encapsulation of the dredge spoils. During the summer of 1978, the state removed portions of the remnant deposits and stabilized the remainder. This appears as option B in Table 2.2. Each of the remaining policy options (C through E) involves more comprehensive and extensive dredging operations than option A, and thus greater costs.

Incremental costs and quantities of PCBs removed have been calculated from data presented by the New York State Department of Environmental Conservation (Hetling et al. 1978). These control costs are computed on the basis of encapsulation of the dredge spoils rather than destructive incineration of sediment contaminated with PCBs. If incineration were employed, costs would be higher. The data show increasing incremental costs as larger quantities of PCBs are removed. These dredging costs are higher than the least costly use and disposal regulations of EPA (options A through E, Table 2.1), but well below the most costly EPA proposed regulations. The cost-effectiveness measures for the Hudson River



2.2 Costs and Quantities Controlled for Dredging in River Sediments

| Policy | Incremental Quantity Controlled (kg)[a] | Incremental Control Costs | Incremental Cost Per kg |
|---|---|---|---|
| A. Maintenance dredging | 23,100 | $ 2,500,000 | $ 110 |
| B. Partial removal of remnants and stabilization of remainder (performed during summer of 1978)[b] | 7,700 | $ 500,000 | $ 65 |
| C. Remove remaining stabilized remnant deposits | 15,100 | $ 3,300,000 | $ 219 |
| D Hot spot dredging | 77,000 | $ 22,400,000 | $ 291 |
| E. Remove all river sediments | 55,600 | $175,300,000 | $3 150 |
| Cumulative Total | 178,500 | $204,000,000 | |

[a]Quantity controlled is based on revised data received from Toffelmire and Quinn, New York State Department of Conservation, personal communication, 1978.
[b]Remnant refers to sediment exposed by removal of the Fort Edward Dam in the summer of 1973.

SOURCE: Modified from New York State Department of Environmental Conservation (1978).

dredging options are based on the assumption that dredging and disposal of spoils do not of themselves pose significant environmental hazards (Hetling et al. 1978).

Limiting PCB Content of Fish for Human Consumption If a limit on the amount of PCBs in fish were imposed, the economic productivity of fisheries would be reduced. Fish with PCB content exceeding the limit would have to be destroyed, and it is likely that fishing for critical species in contaminated areas would cease.

The cost of imposing a maximum limit on allowable PCBs in fish would be the value of the reduction in the economic productivity of fisheries. Precise measurement of this loss is difficult in practice. For simplicity, it can be assumed that the effect of the limitation on the total output of fisheries would be so small that there would be no change in the price of fish. The FDA (1977a) economic analysis is based on this assumption, which seems plausible. At one extreme, the

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assumption would be that fishing activity continues at the same level, but that catches contaminated by PCBs in excess of the limit would be confiscated and destroyed. The resulting reduction in sales revenue at the dock (landed value) can be taken as a measure of the economic cost of the limitation. It has been estimated that the proposed 2 mg/kg temporary limit for fish would result in a loss in landed value of $1 million for marine fisheries and $7 million for freshwater fisheries, the latter located primarily in the Great Lakes region (U.S. FDA 1977a). However, landed values are likely to overestimate real net losses because at least some of the capital and labor, whose productivity is impaired, would be deployed elsewhere.

The assumption at the other extreme would be that particular fishing activities are decreased, but that capital and labor in this fishery would be able to move without productivity losses to other activities, such as fishing for other species or moving to other waters. In this case, the loss in output of fisheries would be exactly offset by the increase in output of other areas as resources would be diverted from one use to another.

Neither extreme is likely to be an accurate representation of the real world. There will be changes in levels of activity and movements of resources in response to any limitation on PCBs. But capital and labor in fisheries are specialized resources and not perfectly mobile. Shifts in resources will occur, but only with a time lag and some reduction in productivity. Precise estimates of cost associated with limiting PCB concentrations in fish would require better data and more careful analysis than we are able to provide. Therefore, for purposes of this study, $8 million is taken as an upper-bound estimate of the economic cost to commercial fisheries resulting from the FDA limit on PCB levels in fish.

The economic impact can be translated into a crude estimate of the cost-per-kg of PCBs avoided in the human diet as a result of the regulation. If, as a first approximation, it is assumed that the average landed price of PCB-contaminated fish is $0.44/kg, an $8 million loss in landed value is equivalent to a reduction of $18.2 \times 10^6$ kg in the catch. Since the FDA proposal would lower the allowable PCB content from 5 mg/kg to 2 mg/kg, we assume that the average concentration in affected fish would be 3.5 mg/kg. Thus, the limitation would reduce total PCB input to the



[illegible] 64 kg. Thus $135,000 would be the [illegible] estimate of the cost per-kg eliminated from [illegible] fish. FDA also estimated that the incremental [illegible] in landed value from lowering the tolerance from 2 mg/kg to 1 mg/kg would be $10 million. By similar calculations, the cost for avoiding an additional 34 kg of PCBs in the human diet would be $293,000/kg.

The FDA limitation applies only to fish entering interstate commerce. If individual states also imposed effective limitations on PCB content of fish caught and consumed by sportsmen, the total cost of controlling the intake of PCBs from fish would be higher but not necessarily the cost-per-kg.

The cost of regulating sports fisheries could take two forms. If the regulation applied only to consumption of fish and not to the fishing activity itself, the utility and value of time spent fishing would be reduced. Even with no change in total sports fishing activity, this reduction in value would constitute a regulatory cost. To the extent that recreational fishing activities were reduced, there would be a further loss equal to their value per day multiplied by the number of days of lost activity. Quantitative estimates of the magnitude of these costs are not available. Total spending on sports fishing activities (for such items as food and lodging during trips, gear, boats, or bait) is not a measure of the potential economic cost of a sports fishing ban, because reduced spending on fishing probably would be offset by increases in other categories of expenditure. Productive resources would be reallocated, and if resources were perfectly mobile, there would be no net loss in the output of the national economy in goods and services.

## Cost-Effectiveness of Alternatives

The preceding sections provide estimates of cost-effectiveness for three different categories of control policy, using different measures of effectiveness and allowing for a ranking of options within each control category. The cost-effectiveness measures would be more useful, however, if they could be made commensurable across categories. If a kilogram of PCBs released through unregulated disposal were equivalent in environmental impact to a kilogram of chemical present

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in river sediment, the cost effectiveness of measures for destroying PCBs and for dredging would be commensurable. If that assumption were to hold, the first six EPA regulations (options A through F) in Table 2.1 would be more cost effective than any of the dredging options. But if EPA were to approve all the proposed regulations (options F though 1, Table 2.1), including those dealing with contaminated transformers and die-casting fluids, removal of all Hudson River sediment also would be economically justified. Dredging activities would produce equivalent results in terms of PCB control at costs considerably below those expected of the most expensive EPA regulation.

However, the control of PCBs through dredging options may not be fully equivalent to regulating PCB use and disposal. The New York State Department of Conservation estimates that about one-third of the mass of PCBs in sediment in the Upper Hudaon is subject to erosion by river currents (Hetling et al. 1978). Given this estimate and assuming that dredging could not be selectively applied to the most mobile sediment containing PCBs, it would be necessary to remove perhaps 3 kg of sediment for an effect equivalent to destroying 1 kg of PCBs presently in service. To make the costa of dredging comparable to values in Table 2.1, incremental measures of cost for complete dredging would be increased by perhaps a factor of 3. Removal of remnanta and hot spot dredging are more selective operationa, and measures of cost effectiveness for these options are probably closely comparable to those for regulation of use and disposal.

To make the measures of cost-effectiveness for dredging and regulations of disposal commensurable with the food tolerance proposals, it is necessary to estimate what fraction of PCBs released to the environment would eventually accumulate in fish and shellfish destined for human consumption. The following calculation was used to estimate this fraction. In the Great Lakes, the total quantity of PCBs in sediment is on the order of $3.5 \times 10^8$ kg (Table 1.16; St. Lawrence drainage, upper bound estimate). Fishing removes about $2.7 \times 10^7$ kg/yr (about $9.0 \times 10^7$ kg contaminated at typical PCB levels of about 3 mg/kg). Assuming that such levels will be maintained for 30 years before PCBs are buried or dissipated, about $8 \times 10^9$ kg of PCBs would be incorporated in fish and consumed by humans. Hence, eliminating 1 kg of PCBs from the human diet would



[illegible] removal of about $4 \times 10^7$ kg from the [illegible]nt. The figure could be reduced if the [illegible]nt deposits most likely to contribute to resid[illegible] fish could be removed selectively, as discussed abo[illegible] for the Hudson River.

On the basis of these admittedly rough calculations, a conservative estimate is made that 0.25 percent of all PCBs released to the environment enter the diet. This figure was derived from data on the Great Lakes, a relatively closed system; similar calculations for the Hudson River biota indicates roughly 0.30 percent of PCBs released into the environment might enter the diet (Hetling et al. 1978). It may be expected that in more open areas, such as estuaries and oceans, a smaller fraction of PCBs would enter the human diet. Thus, the measures of cost effectiveness for the options of regulating PCB use and dredging can be made commensurate with the FDA food tolerance proposals by raising them by 400 or more. In other words, removing 1 kg from the human diet at a cost of $125,000/kg is equivalent to removing 400 kg or more from the environment at a cost of $312/kg. (It should be noted that this adjustment implies that there are no beneficial effects of dredging and regulation other than impacts on the human diet.)

On the basis of these computations, either of the FDA food tolerancea (2 mg/kg or 1 mg/kg) is more cost effective in protecting humans against exposure to PCBs in the diet than the mora costly of the EPA proposed regulationa for use and disposal. However, hot-pot dredging may be a preferred option both because of slightly lower estimatad costs-per-kg removed and because of the possibility of preventing other damages in addition to those to human health. This is not to say that the FDA regulations should be approved or that incinerating contaminated transformer fluids is too costly for policy consideration. Cost-effectiveness data do not constituta an adequate basis for making policy decisions about control options. Although a ranking of options by effectiveness per dollar of resource committed is possible, these data do not tell us how far out and up along the incremental cost curve we should go. To make such determinations, further information is needed on the beneficial effects of controlling PCBs

[illegible] Costs. It is desirable to estimate the [illegible] to allow for comparison with the costs

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of reducing releases of PCBs (Table 2.1) or of reclaiming PCBs already released (Table 2.2). Major categories of benefits derived from control options include: reduced risks to human health, enhanced economic benefits to commerical and recreational fishing activities, and reduction in pollution damage to the environment. We are not able to make any quantitative assessments of the last two catagories, and even a tentative estimate of benefits to human health is incomplete and subject to uncertainty.

Many of the risks to human health resulting from PCB exposure are perceived as subtle physical and behavioral changes. Recently it has also been suggested that exposure to PCBs poses a threat of cancer to humans (Umeda et al. 1978, Urabe 1977). Numerical estimates are available for the latter type of risk only.

The National Academy of Sciences' Safe Drinking Water Committee (National Research Council 1977) estimated that the lifetime carcinogenic risk to a person ingesting p micrograms of PCBs daily (in drinking water) would be approximately $3 \times 10^{4}$ p. The estimate was derived by linear extrapolation of results from a bioassay for carcinogenesis using rats exposed to Aroclor 1260 at 100 mg/kg in the diet. This estimate has a high degree of uncertainty, for three reasons.

1. The estimate was derived by extrapolating over an unusually wide range of exposures (3 orders of magnitude even for highly exposed persons);

2. No scaling factors were incorporated for extrapolation from potential risks of cancer for rats exposed to low doses to expected risks of cancer for humans exposed to corresponding levels of PCBs; and

3. The estimate was derived only for pure grade Arocior 1260, without detectable impurities; environmental residues of technical grade PCBs may be more or less hazardous. (See the discussion of PCB impurities in Appendix D.)

The daily ingestion of p micrograms of PCBs would result in a lifetime intake of about $2.5 \times 10^{4}$ p micrograms. Using the above relationship between exposure and risk, the incrementai cancer risk per microgram of PCBs is approximately $1.2 \times 10^{-9}$. If the exposure-risk relationship is assumed to be linear (at least at the margin), the estimate of individual risk may be extended to a population. Neither the



distribution of exposures within the population nor the distribution with age need be considered. Hence, on average, one additional cancer case would be expected to result from an incremental intake of $8 \times 10^{8}$ µg of PCBs, however this intake may be distributed among the U.S. population.

We assume that the economic benefits to society of reducing a general risk of cancer range from $100,000 to $1,000,000 per cancer death avoided. (The basis for this is discussed in Appendix C.) Given this assumption, the benefits assigned to the reduction of PCB intake are in the range of $12/g to $120/g ($12 \times 10^{3}$/kg to $120 \times 10^{3}$/kg).

To reduce human intake by 1 g, a larger quantity of PCBs must be removed from the environment or withheld from discharge. As discussed above, roughly 400 g or more of PCBs must be withheld to reduce dietary intake by 1 g. Thus, the marginal benefits to be gained from removing PCBs from the environment would be in the range of $30/kg to $300/kg.

In addition to the major qualifications stated above, several uncertainties about this estimate should be emphasized.

- The estimate refers only to carcinogenic risks, although other types of risks to health and other forms of benefits may be more important (see Hazard Evaluation, Appendix D for PCB toxicity).
- The costs of PCB removal are compared with putative benefits that might not be realized for 30 to 80 years. If it is considered appropriate to discount benefits over such lengthy periods (2 to 3 generations), the amount discounted might be in the order of a factor of 100 or more.
- The assumption is made that PCBs released to the environment now will continue to accumulate in fish for approximately 30 years (potential residence time of PCBs in sediment; see Chapter 1). If this estimate is too long, benefits resulting from control options would be correspondingly overestimated. If the estimate is too short, the calculated benefits would be determined primarily by the discount rate and not by persistence of the PCBs.

For these reasons, the benefits of controlling PCBs may be overestimated or underestimated by a factor of 100 or more. Accordingly, although the estimates are

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given as $30/kg to $300/kg for benefits derived from removal of PCBs, it is not possible to state with confidence that these costs are not much higher or much lower.

Considering the great uncertainties in these estimates, health benefits of $30/kg to $300/kg of PCBs removed would economically justify only the least costly EPA disposal regulations on strict benefit-cost grounds (Table 2.1). The FDA proposed food tolerance limit may also be justified at this benefit level because the cost involved, adjusted for the ratio of environmental to dietary levels, is only $312/kg. If we recall that the benefit estimate is only for one category of health effects and that it may underestimate expected benefits by two orders of magnitude, economic justification for incurring incremental costs of $30,000/kg may be possible on the basis of potential benefits. Several of the EPA proposed regulations and the selective Hudson River dredging options also may be justified since their costs lie within the estimated range of benefits. However, given the range of uncertainty, we cannot rule out the possibility that costs may exceed benefits for all control options considered.

When regulatory decisions such as those assessed here must be made in the face of substantial uncertainty, the final choices involve value judgments and implicit trade-offs by the responsible public officials (see National Research Council 1975 for a more comprehensive discussion of this point). But one additional set of calculations can be made in an effort to illustrate some of the value implications of alternative choices.

In 1968 more than 1,000 people in Japan were exposed to high dietary levels of PCBs from the accidental contamination of rice oil in what has become known as the Yusho incident. The average cumulative dose was estimated to be 1 g/person. (See Chapter 3 for a description of this incident.) The cost of preventing 1 kg of dietary intake (sufficient to provide a Yusho dose to 1,000 people) by lowering PCB limits in fish would be $125,000, or $125 per person.

If this were judged as a reasonable price for preventing the magnitude of effects observed in the Yusho victims, the FDA policy as well as maintenance dredging and the other lower cost EPA control options could be economically justified. But by this line of reasoning, a justification for the highest cost EPA disposal regulations would require a willingness to pay $3,200 per person to avoid PCB exposure to the Yusho dose. However, three qualifications should be made.

1. The PCB levels observed in fish would not provide the Yusho dose to small numbers of people at currently typical levels of fish consumption. Rather larger numbers of people would receive only 1 to 2 percent of the Yusho dose. Thus, the above comparison would be valid only if the dose-damage relationship were linear throughout this range.

2. The Kanechlor involved in the Yusho incident was contaminated with unusually high levels of polychlorodibenzofurans, these may have been responsible for some portion of the observed effects on health (Miyata et al. 1977).

3. Accumulation of the 1 g dose by the Yusho victims occurred over a shorter period than would be typical for a person whose diet contained fish contaminated with PCBs.

Tables 2.1 and 2.2 indicate high average or incremental costs with some options for recovering PCBs already discharged or capturing those currently in use to prevent their eventual discharge to the environment. This raises the question of whether it might have been less costly for society to have avoided these problems by preventing the initial production and use of PCBs.

The cost of such an initial prospective ban is measured by the additional cost of achieving equivalent product performance through redesigning equipment and substituting other materials. For those examples where redesign and substitutions are not feasible, we must assess cost factors related to reduced product performance or the maintenance of safety. We cannot accurately estimate these costs, but we can provide an illustrative calculation.

The range of the current cost estimate for banning the use of PCBs in transformers and capacitors is $14.7 million to $32.6 million per year (Westin et al. 1978). This represents primarily the costs of substituting materials, redesigning products, and increasing risks of [illegible] Using the upper limit, we assume for purposes of discussion that the total cost for all segments of the [illegible] PCBs is 4 times this amount. These are [illegible] It is interesting to note the discounted

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present value of a time stream of costs evaluated, hypothetically, at the time of the first introduction of PCBs into commercial use. If we assume infinite time and a discount rate of 8 percent, the current value of costs to ban PCBs would have been $1.6 billion. Cumulative production of PCBs has amounted to about 650 x $10^6$ kg since 1930 (Westin et al. 1978). Thus the cost per kilogram of avoiding that production and its subsequent damage to the environment would have been about $2/kg. This is at the low end of the range of cost-effectiveness measures.

The calculation suggests that prevention can be more cost effective than control and cleaning up measures. But this should not be interpreted as justifying a decision not to have allowed PCB production to begin for two reasons. First, the costs of such a ban would have to be compared with its anticipated benefits. Second, it is not appropriate to think of the choices as being limited to the two alternatives of no production versus production and use in the current patterns. Instead one should attempt to perform separate benefit-cost calculations for each type of use. Use of PCBs in closed systems with strict controls to prevent distribution of PCBs in the environment might be feasible in benefit-cost terms, especially for those uses where PCBs are particularly valuable. As Table 2.1 shows, costs of collection and destruction for closed system uses are relatively low. The high control costs are associated with recovery of those PCBs that have contaminated other materials or escaped to the environment.

## CONCLUSIONS

1. Substantial differences exist in the costs of various options for the control of PCBs. Among the alternatives analyzed--which included EPA proposed regulations on disposal and use, options for dredging and disposal of contaminated river sediment, and FDA regulations limiting permissible residues in fish destined for human consumption--average costs varied by as much as 5 orders of magnitude.

2. Existing EPA regulations on disposal and use are projected to result in the control of 260 x $10^6$ kg of PCBs with average costs of up to $33/kg of PCBs controlled. Proposed EPA regulations would increase the



quantity of PCBs controlled by about 0.1 percent [illegible] those controlled by existing regulations) with average costs becoming as high as $133,000/kg.

3. A case study of the Hudson River indicated that various alternatives for dredging sediment could remove a total of about 178 x $10^3$ kg of PCBs from potentially mobile river sediment at costs ranging from $110/kg to $3,150/kg.

4. It is estimated that up to 0.25 percent of the total PCBs released to the environment enter the human diet; thus at least 400 kg of PCBs would have to be removed from the mobile environmental reservoir in order to achieve a reduction of 1 kg of PCBs in food. This figure (400 kg) is derived using data from the Great Lakes, a closed system; it may be too low for more open systems. FDA limitations on PCB concentrations in commercially distributed fish could reduce PCB input to the American diet by about 64 kg at a cost of less than $125,000/kg. (A reduction of an additional 34 kg would cost less than $293,000/kg.)

5. Based on the figures used in the analysis in this report, FDA regulation of PCB limits in fish destined for human consumption appears a more cost-effective method of controlling dietary intake and human health effects than some of the most costly EPA proposed regulations on use and disposal. Some dredging options may be preferable to either EPA or FDA regulations.

6. It is difficult to assess the benefits of controlling exposure to PCBs with currently available information. The analysis in this report estimates the benefits as $30/kg to $300/kg of PCBs removed from the environment or withheld from discharge, but notes that even these figures could be high or low by a factor of 100 or more.

## RECOMMENDATIONS

1. Policy makers should use available data to calculate the incremental or average costs of PCB control options in conducting economic analyses of proposed regulations.

2. As greater control of PCBs is sought, regulatory efforts should be concentrated on those options which are most cost effective, that is those with the lowest average or incremental costs per comparable effect or benefit.

3. Economic analyses should include cost data on all [illegible] not just those under a single agency's

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regulatory authority. The analysis could then show--as does the analysis presented in this report--that some options beyond a particular agency's authority may be substantially more cost effective than some that lie within the agency's jurisdiction. A comprehensive, coordinated government policy toward control of pollutants should reflect such findings.

## REFERENCES


Haveman, R.H. and B.A. Weisbrod (1975) The concept of benefits in cost-benefit analysis: With emphasis on water pollution control activities. Pages 37-66, Cost-Benefit Analysis and Water Pollution Policy. Edited by Henry M. Peskin and Eugene P. Seskin. Washington, D.C.: Urban Institute.

Hetling, L., E. Horn, and T.J. Tofflemire (1978) Summary of Hudson River PCB Study Results. Technical Paper No. 51. Albany, N.Y.: New York State Department of Environmental Conservation.

Matthews, H., G. Fries, A. Gardner, L. Garthoff, J. Goldstein, Y. Ku, and J. Moore (1978) Metabolism and biochemical toxicity of PCBs and PBBs. Environ. Health Perspect. 24:147-155.

Miyata, H., T. Kashimoto, and N. Kunita (1977) Detection and determination of polychlorodibenzofurans in normal human tissues and Kanemi rice oils caused 'Kanemi Yusho.' J. Food Hyg. Soc. 18(3):260-265.

National Research Council (1975) Decision Making for Regulating Chemicals in the Environment. Committee on Principles of Decision Making for Regulating Chemicals in the Environment, Environmental Studies Board, Commission on Natural Resources. Washington, D.C.: National Academy of Sciences.

National Research Council (1977) Drinking Water and Health. Committee on Safe Drinking Water, Assembly of Life Sciences. Washington, D.C.: National Academy of Sciences.

New York State Department of Environmental Conservation (1978) Toxic Substances Impacting on Fish and Wildlife. Division of Fish and Wildlife Monthly Report 12, March 20, Albany, N.Y.: New York State Department of Environmental Conservation.

Prest, A.R. and R. Turvey (1965) Cost-benefit analysis: A survey. Econ. J. 75(4):683-735.

[illegible], M. [illegible], and K. Yatsuki (197[illegible] Some problems on health effects of polychlorinated biphenyls. Toxicol. Environ. Saf. 1:107-[illegible].

Urabe, H. (1977) Foreword, the Fourth Report of the Study on Yusho and PCB. Fukuoka Acta Medica 65 [illegible].

U.S. Environmental Protection Agency (1978a) Polychlorinated biphenyls (PCBs): Disposal and marking. Federal Register 43(34):7149-7164.

U.S. Environmental Protection Agency (1978b) Polychlorinated biphenyls (PCBs): Manufacturing, processing, distribution in commerce, and use bans. Federal Register 43(110):24802-24818.

U.S. Food and Drug Administration (1977a) Economic Impact Assessment for Proposed Reduction of Temporary Tolerances for Polychlorinated Biphenyls in Food. (Unpublished paper)

U.S. Food and Drug Administration (1977b) Polychlorinated biphenyls (PCBs): Unavoidable contaminants in food and food packaging materials; reduction of temporary tolerances. Federal Register 42(63):17487-17494.

Versar, Inc. (1977) Microeconomic Impacts of the Proposed Marking and Disposal Regulations for PCBs. NTIS Report No. PB-267 813/2WP. Springfield, Va.: National Technical Information Service.

Westin, R.A., L. Fourt, D. Berkey, and B. Woodcock (1978) Microeconomic Impacts of the Proposed PCB Ban Regulation." National Technical Information Service, NTIS Report No. PB 281-881/3WP. Springfield, Va.: Versar, Inc.


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[illegible] Uses of Polychlorinated
[illegible], 1930

| |
|---|
| Protective coatings (Aroclor resins in lacquers) |
| In cement die-casting (to replace natural waxes) |
| Heat-transfer agents |

SOURCE Taken from Penning (1930).

data depended on anticipated exposure or use patterns (e.g., teratogenic, reproduction, or some chronic testing), decisions about whether to include these data in the assessment were made based upon information regarding the original uses of PCBs (Table 3.1), rather than upon our knowledge of the ultimate exposure and environmental distribution. By 1930 commercial PCBs were being considered for use in protective coatings, such as varnishes and lacquers, in investment die-castings, and as heat-transfer agents (Penning 1930). Testing was under way to determine suitability for other uses, such as waterproofing, textile flame-retardant, and artificial leather finishing, but full-scale expansion of PCB use did not occur until much later.

## GENERAL TESTING REQUIREMENTS FOR EVALUATING TOXIC SUBSTANCES

A comparison of the proposed EPA testing guidelines was made to identify those elements common among drafts. Test protocols were similar for most required data. TSCA guidelines followed FIFRA requirements rather closely. The major differences between the guidelines involved the prescribed sequences of hazard evaluation tests and criteria for the choice of particular tests. Summaries of the guidelines data requirements are presented below followed by comments of the Committee wherever appropriate.

### Chemical Identity

Product Identity and Disclosure of Ingredients. Complete information on the components of the product is required. It should include, to the extent possible and relevant, ingredients of manufacture and final product [illegible]nds that can reasonably be identified. The [illegible]ed information includes:



# GUIDELINES FOR EVALUATING THE POTENTIAL TOXICITY OF CHEMICALS

After passage of the Toxic Substances Control Act, a concerted effort was made by various governmental agencies and nongovernmental groups to develop testing guidelines. The guidelines describe the type of data required for an agency evaluation of the potential adverse effects resulting from the manufacture and use of new chemicals. The purpose of this chapter is to determine whether the data required by the guidelines are sufficient for and relevant to making an adequate assessment of potential hazards to human health and the environment.

Draft guidelines were examined that had been prepared by three different sources: the Office of Pesticide Programs (FIFRA, U.S. EPA 1978a), the Office of Toxic Substances (TSCA, U.S. EPA 1978b), and the Conservation Foundation (1978). Basic data requirements were consistent among these drafts; variations generally depended on the intended use of the chemical (e.g., pesticide versus commercial chemical). A summary of the governmental requirements was prepared, and PCB data were compiled accordingly (Appendix D).

The evaluation was performed as though PCBs were new materials. Thus, when the requirements for particular

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1. the chemical name from the Chemical Abstracts 1972 to 1976 Index of Nomenclature, or other well-defined names;
2. the Chemical Abstracts (CAS) registry number;
3. the product name, trade name, and common name (if established);
4. the experimental or internal code number;
5. the molecular (empirical) formula and the weight or the molecular weight range for each ingredient; and
6. the structural formula for each organic ingredient.

Committee's Comments: It is uncertain whether a firm order of presentation will be established for this information. However, because many substances submitted for approval will be mixtures such as PCBs, the following sequence may be most appropriate for identifying a product:

1. the product name, trade name, and common name;
2. the experimental or internal code number;
3. for each ingredient:

- the chemical name from the Chemical Abstracts Index of Nomenclature;
- the Chemical Abstracts (CAS) registry number;
- the molecular formula and structural formula (for organic, organo-inorganic, and inorganic compounds whenever appropriate); and
- the molecular weight or weight range.

Description of Manufacturing Process Guidelines require that application for approval of manufacture or new use, or both, include information regarding the basic manufacturing process as well as information on the composition of the starting and intermediate materials.

Formation of Unintentional Ingredients An indication of whether unintentional ingredients are formed is required along with information on intentionally added ingredients; identification of components that might reasonably be detected as present in amounts of 0.01 percent or more of product weight are to be included.

Committee's Comments: A variety of chemical contaminants is present in manufactured products in amounts less than 0.01 percent of product weight

... For example, as shown in Table 3.2, the ... chlorodibenzofurans (CDFs) reported in ... final samples of commercial PCB products is substantially below this limit, and low-level exposures to CDFs have been reported to produce toxic effects in animals (Zitko et al. 1973, Nisbet 1976, Oishi and Hiraga 1978, Oishi et al. 1978, International Agency for Research on Cancer/IARC, in press). Thus a listing of all identifiable "contaminants" or unintentionally added ingredients should be required to make possible a proper evaluation of the potential toxicity of new compounds. For those ingredients known to be or suspected of being hazardous, an expected maximum limit within the product should be stated, even if this limit is less than 100 ug/g of product weight.

Declaration and Certification of Ingredient Limits Estimates of upper and lower limits should be established for each ingredient as well as for any impurities known to be present. The limits should be presented as percentages of the total product.

Committee's Comments: In addition, quantities should be established for impurities that may be expected to arise during normal use of the product and under specified disposal conditions.

TABLE 3.2 Amounts of Chlorodibenzofurans Detected in Commercial PCB Mixtures

| | | ug/g |
|---|---|---|
| Aroclor | 1248 | 2.0 |
| | 1254 | 1.7 |
| | 1260 | 1.0 |
| | 1016 | ND (less than 0.001) |
| Clophen | A-60 | 8.4 |
| Phenoclor | DP-6 | 13.6 |
| Kanechlor | 300 | 1.0 |
| | 400 | 10.0 |
| | 500 | 4.0 |
| | 600 | 5.0 |

ND indicates not detected

SOURCE Modified from Nisbet (1976)

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Analytical Methods General requirements include descriptions of analytical methods (or references to commonly recognized analytical techniques in the published scientific literature) for detecting and measuring the concentration of ingredients or impurities. The techniques should include extraction procedures, confirmatory methods, and validation procedures, including background (or blank) values and recovery values; and they should be capable of detecting and measuring ingredients or impurities in quantities of 0.01 percent or more of total product weight.

> Committee's Comments: In addition to methods for product analysis, methods for detecting and determining ingredient or impurity levels in environmental media should be required for subsequent use in monitoring programs. Detection and measuring capabilities must be in the range of toxicologically significant quantities, if they are known, rather than at a limit of 0.01 percent of product weight.

Physical and Chemical Data

Data are required on the physical and chemical properties of the technical grade of each active ingredient present in the product. These must include color, odor, melting point, solubility, stability, octanol/water partition coefficient, physical state, density or specific gravity, boiling point, vapor pressure, pH, storage stability, flammability, oxidizing or reducing action, explosiveness, miscibility, viscosity, corrosion characteristics, and dielectric breakdown voltage. The FIFRA guidelines apparently require data on all these properties for each new chemical. TSCA requirements are more flexible, depending on use of the chemical, testing condition, and extent of measurable parameters.

> Committee's Comments: Required data should be relevant to assessing the hazard potential of the substance. They should be presented for known impurities as well as active ingredients. In the context of assessment of the chemical's environmental fate, the reactions of interest are those that determine the stability of a compound

[illegible] conditions (0-40C, normal [illegible] pressure, pH 4 to 10, presence of water, [illegible], ozone, clays or other sediments and minerals, or sunlight).

The stability assessment may be qualitative, based on a comparison of physical and chemical properties of the substance with those of other chemicals whose environmental properties and effects are known, or the analysis may be quantitative, based on models and quantitative structure-activity relationships.

Mobility of the substance in the environment can be judged qualitatively from its physical properties, such as water solubility, boiling point or vapor pressure, or octanol/water partition coefficient. Quantitative estimates may be possible also, based on physical and chemical properties as well as additional data, such as the volatilization rate constant/re-aeration rate constant ratio, and partition coefficients for natural sediments (Smith et al. 1977).

A variety of reactions including oxidation, reduction, hydrolysis, elimination or substitution may affect the substance in the environment. The science of reactions and reactivities is still largely in the empirical stage, but it is generally possible to predict appropriate pathways by considering functional groups present in the molecule. When reaction rates have been measured for suitable model compounds, correlations of reactivity based on electronic and steric factors can be used to estimate reactivity values. Although physical-organic chemistry provides tools for making predictions, more data on model compounds are needed to increase the reliability of the predictions.

A more complex situation occurs with photochemical reactions. Here again, photochemists can generally predict from the functional groups present in a molecule and from the visible and ultraviolet spectrum (which is generally also predictable from the structures), whether the molecule will absorb actinic light and therefore be photoreactive. On the other hand, there are no predictive correlations regarding quantum yield, when several reaction paths are available to an excited molecule, there is little or no ability to [illegible] predictions regarding the relative importance

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of the various reaction channels. Whereas many photochemical reactions have quantum yields less than unity (and most are accompanied by considerable energy wastage, so that a large fraction of the light absorbed is simply degraded to thermal energy), there are some that produce free radicals and may thus involve light-induced chain reactions. Chain lengths in such reactions are complicated functions of initiation, propagation, and termination rates and therefore, are not predictable with any degree of confidence in uncontrolled environments. For these reasons it generally will be necessary to do experimental work on the assessment of persistence and the hazards posed by new chemicals in the environment.

## Environmental Data

The proposed guidelines require a variety of data with which to evaluate the potential environmental fate of a new chemical. Both FIFRA and TSCA require the types of tests described below; differences between the proposed guidelines are noted wherever appropriate.

Chemical Degradation Degradation studies are required to determine the rate of decomposition and to identify chemical residues that may adversely affect nontarget organisms. Oxidation and hydrolysis tests noting pH- and temperature-dependent properties must be made in both soil and aquatic environments. The TSCA requirements specify that only acid- or base-catalyzed reactions need to be considered. Atmospheric oxidation tests are to be performed only on organic chemicals with vapor pressures greater than 10 mm Hg at 30°C. Oxidation tests in aqueous solutions are required for water-soluble chemicals (solubility $\geq$ 1 mg/l at 20°C).

Photolysis should indicate intensity and wavelength of radiation. Data must include a material balance, half-life estimate, and the identification of each photoproduct present in amounts greater than 10 percent of the product weight. TSCA exempts from photochemical degradation testing those chemicals that have extinction coefficients of less than 0.1 $M^{-1}cm^{-1}$ at wavelengths greater than 290 nm.



Committee's Comments: Degradation studies should be [illegible] effects of the chemical on the total [illegible]. Data on hydrolysis, oxidation, reduction, and photodegradation should be presented as applicable. Qualitative predictions of potential degradation and toxicity of new chemicals are possible through (1) identification of functional groups in the molecule, (2) knowledge of general degradation processes, and (3) toxic properties of similar chemicals. The confidence of such predictions will improve with a better data base. However, although they will prove valuable in determining priorities in the sequence of toxicity tests, the predictions cannot replace the toxicological tests themselves.

The fate of the chemical undergoing reaction also must be considered. The environmental problem does not end when the chemical itself is transformed, unless it is completely mineralized or converted to species available for normal biological utilization. Transformation products may be more persistent and more hazardous than the parent compound, and thus must be dealt with in any analysis.

Biodegradation Microbial metabolic transformation may be the major route of degradation in soils and in the aquatic environment. Transformation data must be obtained under aerobic and anaerobic conditions in both soil and water. Information regarding the chemical's effect on microorganism development (bacteria, algae, diatoms, etc.) is required. Data concerning the disruption of such activities as cellulose decomposition, nitrogen fixation, sulfur transformation, and $CO_2$ efflux are particularly important. In addition an assessment of potential disruption of wastewater treatment systems must be made.

Committee's Comments: Microbial degradation is probably the major chemical transformation route in soils and aquatic sediment. In addition to the required data, estimates should be made of the biodegradability index (biological oxygen demand [5 days]/chemical oxygen demand = $BOD_5/COD$) and compatibility with biological waste treatment ($IC_{50}$ = 50 percent inhibition of activated sludge) under both aerobic and anaerobic conditions. Detailed

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studies of biodegradation intermediates should be required for compounds with a low biodegradability index or a low compatibility with waste treatment, or both.

Mobility Movement of a product through the environment may cause contamination of food webs, loss of usable land and water resources, and loss of wildlife habitats; data requirements include information on leaching, volatility, adsorption/desorption, and water-dispersal properties.

Bioaccumulation Data must indicate the extent of accumulation and potential adverse effects on nontarget organisms (in particular, agricultural systems and fish). Residue levels and rate of uptake in plants must be determined according to the TSCA guidelines. Fish bioaccumulation studies should use flow-through and static systems with sunfish and catfish as the FIFRA recommended test species. TSCA requirements specify that fish bioaccumulation should be assessed when octanol/water partition coefficients are greater than 1,000 or when solubility in water is less than 0.9 x $10^{-3}$ M at 20°C and degradation data indicate persistence. Minnows and sunfish are preferred species for the TSCA guidelines.

Committee's Comments: At least two accounts of relationships between octanol/water partition coefficient and solubility in water have been published. The relationships derived by Chiu et al. (1977) and Moriguchi et al. (1976) give solubilities of 0.966 x $10^{-3}$ M and 0.621 x $10^{-3}$ M, respectively, for a partition coefficient of 1,000. The solubilities correspond to an accumulation coefficient of 80 to 100 (Chiu et al. 1977). The guidelines could be modified to require bioaccumulation tests with fish only for those compounds with a partition coefficient of 10,000, corresponding to a solubility of 0.02 x $10^{-3}$ M to 0.03 x $10^{-3}$ M, and an accumulation coefficient of about 500.



Table 3.3 Proposed Testing Levels for Hazard Assessment

| Level I Tests | Level II Tests | Level III Tests |
|---|---|---|
| A. Acute Toxicity | A. Chronic Toxicity | A. Environmental Simulation |
| B. Short-term Screening | B Reproductive Tests | B Metabolic Tests |
| C. Environmental Chemistry | C Bioaccumulation | C. Long-term Chronic Toxicological Tests |

Hazard Assessment

The FIFRA proposed guidelines of June 1978 follow a hierarchical testing scheme for hazard assessment. Table 3.3 summarizes the anticipated movement through the various tests. The extent to which advancement through the hierarchy is required would depend on two factors:

- results from tests in the previous level (e.g., if there is acute toxicity, proceed to level II; if there is no acute toxicity, no further testing will be required unless environmental chemistry suggests persistence, in which case reproductive tests should be performed); and
- anticipated use patterns (e.g., if use does not indicate repeated human exposure, tests are not required for long-term toxicity, teratogenicity, or reproductive impairment).

The guidelines drafted under TSCA offer a basis for assessing potential risk associated with anticipated use of new chemicals. TSCA proposed recommendations do not include a rigid testing sequence. Manufacturers are given the responsibility for determining which tests should be performed to develop the essential "effects-data" with which scientifically sound assessments of risk may be made. Two sets of studies are described: reference studies and hazard evaluation studies. Reference studies are suggested for all new chemicals unless the manufacturer can show that some are not relevant to a particular substance (e.g., chemical intermediates and substances that are produced and used entirely in closed systems). The reference set [illegible].

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- acute toxicity studies,
- subchronic toxicity studies,
- reproduction studies,
- teratogenic studies,
- mutagenic studies, and
- short-term predictive oncogenic studies.

Hazard evalustion studies include chronic toxicity and oncogenicity tests. Draft guidelines recommend these when data from reference studies are not adequate to predict potential toxicity of the new chemical, or when extensive hazard evaluation is considered essential.

The difference between FIFRA and TSCA proposed requirements can be summarized as follows:

- FIFRA establishes a set sequence of tests with trigger points for movement through the sequence;
- TSCA allows the manufacturer to determine the extent and sequence of testing deemed necessary for scientifically sound hazard evaiuation; exemptions or departures from recommended tests may be permitted if adequate justification can be provided.

General descriptions of the tests specified in both guidelinea are presented in Table 3.4.

Tests within the FIFRA scheme proceed from basic laboratory tests to applied field tests. Beyond tests of acute toxicity, wildlife data are required only when patterns of use suggest exposure in the habitat. When use patterns indicate no anticipated exposure or a minimal level of risk, data from chronic tests are not required.

As with the hazard evaluation tests for wildlife exposure risk, the data required for assessing risk to domestic animals and humans also would depend largely upon anticipated use patterns and results of acute toxicity tests.

In addition to the FIFRA tests for animal toxicity evaluation, TSCA requires assessments of effects on plants, both monocotyledons and dicotyledons. The recommended tests determine effects on plant growth and seed germination. The TSCA guidelines concentrate testing efforts on assessment of risk to human health and do not consider risk to domestic or commercial animals.

[illegible] for hazard evaluation appear to be adequate. However, there remain two basic problems in the FIFRA evaluation scheme:

1. Physical and chemical properties apparently are not used to determine the appropriate testing route for each chemical except for deciding whether reproductive toxicity tests should be performed; and
2. Use patterns combined only with results of acute toxicity tests are inadequate means for selecting further appropriate tests of new chemicals. For example, in this evaluation format chronic data are required for human health assessment only if use patterns suggest repeated exposure over a significant portion of the human lifespan and acute toxicity results are positive; similarly, teratogenicity and reproductive studies are necessary only if significant exposure to humans is expected. The problems associated with these two points are discussed in the following pages using PCBs as the test case.

The relative flexibility of the TSCA guidelines reflects the fact that no single, rigid testing scheme is suitable for the evaluation of all chemicals. In the opinion of this Committee, the flexibility is good. However, the decision to allow exemptions from the test requirements or to require additional testing should not be left to an individual but should be made by a group of experts with backgrounds in chemistry, toxicology, ecology, and agriculture.

## ASSESSMENT OF PCB HAZARD

An analysis of PCB data (Appendix D) compiled following the criteria of FIFRA and TSCA proposed guidelines, leads to the conclusion that PCBs are persistent, and are likely to accumulate. PCBs do not appear particularly toxic for short-term exposure, but results are subject to interpretation (Roberts et al. 1978). Based on anticipated patterns of use (Table 3.1), one expect PCB distribution in the environment and a [illegible] risk for humans through use of lacquers and [illegible] containing PCBs.

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TABLE 3.4 FIFRA and TSCA Data Requirements for Product Hazard Evaluation (Spp - species)

| TERRESTRIAL AND AQUATIC WILDLIFE: Tests | FIFRA Criteria | TSCA Tests | TSCA Reference Set Criteria |
|---|---|---|---|
| Acute Toxicity | $LD_{50}$ - one spp each avian, mammal, fish<br>$LC_{50}$ - 2 spp each avian, mammal, fish | Product dependent and 1 spp of invertebrates from different classes or phyla<br>$LC_{50}$ - 1 spp fish; preferred spp are rainbow trout or bluefish | Product expected to enter aquatic system or be dispersed by water |
| | Life Cycle - fish embryo-larval or invertebrate | $LC_{01}$ or 1mg/l and water concentration >0.1 of $LC_{50}$ use pattern result in water concentrations >0.1st of $LC_{10}$ with significant reproduction effects physical chemical accumulation test data, use pattern expect continuous water exposure | $LC_{50}$ - quail dietary | Product not significantly volatilized from food prior to consumption |
| | Life Cycle - fish preferred spp minnow (fresh water or estuarine) | If embryo/larvae tests indicate adverse effects environmental concentration is 0 at 'no effect' level of embryo/larvae tests if effects observed in mortality or avian reproduction with low concentration | Life Cycle - Daphnia/ fish embryo/juvenile | Product within aquatic systems and is soluble or dispersed |
| | Organism toxicity and residue studies | Pre tests test results suggest adverse effects on non-target organisms considered on case-by-case basis | |
| Reproduction | One generation reproductive studies-quail and mallard | Physical chemical and environmental data indicate product is persistent compound or metabolites bioaccumulate use patterns suggest repeated avian exposure | See Human Health Effects |
| Field Studies | Tests for direct or indirect poisoning behavioral impairment reproductive effects and evidence of ecological disturbances | If acute & reproductive data indicate adverse effects | |

| HUMANS, ANIMALS AND HUMAN HEALTH: Tests | FIFRA Criteria | TSCA Tests | TSCA Reference Set Criteria |
|---|---|---|---|
| Acute Toxicity | Oral and inhalation studies - rats | Required for all new compounds critical levels or mortality-dose level 5mg/kg 5 animals eac | Same tests as FIFRA | Recommended for all new chemicals that require TSCA notification |
| | Dermal studies - rabbits | Required for all new compounds critical level 5mg/l for 4 hours | | |
| | Dermal and eye irritation studies - rabbits | Required for all new chemicals | Same tests as FIFRA | Recommended for all new chemicals that require TSCA notification |
| | Delayed neurotoxicity - hens test parent compound and metabolites | Required for all new chemicals | Not recommended in Reference Tests | |
| Subchronic Toxicity | Oral dose - 3 mammalian spp (1 rodent, 1 nonrodent. | Required if use patterns indicate repeated human exposure is expected | Same as FIFRA | Recommended for all new chemicals that require TSCA notification |
| | Dermal studies - 21 day, rabbit | Required if use patterns and data indicate repeated skin contact is anticipated | | |
| | 90 day, rabbit | Required if dermal pulmonary skin applications use will result in human exposure or absorbed to by metabolized in animal manner dermal or inhalation routes | Dermal - 90 day studies, based on FIFRA | Recommended for all new chemicals that require TSCA Notification |
| | Rats inhalation studies - rat | Required if acute inhalation results are positive and use patterns indicate repeated inhalation exposure at these levels | Same as FIFRA | Recommended for all new chemicals that require TSCA notification |
| | Neurotoxicity | Required if a pre ingestion or inhalation activity or chemical structural similarity to known neurotoxins | Not recommended in Reference Tests | |
| Reproductive Toxicity | Two generation studies Rabbits or rats same as teratogenic studies | Required if use patterns indicate repeated human exposure | Three generation studies same as FIFRA | Recommended for all new chemicals that require TSCA notification |



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TABLE 3.4 continued

| DOMESTIC ANIMALS AND HUMAN HEALTH | FIFRA Tests | FIFRA Criteria | TSCA Tests | TSCA Reference Set | TSCA Criteria |
|---|---|---|---|---|---|
| Teratogenic Toxicity | Studies done using two species of rodents and one species of non-rodent if warranted by toxicity tests | Required if use patterns indicate significant potential for human exposure to chemical | Same as FIFRA | | Recommended for all new chemicals that require TSCA Reinitiation |
| Mutagenic Toxicity | Short-term studies to establish mechanism of action, possible synergistic, additive or antagonistic effects in presence of other chemicals, and effects of chemical on human lymphocytes, or if chemically related to a human mutagen | Required if use patterns indicate significant potential for human exposure to chemical | Same as FIFRA | | Recommended for all new chemicals that require TSCA Reinitiation |
| | | | Tiered Evaluation Set | | |
| Chronic Toxicity | Oral, dermal or inhalation studies - six months duration - to determine dose-related mortality and morbidity (organ-wise terminal blood, urine, specific tissues and eye examinations) in rodents and non-rodents | Required if acute results are positive and/or statistically significant evidence of human exposure over a significant portion of the human lifespan | Same Tests as FIFRA - 1 spp | | Recommended if projected human exposure is great and if results from electron capture potential human health hazard, or if anticipated effects cannot be predicted by data in the Extension Set. |
| | | | Continuous testing over animals lifespan; route of exposure to be the same as major anticipated human exposure - 2 spp | | Recommended if projected human exposure is great and if results from electron capture potential human health hazard, or if anticipated toxic mount predict unacceptable effects, or if chemical is structurally related to a known carcinogen. |
| Oncogenic Toxicity | Tests with mammalian species to be determined for animals lifespan - 2 mammalian spp | Required if human metabolite or degradation products structurally related to a known carcinogen, have metabolic effects that may lead to oncogenic changes | no metabolic studies recommended | | |
| Metabolic Studies | Generally no specific tests, but metabolic studies, dose-related studies, absorption, binding properties with tumor organ distribution, storage and excretion, and genotoxicity studies | Required whenever chronic data are required or suspect for inducing an oncogenic lesion | | | |
| | Data from the above tests may be required for domestic animals if exposure to the chemical is anticipated for humans, as through direct application or through consumption of food | | | | No recommended tests for domesticated or commercially used animals |

[illegible] hazard assessment predicate [illegible] on two factors (1) evidence for [illegible], and (2) use patterns as an indication of anticipated exposure. The TSCA requirements call for more extensive testing (i.e., reproduction, subcarcinogenic, etc.) without consideration of use patterns or results of other tests.

## FIFRA Testing Results

The conclusion reached regarding the potential hazard posed to the environment by PCBs based on FIFRA requirements is questionable. Acute toxicity is observed only for some aquatic invertebrates, but these species are not required test animals in the FIFRA guidelines. Based on anticipated use patterns, the expected human and environmental exposure to the chemical would be primarily from protective coatings. Some risk would be anticipated in occupational situations, but these exposures could be carefully controlled. If the anticipated use had been limited to heat-exchangers and closed electrical equipment, the testing data might not have suggested potential hazard to human health or the environment. In such a case PCBs might have "passed" the toxic substance evaluation. The aggregate data presented in Appendix D and the "real world" experience of more than 40 years of use of this chemical indicate widespread environmental exposure and possible serious health problems. The discrepancy is due to the fact that the current FIFRA protocols for hazard evaluation relate the degree of required testing to proposed uses. The PCB experience provides a useful test of the appropriateness of relating test requirements to projections of chemical uses.

The question of whether projected use patterns give an adequate indication of environmental exposure is complex. The use patterns anticipated prior to manufacture clearly do not give an adequate impression of the hazards associated with accidental release to the environment during manufacture and use, or of the extended problems of environmental release from disposal of equipment containing PCBs. The expanded uses of PCBs after 1930, illustrated in Table 3.5, may suggest [illegible] exposure through disposal of products [illegible] PCBs, but these uses still do not suggest the

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TABLE 3.5 Uses of Aroclor by Type

| Current Uses (since 1970) | 1016 | 1221 | 1232 | 1242 | 1248 | 1254 | 1260 | 1262 | 1268 |
|---|---|---|---|---|---|---|---|---|---|
| Capacitors | X | X | | X* | | X* | | | |
| Transformers | | | | X | | X | X* | | |
| Die casting -- Fenoclor-79ICl | | | | | | | | | |
| Former Uses | | | | | | | | | |
| Heat transfer | | | | X | | | | | |
| Hydraulic/lubricants | | | X | X | XX | XX | X | X | XX |
| • Hydraulic fluids | | | | | | | | X | |
| • Vacuum pumps | | X | | X | | | | | |
| • Gas-transmission turbines | | | | | | | | | |
| Plasticizers | | X | X | X | XX | XX | X | | XX |
| • Rubbers | | | | | | | | | |
| • Synthetic resins | | | | X | | | | | |
| • Carbonless paper | | | | | | | | | |
| Miscellaneous | | X | X | XX | X | XXXXXXXX | X | | X |
| • Adhesives | | | | | | | | | |
| • Wax extenders | | | | | | | | | |
| • Dedusting agents | | | | | | | | | |
| • Inks | | | | | | | | | |
| • Cutting oils | | | | | | | | | |
| • Pesticide extenders | | | | | | | | | |
| • Sealants and caulking compounds | | | | | | | | | |

*Discontinued use of these types.

SOURCE Modified from IARC (in press).

sterility in Mink in North Central United States and Canada (Aulerich and Ringer 1977). In early 1960, mink ranchers began noticing reproductive complications and excessive kit mortality among their stock. An acute problem was evident by 1967, resulting in an unprecedented 80 percent increase in newborn mortality. Subsequent investigation indicated a strong relationship between kit mortality and the percentage of coho salmon in the mother's diet, as well as the duration of feeding with a salmon diet. Factors such as rancidity, pesticide contamination and mercury poisoning were suspected but were shown not to contribute to the problem.

Experiments with the mink diet were conducted, and the results indicated that diets of 30 percent salmon produced the reproductive problems. Further investigation ruled out salmon per se as the cause, because other species of Great Lakes fish caused similar adverse effects. The following evidence strongly implicated PCBs as the causative factor:

- high PCB residue levels of Lake Michigan coho salmon;
- sensitivity of mink to PCBs;
- similarity of reproductive complications in mink fed coho salmon and those fed diets contaminated with PCBs;
- similarity of clinical signs and lesions observed in mink that died while on diets containing coho salmon and diets containing PCBs; and
- accumulation of PCB residues in mink tissue.

Rice Oil Contamination in Japan (Kuratsune et al. 1972, World Health Organization 1976). In June 1968, Japanese patients suffering from chloracne began appearing at the Kyshu University Hospital. By October of that year the disease was of epidemic proportions, involving more than 1,000 people in 21 prefectures in the western section of Japan. The exposure factor common to these patients was consumption of rice oil produced in February, 1968. Investigation of the manufacturing process identified a leak in a heat-exchanger, which resulted in oil contamination with Kanechlor 400, a 48 percent

chlorinated biphenyl. In addition to symptoms associated with chloracne, clinical signs included a rise in serum transaminases and alkaline phosphatase, respiratory symptoms and chronic bronchitis-like problems, decrease in conduction velocity in peripheral sensory nerves, diminished growth in young males, and pigmented skin on newborns.

Between February and March of 1968, a disease occurred in chickens in the same section of Japan. More than 400,000 animals died. Clinical signs included labored breathing, droopiness, and decreased egg production. Autopsy indicated subcutaneous edema, hydropericardium, ascites, pulmonary edema, muscular ecchymosis, and a mottled appearance of the liver. Investigations revealed that the chicken feed had been prepared with the rice oil contaminated with Kanechlor 400. Subsequent study of the Kanechlor indicated large amounts of chlorodibenzofurans present. The extent of this chemical's involvement in the Yusho symptoms is not fully resolved.

Sewage Sludge Problem in Bloomington, Indiana (McCloskey et al. 1978) Residents of Bloomington, Indiana, had been using sewage sludge in fertilizing their gardens until high concentrations of PCBs were reported in the sludge in 1976. An investigation to locate the source revealed that a local electrical firm was using PCBs as a dielectric in capacitors. The company had been discharging wastes contaminated with PCBs into the municipal sewage system. By January 1976, sludge samples had mean PCB levels of 479 mg/kg and samples of soil treated with the sludge had mean concentrations of 17 mg/kg.

Health officials conducted clinical evaluations on workers in the electrical firm, their families, and Bloomington residents who had used the PCB-contaminated sludge for fertilizer. Serum PCB levels, alteration in hepatocellular function, and plasma triglyceride levels were examined. Although mean serum levels were relatively high compared to U.S. population groups (workers 72 mg/kg; families 34 mg/kg; residents 18 mg/kg; U.S. population 5 mg to 20 mg/kg) no correlation could be detected between the serum levels and the number of years using sludge, total pounds used, or time since the last use. Serum levels of selected liver enzymes increased significantly with PCB levels in serum as did plasma triglyceride levels.



[illegible] General Electric Plants in New York(Horn [illegible] [in press]) The Hudson River had been contaminated between 1942 to the mid 1970s by PCB effluent from General Electric manufacturing plants at Fort Edward and Hudson Falls, averaging about 14 kg per day. The long-term contamination has polluted virtually the entire river bed for many miles downstream. Bottom sediment below Fort Edward reportly contains 540 µg/kg to 2,980 µg/kg. The average concentration of PCBs in bass, perch, and sucker is 62 mg/kg to 135 mg/kg.

Contamination of Fish Meal at a Ralston Purina Plant (Jaroslovsky 1978) In 1977, a poultry firm began noticing high chemical levels in tissue samples taken from their chickens. The feed supplier was asked to help in pinpointing the cause, and the culprit was found to be fish meal contaminated by PCBs that had been used in formulating the chicken feed.

The fish meal had been prepared and stored in Puerto Rico at a Ralston Purina plant warehouse where two electrical tranaformers containing PCBs were stored also. Contamination of the meal occurred during a fire in April 1977, which damaged the electrical equipment and allowed PCBs to leak out. Water from fire hoses evidently mixed with the PCB fluid and soaked into the stored fish meal. The contamination was not detected and the fish meal was subsequently sold. As a consequence, 400,000 chickens and 15,000 dozen eggs were destroyed. Some of the contaminated feed did not actually contain the PCB fish meal, but the feed had been processed with the same machinery as the fish meal.

It is clear from these illustrations that projected patterns of use may not provide an adequate indication of the eventual distribution of PCBs in the environment or the extent of the hazard this distribution can pose. These incidents were the result of manufacturing and disposal procedures--factors not considered in proposing guidelines for testing new chemicals.

Toxic effects of PCBs in workers exposed in their occupations were noted as early as 1917 and might have been anticipated even earlier from the effects of chlorinated naphthalenes detected during World War I (Jacobs 1949). Had the awareness of toxic chemicals [illegible] been as widespread as it is currently, these [illegible] would have been more than sufficient to have [illegible] the manufacture and use of PCBs or, at least,

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to have demanded detailed testing that would have revealed the chemical's chronic and more subtle toxic effects.

## TSCA Testing Results

Physical and chemical properties of PCBs indicate the tendency of these compounds to persist as well as their high bioaccumulation potential. The data compiled following the Reference Set requirements and the proposed use of PCBs in protective coatings, which would result in some exposure of humans, justify the requirement for Hazard Evaluation Studies. These studies probably would detect at least some of the toxic effects of PCBs. However, it should be noted that most of the chronic tests proposed to assess human health hazards require use of species (rodent plus one other mammal) that would not adequately reflect the human toxic response to PCB exposure. The common laboratory test animals (rats and dogs) do not always respond to PCB exposure in the same manner as humans, and Table 3.6 suggests that no one species exhibits "typical" human symptoms of PCB poisoning except in the case of reproductive tests. Thus proper choice of the test species becomes important when evaluating the potential hazard of any substance.

Although PCBs have been detected in the U.S. human population (Kutz and Strassman 1975, Humphrey 1977), there have been no documented cases of PCB intoxication of humans other than those related to industrial exposure. Effects, when detectable, have been subtle rather than overt. For the most part, the proposed sets of tests for FIFRA and TSCA are designed to detect only gross intoxication. Subtle effects are not considered at all in the scheme proposed for TSCA and only peripherally in those drafted by FIFRA.

## RECOMMENDATIONS

Perhaps the two most important properties of a chemical that relate to potential hazards to health and the environment are its persistence and lipophilicity. "High scores" in those tests that identify these properties should alert officials to the inevitability of environmental contamination and health problems. It

TABLE 3.6 Summary of Chronic Toxic Effects of PCBs

| Test | Effects |
|---|---|
| Chronic Feeding | |
| Aquatic Species | Threshold effects in egg hatchability of vertebrates and invertebrates at levels of 2-5 µg/l |
| | Embryo toxicity evident at 50 µg/l |
| Terrestrial Species | Mouse - some liver change with exposure to high chlorine containing products, 300-500 µg/g |
| | Rat - some liver changes, minimal reproductive effects, 100-500 µg/g |
| | Monkey - Yusho symptoms, altered reproduction cycles, hyperplastic gastritis and ulceration, 2.5-5 µg/g |
| | Chicken - some morphologic deformity, reproduction decline, subcutaneous edema, 20-50 µg/g |
| | Mink - dose response relationship in growth and reproduction, 10 µg/g |
| | Pelican - some hepatocellular changes, 100 µg |
| | Dogs - reduced growth, some liver changes, 100 µg |
| | Wildfowl - some reproduction changes, varies with species, 50-200 µg/g |
| Teratogenicity | Effects seen in avian species, 50-200 µg/g |
| Mutagenicity | Chromosomal aberrations - negative results |
| | Dominant lethal mutations - negative results |
| | Ames test - 1221, 4 chlorobiphenyl significantly mutagenic |
| Oncogenicity | High chlorinated compounds produced tumors in rats and mice, relationship with PCB not known, clear |

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should be emphasized, however, that evidence of nonpersistence or nonlipophilicity does not suggest lack of hazard potential.

Physical, chemical, and environmental chemistry data are sufficient to determine persistence and lipophilic properties of chemicals and can be used to predicate potential bioaccumulation and environmental distribution. Therefore, the Committee recommends a multitiered evaluation scheme as illustrated in Figure 3.1. This scheme requires first:

- chemical identification of the parent compound, impurities, metabolites, and degradation products;
- determination of physical and chemical properties; and
- assessment of environmental and bioscreening (short-term toxicity) test results for parent compound and metabolites.

At this point, if the data indicate that the chemical is highly persistent or lipophilic, or both, its development for large-scale use should be scrutinized carefully and possible alternatives considered. On the basis of the scrutiny, the chemical may be either rejected or subjected to additional testing. If the data indicate that the chemical is not highly persistent or lipophilic but there are indications of potential hazards, the chemical should be subjected to additional testing. If there are no indications of potential hazards, the chemical may be approved for manufacture and use.

Additional assessment of hazard potential for all suspect substances should include:

- development of a mass balance model of the potential movement of the substance through the environment using data regarding physical and chemical properties and anticipated source function parameters;
- identification of "problem areas" or sites of potential chemical accumulation (e.g., agricultural lands, urban areas, marine or freshwater sources, or combinations of these);
- specification of tests to assess toxic effects within the "problem areas."

Test guidelines should be formulated specifying representative species for each environmental medium and

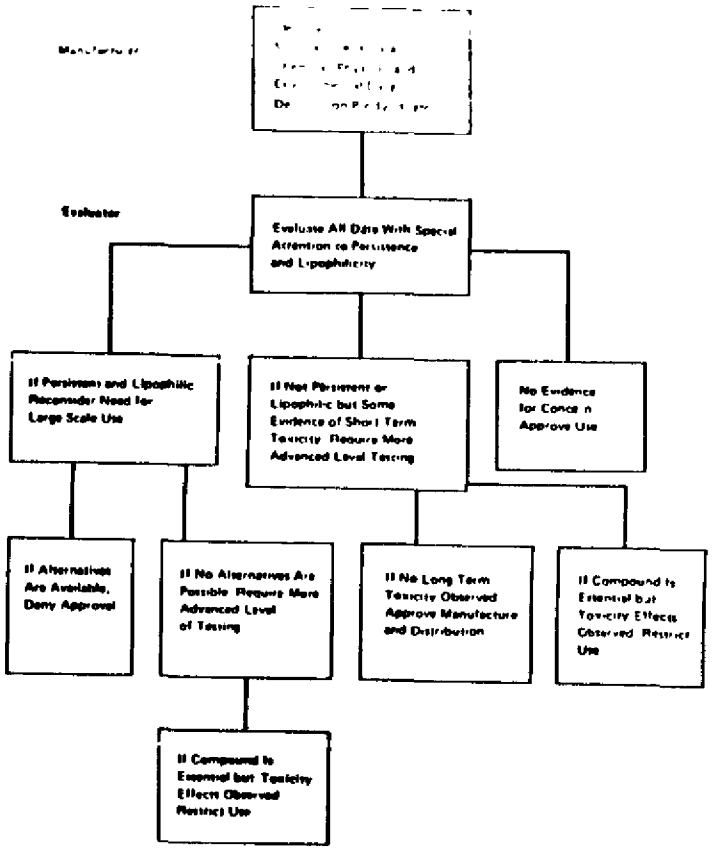


FIGURE 3.1 Evaluation scheme for assessing potential hazards to health and the environment

those species most appropriate for assessing toxicity to humans. Specified tests should include assessment of:

1. reproductive alterations;
2. pathological effects, and
3. behavioral and learning impairment.

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Using such an approach, industry could concentrate its testing efforts on identifying effects in specific ecosystems. For example, if the model predicts primary chemical accumulation in the sediment of rivers and lakes, the manufacturer could concentrate on assessment of hazards within these systems, using quick screening tests for other media when results indicated possible contamination of land or air. Monitoring after a product has been marketed should be required to keep industry and government alert to unexpected accumulations, and additional tests and the implementation of appropriate control measures could be demanded as necessary.

REFERENCES


Aulerich, R.J. and R.K. Ringer (1977) Current status of PCB toxicity to mink, and effect on their reproduction. Arch. Environ. Contam. Toxicol. 6:279-292.

Chiu, C.T., V.H. Freed, D.W. Schmedding, and R.L. Kohnert (1977) Partition coefficient and bioaccumulation of selected organic chemicals. Environ. Sci. Technol. 11:475-478.

Conservation Foundation (1978) Approaches for Developing Testing Guidelines Under the Toxic Substances Control Act. (Unpublished)

Horn, E.G., L.J. Hetting, and J. Tofflemire (In press) The problem of PCBs in the Hudson River system. The scientific basis for public control of environmental health hazards, symposium, 1976. New York Acad. Sci. Science Week. (To be published Spring 1979)

Humphrey, H.E.B. (1977) Evaluation of Changes in the Level of Polychlorinated Biphenyl in Human Tissue. Final Report on FDA Contract No. 223-73-2209. Michigan Department of Public Health.

International Agency for Research on Cancer (IARC) (In press) Monographs on the evaluation of the carcinogenic risk of chemicals to humans 18:43-103. (Due for release in or after 1979)

Jacobs, M.B. (1949) The Analytical Chemistry of Industrial Poisons, Hazards, and Solvents. New York: Interscience Publishers, Inc.

Jaroslovsky, R. (1978) Contamination of fish meal with PCB sparks FDA study of mishap at a Ralston Purina plant. Wall St. J. 46, July 12.





[illegible], T. Yorhimura, J. Matsuzaha, and A. [illegible]hi (1972) Epidemiological study of Yusho, a [illegible]sing caused by ingestion of rice oil contaminated with a commercial brand of polychlorinated biphenyls. Environ. Health Perspect. 1:119-128.

Kutz, F.W. and S.C. Strassman (1975) Residues of polychlorinated biphenyls in the general population of the United States. Pages 138-143, Proceedings of National Conference on Polychlorinated Biphenyls, November 19-21, 1975, Chicago, Illinois. Report No. EPA-560/6-75-004. Washington, D.C.: U.S. Environmental Protection Agency.

McCloskey, F.X., T.O. Middleton, R. Schneider, J. VanFleet, W.T. Paynter, R.D. Telle, and C.J. Glueck (1978) Polychlorinated biphenyl exposure - Indiana. Morbidity and Mortality Weekly Report 27(12):99-100.

Moriguchi, I., Y. Kanada, and K. Komatsu (1976) Van Der Waals volume and the related parameters for hydrophobicity in atructure-activity studies. Chem. Pharmacol. Bull. 24:1799-1806.

Niabet, I.C.T. (1976) Criteria Document for PCBs. Report No. PB-225 397. Washington, D.C.: U.S. Department of Commerce.

Oishi, S. and K. Hiraga (1978) Is a mixture of polychlorinated dibenzofurans an inducer of hepatic porphyria? Food Cosmet. Toxicol. 16(1):47-48.

Oishi, S., M. Morita, and H. Fukuda (1978) Comparative toxicity of polychlorinated biphenyls and dibenzofurans in rats. Toxicol. Appl. Pharmacol. 43:13-22.

Penning, C.H. (1930) Physical characteristics and commercial possibilities of chlorinated diphenyl. Ind. Eng. Chem. 22(11):1180-1182.

Roberts, J.R., D.W. Rodgers, J.R. Bailey, and M.A. Rorke (1978) Polychlorinated Biphenyls: Biological Criteria for an Assessment of Their Effects on Environmental Quality. Associate Committee on Scientific Criteria for Environmental Quality. Ottawa, Canada: National Research Council of Canada.

Smith, J.H., W.R. Mabey, N. Bohonos, B.R. Holt, S.S. Lee, T.W. Chou, D.C. Bomberger, and T. Mill (1977) Environmental Pathways of Selected Chemicals in Freshwater Systems. Part I: Background and Experimental Procedures. Report No. EPA-600/7-77-113. Washington, D.C.: U.S. Environmental Protection Agency.


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U.S. Environmental Protection Agency (1978a) Draft Guidelines for Testing Toxic Substances. Office of Toxic Substances. (Unpublished)

U.S. Environmental Protection Agency (1978b) FIFRA Proposed Guidelines for Registering Pesticides in the United States. Subpart B-Introduction, Subpart D-Chemistry Requirements, Subpart F-Hazard Evaluation: Wildlife and Aquatic Organisms, Subpart F-Hazard Evaluation: Humans and Domestic Animals. Federal Register 43(132):29696-29741 and 43(163):37351-37403.

World Health Organization (1976) Polychlorinated Biphenyls and Terphenyls. Environmental Health Criteria 2. Geneva: World Health Organization.

Zitko, V., D.J. Wildish, O. Hutzinger, and P.M.K. Choi (1973) Acute and chronic oral toxicity of chlorinated dibensofurans to salmonid fishes. Environ. Health Perspect. 2:187-189.

APPENDIX A

AREAS, VOLUMES, AND MISCELLANEOUS QUANTITIES FOR CHAPTER 1 CALCULATIONS[1]

[1]The values in this table were used in calculating fractional body burdens of PCBs.

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TABLE A.1 Areas, Volumes, and Miscellaneous Quantities

| U S Area | Area of Drainage Basin ($km^2$) | Agricultural Land ($km^2$)[b] | Discharge of Water ($km^3$/yr) | Discharge of Suspended Sediment ($10^6$ metric tons/yr)[c] | Area of Sediments Origin ($km^2$)[c] | Sediment Yield (metric tons/$km^2$/yr)[c] |
|---|---|---|---|---|---|---|
| Atlantic[a] | 1,361,664 | 1,100,288 | 321 258 | 12.886 | 741,760 | 17 3 |
| Gulf of Mexico | 4,315,392 | 3,652,766 | 788 99 | 341 080 | 4,504,528 | 76 1 |
| Pacific[a] | 2,108,672 | 1,523,256 | 445 97 | 84 872 | 1,637,942 | 54 9 |
| Total | 7,785,728 | 6,276,310 | 1,556,318 | 445 838 | 6,886,210 | |

| | Streams[d] | | Reservoirs having Capacity > 5000 acre-feet | | Lakes > 10 $mi^2$[d] | |
|---|---|---|---|---|---|---|
| U S Area | Surface Area ($km^2$) | Volume of Water ($km^3$) | Surface Area ($km^2$) | Volume of Water ($km^3$) | Surface Area ($km^2$) | Volume of Water ($km^3$) |
| Atlantic | 21,299 | 106.5 | 11,655 | 48 393 | 163,264 | 11,623 |
| Gulf of Mexico | 25,728 | 128.6 | 35,274 | 227 5 | 12,966 | 83.8 |
| Pacific | 5,458 | 27.2 | 8,376 | 153 741 | 3,080 | 341.8 |
| Total | 52,485 | 262 3 | 55,305 | 429.854 | 177,310 | 12,048.8 |

| | Sediments | | |
|---|---|---|---|
| U S Area | Volume of Stream Sediments Containing PCBs ($km^3$)[e] | Volume of Reservoir Sediments Containing PCBs ($km^3$)[f] | Lake Sediments Containing PCBs ($km^3$)[g] |
| Atlantic | 4.472 | 5.827 | 16.326 |
| Gulf of Mexico | 5 402 | 17.637 | 1 297 |
| Pacific | 1 146 | 4 188 | 0 308 |
| Total | 11.020 | 27.652 | 17 931 |

[a] In general, our boundaries coincide with those in the Water Resources Council's report. Two differences are (1) with Atlantic we included the Souris-Rainy-Red drainage into the Hudson Bay and the Great Lakes area, which drains via the Gulf of St [illegible]. [b] The value for area in the Great Lakes Region excludes the water-surface area of the U.S portion of the Great Lakes [illegible]. [c] Pacific column includes the area of the Great Basin. The data in the Water Resources report did not enable us to [illegible]

[illegible]s that are unified in these drainages from the useas in the Water Resources Council's book [illegible]ng includes cropland, both that used for crops and that left idle and fallow, permanent pas[illegible]. [illegible] the Water Resources report, these areas were tabulated in acres. They were converted to square k[illegible] [illegible] proportion of the total land area, which is given both as $10^3$ acres and square miles, and then by conv[illegible] [illegible] meters (using 2.56 $km^2$ = 1 $mi^2$). The figures shown are the areas as projected for 1980 [illegible]s for discharge of suspended sediment from the three areas are from Curtis, Culbertson, and Chase, 1973, [illegible] Surv ey Circ 670, Table 3, p 11. The reasons for the discrepancies between the areas of the drainage basins [illegible] S Geological Survey Circ 670 and the figures shown here from the Water Resources Council are not known [illegible] of the information required for this study included the surface areas of lakes and rivers within the United State[illegible] [illegible] of water in the lakes and rivers, volumes of sediments containing PCBs, and the concentration of PCBs in th[illegible]. For the Great Lakes area, only the U.S portion was included.

In contrast to the large array of data on the areas of drainage basins, lengths of river systems, and quantities o[illegible] water and suspended sediment discharged, data on the areas and volumes for the present purpose are extremely scarce. The basis for the areas of the water bodies used in this study is the 1940 census compilation of "Inland Water o[illegible] United States." These figures are summarized state by state in the National Atlas, but in the Atlas, the definition of inland water was not included. A tabulation with the definition, but attributed incorrectly to the 1960 census, appears on page 1[illegible] of the Water Encyclopedia. In the 1940 census, inland water was defined as lakes, reservoirs and ponds having an area of 40 acres or more, and streams, sloughs, and channels having widths of 1/8 mile and greater.

From the total area of inland water were subtracted the areas of the following: (1) lakes having areas of 10 square miles or more (from D.K Thomas, ed., 1973, The Water Encyclopedia, p 122-124); (2) large saline lakes (from Water Encyclopedia, p 128-129), and (3) reservoirs having a usable capacity of 5000 acre-feet or more (from Water Encyclopedia p 410).

The category for streams does not include those narrower than 1/8 mile but does include lakes and ponds having areas of greater than 40 acres but less than 10 square miles.

The areas in the "stream" column are accurate to the extent that the cumulative areas of lakes and ponds smaller than 10 square miles but larger than 40 acres equals the cumulative area of streams narrower than 1/8 of a mile.

In a few states, the subtraction of the areas of the reservoirs (after deducting inland water by subtracting the [illegible] of [illegible] greater than 10 square miles and the areas of saline lakes yielded a negative number. We took this [illegible] mean that in these states, large reservoirs had been built since 1940. We have not tried to check this independently. Where a negative number would have resulted, we did not subtract the areas of the reservoirs.

To get the mean volume of water in the rivers, we assumed that the mean depth of streams is 2.5 meters [illegible] [illegible] has been corrected by assuming that 60 percent of the "streams" area is [illegible] [illegible] mis, that 0.5 meters contains PCBs. This figure is based on the supposition that the [illegible] [illegible] is similar to all other streams.

The [illegible] reservoir sediments was computed using an average thickness of 0.5 meters times the areas of the [illegible].

The [illegible] lake sediments was computed using an average thickness of 0.1 meters times the areas of the lakes.



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REFERENCES

Curtis, W.F., J.K. Culbertson, and E.B. Chase (1973) Fluvial-sediment discharge to the oceans from the conterminous United States. U.S. Geological Survey Circ. 670. Washington, D.C.: U.S. Geological Survey.

Todd, D.K., ed. (1970) The Water Encyclopedia. Port Washington, N.Y.: Water Information Center, Inc.

U.S. Department of the Interior Geological Survey (1970) The National Atlas of the United States of America. Washington, D.C.: U.S. Government Printing Office.

U.S. Water Resources Council (1968) The Nation's Water Resources: The First National Appraisal. Washington, D.C.: U.S. Government Printing Office.

APPENDIX B

COMPUTATION OF INCREMENTAL COSTS PER KILOGRAM CONTROLLED FOR EXISTING AND PROPOSED EPA REGULATIONS

This Appendix provides computations for each of the average cost figures of Table 2.1. The sources for the data are Versar, Inc. (1977), Westin et al. (1978), and Steven Malkenson, Office of Planning and Management, U.S. Environmental Protection Agency.

A. Drain and Incinerate Askarel Fluids from PCB Transformers.

Each transformer contains approximately 1,364 kg of fluid of which 977 kg are PCBs. Draining is 90 percent effective, yielding 1,228 kg of fluid (880 kg of PCBs) for handling and processing. A cost of $536 or $0.61/kg of PCB is derived based on the following:

| | |
|---|---|
| Transportation charge, $0.07/kg | $ 81 |
| Incineration charge, $0.33/kg | 405 |
| Handling charge per unit | 50 |
| | $536 |

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B. Shred and Incinerate High Voltage Capacitors.

If each capacitor weighs 55 kg and contains 11 kg of PCBs, the cost-per-kg destroyed is $3.86.

| | |
|---|---|
| Handling and transportation charge, $0.13/kg | $ 7.15 |
| Incineration charge, $0.55/kg | 30.35 |
| Handling charge per unit | 5.00 |
| | $42.50 |

C. Shred and Incinerate Low Voltage Capacitors.

If each capacitor weighs 9.1 kg and contains 1.6 kg of PCBs, a cost of $7.19 or $4.49/kg is derived.

| | |
|---|---|
| Handling and transportation charge, $0.13/kg | $1.18 |
| Incineration charge, $0.55/kg | 5.00 |
| Handling charge per unit | 1.00 |
| | $7.19 |

D. Flush and Drain Transformers and Incinerate Fluid.

Each transformer uses 1,022 liters of solvent at $0.10/l. Contaminated solvent weighing 1,350 kg must be shipped and incinerated at a cost of $0.40/kg (see A). If the process removes 90 percent (or 88 kg) of the PCBs remaining after draining, the cost-per-kg of PCBs destroyed will be $8.50/kg.

| | |
|---|---|
| Solvent | $108.00 |
| Transportation and incineration | 540.00 |
| Handling charge per unit | 100.00 |
| | $748.00 |

F. Place Transformer Carcasses in Chemical Waste Landfill.

Disposal costs are estimated at $3/ft³ or $0.07/kg with transportation costs of $0.04/kg. The average weight of the transformer is 2,955 kg and cost per unit is $325. If each unit contains 9.8 kg of PCBs, the cost-per-kg of PCBs immobilized will be $33.16.

[illegible] Contaminated [illegible]

Westin et al. (1978) estimates that 200 units containing 500 kg to 750 kg of PCBs per unit would require fluid replacement at a total cost of $2.8 million to $3.4 million or $18.67/kg to $35.00/kg.

G. Replace Motors Containing PCBs in Mining Equipment.

The following estimates are based on Westin et al. (1978). The regulations would force the rebuilding of 652 loader motors containing 20 kg of PCBs each at a cost of $3,100 or $155/kg. Equipment in service contains a total of 1,145 kg of PCBs. Costs of premature scrapping or replacement of these machines would range from $640,000 to $2,240,000 or $560/kg to $1,955/kg.

H. Drain and Incinerate Mineral Oil Transformer Fluids if Contaminated by More Than 50 mg/kg PCBs.

Westin et al. (1978) estimates that the cost of testing transformer fluids and subsequent incineration if necessary would range from $613 million to $769 million. It is estimated that transformer fluids may be contaminated by 175,000 kg of PCBs. If all of the PCBs were detected and removed, the average cost would be $3,500/kg to $4,400/kg. However, this calculation may underestimate the average cost of destruction of PCBs if in the absence of regulation, contaminated fluids were normally used as utility boiler fuel, some fraction of the PCBs would be destroyed in any event. If 95 percent of the PCBs are destroyed when combustion occurs in utility boilers, then the average destruction attributable to the regulation is only 8,750 kg (0.05 x 175,000), in which case the average cost of destruction of PCBs is $70,000/kg to $88,000/kg. It should be noted however that the evidence on effective destruction of PCBs in utility boilers is scanty. If for example the destruction rate were as low as 50 percent, the average cost in Table 2.1 would be lower by an order of magnitude.

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1. Flush and Replace Contaminated Hydraulic Fluids in Die-Casting.

Westin et al. (1978) estimate a cost per machine of $14,600 and an average PCB contamination of 0.11 kg/machine or $133,000/kg. However, there may be substantial variation in the degree of contamination in different machines. True average costs would be much lower for a policy directed only at the most contaminated machines, but would rise as the policy was extended to machines with lesser contamination.

REFERENCES

Versar, Inc. (1977) Microeconomic Impacts of the Proposed Marking and Disposal Regulations for PCBs. NTIS Report No. PB-267 833/2WP. Springfield, Va.: National Technical Information Service.

Westin, R.A., L. Fourt, D. Berkey, and B. Woodcock (1978) Microeconomic Impacts of the Proposed "PCB Ban Regulations." National Technical Information Service, NTIS Report No. PB 281-881/3WP. Springfield, Va.: Versar, Inc.

APPENDIX C

THE BASIS FOR THE VALUE OF STATISTICAL LIFE

A number of approaches to assigning moretary values to the loss of life have been proposed or used in the literature on the economics of health and safety. The approaches can be broadly categorized by the way they determine values, i.e., either according to individual preferences (willingness to pay) or according to resource or opportunity costs. The following is a brief evaluation of the two approaches. (For more extensive discussions of alternative concepts of the value of life or safety, see Schelling 1968, Mishan 1971, Acton 1973, and Jones-Lee 1976.)

Willingness to Pay

Conceptually, the more attractive approach is to value increases in longevity or reductions in the probability of death due to accident or illness according to what an individual is willing to pay to achieve them. Individuals often act as if life expectancy were like any good, that is, they are willing to trade life money for goods and services that they value more and vice versa. Individuals make decisions that

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can result in reductions in life expectancy or increased probability of death in return for increases in income or other goods and services, and perceive themselves to be better off because of these choices. Examples would be accepting risky or hazardous jobs because of higher wages, or travelling by air instead of on the ground because air travel is quicker and more convenient.

As these examples make clear, the question is not how much an individual would be willing to pay to avoid certain death tomorrow, but rather how much he would be willing to pay to achieve a small decrease in the probability of death during a given period. This willingness to pay can be translated into a more useful figure for policy evaluation, namely, the value of statistical life or the value of a statistical death avoided. Suppose that in a group of 1,000 similar individuals, each would be willing to pay $1,000 for a policy that would reduce the probability of death by 0.01. This policy is a form of collective good for the individuals involved. The benefit to the group is found by adding across all individuals. Thus, the aggregate willingness to pay would be $1 million (1,000 x $1,000), and the expected number of deaths avoided would be 10. The statistical value per life would be $1 million divided by 10 or $100,000.

Two approaches to the empirical measurement of willingness to pay have been used in the literature. One is to observe market transactions where individuals actually purchase or sell changes in their risk levels. For example, if wage differentials among occupations are related to differences in occupational risk levels, these differences might be interpreted as reflecting, at the margin, the individuals' trade-off between risk/safety and money. The other approach is to conduct surveys that ask individuals a series of questions about hypothetical situations involving risk/money trade-offs. If the questions are carefully designed, and if individuals are truly capable of predicting how they would act if placed in these hypothetical situations, their answers may reveal the monetary values they attach to reductions in risk.

There have been two major studies using wage rates to estimate willingness to pay. Both studies are based on the same wage data but they employ different sets of occupational risk data and thus present quite different results. Thaler and Rosen (1976) concluded that the statistical value of life lies between $140,000 and

[illegible] estimates ranged between $[illegible]

There are also differences in the [illegible] efforts to use the survey approach to obtain values for statistical life. Acton (1973) obtained 36 responses from a stratified random sample of residents in the Boston area. The survey instrument contained a number of questions about attitudes and value judgments with respect to efforts to save lives in emergency situations. The question of greatest interest concerned the willingness of the individual to pay for a program of emergency coronary care that would reduce the probability of his death from heart attack. Two different forms of the question implied values for statistical life of $25,000 and $45,000. Jones-Lee (1976) asked a similar small sample of individuals several questions about their willingness to accept higher air fares to travel on lines that had lower probabilities for a fatal crash. The value of statistical life implied by the respondents was about $5 million. The disparities suggest that substantially more research is required before reliable estimates of the value of life can be obtained with such techniques. But the results may suggest order of magnitude bounds on appropriate figures.

## Opportunity Cost/Human Capital

The more common approach to the valuation of life is the opportunity cost, productivity, or human capital approach which values each life lost at the present level of the earnings that an individual could have expected had the death not occurred. The approach is based on the assumption that earnings reflect the individual's marginal productivity, or the individual's contribution to total economic output.

There are three criticisms of this approach. First, it has no necessary relationship to individual willingness to pay. Second, it allows no role for the probabilistic nature of death and the avoidance of death in the health and safety areas, or for different individual attitudes or preferences toward risk and its [illegible]. By definition an individual could pay no [illegible] the current value of his earnings to avoid [illegible], although the statistical value of life

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based upon willingness to pay for small probability changes could be several times the discounted expected earnings stream. Third, the implicit judgment underlying the approach is that an individual is worth what he or she does, or that productivity is the measure of worth.

Because of variation in patterns of earnings over a lifetime and differences among individuals in their experiences in the labor market (including the factor of discrimination), values derived from this approach relate crucially to age, sex, and race. For example, using a 6 percent discount rate and 1972 patterns of earnings, saving the life of a white male between 30 and 34 years of age would prevent the loss of about $180,000 in discounted lifetime earnings. For a nonwhite woman of the same age, the earnings loss prevented would be only about $70,000 (Cooper and Rice 1976). Some benefit-cost analyses using productivity values have attempted to adjust for these factors. Others have used averages for the population as a whole, while still other studies have added estimates of the direct costs of illness as represented by such items as doctors' fees and hospital bills.

Judgmentaliy Derived Shadow Prices

As indicated above, empirical estimates of the value of statistical life vary within two orders of magnitude, and any of them can be questioned on statistical and conceptual grounds. An alternative to the empirical estimate is the judgmentally determined value or range of values. There are several instances of this approach in the literature on the evaluation of health and safety benefits.

The earliest official use of an explicit value of life in the analysis of proposed governmental strategies appeared in the RECAT report (Office of Science and Technology 1972) in which a life was valued at $140,000 in estimating the costs of fatalities associated with automobile accidents. The figure represents the average foregone participation in the personal income of the nation; it is the product of per capita personal income in 1970 times the estimated reduction of the expected life span of the average accident victim. Another use of a value of life occurred in a report by the National Research Council (1974) where the value of improved

[illegible] yield from reducing gaseous residual [illegible] from automobiles was arrived at on the basis [illegible],000 per death avoided. Another report by the National Research Council (1978) used a range of $100,000 to $1,000,000 per death avoided. Most of the more broadly based empirically derived estimates cited above reside within this range. For these reasons we chose to use this range for the calculations in the text.

REFERENCES


Acton, J.P. (1973) Evaluating Public Programs to Save Lives: The Case of Heart Attacks. Santa Monica, Calif.: Rand Corporation.

Cooper, B.S., and D.P. Rice (1976) The economic cost of illness revisited. Soc. Security Bull. 39(2):21-36.

Jones-Lee, M.W. (1976) The Value of Life: An Economic Analysis. Chicago: University of Chicago Press.

Mishan, E.J. (1971) Evaluation of life and limb: A theoretical approach. J. Political Econ. 79(4):687-705.

National Research Council (1974) Air Quality and Automobile Emission Control, Volume 4: The Costs and Benefits of Automobile Emission Control. A report by the Coordinating Committee on Air Quality Studies, Commission on Natural Resources, National Academy of Sciences/National Academy of Engineering for the Committee on Public Works, U.S. Congress Senate. Serial No. 93-24, 93rd Congress, 2nd Session.

National Research Council (1978) Chloroform, Carbon Tetrachloride, and Other Halomethanes: An Environmental Assessment. Committee for Scientific and Technical Assessments of Environmental Pollutants, Environmental Studies Board, Commission on Natural Resources, Washington, D.C.: National Academy of Sciences.

Office of Science and Technology (1972) Cumulative Regulatory Effects on the Cost of Automotive Transportation. Report of the Ad Hoc Committee. Washington, D.C.: U.S. Government Printing Office.

[illegible]ling, T.C. (1968) The life you save may be your [illegible]. In Problems in Public Expenditure Analysis, [illegible]ted by S.B. Chase, Jr. Washington, D.C.: The [illegible] Institution.


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Smith, R.S. (1976) The Occupational Safety and Health Act: Its Goals and Its Achievements. Washington, D.C.: American Enterprise Institute for Public Policy Research.

Thaler, R.H. and S. Rosen (1976) The value of saving a life: Evidence from the labor market. In Household Production and Consumption. Edited by N.E. Terleckyj. New York: Columbia Univeraity Press.

APPENDIX D

CHEMICAL AND TOXICITY DATA AS REQUIRED BY FIFRA AND TSCA GUIDELINES

*CHEMICAL IDENTITY*

Product Identity

The PCB product contains a mixture of chlorinated biphenyls; each biphenyl can be present as any one of a number of isomers. Table D.1 lists the chemical name and registry number for the most common (IARC, in press).

Synonyms for PCBs include chlorinated biphenyl, chlorinated diphenyl, chlorobiphenyl, polychlorinated biphenyl, and polychlorobiphenyl.

The following trade names have been used:

| | | | | | |
|---|---|---|---|---|---|
| USA: | Aroclor | France: | Phenoclor | Italy: | Fenclor |
| | Chloretol | | | | |
| | Dykanol | Japan: | Kanechlor | USSR: | Soval |
| | Inerteen | | Santotherm | | |
| | Noflamol | | | | |
| | Pyranol | FRG: | Clophen | | |

The molecular formula and weight, percent chlorine [illegible] of isomers are shown in Table D.2 (Durfee et

# TABLE D.1 Chemical Abstracts Register Number for PCBs

Biphenyl

Chem Abstr Services Reg. No : 92-52-4
Chem Abstr. Name 1,1'-biphenyl

2-Chlorobiphenyl

Chem Abstr Services Reg. No : 2051-60-7
Chem Abstr Name 2-Chloro-1,1'-biphenyl

4-Chlorobiphenyl

Chem Abstr Services Reg No : 2051-62-9
Chem Abstr Name 4-Chloro-1,1'-biphenyl

2,2'-Dichlorobiphenyl

Chem Abstr Services Reg No 13029-08-8
Chem Abstr Name 2,2 -Dichloro-1,1 -biphenyl

3,3'-Dichlorobiphenyl

Chem Abstr Services Reg No 25569-80-6
Chem Abstr Name: 3,3'-Dichloro-1,1'-biphenyl

2,4'-Dichlorobiphenyl

Chem Abstr Services Reg No 34883-43-7
Chem Abstr Name 2,4'-Dichloro-1,1'-biphenyl

4,4'-Dichlorobiphenyl

Chem Abstr Services Reg No 2050-68-2
Chem Abstr Name 4,4'-Dichloro-1,1'-biphenyl

2,2',3-Trichlorobiphenyl

Chem. Abstr Services Reg. No 38444-78-9
Chem Abstr Name 2,2',3-Trichloro-1,1'-biphenyl

2,3',4-Trichlorobiphenyl

Chem Abstr Services Reg No none available
Chem Abstr Name 2,3',4-Trichloro-1,1'-biphenyl

,4,4 -Trichlorobiphenyl

-- Abstr Services Reg. No : 7012-37-5
- Abstr Name 2 4,4'-Trichloro-1,1'-biphenyl

[illegible]-Trichlorobiphenyl

-- Abstr Services Reg No 37680-65-2
- Abstr Name: 2,2',5-Trichloro-1,1'-biphenyl

,4 5-Trichlorobiphenyl

Chem Abstr Services Reg No.: 16606-02-3
Chem Abstr Name 2,4',5-Trichloro-1 1'-biphenyl

2,2',3,5',6-Pentachlorobiphenyl

Chem Abstr. Services Reg No : 38379 99-6
Chem Abstr Name 2,2' 3,5',6-Pentachloro-1,1'-biphenyl

2,3,3',4',6-Pentachlorobiphenyl

Chem Abstr Services Reg No 38380 01-9
Chem Abstr Name 2,3,3',4',6-Pentachloro-1,1'-biphenyl

2,3 ,3 4,5-Pentachlorobiphenyl

Chem Abstr Services Reg No.: 41464-51-1
Chem Abstr Name 2,3',3',4,5-Pentachloro-1,1'-biphenyl

2,2' 4,4',5-Pentachlorobiphenyl

Chem Abstr Services Reg. No.: 38380-01-7
Chem Abstr Name 2,2',4,4',5-Pentachloro-1,1'-biphenyl

2,3 4 5 5'-Pentachlorobiphenyl

Chem Abstr Services Reg No : 37680-73-2
Chem Abstr Name 2,2',4,5,5'-Pentachloro-1,1'-biphenyl

,3 4 4',5-Pentachlorobiphenyl

Chem Abstr Services Reg No 31508-00-6
Chem Abstr Name: 2,3',4',5-Pentachloro-1,1 -biphenyl

2,3',3,4 4 5-Hexachlorobiphenyl

[illegible] Abstr Services Reg No 35694-06-5
[illegible] Abstr Name 2 3 4 4 ,5-Hexachloro-1,1'-biphenyl

[illegible] 3 3 4 [illegible] 5-Hexachlorobiphenyl

[illegible] Abstr Services Reg No 38380-04-1
[illegible] Abstr Name 3,3 4,5-Hexachloro-1,1'-Biphenyl

2' 1,4-Trichlorobiphenyl

Chem Abstr Services Reg No 38444-86-9
Chem Abstr Name 2',3,4-Trichloro-1,1'-biphenyl

3,3,4,4'-Tetrachlorobiphenyl

Chem Abstr Services Reg. No.: 32025-41-1
Chem Abstr Name 3,3,4,4'-Tetrachloro-1,1'-biphenyl

2,3',3,5'-Tetrachlorobiphenyl

Chem. Abstr Services Reg. No. 41464-39-5
Chem Abstr Name: 2,3',3,5'-Tetrachloro-1,1'-biphenyl

2,3',4,5'-Tetrachlorobiphenyl

Chem Abstr. Services Reg. No : 41464-40-8
Chem Abstr Name: 2,3',4,5'-Tetrachloro-1,1'-biphenyl

2,3',4,4'-Tetrachlorobiphenyl

Chem Abstr Services Reg No.: 32598-10-0
Chem Abstr Name: 2,3',4,4'-Tetrachloro-1,1'-biphenyl

2,3',5,5'-Tetrachlorobiphenyl

Chem Abstr Services Reg. No.: 25693-99-3
Chem. Abstr. Name: 2,3',5,5'-Tetrachloro-1,1'-biphenyl

2,3',4',5-Tetrachlorobiphenyl

Chem Abstr Services Reg. No.: 32598-11-1
Chem. Abstr. Name: 2,3',4',5-Tetrachloro-1,1'-biphenyl

3,3',4,4'-Tetrachlorobiphenyl

Chem Abstr Services Reg. No. 32598-13-3
Chem Abstr Name: 3,3',4,4'-Tetrachloro-1,1'-biphenyl

2,3,3',4,4'-Pentachlorobiphenyl

Chem Abstr. Services Reg. No.: 32598-14-4
Chem. Abstr. Name: 2,3,3',4,4'-Pentachloro-1,1'-biphenyl

2,3',3,4,5'-Pentachlorobiphenyl

Chem Abstr Services Reg. No.: 38380-02-8
Chem. Abstr Name: 2 3',3,4,5'-Pentachloro-1,1'-biphenyl

2,3',3,3',6-Pentachlorobiphenyl

Chem Abstr. Services Reg. No. 52663-60-
Chem. Abstr. Name: 2 3',3,3',6-Pentachloro-1,1 -biphenyl

2,3',3,4',5-Pentachlorobiphenyl

Chem Abstr. Service: Reg. No none available
Chem. Abstr Name: 2,3',3,4',6-Pentachloro-1,1'-biphenyl

2,3',3 4,4',5'-Hexachlorobiphenyl

Chem Abstr Services Reg. No . 35065-28-2
Chem. Abstr Name: 2,3',3,4,4',5'-Hexachloro-1 1'-biphenyl

2,2',3,3',6,6'-Hexachlorobiphenyl

Chem Abstr Services Reg No.: 38411-22-2
Chem Abstr Name: 2,2',3,3',6,6'-Hexachloro-1,1'-biphenyl

2,2',4,4',5,5'-Hexachlorobiphenyl

Chem Abstr Services Reg No 35065-27-1
Chem Abstr. Name: 2, ',4,4',5,5'-Hexachloro-1,1 -biphenyl

2,2',3',4,5,6'-Hexachlorobiphenyl

Chem Abstr Services Reg. No 38380-04-0
Chem Abstr Name 2, ,3',4,5,6'-Hexachloro-1,1'-biphenyl

2 2',3,3',4,4',5-Heptachlorobiphenyl

Chem Abstr Services Reg No 35065-30-6
Chem Abstr Name 2,2 ,3,3',4,4',5-Heptachloro-1,1 -biphenyl

2,2',3 3',4 5,6'-Heptachlorobiphenyl

Chem Abstr Services Reg No 38411-25-5
Chem Abstr Name 2,2 ,3,3',4,5,6'-Heptachloro-1,1 -biphenyl

2,2',3,4,4',5 5'-Heptachlorobiphenyl

Chem Abstr Services Reg No 35065-29-1
Chem Abstr Name 2,2 ,3,4,4',5,5'-Heptachloro-1,1 -biphenyl

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TABLE D.2 Composition of Chlorinated Biphenyls

| Empirical Formula Chlorobiphenyls | Molecular Weight[a] | Percent Chlorine[a] | No. of Isomers |
|---|---|---|---|
| $C_{12}H_{10}$ | 154 | 0 | 1 |
| $C_{12}H_9Cl$ | 188 | 18 | 3 |
| $C_{12}H_8Cl_2$ | 222 | 31 | 12 |
| $C_{12}H_7Cl_3$ | 256 | 41 | 24 |
| $C_{12}H_6Cl_4$ | 290 | 48 | 42 |
| $C_{12}H_5Cl_5$ | 324 | 54 | 46 |
| $C_{12}H_4Cl_6$ | 358 | 58 | 42 |
| $C_{12}H_3Cl_7$ | 392 | 62 | 24 |
| $C_{12}H_2Cl_8$ | 426 | 65 | 12 |
| $C_{12}HCl_9$ | 460 | 68 | 3 |
| $C_{12}Cl_{10}$ | 494 | 79 | 1 |

[a]Based on $Cl^{35}$

SOURCE Modified from Durfee et al. (1976).

al. 1976). The composition of some commercial mixtures from the United States, Italy, Germany, and Japan are illustrated in Table D.3 (IARC, in press; Durfee et al. 1976; Nisbet 1976).

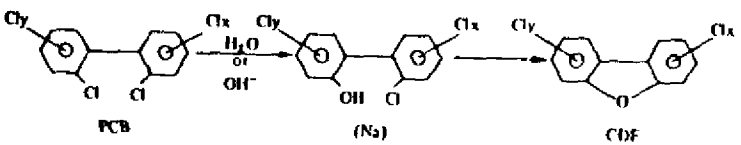
## Manufacturing Process

PCBs are produced by chlorination of biphenyl. The process, illustrated in Figure D.1, requires biphenyl, anhydrous chlorine, a catalyst such as iron filings or ferric chloride, and elevated temperatures. The crude product is purified by alkali wash, sometimes followed by distillation (Durfee 1976; IARC, in press; Nisbet 1976; Durfee et al. 1976). A detailed discussion of the manufacturing process of Monsanto (the U.S. producer) is given by Durfee.

The chemical composition of the final product varies with the degree of chlorination and with the blending process employed. Batches of PCBs with the same chlorine content may vary in composition and the

process does not lead to the formation of [illegible] ingredients measurable at 0.01% product weight or more of product weight (Nisbet 1976). However, small amounts (see Table 3.2) of chlorodibenzofuran are detected in PCB mixtures, and quantities may vary from batch to batch.

The fact that different batches of PCBs, even from the same manufacturer, have different amounts of chlorodibenzofurans (CDFs) appears to be a reflection of variabilities in the production process. Whereas the primary process in the production of PCBs by the chlorination of biphenyl would not give CDFs, the aqueous alkaline washes and steam distillations could be anticipated to give the small amounts of CDFs observed, as shown in the following example.




Such a scheme could have been involved in the Japanese "Kanemi Yusho" incident, where the PCB-contaminated rice oil was heated with steam at 200°, and where a relatively high ratio of CDFs to PCBs was observed, although Japanese investigators of the problem suggested an alternative route of contamination (Miyata et al. 1977).

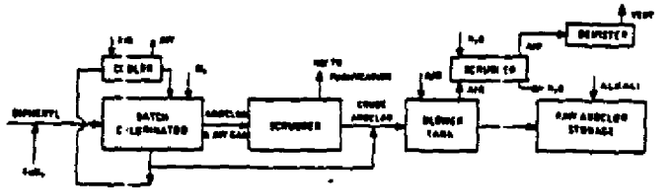
## PHYSICAL AND CHEMICAL DATA

These data have not been reported in a consistent manner in the scientific literature. Chemical and physical properties have been investigated for individual chlorobiphenyls in some instances and in others for the commercial product or mixture. An additional problem [illegible] because two grades of PCBs existed. For pearl/Aroclor mentioned, a darker, less pure grade was [illegible]. The physical and chemical characteristics of [illegible] technical grades were about the same, but

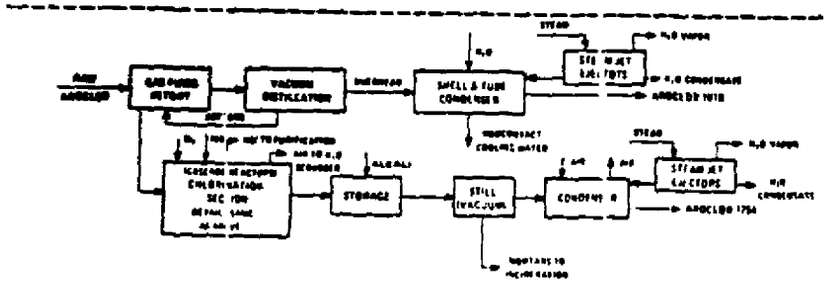
TABLE D.3 Approximate Percent Composition of Some Commercial PCB Products

| Chlorobiphenyl | Aroclor Type or Grade 1016 | 1221 | 1232 | 1242 | 1248 | 1254 | 1260 | Kanechlors KC-300 | KC-400 | KC-500 | Fenclors DK | Clophen A60 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| $C_{12}H_{10}$ | <0.1 | 11 | 6 | <0.1 | - | <0.1 | - | - | - | - | - | - |
| $C_{12}H_9Cl$ | 1 | 51 | 28 | 1 | - | <0.1 | - | - | - | - | - | - |
| $C_{12}H_8Cl_2$ | 20 | 32 | 28 | 16 | 2 | 0.5 | - | 17 | 1 | - | - | - |
| $C_{12}H_7Cl_3$ | 57 | 4 | 24 | 49 | 18 | 1 | - | 60 | 53 | 9 | - | 7 |
| $C_{12}H_6Cl_4$ | 21 | 2 | 15 | 25 | 40 | 21 | - | 23 | 44 | 37 | - | 55 |
| $C_{12}H_5Cl_5$ | 1 | 0.5 | 0.5 | 8 | 36 | 48 | 12 | 0.6 | 14 | 35 | - | 35 |
| $C_{12}H_4Cl_6$ | <0.1 | - | - | 1 | 4 | 23 | 38 | - | 5 | 13 | - | 3 |
| $C_{12}H_3Cl_7$ | - | - | - | <0.1 | - | 4 | 41 | - | - | - | - | - |
| $C_{12}H_2Cl_8$ | - | - | - | - | - | - | 8 | - | - | - | - | - |
| $C_{12}H_1Cl_9$ | - | - | - | - | - | - | 1 | - | - | - | - | - |
| $C_{12}Cl_{10}$ | - | - | - | - | - | - | - | - | - | - | 100 | - |

SOURCE: IARC (in press), Nisbet (1976), Durfee et al. (1976).



Preparation of crude chlorinated biphenyls



Distillation of crude products

SOURCE: Durfee (1976)

FIGURE D.1 Monsanto-Swift chemical process for PCBs



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the technical grades were lower in price (Monsanto Technical Bulletin O/PL 306A). Impurities in lower grade Aroclors have not been determined, nor have these grades been characterized toxicologically. It is not known what percentage of total Aroclor usage in the United States was represented by the technical grade product. As far as we know, all toxicological studies have been done with the purified grades, and therefore data on technical grades are not available. Thus it is not possible to assess the toxicological effects that may be realized from exposure to technical grades of PCBs. (See Tables D.4-D.7.)

## ENVIRONMENTAL CHEMISTRY DATA

### Physical-Chemical Degradation

Hydrolysis PCBs may enter natural water through industrial discharge or as a result of disposal and cleanup of equipment. However, they are inert to hydrolysis except under extreme conditions (Hutzinger et al. 1974, Nisbet 1976).

Chemical Degradation PCBs are extremely stable compounds, not reactive chemically under environmental conditions. Oxidation, reduction, nitration, isomerization and nucleophilic reactions (reactions with alkali, alkoxides, amines) have been described (Hutzinger et al. 1974). The conditions required for these reactions make it unlikely that they will take place in the environment at any significant rate.

Some applications of PCBs, such as quenching of heated metals, heat-transfer media, and transformer oils, may lead to the formation of degradation products that can escape into the environment. The formation of chlorinated dibenzofurans on heating of PCBs to 500-600 C is one example (Buser et al. 1978). Polymeric products of unknown composition, possibly incorporating oxygen, are also formed from PCBs upon heating (Zitko and Choi 1971). Chemical contaminants (penta- to decachloroquaterphenyls, hepta- to nonachloro-quaterphenyl ethers, and penta- to octachloroterphenyls), considered to be a result of dimerization were identified in Kanemi (Yusho) rice oil (Miyata et al. 1977). About 90 percent of this mixture consisted of the quaterphenyls, and their concentration, [illegible] exceeded, by 4 times, the concentration of [illegible], the analyses and identifications were [illegible] using saponified oil samples, and further [illegible]rmation of the results is required.

Photodegradation Photoreaction of PCBs produces a variety of products. Loss of chlorine, with replacement by hydrogen or hydroxyl groups, rearrangement, condensation, and "polar" products have been observed (Hutzinger et al. 1974.)

One of the reaction products of sunlight irradiation of PCB mixtures is the replacement of chlorine by hydroxy groups at the ortho positions. This allows oxygen to bind in a similar position on the other ring and to produce chlorodibenzofurans. Both heat and light can accelerate the transformation of PCBs to [illegible] (Hutzinger et al. 1974).

It is not possible to assess quantitatively the importance of photodegradation of PCBs in the environment. A major portion of PCBs in the environment is not accessible to sunlight and, consequently, the amount of PCBs decomposed by photodegradation is likely to be very small. Table D.8 provides extinction coefficients and quantum yield data for the loss of a given PCB isomer (i.e., transformation by loss of a chlorine atom and formation of a PCB molecule with one less chlorine).

### Biodegradation

Microbial degradation of PCBs depends on the degree of chlorination and the position of the chlorine atom on the biphenyl molecule. Lower chlorinated biphenyls are readily transformed by bacteria, but the higher chlorinated compounds are not (Nisbet 1976). Achromobacter isolated from sewage sludge rapidly degrades PCBs (Ahmed and Focht 1973). Kaiser and Wong (1974) also established microbial degradation using a bacterial culture isolated from lake water. Bacterial growth was not affected adversely when up to 0.1 percent Aroclor was added to the culture (Wong and Kaiser 1975). (See Table D.9.)

Biodegradation rates of chlorobiphenyls may be derived from the data of Furukawa et al. (1978b) using [illegible] of Alcaligenes and Acinetobacter. From these

TABLE D.4 Properties of Selected Aroclors

| Property | 1016 | 1221 | 1232 | 1242 | 1248 | 1254 | 1260 |
|---|---|---|---|---|---|---|---|
| Color (APHA) | - | 100 | 100 | 100 | 100 | 100 | 150 |
| Physical state | mobile oil | mobile oil | mobile oil | mobile oil | mobile oil | viscous liquid | sticky resin |
| Stability | inert | inert | inert | inert | inert | inert | inert |
| Density (lb/gal 25°C) | 11.40 | 9.85 | 10.55 | 11.50 | 12.04 | 12.82 | 13 50 |
| Specific gravity x/15 5°C | 1.36-1.37 x-25° | 1.18-1.19 x-25° | 1.27-1.28 x-25° | 1.30-1.39 x-25° | 1.40-1.41 x-65° | 1.49-1.50 x-65° | 1.55-1 56 x-90° |
| Distillation range (°C) | 323-356 | 275-320 | 290-325 | 325-366 | 340-375 | 365-390 | 385-420 |
| Acidity mg KOH/g, maximum | - | .014 | .014 | .015 | .010 | .010 | .014 |
| Fire point (°C) | none to boiling point | 176 | 238 | none to boiling point | none to boiling point | none to boiling point | none to boiling point |
| Viscosity (100°F) | 71-81 | 38-41 | 44-51 | 82-92 | 185-240 | 1400-2500 | - |

SOURCE: Durfee et al. (1976).

Melting Points (°C) for Some PCB Isomers

| | | | |
|---|---|---|---|
| Biphenyls | | Pentachlorobiphenyls | |
| | 70 | | |
| | 34 | 2,3,3',4,4'- | 117-118.5 |
| | 77 7 | 2,2',3,4,5'- | 111 5-113 |
| | | 2,2',3,3',6- | |
| Fluorobiphenyls | | 2,2',3,4',6- | |
| | | 2,2',3,5',6- | 98.5-100 |
| 2,2'- | 60.5 | 2,3,3',4',6- | |
| 2,3'- | | 2,2',3',4,5- | 81-82 |
| 2,4'- | 44.5-46 | 2,2',4,4',5- | |
| 4,4'- | 149-150 | 2,2',4,5,5'- | 76.5-77.5 |
| | | 2,3',4,4',5- | 112-113 |
| Trichlorobiphenyls | | Hexachlorobiphenyls | |
| 2,2',3- | 28.1-28.8 | | |
| 2,3',4- | | 2,2',3,4,4',5- | 77-78 |
| 2,4,4'- | 57-58 | 2,2',3,3',4,6- | |
| 2,2',5- | 43-44 | 2,2',3,4,4',5'- | 78.5-80 |
| 2,4',5- | 67 | 2,2',3,3',6,6'- | 114-114 5 |
| 2',3,4- | 65-66 | [illegible] | 103-104 |
| | | 2,2',3',4,5,6'- | |
| Tetrachlorobiphenyls | | Heptachlorobiphenyls | |
| 2,3,4,6'- | 182 | | |
| 2,2',3,5'- | 49-50 | 2,2',3,3',4,4',5- | 134.5-135.5 |
| 2,2',4,5'- | 66-68.5 | 2,2',3,3',4,5,6'- | 130 5-130.7 |
| 2,3',4,4'- | 127-127.5 | 2,2',3,4,4',5,5'- | 109-110 |
| 2,2',5,5'- | 87-89 | | |
| 2,3',4',5- | 104-105 | | |
| 3,3',4,4'- | 182-184 | | |

SOURCE: Modified from IARC (in press).

data, the mean primary biodegradation rates were calculated (Table D.10).

Considerable differences exist between the primary biodegradation rates of isomers (Furukawa et al. 1978b). Those substituted in 2 and 6 positions" are degraded much more slowly than others, but the rates given in Table D.10 would probably be useful in designing a model.

According to Baughman and Lassiter (1978), representative concentrations of microbes in different water bodies are:

| Water Body | cell/ml |
|---|---|
| Pond | $10^5$ |
| Stream | $10^4$ |
| Eutrophic Lake | $10^5$ |
| Oligotrophic Lake | 10 |
| Sediments | $10^6$ |

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TABLE D.6 Solubility of Chlorobiphenyl Isomers and Aroclors in Water (mg/l)

| | | | |
|---|---|---|---|
| **Monochlorobiphenyls** | | **Pentachlorobiphenyls** | |
| 2- | 5.90 | 2,2',3,4,5'- | $2.2 \times 10^{-2}$ |
| 3- | 3.50 | 2,2',4,5,5'- | $3.1 \times 10^{-2}$ |
| 4- | 1.19 | | |
| **Dichlorobiphenyls** | | **Hexachlorobiphenyl** | |
| | | 2,2',4,4',5,5'- | $0.8 \times 10^{-2}$ |
| 2,4- | 1.40 | | |
| 2,2'- | 3.50 | **Octachlorobiphenyl** | |
| 2,4'- | 1.80 | | |
| 4,4'- | 0.08 | 2,2',3,3',4,4',5,5'- | $0.7 \times 10^{-2}$ |
| **Trichlorobiphenyls** | | **Decachlorobiphenyl** | $1.5 \times 10^{-2}$ |
| 2,4,4'- | $0.5 \times 10^{-2}$ | | |
| 2',3,5- | $7.8 \times 10^{-2}$ | | |
| **Tetrachlorobiphenyls** | | | |
| 2,2',5,5'- | $4.6 \times 10^{-2}$ | | |
| 2,2',3,3'- | $3.4 \times 10^{-2}$ | | |
| 2,2',3,5'- | $17.0 \times 10^{-2}$ | | |
| 2,2',4,4'- | $6.8 \times 10^{-2}$ | Aroclor 1242 | 0.24 |
| 2,3',4,4'- | $5.0 \times 10^{-2}$ | 1248 | $5.40 \times 10^{-2}$ |
| 2,3',4',5- | $4.1 \times 10^{-2}$ | 1254 | $1.10 \times 10^{-2}$ |
| 3,3',4,4'- | $10.0 \times 10^{-2}$ | 1260 | $0.30 \times 10^{-2}$ |

SOURCE Modified from Durfee et al. (1976)

Attempts to extrapolate the rates to environmental conditions introduce considerable uncertainty. The rates were measured under optimum conditions for the microorganism, at cell concentrations from $4.4 \times 10^8$ to $2 \times 10^9$ cells/ml and chlorobiphenyl concentrations of 50 nmol/ml, which is 11 mg/l to 15 mg/l for mono- to tetrachlorobiphenyls, respectively, well above the solubility of di- through tetrachlorobiphenyls (Weil et al. 1974).

If we assume that the rate depends linearly on the concentration of chlorobiphenyls, extrapolation to the "normal" ambient concentration in the ng/l range would involve multiplication by $10^6$. About 10 percent of the cells may be degrading PCBs (Wong and Kaiser 1975). Therefore, the rates should be multiplied by 10 percent of the normal cell concentration. The results of these computations are given in Table D.11.

Note that the biodegradation of most pentachlorobiphenyls may be extremely slow, and that of hexa- and more substituted chlorobiphenyls practically



[illegible] rates are comparable to the rates of [illegible] activated sludge illustrated in Figure

[illegible]ects of PCBs on microorganisms have been studied by many investigators and results are presented in Table D.12 (Bourquin and Cassidy 1975, Ewald et al. 1976, Nisbet 1976, Powers et al. 1977, Blakemore 1978, Blakemore and Carey 1978, Furukawa et al. 1978a).

PCBs may induce changes in the ecological role of microorganisms because of different sensitivities. Thus, community composition may change depending on level of PCBs, temperature, and the ability of the organisms to metabolize these compounds (O'Connors et al. 1978).

## Mobility Through the Environment

**Adsorption and Leaching** Laboratory tests indicate that Aroclor 1254 is adsorbed most efficiently on soils with a high organic content and on clays. Aroclor 1016 was adsorbed strongly on silty clay loam soil and could be leached only with difficulty from silt and sandy loams. Less than 0.05 percent was leached after 4 months, and it consisted of mainly mono- and dichlorobiphenyls (Haque et al. 1974, Muto et al. 1974, Tucker et al. 1975a, Haque and Schmedding 1976, Nisbet 1976).

Transfer of PCB isomers from soil to water follows closely their physical properties, especially water solubility and partition coefficients as indicated in Table D.13 (Halter and Johnson 1977).

**Volatility** Monsanto Technical Bulletin O/PL-306A gives a graph of vapor pressure (VP, mm Hg) of Aroclors 1242, 1248, 1254, and 1260 over the temperature range 150° to 300°C. From the graph, the logarithm of vapor pressure is a linear function of 1/T (T = temperature, °K):

$\log(VP) = A - B/T$

| Aroclor | A | B | VP extrapolated to 20°C, mm Hg |
|---|---|---|---|
| 1242 | 8.8 | 3,500 | $9.0 \times 10^{-4}$ |
| 1248 | 8.4 | 3,400 | $8.3 \times 10^{-4}$ |
| 1254 | 8.8 | 3,700 | $1.8 \times 10^{-4}$ |
| 1260 | 8.5 | 3,700 | $0.9 \times 10^{-4}$ |

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**TABLE D.7 Electrical Properties of Some Aroclors**

| Aroclor | Dielectric constant at 1000 cycles 25°C | 100°C | Volume resistivity, Ω cm at 100°C 500 V, dc | Dielectric strength, | Power factor 100°C, 1000 cycles, % |
|---|---|---|---|---|---|
| 1232 | 5.7 | 4.6 | | | |
| 1242 | 5.8 | 4.9 | above $500 \times 10^9$ | >35 | <0.1 |
| 1248 | 5.6 | 4.6 | above $500 \times 10^9$ | >35 | <0.1 |
| 1254 | 5.0 | 4.3 | above $500 \times 10^9$ | >35 | <0.1 |
| 1260 | 4.3 | 3.7 | above $500 \times 10^9$ | >35 | <0.1 |
| 1268 | 2.5 | | | | |

SOURCE: Modified from Durfee et al. (1976).



Photochemical Data for Selected PCBs

in Acetonitrile

| | Molecular Extinction Coefficients | Quantum Yield |
|---|---|---|
| 4-Cl | 120 | 0.0037 |
| 2,4-$Cl_2$ | 25 | 0.22 |
| 2,4,6-$Cl_3$ | 7 | 0.25 |
| 2,2',5,5'-$Cl_4$ | 30 | 0.017 |
| $Cl_{10}$ | 1200 | >0.21 |
| Aroclor 1232 | 60 | 0.036 |
| Aroclor 1268 | 770 | 0.16 |

SOURCE: Reprinted (with modifications) with permission from [Chemosphere 2:155-164, Bunce, N.J., V. Kumar and B.G. Brownlee, An assessment of the impact of solar degradation of polychlorinated biphenyls in the aquatic environment], Copyright [1978], Pergamon Press, Ltd.

**TABLE D.9 Microbial Species Used in Degradation Experiments**

| Species | Time | Degradation of PCB |
|---|---|---|
| Achrombacter | | + (unsubstituted ring) |
| Nocardia | 20 days | + (low chlorinated) <tetra CB |
| Aspergillus | | - |
| Pseudomonces | 73 days | + (<tetra CB) |
| Acinetobacter | | + |

+ indicates degradation.
- indicates no effect.

**TABLE D.10 Chlorobiphenyls Primary Biodegradation Rate**

| | nmol/cell hour |
|---|---|
| Mono | $7 \times 10^{-8}$ |
| Di-(both rings subst.) | $6 \times 10^{-8}$ |
| Tri-(both rings subst.) | $5 \times 10^{-8}$ |
| Tetra-(both rings subst.) | $2.5 \times 10^{-8}$ |
| Penta-*(both rings subst.) | $1.5 \times 10^{-8}$ |

*Only one isomer, 2,4,5,2',5' studied.

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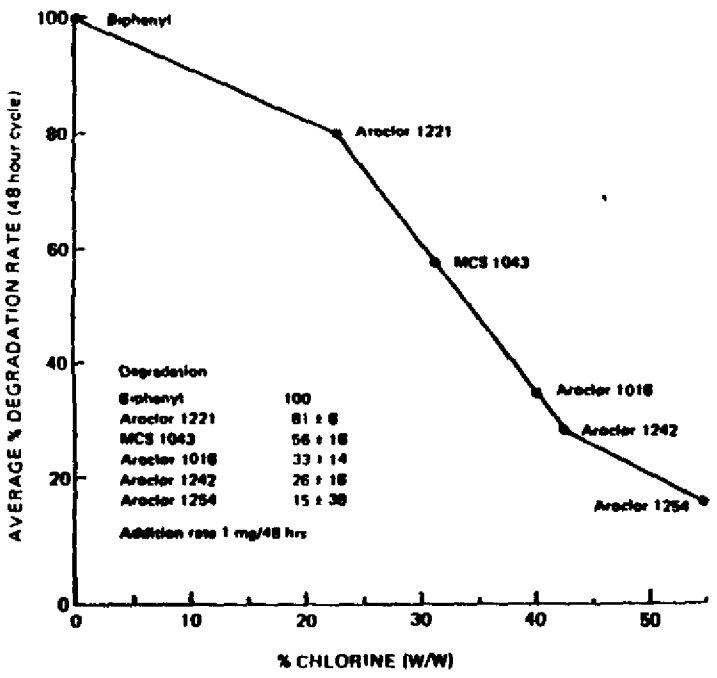
TABLE D.11 Primary Biodegradation Rate (ng/ml-hr)

| Chlorobiphenyls | Eutrophic Lake | Oligotrophic Lake | Sediment |
|---|---|---|---|
| Mono- | $1.3 \times 10^{-11}$ | $1.3 \times 10^{-15}$ | $1.3 \times 10^{-10}$ |
| Di-(both rings subst.) | $1.3 \times 10^{-11}$ | $1.3 \times 10^{-15}$ | $1.3 \times 10^{-10}$ |
| Tri-(both rings subst.) | $1.3 \times 10^{-11}$ | $1.3 \times 10^{-15}$ | $1.3 \times 10^{-10}$ |
| Tetra-(both rings subst.) | $7.2 \times 10^{-11}$ | $7.3 \times 10^{-17}$ | $7.3 \times 10^{-12}$ |
| Penta-(both rings subst.) | $<5 \times 10^{-14}$ | $<5 \times 10^{-18}$ | $<5 \times 10^{-13}$ |
| More than penta | | | |
| no biodegradation | | | |




SOURCE Tucker et al (1975b)

FIGURE D.2 Semi-continuous activated sludge primary biodegradation rates of commercial PCBs as a function of the weight percent chlorine

Effect of PCB on Selected Microorganisms

| | Amount PCB | Effect |
|---|---|---|
| [illegible] | | inhibits growth |
| [illegible]bacter | | inhibits growth |
| Micrococcus/ Acrococcus | | modified growth rate (reversible condition) |
| Scenedesmus | 300-600µg/l | inhibit growth |
| Euglena | 300-800µg/l | inhibit growth |
| Pseudomonad | 10-100µg/l | reduced rate of cell division, reduced total amount and synthetic rate of nucleic acids |
| Cyclosterium | .1µg/l | reduced growth rate and chlorophyll levels |
| Thalassiosira | 10µg/l | suppressed growth rate and photosynthesis |

Dispersal through Water PCBs are strongly adsorbed to sediment and their dispersal through water depends on the movement of the sediment. (See discussion of the PCBs in Sediment in Chapter 1.)

Bioaccumulation

PCBs are remarkable among organic industrial chemicals for their low solubility in water (Table D.6), their high octanol/water partition coefficients, accumulation coefficients, (Tables D.14-D.16), and their resistance

TABLE D.13 Equilibrium PCB Concentrations and Distribution Coefficients (Kd) for Aroclor 1254

| | Soil Concentration (mg/g) | $H_2O$ Concentration (µg/l) | Mean (Kd) |
|---|---|---|---|
| Static Condition | 500 | 7.60 | $5.6 \times 10^4$ |
| | 350 | 7.10 | |
| | 200 | 3.90 | |
| | 100 | 1.70 | |
| Continuous Flow | 500 | .54 | $93 \times 10^4$ |
| | 350 | .37 | |
| | 200 | 21 | |
| | 100 | .11 | |

SOURCE: Modified from Halter and Johnson (1977)

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TABLE D.14 Octanol/Water Partition Coefficients and Ecological Magnification Values for Selected PCBs

| | Octanol/Water Partition Coefficient | Ecological Magnification Value |
|---|---|---|
| 4-chloro | 390 | 480 |
| 4,4'-dichloro | 5,700 | 1,200 |
| 2,5,2'-trichloro | 7,800 | 6,400 |
| 2,5,2',5'-pentachloro | 8,100 | 12,000 |
| 2,4,5,2',4',5'-hexachloro | 17,000 | 12,000 |
| decachloro | 30,000 | 42,000 |
| | 190,000 | 97,000 |

to in vivo degradation (Table D.17). As a result, they exhibit extraordinarily high values for bioaccumulation in animal tissues, especially in fish and other aquatic organisms. For example, the average concentration of PCBs in Lake Michigan is about 0.008 µg/kg, yet mature lake trout, Salvelinus namaycush, contain an average of 28 mg/kg, equivalent to a bioconcentration factor of $1.41 \times 10^6$ (Metcalf 1977, U.S. EPA 1972).

Bioaccumulation of PCBs by fish and other aquatic organisms may take place either through ingestion of contaminated food organisms or by direct absorption through the integument. Coho salmon, Oncorhynchus kisutch, were fed food pellets contaminated with 3,4,3',4'-tetrachlorobiphenyl, 2,4,6,2',4',6' and 2,4,5,2',4',5'-hexachlorobiphenyl. After 108 days, the salmon contained an average of 1.4 mg/kg of each of the two hexachlorobiphenyls and 0.65 mg/kg of tetrachlorobiphenyl. From 30 to 60 percent of the

TABLE D.15 PCB Accumulation Coefficients in Selected Animals

| Animal | Accumulation Coefficients |
|---|---|
| oysters (Aroclor 1254)[1] | $101 \times 10^3$ - $165 \times 10^3$ |
| shrimp (Aroclor 1254)[1] | $26 \times 10^3$ |
| fish (estuarine) (Aroclor 1254)[1] | $37 \times 10^3$ |
| zooplankton (dichloro-)[2] | 3 - 100 (% body wt $\times 10^{-4}$) |
| trout (tetrachloro-)[3] | $9 \times 10^3$ |

SOURCE:
[1]Modified from Hansen (1976).
[2]Modified from McNaught (1978).
[3]Modified from Branson et al. (1975)



TABLE D.16 Accumulation Coefficient of Aroclor 1254 by Aquatic Invertebrates

| Organism | Organisms Per Sample[a] | Water Concentration[b] (µg/kg) | Organism Concentration[b] (4-day exposure) (mg/kg) | Accumulation Factor[c] ($\times 10^3$) 1 day | 4-day | 7-day | 14-day | 21-day |
|---|---|---|---|---|---|---|---|---|
| Daphnia | | | | | | | | |
| Daphnia magna | 60 | 1.1 ± 0.2 | 52.0 ± 2.0 | 21.0 | 47.0 | --- | --- | --- |
| Midge | | | | | | | | |
| Chaoborus punctipennis | 5 | 1.3 ± 0.1 | 30.0 ± 1.6 | 22.0 | 23.0 | 24.0 | 25.0 | --- |
| Scud | | | | | | | | |
| Gammarus pseudolimnaeus | 6 | 1.4 ± 0.1 | 35.0 ± 3.0 | 1.0 | 24.0 | 26.0 | 29.0 | 27.0 |
| Mosquito larvae | | | | | | | | |
| Culex tarsalis | 10 | 1.5 ± 0.3 | 27.0 ± 2.0 | 13.0 | 18.0 | 20.0 | --- | --- |
| Glass shrimp | | | | | | | | |
| Palaemonetes kadiakensis | 3 | 1.3 ± 0.1 | 16.0 ± 2.4 | 10.0 | 12.0 | 14.0 | 14.0 | 17.0 |
| Stonefly | | | | | | | | |
| Pteronarcella badia | 3 | 2.8 ± 0.8 | 7.0 ± 0.3 | 2.1 | 2.5 | 2.8 | 2.9 | 1.8 |
| Oligochaete | | | | | | | | |
| [illegible] | 3 | 1.1 ± 0.1 | 5.1 ± 0.2 | 1.4 | 4.6 | 5.7 | 6.6 | 6.8 |
| Crayfish | | | | | | | | |
| [illegible] | 2 | 1.2 ± 0.1 | 0.2 ± 0.2 | 0.57 | 1.7 | 1.4 | 4.5 | 5.1 |

[illegible] in triplicate.
[illegible] expressed as mean value ± standard error ($p = .05$)
[illegible] in water

[illegible] from Metcalf (1976)

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TABLE D.17 Ecological Magnification (EM) and Biodegradability Index (BI) Compared with Water Solubility and Partition Coefficients

| Chemical | Water Solubility (ppb) | Partition Coefficient octanol/water | EM ($\times 10^3$) | | | | BI ($\times 10^{-2}$) | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Algae | Snail | Mosquito | Fish | Algae | Snail | Mosquito | Fish |
| tri-Cl-PCB (2,5,2') | 16 | 7,000 | 7.3 | 5.0 | .8 | 6.4 | 36.9 | 17.0 | 35.0 | 60.0 |
| tetra-Cl-PCB (2,5,2',5') | 16 | 8,100 | 10.0 | 39.0 | 10.6 | 12.9 | 1.5 | 8.2 | 7.6 | 6.0 |
| penta-Cl-PCB (2,4,5,2',5') | 19 | 16,000 | 5.5 | 60.0 | 17.0 | 12.9 | 2.9 | 2.7 | 1.3 | 1.9 |
| hexachloro (2,4,5,2',4',5') | 9 | 30,000 | 9.4 | 101.0 | 105.0 | 42.0 | 0.8 | 2.1 | 0.7 | 0.4 |
| decachloro | | 100,000 | 100.0 | 930.0 | 22.0 | 97.0 | --- | --- | --- | 0.5 |

is in the diet was retained in the fish
ion of feeding of the chlorobiphenyls
of the tetrachlorobiphenyl in the adipose
decreased from 10.8 mg/kg to 2.5 mg/kg over a
of 48 days, compared with 2,4,5,2',4',5'-
hexachlorobiphenyl, which decreased from 20.1 mg/kg to 6.5 mg/kg; and with 2,4,6,2',4',6'-hexachlorobiphenyl, which decreased from 20.2 mg/kg to 6.0 mg/kg (Gruger et al, 1975).

The direct uptake of $^{14}$C-2,5,2'-trichloro-, 2,5,2';5'-tetrachloro-, and 2,4,5,2',5'-pentachlorobiphenyls by the green sunfish, Lepomis cyanellus, was studied by Sanborn et al. (1975). After the fish had spent 15 days in atatic water, the bioconcentration factors were determined as trichloro-54, tetrachloro- 400, and pentachlorobiphenyl 1,510.

Model ecosystem studies Bioaccumulation and biodegradation studies with a series of $^{14}$C-labeled chlorobiphenyls have been made using the terrestrial aquatic model ecosystem involving both food-chain tranafer and direct absorption in aquatic organisms, including the snail, Physa sp., and the fish Gambusia affinis (Metcalf et al. 1975, Metcalf and Lu 1978). The degree of bioaccumulation, as determined from the ratio of amount of parent compound in snail or fish to amount in water, increased in proportion to the degree of chlorination; and it reached high values for hexachlorobiphenyl--42 x $10^3$ in fish and 100 x $10^3$ in snail--and for decachlorobiphenyl--97 x $10^3$ in fish and 930 x $10^3$ in anail.

Correspondingly, the higher chlorobiphenyls were more stable in the tissues of the organisms, as determined by biodegradability index or ratio of amount of polar compound stored. The biodegradability index in the fish decreased from 0.60 for trichlorobiphenyl to 0.005 for decachlorobiphenyl.

## HAZARD EVALUATION

### Acute Toxicity Tests

Lethal Dose Studies $LD_{50}$ and $LC_{50}$ for aquatic and terrestrial wildlife are presented in Tables D.18-D.22. Oral doses range from 4,000 mg/kg to 11,000 mg/kg, and this level is required to obtain $LD_{50}$s depending upon

TABLE D.18 Toxicity of PCBs and DDT in Continuous Flow Tests

| | | $LC_{50}$ (µg/l) | | | | | |
|---|---|---|---|---|---|---|---|
| Aroclor | Species | 5 days | 10 days | 15 days | 20 days | 25 days | 30 days |
| 1254 | Rainbow Trout | 150.0 | 8 0 | ---- | ---- | ---- | ---- |
| 1260 | | ---- | 240 0 | 94.0 | 21.0 | ---- | ---- |
| DDT | | 2.3 | 0.9 | 0.3 | ---- | ---- | ---- |
| 1242 | Bluegills | 150.0 | 72.0 | 54.0 | ---- | ---- | ---- |
| 1248 | | 320.0 | 160 0 | 76.0 | 10.0 | ---- | ---- |
| 1254 | | ---- | 440 0 | 200.0 | 140.0 | 54.0 | ---- |
| 1260 | | ---- | ---- | ---- | 240.0 | 210.0 | 150.0 |
| 1242 | Channel Catfish | ---- | 170 0 | 110 0 | ---- | ---- | ---- |
| 1248 | | ---- | 230.0 | 130.0 | ---- | ---- | ---- |
| 1254 | | ---- | ---- | 740.0 | 300.0 | 110.0 | ---- |
| 1260 | | ---- | ---- | ---- | 300.0 | 170.0 | 140.0 |
| 1248[a] | Bluegills | 140.0 | 76.0 | ---- | ---- | ---- | ---- |
| 1248 | Channel Catfish | ---- | 94.0 | 57.0 | ---- | ---- | ---- |

[a]Temperature, 27°C.

SOURCE: Stalling and Mayer (1972).

TABLE D.19 Toxicity of PCBs and DDT to Invertebrates

| Compound | Organism | Bioassay Type[a] | Exposure (days) | $LC_{50}$ (µg/l) |
|---|---|---|---|---|
| Aroclor 1242 | Crayfish | static | 7 | 30 |
| Aroclor 1254 | Crayfish | static | 7 | 100 |
| Aroclor 1254 | Crayfish | continuous-flow | 7 | 80 |
| DDT | Crayfish | static | 4 | 100 |
| Aroclor 1242 | Scud | continuous-flow | 4 | 10 |
| Aroclor 1242 | Scud | continuous-flow | 10 | 5.0 |
| Aroclor 1248 | Scud | static | 4 | 52 |
| Aroclor 1254 | Scud | static | 4 | 2,400 |
| DDT | Scud | static | 4 | 3.2 |
| DDT | Scud | continuous-flow | 5 | 0.6 |
| Aroclor 1254 | Glass Shrimp | continuous-flow | 7 | 3.0 |
| DDT | Glass Shrimp | static | 5 | 1.0 |
| DDT | Glass Shrimp | continuous-flow | 5 | 1.3 |
| Aroclor 1242 | Dragonfly | static | 7 | 800 |
| Aroclor 1254 | Dragonfly | static | 7 | 1,000 |
| Aroclor 1242 | Damselfly | continuous-flow | 4 | 400 |
| Aroclor 1254 | Damselfly | continuous-flow | 4 | 200 |
| DDT | Damselfly | static | 4 | 56 |

[a]Temperature, 15.6°C, alkalinity, 35ppm, pH, 7.1.

SOURCE: Stalling and Mayer (1972).



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TABLE D.20 Toxicity of PCBs to Rats and Rabbits, $LD_{50}$ (mg/kg of body weight)

| | Aroclor 1221 | Aroclor 1232 | Aroclor 1242 | Aroclor 1248 | Aroclor 1254 | Aroclor 1260 | Aroclor 1268 |
|---|---|---|---|---|---|---|---|
| Rat (oral) | 4000 | 4500 | 8700 | 11,000 | ---- | 10,000 | 11,300 |
| Adult rats (oral) | | | | | 4000-10,000 | 4000-10,000 | |
| Weaning rats (oral) | | | | | 1200 | 1300 | |
| Adult rats (i.v.) | | | | | 400 | | |
| 30 day old rats (oral) | | | | | 1400 | | |
| 120-day old rats (oral) | | | | | 2000-2500 | | |
| Rabbit (skin) | >2000 <3200 | >1300 <2000 | >800 <1300 | >800 <1300 | | >1300 <2000 | <2500 |

SOURCE: Modified from IARC (in press).

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TABLE D.21 Acute Oral and Dermal Toxicity Effects of PCB Exposure

| Preparation | Animal | Treatment | Percent Mortality | Liver Effects | Skin Effects |
|---|---|---|---|---|---|
| Oral | | | | | |
| Unknown | Mouse | Single dose 2000mg/kg | | | |
| 54%Cl | Rat | Single dose 500mg/kg | 0 | Increase of weight and lipid; potentiation of $CCl_4$ toxicity | |
| 42, 54, 60, and 68%Cl | Mallard | Single dose 2000mg/kg | 0 | | |
| 42%Cl | Rat | 20 daily doses 130mg | 0 in 3 months | Hyalin bodies in liver cells | |
| 42%Cl | Guinea Pig | 2 doses (1 week apart) 69mg | 100 between 11 and 19 days | Fatty metamorphosis; central atrophy | |
| 54%Cl | Rat | 8 daily doses 300mg | 70 in 14 days | Increase of weight; cell swelling; hyalin granules | |
| 60%Cl | Rat | 50mg every second day | 60 in 5 weeks | 1/3 weight increase; cell swelling; hyalin granules | |
| Dermal | | | | | |
| 42%Cl | Guinea Pig | 11 daily, 34.5mg | 100 between 11 and 21 days | Fat; central atrophy perinuclear basophilic granulation, focal necrosis in a few animals | Occasional thickening of the epidermis |
| 42%Cl | Rabbit | Alternate days; total dose from 94g to 1980mg | 100 between 17 and 98 days | Fatty degeneration, central atrophy | Thinning of prickle cell layer and thickening of outer cornified layers |
| Aroclor | Rabbit | Daily, 0.3, 0.6, and 0.9g | With 30% dead before liver necrosis developed | Moderate [illegible] mottled liver, [illegible] atrophy [illegible] degeneration and [illegible] | Reddening, formation of small papules and blisters, finally desquamation of external epidermal layers |

SOURCE: Modified from Kimbrough (1976).

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TABLE D.22 $LC_{50}$ Comparisons of Aroclors in Diets (mg PCB/kg Diet) of 2-Week-Old Birds

| | Mallard | Pheasant | BobWhite Quail | Japanese Quail |
|---|---|---|---|---|
| Aroclor 1232 | ---- | 3200 | 3000 | >5000 |
| Aroclor 1242 | 3200 | 2100 | 2100 | >5000 |
| Aroclor 1248 | 2800 | 1300 | 1200 | 4800 |
| Aroclor 1254 | 2700 | 1100 | 600 | 2900 |
| Aroclor 1260 | 2000 | 1100 | 750 | 2200 |
| Aroclor 1262 | 3000 | 1200 | 870 | 2300 |

SOURCE: Modified from Nisbet (1976).

... PCBs are virtually nontoxic to ..., toxic to some invertebrates ($LC_{50}$ ... less than 1 mg/l).

... Data Tests using Daphnia, fathead minnow, ...nella, and Gammrus indicated that the larval stage is most sensitive to PCB effects (Nebeker et al. 1977). (See Tables D.19, D.23, and D.24.) Life-cycle effects on birds are given in Table D.25.

Metabolic and Chronic Feeding Studies The predominant route of PCB metabolism is hydroxylation (Table D.26). Some chlorobiphenyls may be hydroxylated via arene oxide intermediates which suggests potential carcinogenicity and mutagenicity. The hydroxylated chlorobiphenyls are more toxic than the parent compounds. The toxicological significance of the recently discovered chlorobiphenyl methyl sulfones is not known. Results of chronic feeding studies are presented in Table D.27.

Mutagenicity For Aroclors 1242 and 1254 no statistical evidence has been presented for mutagenic effects in bone marrow or spermatogonial cells, for mutagenic effects in dominant lethal assays, or for chromosomal aberrations in human lymphochromosomal cultures. Ames test results with Aroclor 1221 and 4-chlorobiphenyl indicated significant mutagenicity (Wyndham et al. 1976).

Oncogenicity Tests Data indicate that Kaneclor 500, Aroclor 1260, and perhaps Aroclor 1254 are carcinogenic in mice and rats (Table D.27).

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TABLE D.23 Spawning and Egg Production of Fathead Minnows Exposed to PCBs

| Aroclor concentration (ug/l) | Spawnings/ female | Eggs/ spawning | Eggs/ female | Percent hatched |
|---|---|---|---|---|
| Aroclor 1242 | | | | |
| 51.0 | 0 | 0 | 0 | 0 |
| 15.0 | 0 | 0 | 0 | 0 |
| 5.4 | 2.5 | 30 | 150 | 81 |
| 2.9 | 3.9 | 63 | 280 | 38 |
| 0.9 | 1.3 | 28 | 35 | 84 |
| 0.0 | 6.6 | 90 | 440 | 53 |
| Aroclor 1254 | | | | |
| 15.0 | 0 | 0 | 0 | 0 |
| 4.6 | 0 | 0 | 0 | 0 |
| 1.8 | 1.8 | 63 | 110 | 79 |
| 0.5 | 5.2 | 100 | 560 | 63 |
| 0.2 | 3.0 | 64 | 220 | 55 |
| 0.0 | 2.4 | 100 | 250 | 75 |

SOURCE: Modified from Nisbet (1976).

... Incubation Times and Hatchabilities of ... Eggs, and Survival and Sizes of Alevins ... days After Hatching, After Exposure to Aroclor 1254 for 6 and 2 Weeks at 12-14°C

| | Incubation | | Mean Alevin Size | | |
|---|---|---|---|---|---|
| PCB (ug/kg) | Days | Percent Hatchability | Percent Survival | Length (mm) | Weight (g) |
| Exposure for 6 weeks (2 before and 4 after hatching) | | | | | |
| 0 | 510 | 96 | 91 | 36 | 0.30 |
| 5 | 490 | 88 | 76 | 35 | 0.30 |
| 10 | 490 | 78 | 64 | 34 | 0.30 |
| 15 | --- | 38 | 24 | 32 | 0.27 |
| 25 | 460 | 96 | 77 | 30 | 0.28 |
| 55 | 460 | 63 | 7 | 28 | 0.24 |
| Exposure for 2 weeks (to 2 days before hatching) | | | | | |
| 0 | 500 | 93 | 93 | 35 | 0.30 |
| 5 | 480 | 96 | 70 | 35 | 0.30 |
| 10 | 490 | 90 | 88 | 35 | 0.30 |
| 15 | 490 | 92 | 84 | 33 | 0.26 |
| 25 | 488 | 98 | 75 | 35 | 0.29 |
| 55 | 490 | 66 | 48 | 34 | 0.26 |

SOURCE: Modified from Nisbet (1976).

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TABLE D.25 Life Cycle Effects of Aroclor 1254

| Species | Dose (mg/kg) | Effects |
|---|---|---|
| Pheasants | 50 and 200 | reduced egg production; increase in no. of eggs pipped but not hatched; reduced survival of chicks and overall reproductive success |
| Mallard* | 25 | no reproductive effects; PCB residue in eggs equalled 33 ppm wet weight |
| Bobwhite Quail* | 50 | no reproductive effects |
| Ring Doves | 10 | high embryonic mortality in 2nd generation; chromosomal aberrations; abnormal incubation behavior seen in parents |
| Am Kestrels | 10 | slight increase in egg shell thickness |
| Japanese Quail | 78 | slight reduction in egg production; 12% reduction in egg shell thickness |
| | 50 | non significant reproductive effects |

*Required test species in proposed guidelines.



TABLE D.26 Metabolism of Polychlorinated Biphenyl Isomers

| Species | Parent Compound | Products |
|---|---|---|
| rat | 4-chlorobiphenyl | monohydroxylated derivative<br>dihydroxylated derivative |
| rat | 4-chlorobiphenyl | 4'-chloro-3,4-biphenyldiol<br>4'-chloro-3-methoxy-4-biphenyldiol<br>4'-chloro-4-methoxy-3-biphenyldiol<br>4'-chloro-4-methoxy-3,5-biphenyldiol |
| rat | $^{14}$C-2,3 dichlorobiphenyl | 2',5'-dichloro-4-biphenyl |
| rat | $^{14}$C-2,4,6-tr.chlorobiphenyl | 4'-hydroxylated derivative<br>3',4'-dihydroxylated derivative<br>3'-hydroxy-4'-methoxylated derivative<br>4'-hydroxy-3'-methoxylated derivative |
| rat microsomes | 2,2',4-trichlorobiphenyl | monohydroxylated derivative |
| rat | $^{3}$H-2,2',5,5'-tetrachlorobiphenyl | 3-monohydroxylated derivative |
| rat | $^{3}$H-2,2',5,5'-tetrachlorobiphenyl | 2,2',5,5'-tetrachloro-3-biphenylol<br>trans-3,4-dihydro-2,2',5,5'-tetra-chlorobiphenyl-3,4-biphenylol |
| rat | 3,3',4,4'-tetrachlorobiphenyl | 2 and 5-hydroxylated derivative<br>3 and 5-hydroxylated derivative |
| rat | $^{14}$C-2,3,4,5,6-pentachlorobiphenyl | 3' and 4'-hydroxylated derivative<br>3',4'-dihydroxylated derivative |
| rat | 2,2',4,5,5'-pentachlorobiphenyl | 2,2',4,5,5'-pentachloro-3'-biphenylol<br>3',4'-dihydro-3,4'-diol<br>2,2',4,5,5'-pentachloro-3,4-biphenylol |
| rat | 2,2',4,4',6-pentachlorobiphenyl | monohydroxylated derivative |
| | 2,2',4,6,6'-pentachlorobiphenyl | monohydroxylated derivative |
| | 2,3,4,4',6-pentachlorobiphenyl | monohydroxylated derivative |
| | 2,3',4,5',6-pentachlorobiphenyl | monohydroxylated derivative |
| rat | 2,3,3',4,4'-pentachlorobiphenyl | no hydroxylated products |

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TABLE D.26 continued

| Species | Parent Compound | Products |
|---|---|---|
| rat | 2,2',4,4',5,5'-hexachlorobiphenyl | no hydroxylated products |
| rat | $^{14}$C-2,4,6,2',4',6'-hexachlorobiphenyl | monohydroxylated derivative |
| rat | 2,2',4,4',5,5'-hexachlorobiphenyl | 2,2',4,4',5,5'-hexachloro-3-biphenylol |
| rabbit | 4-chlorobiphenyl | 4-chloro-4'-biphenylol |
| rabbit (liver microsomes) | 4-chlorobiphenyl | 4-chloro-4'-biphenylol<br>4'-chloro-3,4-biphenyldiol |
| rabbit | 4,4'-dichlorobiphenyl | 4,4'-dichloro-3-biphenylol<br>3,4'-dichloro-4-biphenylol<br>4'-chloro-4-biphenylol |
| rabbit | 2,2',5,5'-tetrachlorobiphenyl | 2,2',5,5'-tetrachloro-3-biphenylol<br>2,2',5,5'-tetrachloro-4-biphenylol<br>trans-3,4-dihydro-2,2',5,5'-tetrachloro-3,4-biphenyldiol |
| rabbit | 2,2',4,4',5,5'-hexachlorobiphenyl | monohydroxylated derivative<br>monohydroxy with chlorine shift<br>monohydroxy methoxylated derivative |
| pigeons | 4-chlorobiphenyl | monohydroxylated derivative |
| pigeons | 4,4'-dichlorobiphenly | monohydroxylated derivative |
| pigeons | 2,2',5,5'-tetrachlorobiphenyl | monohydroxylated derivative |
| pigeons | 2,2',4,4',5,5'-hexachlorobiphenyl | no hydroxylated products |
| chickens | 2,2',4,4',5,5'-hexachlorobiphenyl | metahydroxylated derivative<br>pentachlorobiphenyl derivative<br>pentachloro trihydroxylated derivative |
| trout | 4-chlorobiphenyl | no hydroxylated products |
| trout | 4,4'-dichlorobiphenyl | no hydroxylated products |
| trout | 2,2',5,5'-tetrachlorobiphenyl | no hydroxylated products |
| trout | 2,2',5,5'-tetrachlorobiphenyl | conjugated metabolites<br>4-hydroxy-2,2',5,5'-TCB |
| trout | 2,2',4,4',5,5'-hexachlorobiphenyl | no hydroxylated metabolites |
| goat | 4-chlorobiphenyl | 4'-chloro-4-biphenylol<br>4'-chloro-3,4-biphenyldiol |
| goat | 4,4'-dichlorobiphenly | 4,4'-dichloro-3-biphenylol |
| cow | 4-chlorobiphenyl | 4'-chloro-4-biphenylol |
| rhesus monkey | 2,2',5,5'-tetrachlorobiphenly | monohydroxylated derivatives<br>dihydroxylated derivatives<br>trans-3,4-dihydro-3,4-TCB diol<br>hydroxy-3,4-dihydro-3,4-TCB diol |
| mice | 2,2',5,5'-tetrachlorobiphenly | methylsulfone derivative |
| mice | pentachlorobiphenly | methylsulfone derivative |
| mice | 2,2',4,4',5,5'-hexachlorobiphenyl | 2,2',4,4',5,5'-hexachloro-3-biphenylol |

SOURCE Modified from IARC (in press).



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TABLE D.27 Results of Chronic Feeding Studies Using Various PCB Mixtures

| Species | Dose | PCB | Time | Effect |
|---|---|---|---|---|
| Mouse | 500µg/g or less | Kanechlor 500 | 32 weeks | Hepatocellular carcinomas; some widespread nodular hyperplasia (Nagasaki et al. 1972, Ito et al. 1973) |
| | 500µg/g | Kanechlor 300, 400 | 32 weeks | No effect (Nagasaki et al. 1972) |
| | 300µg/g | Aroclor 1254 | 11 months | Adenofibrosis of liver (Kimbrough and Linder 1974) |
| | 300µg/g | Aroclor 1254 | 6 months | No effect (Kimbrough 1974) |
| Rat | 100 & 500 µg/g | Aroclor 1254 | 8 months | Adenofibrosis (Kimbrough et al. 1972) |
| | 38 - 616 µg/g | Kanechlor 400 | 57 weeks | Females developed neoplastic nodules in liver (Kimura and Baba 1973) |
| | 100µg/g | Aroclor 1260 | 21-23 months | Increased number of stillborns and reduction of fetal weight; neoplastic nodules of liver (Burke and Fitzhugh 1970) |
| | 100µg/g | Aroclor 1254 & 1260 | 1 year | No illness; increase in total serum lipids & cholesterol; slight increase in triglycerides; liver hypertrophy and focal areas of hepatocellular degeneration (IARC [in press]) |
| | 20 - 100 µg/g | Kanechlor 500 | 2 weeks (gestation) | Offspring with reduced learning ability (Shiota 1976) |
| Monkey | 2.5 - 5.0 µg/g | 1248 | 6 months | Females developed acne, alopecia, erythema and swelling of eyelids; alteration of menstrual cycles; increased frequency of abortion (IARC [in press]) |
| | | | 14 months | Males developed only slight periorbital oedema and erythema (Barsotti et al. 1976) |
| | | | | Hypertrophic hyperplastic gastritis, ulceration and anemia (IARC [in press]) |
| | | | 2 months | Infants with evidence of severe hyperplastic gastritis and hyperplasia of hair follicle epithelium and epidermis (IARC [in press]) |
| Chickens | 50µg/g | 1254 | 39 weeks | Reduced egg production and decline in fertility (Platonow and Reinhart 1973) |
| | 50mg/l in $H_2O$ | 1254 | 6 weeks | Deformities in neck and toe (Bush et al. 1974) |
| | 100µg/g | 1254 | | Decrease hatchability (Keplinger et al. 1971) |
| | 20µg/g | | 8 weeks | Subcutaneous edema; atrophy of the spleen, mild necrosis; congestion and fatty infiltration of the liver 30% mortality (Bird et al 1978) |
| Mink | 10µg/g | 1254 | 4 months | Dose response relationship in depressed weight gain reduced reproduction (Aulerich and Ringer 1977) |
| | | | 5.5 months | Mortality occurred (Aulerich and Ringer 1977) |
| Pelican | 100mg | | 10 weeks | Increased size of hepatocytes; number of vacuoles per hepatocyte and number of mitochondria (Stotz and Greichus 1978) |
| Dogs | 100µg/g | 1260 | 12 months | Males - reduced growth rate, increased liver weights; elevated serum alkaline phosphate (Keplinger et al. 1971) |





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REFERENCES


Ahmed, M. and D.D. Focht (1973) Degradation of polychlorinated biphenyls by two species of Arochromobacter. Can. J. Microbiol. 19:47-52.

Aulerich, R.J. and R.K. Ringer (1977) Current status of PCB toxicity to mink, and effect on their reproduction. Arch. Environ. Contam. Toxicol. 6:279-292.

Barsotti, D.A., R.J. Marlar, and J.R. Allen (1976) Reproductive dysfunction in rhesus monkeys exposed to low levels of polychlorinated biphenyls (Aroclor 1248). Food Cosmet. Toxicol. 14:99-103.

Baughman, G.L. and R.R. Lassiter (1978) Prediction of environmental pollutant concentration. Pages 35-54, Estimating the Hazard of Chemical Substances to Aquatic Life. Edited by John Cairns, Jr., K.L. Dickson, and A.W. Maki. Philadelphia: American Society for Testing and Materials.

Bird, F.H., C.B. Chawan, and R.W. Gerry (1978) Response of broiler chickens to low level intake of polychlorinated biphenyl isomers. Poultry Sci. 57(2):538-541.

Blakemore, R.P. (1978) Effects of polychlorinated biphenyls on macromolecular synthesis by heterotrophic marine bacterium. Appl. Environ. Microbiol. 35(2):329-336.

Blakemore, R.P. and A.E. Carey (1978) Effects of polychlorinated biphenyls on growth and respiration of heterotrophic marine bacteria. Appl. Environ. Microbiol. 35(2):323-328.

Bourquin, A.W. and S. Cassidy (1975) Effect of polychlorinated biphenyl formulations on the growth of estuarine bacteria. Appl. Microbiol. 29(1):125-127.

Branson, D.R., G.E. Blau, H.C. Alexander, and W.B. Neely (1975) Bioconcentration of 2,2',4,4'-tetrachlorobiphenyl in rainbow trout as measured by an accelerated test. Trans. Am. Fish Soc. 4:785-792.

Bunce, N.J., Y. Kumar, and B.G. Brownlee (1978) An assessment of the impact of solar degradation of polychlorinated biphenyls in the aquatic environment. Chemosphere 2:155-164.

Burke, J. and O.G. Fitzhugh (1970) Status Report of Chemistry and Toxicology of PCBs, Suppl. I, December 1. Washington, D.C.: U.S. Food and Drug Administration.





Buser, H.R., H.P. Boffhardt, and C. Rappe (1978) Formation of polychlorinated dibenzofurans (PCDFs) from the pyrolysis of PCBs. Chemsphere 7:109.

Bush, B., C.F. Tumasonis, and F.D. Baker (1974) Toxicity and persistence of PCB hemologs and isomers in the avian system. Arch. Environ. Contam. Toxicol. 2:195-212.

Durfee, R.L. (1976) Production and usage of PCBs in the United States. Pages 103-107, Proceedings of National Conference on Polychlorinated Biphenyls, November 19-21, 1975, Chicago, Illinois. Report No. EPA-560/6-75-004. Washington, D.C.: U.S. Environmental Protection Agency.

Durfee, R.L., G. Contos, F.C. Whitmore, J.D. Barden, E.E. Hackman III, and R.A. Westin (1976) PCBs in the United States--Industrial Use and Environmental Distribution. Report No. EPA 56016-16-005. Springfield, Va.: Versar, Inc.

Ewald, W.G., J.E. French, and M.A. Champ (1976) Toxicity of polychlorinated biphenyls (PCBs) to Euglena gracilis: Cell population growth, carbon fixation, chlorphyll level, oxygen consumption, and protein and nucleic acid synthesis. Bull. Environ. Contam. Toxicol. 16(1):71-80.

Furukawa, K., F. Matsumura, and K. Tonomura (1978a) Alcaligenes and Acinatobacter strains capable of degrading polychlorinated biphenyls. Agric. Biol. Chem. 42(3):543-548.

Furukawa, K., K. Tonomura, and A. Kamibayashi (1978b) Effect of chlorine substitution on the biodegradability of polychlorinated biphenyls. Appl. Environ. Microbiol. 35(2):223-227.

Gruger, H.E., N.L. Karrich, A.I. Davidson, and T. Hruby (1975) Accumulation of 3,4,3',4'-tetrachlorobiphenyl and 2,4,5,2',4',5'- and 2,4,6,2',4',6'-hexachlorobiphenyl in juvenile Coho salmon. Environ. Sci. Technol. 9:121-127.

Halter, M.T. and H.E. Johnson (1977) A model system to study the desorption and biological availability of PCB in hydrosoils. Aquat. Toxicol. Haz. Eval. ASTM-STP 634-178-195.

Hansen, D.J. (1976) PCBs: Effect on and accumulation by estuarine organisms. Pages 282-283, Proceedings of National Conference on Polychlorinated Biphenyls, November 19-21, 1975, Chicago, Illinois. Report No. EPA-560/6-75-004. Washington, D.C.: U.S. Environmental Protection Agency.


MONS 219207

Haque, R. and D. Schmedding (1976) Studies on the adsorption of selected polychlorinated biphenyl isomers on several surfaces. J. Environ. Sci. Health B11(2):129-137.

Haque, R., D.W. Schmedding, and V.H. Freed (1974) Aqueous solubility, adsorption, and vapor behavior of polychlorinated biphenyl Aroclor 1254. Environ. Sci. Technol. 8(2):139-142.

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

International Agency for Research on Cancer (IARC) (In press) Monographs on the evaluation of the carcinogenic risk of chemicals to humans 18:43-103. (Due for release in or after 1979)

Ito, N., H. Nagasaki, M. Arai, S. Makiura, S. Sugihara, and K. Hirao (1973) Histopathologic studies on liver tumorigeneais induced in mice by technical polychlorinated biphenyls and its promoting effect on liver tumors induced by benzene hexachloride. Gann 51:1637-1646.

Kaiser, K.L.E. and P.T.S. Wong (1974) Bacterial degradation of polychlorinated biphenyls. I. Identification of some metabolic products from Aroclor 1242. Bull. Environ. Contam. Toxicol. 11(3):291-296.

Keplinger, M.L., O.E. Fancher, and J.C. Calandra (1971) Toxicologic studies with polychlorinated biphenyls. Toxicol. Appl. Pharmacol. 19:402-403.

Kimbrough, R.D. (1974) Tha toxicity of polychlorinated polycyclic compounds and related chamicals. In, CRC Crit. Rev. Toxicol. Pages 445-498, Cleveland, Ohio: CRC Press.

Kimbrough, R.D. and R.E. Linder (1974) Induction of adenofibrosis and hepatomas of the liver in BALB/cJ mice by polychlorinated biphenyls (Aroclor 1254). J. Nat. Cancer Inst. 53:547-552.

Kimbrough, R.D., R.E. Linder, and T.B. Gaines (1972) Morphological changes in livers of rats fed polychlorinated biphenyls. Arch. Environ. Health 25:354-364.

Kimura, M.T. and T. Baba (1973) Neoplastic changes in the rat liver induced by polychlorinated biphenyl. Gann 64:105-108.

Mc'laught, D.C. (1978) Uptake Rates and Effects of PCBs on Phytoplankton and Zooplankton. (Unpublished paper)



Metcalf, R.L. (1977) The biological fate and transformation of pollutants in water. Pages 195-221, Fate of Pollutants in the Air and Water Environment, Part II. Edited by I.H. Suffet. New York: John Wiley & Sons, Inc.

Metcalf, R.L. and Po-Yung Lu (1978) Partition Coefficients as Measures of Bioaccumulation Potentials for Organic Compounds. Final Report to U.S. Environmental Protection Agency, Contract No. 68-014189, July 15. Urbana, Ill.: University of Illinois.

Metcalf, R.L., J.R. Sanborn, Po-Yung Lu, and D. Nye (1975) Laboratory model ecosystem studies of the degradation and fate of radiolabeled tri-, tetra-, and pentachlorobiphenyl compared with DDE. Arch. Environ. Contam. Toxicol. 3:151-165.

Miyata, H., T. Kashimoto, and N. Kunita (1977) Detection and determination of polychlorodibenzofurans in normal human tissues and Kanemi rice oils caused 'Kanemi Yusho.' J. Food Hyg. Soc. 18(3):260-265.

Muto, T., T. Kabodera, and Y. Shinozaki (1974) The solubility of PCBs (polychlorinated biphenyls) in water and adsorption from water. Radioisotopes 23(4):19-22.

Nagasaki, H., S. Tomii, T. Mega, M. Marugami, and N. Ito (1972) Hapatocarcinogenicity of polychlorinated biphenyls in mice. Gann 63:805.

Nebeker, A.V., F.A. Puglisi, and D.L. DeFoe (1977) Toxicity of Polychlorinated Biphanyls (PCBs) to Fish and Other Aquatic Life. Report No. PB-264 804. Springfield, Va.: National Technical Information Service.

Nisbet, I.C.T. (1976) Criteria Document for PCBs. Report No. PB-225 397. Washington, D.C.: U.S. Department of Commerce.

O'Connors, H.B., Jr., C.F. Wurster, C.D. Powers, D.C. Biggs, and R.G. Rowland (1978) Polychlorinated biphenyls may alter marine trophic pathways by reducing phytoplankton size and production. Science 201:737-739.

Platonou, N.S. and B.S. Reinhart (1973) The effects of polychlorinated biphenyls (Aroclor 1254) on chicken egg production, fertility and hatchability. Can. J. Comp. Med. 37:341-346.

Powers, C.D., R.G. Rowland, H B. O'Connors, Jr., and C.F. Wurster (1977) Response to polychlorinated

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biphenyls of marine phytoplankton isolates cultured under natural conditions. Appl. Environ. Microbiol. 34(6):760-764.

Sanborn, J.R., W.F. Childers, and R.L. Metcalf (1975) Uptake of three polychlorinated biphenyls, DDT, and DDE by the green sunfish, Lepomis cyanellus Raf. Bull. Environ. Contam. Toxicol. 13:209.

Shiota, K. (1976) Embryotoxic effects of polychlorinated biphenyls (Kaneclora 300 and 500) in rats. Okajimas Fol. Anat. Jap, 53:93-104.

Stalling, D.L. and F.L. Mayer, Jr, (1972) Toxicities of PCBs to fish and environmental residues. Environ. Health Perspect. 1:159-164.

Stotz, J.J. and Y.A. Greichus (1978) The effects of a polychlorinated biphenyl Aroclor 1254 on the white pelican: ultrastructure of hepatocytes. Bull. Environ. Contam. Toxicol. 19(3):319-325.

Tucker, E.S., W.J. Litachgi, and W.M. Mesa (1975a) Migration of polychlorinated biphenyls in soil induced by percolating water. Bull. Environ. Contam. Toxicol. 13(1):86-93.

Tucker, E.S., V.W. Saeger, and O. Hicks (1975b) Activated sludge primary biodegradation of polychlorinated biphenyls. Bull. Environ. Contam. Toxicol. 14(6):705-713.

U.S. Environmental Protection Agency (1972) An Evaluation of DDT and Dieldrin in Lake Michigan. Ecol. Res. Ser. Report No. EPA R3-72-003, Washington, D.C.: Environmental Protection Agency.

Weil, L., G. Dure, and K.E. Quentin (1974) Water solubility of chlorinated hydrocarbon insecticides and PCBs in respect to water loading by these compounds. Wasser Abwasser Forschung 7:169-175.

Wong, P.T.S. and K.L.E. Kaiser (1975) Bacterial degradation of polychlorinated biphenyls II. Rate studies. Bull. Environ. Contam. Toxicol. 13(2):249-255.

Wyndham, C., J. Devenish, and S. Safe (1976) The in vitro metabolism, macromolecular binding and bacterial mutagenicity of 4-chlorobiphenyl, a model PCB substrate. Res. Commun. Chem. Pathol. Pharmacol. 15:563-570.

Zitko, V. and P.M.K. Choi (1971) PCB and Other Industrial Halogenated Hydrocarbons in the Environment. Technical Report No. 272. St. Andrews, New Brunswick, Canada: Fisheries Research Board of Canada.

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# FINAL REPORT OF THE
# SUBCOMMITTEE ON THE HEALTH EFFECTS
# OF
# POLYCHLORINATED BIPHENYLS
# AND
# POLYBROMINATED BIPHENYLS

JULY 1976

DEPARTMENT OF HEALTH, EDUCATION AND WELFARE

WASHINGTON, D.C.

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OF
POLYCHLORINATED BIPHENYLS
AND
POLYBROMINATED BIPHENYLS

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OF
POLYCHLORINATED BIPHENYLS
AND
POLYBROMINATED BIPHENYLS

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MONS 219220

Subcommittee on the Health Effects

of PCBs and PBBs

Purpose:

1. "Assemble, review and interpret data that assess the health significance of polychlorinated biphenyls."

2. "Formulate recommendations as to future research needs."

Specific Objectives:

1. Chemistry:

   Assess the impurities that may be found in PCBs and PBBs, the "hardness" of the data giving their potential for accumulating in the food chain.

2. Metabolism and Biochemical Toxicity:

   Assess the Metabolism of PCBs and PBBs and the possible contribution of specific isomeric homologs to the observed toxicity; evaluate possible effects at the cellular level; and define existing dose-response relationships for the phenomena.

3. Animal Toxicity and Carcinogenicity

   Assess the short and long term effects, including carcinogenis, of exposu.. to PCBs and PBBs in various animal models, their relationship to known human effects, and the establishment of dose-response curves.

x

4. Human Exposure

Quantity known human exposures, accidental pulsed, and long range, their relationship to possible overt symptomatology, and the establishment of possible dose-response curves.



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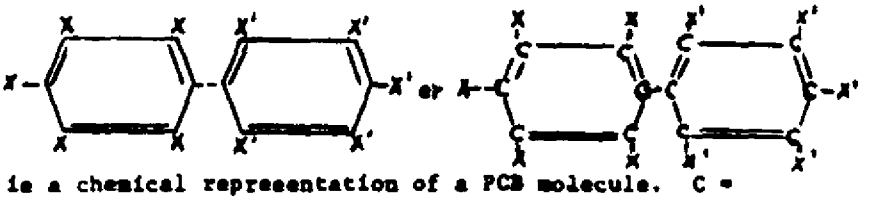
## INTRODUCTION

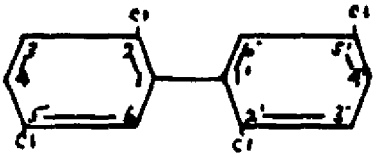
PCBs were reportedly first synthesized in 1881 by Schmidt and Schultz, but were not available commercially until 1930[1] (Hubbard, 1964). From 1930 until 1970, when their distribution in the U.S. was voluntarily restricted by Monsanto to closed systems, the use of PCBs in a wide variety of industrial applications had steadily increased.

---

1. Commercial PCBs are complex mixtures. Arochlor 1254, for example, contains 69 different molecules (Zitko, 1971), which differ in the number and position of chlorine atoms.



This is a chemical representation of a PCB molecule. C = carbon atom, X = Cl (chlorine atom) or H (hydrogen atom). The basic parent molecule contains two benzene rings connected by a carbon-carbon bond. When all the X' and X are hydrogen, the molecule is called biphenyl. When several of the hydrogen atoms are replaced by chlorine atoms, the molecule is called a PCB. X and X' are used to indicate that the chlorine atoms are on different benzene rings. For example, the chemical structure for 2,5,2',5'-tetrachlorobiphenyl is:



Positional isomers have the same number of chlorine atoms, but they are located at different positions on the ring. 2,5,2',5'-tetrachlorobiphenyl and 3,4,5',4'-tetrachlorobiphenyl (TCB) are isomers. They are different molecules with different chemical, physical, and biological properties. The chemical structure for 3,4,3',4'-TCB is:



The problem of polychlorinated biphenyls (PCBs) became of national concern in 1971, when several accidental contaminations of foods were reported. In addition, the extent of the environmental contamination, and its persistence, were not precisely known. Subsequently, various regulatory actions were taken by the concerned agencies involved and, with the cooperation of the only U.S. producer, the situation was felt to be under control.

Then, in 1975, the reported high levels of contamination, with PCBs, of Hudson River fish, refocused national attention on this environmental contaminant. It was soon apparent that the actions and control measures of the early 1970's had not succeeded in totally reducing, or even substantially alleviating the problems associated with this environmental pollutant.

On November 17-19, 1975, a National Conference on Polychlorinated Biphenyls, sponsored by the Environmental Protection Agency in cooperation with the Department of Agriculture, Council on Environmental Quality, Department of Health, Education and Welfare and Department of the Interior, was held in Chicago, Illinois. This conference brought together the latest data, and scientific expertise, to consider all aspects of the various problems associated with PCBs, and the means for their possible resolution.

In his introductory remarks to the Session on Health Effects and Human Exposure, Dr. David P. Rall, Session Chairman, announced the formation of a Subcommittee on PCBs of the DHEW Committee to



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Coordinate Toxicology and Related Programs. The Subcommittee was given the charge to:

1. Assemble, review and interpret data that assesses the health significance of polychlorinated biphenyls.

2. Formulate recommendations as to future needs.

The accidental contamination of animal feeds with polybrominated biphenyls (PBBs), and consequent human exposures, that occurred in the State of Michigan, has been approached not only on its own basis, but in comparative relationship to the PCBs, as well. Therefore, the scope of this Subcommittee was expanded to the Subcommittee on the Health Effects of PCBs and PBBs. Members of this Subcommittee came from within the Department of Health, Education and Welfare, as well as including consultants from other Federal Agencies concerned with this difficult problem.

The report which follows represents the culmination of this Subcommittee's efforts.



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# GENERAL
# SUMMARY AND CONCLUSIONS

## CHEMISTRY

The great complexity of PCB commercial mixtures has provided the difficult task of separating the components of these mixtures and determining the complete identity of the compounds. Efforts to define these mixtures have been quite successful over the past several years (Hutzinger et al., 1974). Identification of the component chemicals in PCB residues presents another difficult problem. Little progress has been reported in this area. Since the fate of components of commercial PCB mixtures entering the environment cannot be followed directly, inferences must be drawn about chemical alteration of chlorinated biphenyl compounds in the environment mainly based on laboratory studies.

The known resistance of aryl chlorides to chemical oxidation and hydrolysis presumably was a major factor in the promotion of PCBs for industrial uses. Exposure to non-metabolic environmental agents is unlikely to result in significant oxidation or hydrolysis of chlorinated biphenyls. However, since appreciable photoalteration of chlorinated biphenyls has been observed using either sunlight or sunlight-simulating lamps in the laboratory (Hutzinger et al., 1974) as energy sources, it is highly probable that photochemical changes occur in the environment. These photochemical studies have indicated that reductive dechlorination to biphenyls



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of lower chlorine content is a predominant alteration route although formation of more polar compounds, including a chlorinated dibenzofuran, has also been reported (Hutzinger et al., 1974, Crosby and Moilanen, 1973). The diverse conditions of the environment make it difficult to predict accurately the end products and rates of photoalteration of chlorinated biphenyls. From a comparison of carbon-halogen bond-energies, brominated biphenyls might be expected to be more readily altered by light than chlorobiphenyls. But because of the lower volatility and different end uses of bromobiphenyls, they may not be as readily exposed to light as the chlorinated analogs.

A typical PCB residue from fish resembles the Aroclor 1254 mixture more closely than it does other Aroclors (Zitko et al., 1972; Veith, 1975). Considering the major components of Aroclor 1254 (Sissons and Welti, 1971) it appears that penta-, hexa-, and heptachlorinated biphenyls tend to concentrate at this trophic level in the biosphere. It is of considerable interest, therefore, to carry out comparative accumulation, metabolism and toxicity studies emphasizing these higher chlorinated biphenyls using compounds of known chlorine substitution pattern. Individual chlorinated biphenyls are now quite accessible through synthesis and recent studies have begun to obtain these types of biological data.

Studies using a group of five symmetrical hexachlorobiphenyl isomers in chicks (McKinney et al., in press) and in mice (Biocca,



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1975) indicated that distinct difference in toxicity are possible and these differences could be related to structure. The isomers with chlorine substituents in the 4,4' positions appeared to accumulate more rapidly in adipose tissue, showed increased activity in the liver and greater overall toxicity (McKinney, 1975; McKinney et al., in press). Comparison of two pentachlorbiphenyls in laying chicks again indicated the one isomer with 4,4'-substitution to have greater toxic effects (Ax and Hansen, 1975; Ax et al., 1976). Structure-activity relationships, particularly for biphenyls known to be major components of commercial PCB mixtures, could be useful in assessing the potential hazard of these compounds.

A major part of determining human exposure to PCB residues in foods is evaluating the adequacy of the analytical methodology used and the significance of the data obtained. The methods applicable to PCB determination are complex and to be judged adequate should give acceptably reproducible results in the hands of experienced analysts. The degree of success of inter-laboratory collaborative studies helps measure the adequacy of a method.

PCB residues are multicomponent mixtures. Many common chlorinated pesticides are extracted from samples along with the PCBs. Procedures for separation of PCB residues from interfering pesticides therfore bec me important as a prerequisite to quantitation. Several different quantitation techniques have been used (Hutzinger et al., 1974). Comparison of PCB residue data is difficult where standardization of the quantitation procedure is lacking.



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It has been shown that recovery of PCB mixtures purposely added to food samplee, using commonly applied analytical methodology, varies with the average level of chlorination. Recoveries tend to decrease with increasing chlorine level (Stalling et al., 1972).

Analytical procedures for determination of PCB and PBB (the latter is predominantly a hexabromobiphenyl) residues are essentially similar. They differ somewhat in cleanup of the extract and in the use of a higher column temperature for gas chromatographic determination of the PBB residue. The limit of quantitation, for PCBs is generally 0.2 ppm for individual foods and about 0.05 ppm for Total Diet composites using Food and Drug Administration methodology. For PBB residues, the limit of quantitation is approximately 0.05 ppm in fats and 0.01 ppm in non-fatty foods. Interlaboratory studies of analytical methods for PBB determination have not been reported.

The chlorinated dibenzofurans (Cl-DBFs) have become a focus of concern as the class of contaminant compounds in commercial PCBs most likely to contribute significantly to the toxicity of the PCB mixture. This concern presumably stems mainly from the demonstrated toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin and the similarity in structure between dibenzo-p-dioxin and dibenzofuran (NIEHS Conference, 1973; also see Appendix of Chemistry Report) and from the little toxicity data that has been published for Cl-DBFs (Bauer et al., 1961). Recent data indicated



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the toxicity of 2,3,7,8-tetrachlorodibenzofuran approaches that of the analogous dibenzo-p-dioxin at least in chicks and in guinea pigs (Moore et al., 1976). If the chlorinated dibenzo-p-dioxins can serve as an example, chlorinated dibenzofuran toxicity can be expected to vary with number and position of chlorine atoms on the parent ring system. Relationships between structure and toxicity will emerge as more Cl-DBFs are synthesized and tested for toxicity.

A range of Cl-DBFs (from dichloro through hexachloro) have been reported in various Aroclors (Roach and Pomerantz, 1974b; Bowes et al., 1975a) and two specific compounds, the 2,3,7,8-tetrachloro- and 2,3,4,7,8-pentachloro-dibenzofurans, have been identified (Bowes et al., 1975b). Quantitation of Cl-DBF contaminants in PCB mixtures, a difficult procedure, suggests total Cl-DBF levels in the low parts per million range (Nagayama et al., 1975; Bowes et al., 1975a). The presence of Cl-DBF in a synthesized, individual, asymmetrical chlorinated biphenyl has been reported (Moron et al., 1973). Chemical analysis of these individual chlorobiphenyls may be necessary prior to their use in toxicological and other biological studies.

There have been no published positive findings of Cl-DBFs in environmental samples or in foods other than rice oil. The finding of Cl-DBFs in such samples would not necessarily implicate commercial PCBs as the source of these contaminants since Cl-DBFs (generally the higher chlorine levels) have been reported in other



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industrial chemicals (Firestone et al., 1972; Schwetz et al., 1974; Villanueva et al., 1974). It is also possible that Cl-DBFs may be formed or altered in the environment.

METABOLISM AND BIOCHEMICAL TOXICITY

The effect of PCBs and PBB on the hepatic mixed-function oxidase (MFO) enzymes has been the most thoroughly studied of any biochemical parameter that they are known to alter. On a molar basis, the PBBs are approximately five-fold more potent than the PCBs in inducing increased levels of the MFO enzymes (Farber and Baker, 1974). Among the PCBs it appears as if their potency increases with increasing chlorination and chlorine substitution in the para > ortho > meta positions respectively (Echobichon, 1975). However, this induction of the MFO enzymes is not unique to these compounds and the induction observed is well within the range observed with many other xenobiotics. The PCBs are somewhat unique as MFO inducers in that they induce the formation of both Type I and Type II P-450 (Alvares et al., 1973), but this induction may have been due to the fact that the commercial formulation used in the study was a mixture of twenty or more PCBs which were metabolized to an even greater number of metabolites. In addition, many, if not most co-mercial PCB formulation contain trace amounts of chlorinated dibenzofurans (Araki, 1974). These compounds may be up to 170 times more potent as MFO inducers than the PCBs. The chlorinated dibenzofuran concentration of American



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PCB formulations is usually quite low when they are produced, however, the effect of long-term is unknown. In any case, since induction of the MFO enzymes may result in increased hormone metabolism or carcinogen activation, exposure to the PCBs and PBBs should be limited on that basis alone.

PCBs and PBBs administered at relatively high concentrations, usually 50 ppm or higher in the diet, to laboratory animals have been shown to cause prophyria (Goldstein et al., 1975), disfunction of the thyroid (Rastomsky, 1974), inhibition of various enzymes (Pardini, 1971), changes in the liver to body weight ratios (Garthoff et al., 1975) various disorders of the liver (Kimbrough et al., 1975), and to alter the level or utilization of corticosteroids (Wasserman et al., 1973) and vitamins A, D and E (Cecil et al., 1973; Wong et al., 1974; Combs et al., 1975). Certain of the less chlorinated PCBs have also been shown to have a mild estrogenic effect when tested in the immature rat and mouse (Kihlstrom, 1973; Nelson, 1974). However, most of these parameters have not been studied in sensitive species or demonstrated at low exposure levels in laboratory animals.

The available data imply that the PCBs and PBBs containing six or fewer halogen atoms are readily absorbed from the gut of higher animals. The available data also imply that the PCBs are not excreted to an appreciable extent prior to metabolism to more polar compounds, and that long term PCB storage is in the skin and adipose tissue (Matthews and Anderson, 1975a). Studies of PBB



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metabolism are not yet available. Since tissue samples from birds and mammals with known exposures to PCBs usually contain only those PCBs with five or more chlorine atoms, it is assumed that the less chlorinated PCBs have been metabolized and excreted. On the other hand there is little evidence that fish can metabolize any PCB and an analysis of fish tissue usually shows a PCB pattern very similar to that to which the fish were exposed (Stalling and Mayer, 1972).

Laboratory studies have demonstrated that the rate of PCB metabolism and thus excretion is approximately inversely proportional to the degree of PCB chlorination so long as there are two adjacent unsubstituted carbon atoms on the biphenyl molecule. When two adjacent unsubstituted carbon atoms are not present the biological half-life of the given PCB may be a matter of years and accumulation of high tissue concentrations with continued exposure is inevitable (Matthews and Anderson, 1975a). It should be noted that the major constituent of Firemaster BP-6 does not have two adjacent unsubstituted carbon atoms and that the corresponding PCB has been shown to have an extremely long half-life in the laboratory rat and probably the human population as well.

It should also be noted that metabolism can also result in further complications of the PCB problem because those PCBs which are most readily metabolized and excreted are those which are most likely to be metabolized via arene oxide intermediates. And where as the reactivity of arene oxides varies greatly and often



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ultimately determines the toxicity, mutagenicity or carcinogenicity of the parent compound (Jerina and Daly, 1974), the PCBs offer such a range of degree and position of substitution that it would not be unlikely to find that one or more of these PCB metabolites would have the proper stability to be a mutagen or a carcinogen. Thus a move from the more highly chlorinated PCB formulations to the less chlorinated ones may help solve the long-term residue problem only to intensify other problems.

## ANIMAL TOXICOLOGY

Gleaning the information that is now available on animal toxicology makes it obvious that different commercial mixtures of PCBs elicit different toxic responses in animals, and that different animal species vary in their susceptibility to the toxic effects of PCBs. Reproduction is severely affected in mink at a dietary level of 5 ppm Aroclor 1254, and a slight effect is still noted at a dietary level of 1 ppm (Ringer et al., 1972). In rhesus monkeys reproduction was reduced at a dietary level of 2.5 ppm of Aroclor 1248 (Allen 1975). In rats a dietary level of 20 ppm Aroclor 1254 depressed reproduction while in the same rat strain (Sherman), in a study conducted simultaneously in the same laboratory, a dietary level of 500 ppm Aroclor 1260 was necessary to reduce reproduction (Linder et al., 1974). In comparative studies done with European products, Phenoclor DB6 and Clophen A60, and the American product, Aroclor 1260, the



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European products were more toxic to chickens than the American product (Vos & Koeman, 1970). The difference in toxicity in this study was attributed to contamination of the European products with chlorinated dibenzofurans and perhaps chlorinated naphthalenes. However, if the differences in the effect on reproduction by the different Aroclors are compared, the contamination with chlorinated dibenzofurans may not be the decisive factor.

Hepatic porphyria has been produced in a number of species, namely the chicken, rabbit, Japanese quail and rat, by a number of commercial mixtures, such as Clophen A60, Phenoclor DP6, Aroclor 1016, 1242, 1254, and 1260 (Vos and Koeman, 1970, Vos and Beems, 1971, Iverson et al., 1975, Goldstein et al., 1974, 1975). It (hepatic porphyria) has not been reported in the monkey, mink or human (see Yusho) mammalian species quite susceptible to the toxic effects of PCBs in other ways. Aroclor 1254 and Aroclor 1242 produced hepatic porphyria in the female rat at doses lower than Aroclor 1016. Again, different commercial mixtures produce this toxic effect at different dosage levels. The hepatic porphyria occurs concomitantly with an increase in ALA synthetase in the liver. Mixed function oxidases are also induced in the liver and comparative studies with PCB isomers have suggested that if the 4,4' positions on the biphenyl ring are occupied by chlorine atoms the effect is most pronounced (Escobichon and Couseau, 1975).

The liver is the primary target organ for PCBs in the rat. Early changes include hepatomegaly with a concomitant increase



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in smooth endoplasmic reticulum. Lipid accumulation at higher

noted with either higher

These changes are not as pronounced with Aroclor 1242 or 1016.

Similar changes have been reported in the primate (Allen, 1975).

PCB mixtures do not represent a no effect level. It is presently not known what the no effect levels are. Other uncertainties



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contribute to these problems. The Yusho oil which caused a poisoning outbreak in Japan was not only contaminated with PCBs but also with high levels of chlorinated dibenzofurans.

In the primate, in addition to the effect on the liver, the gastric mucosa (Allen and Norback, 1973), the skin, and the Meibomian glands are also affected at comparatively low dietary levels and the bone marrow is depressed (Allen 1975) while the gastric mucosa of the rat is only affected at exceedingly high doses (Kimbrough, 1974). Whether the dog also shows an effect on the gastric mucosa needs clarification. In the rabbit, atrophy of the thymus is a toxic manifestation in addition to liver pathology (Vos and Beems, 1971). Fluid accumulation occurs in primates, chickens, and finches. The lymphatic system is also affected in minks.

The few studies conducted with some hexachlorobiphenyl isomers demonstrated that the 3,4,5,3',4',5' hexachlorobiphenyl was the most toxic while 2,3,6,2',3',6' hexachlorobiphenyl was the least toxic isomer (Biocca et al., 1975). Penta-, hexa- and heptachlorobiphenyls are preferentially retained in mammalian adipose tissue for extremely long periods of time (Kimbrough, 1975) (at fairly high concentrations in the rat for a recovery period of 16 months). Whether this has an influence on the toxicity of PCB mixtures has not been determined.

The toxic effects of the contaminants of PCBs have not been extensively studied. While it is assumed that chlorinated



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naphthalenes are toxic within the same dosage range as the chorinated biphenyls, the chlorinated dibenzofurans probably have a greater toxicity. It is assumed that 2,3,7,8 tetrachlorodibenzofuran is the most toxic of this group of compounds. This compound also shows marked species variation. While the single oral $LD_{50}$ in guinea pigs is between 5 and 10 ug/kg bodyweight, 1000 ug/kg TCDF given orally had no effect on rats. Mice are equally insensitive to the toxic effects of TCDF (Moore et al., 1976).

For the brominated biphenyls limited toxicity data are only available on mixtures containing predominately hexa- and octabromobiphenyl. Both mixtures differ sufficiently in isomeric composition so that their toxic effects may quantitatively be quite different and also different from the PCBs. Again the problem of toxic contaminants has not been resolved. If toxic effects are similar to PCBs then, at least in some species such as mink and monkey, longterm low level exposure should result in measureable toxicity. Additional animal studies are needed to resolve some of these problems. Poor metabolism and excretion of the brominated biphenyls may lead to long retention of these compounds predominantly in adipose tissue with accumulation to very high levels on continued exposure. Whether this would lead to sufficient recirculation of the chemicals to cause toxic effects on target organs is presently not known.



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## HUMAN EXPOSURE

Several reports (Jelinek & Corneliussen, 1975; Humphrey, 1976; Kutz and Yang, 1975; Dennie, 1975; Kleinhert, 1975; Hesse, 1975; Yobs, 1972; Kutz and Strassmman, 1975) provide evidence that would indicate that a substential proportion of the population of the United States has been exposed to PCB's. Minimal human exposure of the population to PCB's has occurred from food, air and water, while significant human exposure appears to be limited to sports fishermen consuming fresh water fish from contaminated streams and lakes, and to occupational exposure in industrial workers.

Jelinek & Corneliussen (1975), in reviewing data from the FDA Total Diet Study (1971-1975) report that all food classes of the total diet have declined to no PCB occurrences except in these meat-fish-poultry composites. About 40 percent of these composites continue to contain detectable residues of PCB's, although only traces have been detected in the latter years. The fact that levels in these composites have declined to only traces further support the inference that the meat, poultry and eggs no longer contain detectable PCB's and that the low level findings are probably due to the fish in these composites. This would imply that the PCB levels in the diet may have "bottomed out" and may remain static until such time as there is a change in the PCB residues in fish.



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Data compiled from studies sponsored by the National Marine Fisheries Service (NMFS) also support the continuing presence of PCB residues in fish. A compilation of PCB data, representing the results of all the measurements known to NMFS on PCB's in fish used in the U.S. diet, indicates several important points: (1) while at one time or another, some PCB measurements have been made on many fish, there is an inadequacy of information on PCB residues in the most important fish items in the fish diet; (2) sampling and analysis have been sporadic and not designed to measure trends in human exposure; and (3) systematic surveys of neither the important items nor the species most likely to be contaminated have been undertaken.

However, these survey data do show that in general U.S. fish eaters include a wide variety of fish items in their diet and that some 93 percent of the U.S. population (197 million) consume fish, with the average annual consumption of fish being 15 pounds/year/fish eater.

At present, it is difficult to estimate all human exposure to PCB from eating fish, either from the population as a whole or subgroups at higher risk of consuming large quantities of fish with higher PCB residues. Fragmentary evidence from NMFS data suggests that the exposure of the population as a whole from PCB residues in ingested fish is probably well below 19 mcg/day/consumer, based on PCB levels which are estimated to be below 1 ppm and 19 g of fish consumed/day. This can be compared to



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the results of the FDA Total Diet Study where it has been estimated that the overall PCB daily intake is on the order of 5-10 mcg/day for the general population. The lower FDA estimate is based on the methodology of the Market Basket Survey where fish are purchased at the consumer level and would not be applicable to diets which include a high consumption of fish from certain areas with high PCB residues.

A recently completed study (Humphrey et al., 1976) has attempted to assess some of the consequences of human exposure to PCBs from the high consumption of fish from contaminated areas. The results of this study show that a group of sports fishermen consumed an average of 24-25 pounds of fish/person/year, with the highest individual exposure for a two year period reported as 180 pounds per year. PCB residues in cooked fish ranged from 0.36 to 5.38 ppm. Although there was a wide range of blood PCB levels for each quantity of fish consumed, there was a highly significant correlation between the reported quantity of Lake Michigan fish consumed and the concentration of PCB in the blood of study participants, with the higher reported fish consumption being associated with higher PCB blood levels. The blood values ranged from a low of 0.007 ppm in the control group (fish consumption less than 6 pounds/year) to a high of 0.366 ppm in the exposed group (fish consumption 24-25 pounds/year).

These investigators calculated that the amount of PCB ingested by the exposed group could average 46.5 mg/year and ranged from



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14.17 to 114.31 mg/year. While no systematic adverse health effects could be demonstrated in the exposed group when compared to controls, the investigators caution that any long-term chronic effects are unknown at the present time. Additionally, it can be concluded that exposure similar to those reported in this special group will continue and there is the likelihood that as sportsfishing becomes more popular, larger numbers of people may be exposed in a similar way.

Although human exposure to PCBs from air and water is probably minimal, there seems little question that such exposure does occur. Samples of ambient air collected in Florida, Mississippi and Colorado, show that PCBs were present at all locations. The average concentration at each of the three locations was approximately 100 nanograms per cubic meter of air. Studies of surface water from the major drainage basins of the United States report the widespread occurrence of PCBs in both surface water and bottom sediments. Mean residue levels of PCBs in the surface water ranged from 0.01 to 0.05 mcg/liter, with a maximum residue level of 20.0 mcg/liter.

In Wisconsin, effluents from cooling water in aluminum foundries contained PCBs ranging from 11.5 to 335 ppb. Effluents from paper mills ranged from 0.01 to 25 ppb. Analysis of snow melt water from Wisconsin cities showed PCB residue levels of 0.17 to 0.24 ppb, suggesting that fallout of PCBs from the air may be an important source of PCBs entering the waters of the state.



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In Michigan, testing of 900 samples of industrial effluents showed 22 percent had PCB residues greater than 0.5 mcg/liter, 8 percent greater than 1.0 mcg/liter, 6 percent greater than 10 mcg/liter and 2 percent greater than 100 mcg/liter. With sludge disposal taking place by incineration, landfill and crop or pasture application, the continuation of PCBs in the environment seems obvious.

Adverse human health effects resulting from PCB exposure have come primarily from studies of occupational exposure and from human exposure through the ingestion of contaminated rice oil in Japan.

Schwartz (1936) provided some of the earliest reports of adverse health effects due to occupational exposure in the U.S., in which he described skin lesions and symptoms of systematic poisoning among workers who were reported to have inhaled chlorodiphenyls. There have been numerous reports over the ensuing years describing cutaneous eruptions and of systematic manifestations as well, among marine electricians, machinists, capacitor and transformer manufacturing workers, and others occupationally exposed to PCBs. The skin lesions described by Schwartz in 1936 have come to be designated as "chloracne." Part of the chloracne lesion resembles adolescent acne, but is generally more severe and the lesion distribution is inconsistent with adolescent acne.

Hara (1969) and Hasegawa et al., (1973) have reported dermatologic ailments which include "brown chromodermatosis" of the



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dorsal joints of the hands and purple-like eruptions of the face and neck. However, Hasegawa et al., (1973) performed a health survey of workers in carbonless copy paper factories, two years after the use of PCB in such processes had ceased, and reported no dermal effects nor liver function, urine or blood test abnormalities. PCB blood levels were reported as 0.01 - 0.02 ppm.

Typical clinical findings in the human exposure to PCB which occured in Japan in 1968, and which resulted from the ingestion of rice oil contaminated with Kanechlor 400 included chloracne and increased pigmentation of the skin, increased eye discharge, transient visual disturbances, feeling of weakness, numbness in limbs and some disturbance in liver function. Adult Yusho patients had protracted clinical disease with a slow regression of symptoms and signs. In the dose-response epidemiologic study, the average cumulative intake of PCB's leading to overt symptoms was 2,000 mg, with the lowest dose leading to overt symptoms being 500 mg.

However, Kuratsune et al., (1975) have introduced a new factor into the Yusho incidence with the finding that the rice oil contained chlorinated dibenzofurans (Cl-DBF) at 5 ppm. Nagayama et al., (1975) reports that the toxicity of (Cl-DBF) is said to be from 200 to 500 times that of PCB. Whether or not the (Cl-DBF) contaminant is the crucial toxic substance producing the symptoms observed in the Yusho incident, or whether the exposure to the high levels of Kanechlor 400 produced the observed effects, or



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whether there was an inter-active process taking place is unknown.

The data necessary for determining reasonably accurate time and dose exposure to PBB in individuals in Michigan is either unavailable or non-existent. Attempts to secure accurate dietary intake with the PBB levels in food and the duration of consumption have been unsuccessful. It is hoped that data to be received from the Michigan Department of Public Health may provide some basis for crude estimates.

While there appears to be no evidence at the moment to indicate acute health effects from exposure to PBB, any chronic effects remain largely unknown. A large scale epidemiological study is expected to get underway shortly to identify all the farm family members from quarantined farms; a large group of study subjects secondarily exposed to PBB through the purchase of farm products on a regular basis from quarantined farms and a control group of individuals not exposed to PBB contamination. This study will continue efforts to identify any acute or chronic effects of PBB exposure through physical examination, biochemistry tests, and dietary histories. Efforts will continue to assess the original Firemaster BP-6 for the presence, if any, of chemical contaminants which might present human health hazards. In the meantime, a ten-fold safety factor for PBBs when compared to PCBs appears both reasonable and acceptable based on all currently available scientific data.



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Much work remains to be done concerning the toxicity of PCBs and PBBs and the association of these compounds with any demonstrable adverse human health effects. To accomplish some of these needs a list of recommendations follows.



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## GENERAL RECOMMENDATIONS

More complete and additional metabolism studies with various individual PCBs and PBBs should be undertaken. Attention to the formation of possible toxic metabolites of these compounds should be considered, as well as detecting and quantifying any specific biochemical parameters which may be affected as a result of such exposures. Based on the above, lifetime feeding studies should be conducted with selected commercial PCBs and PBBs mixtures, as well as selected individual PCBs and PBBs.

Why some species seem to be more susceptible to the toxic effects of PCBs, and which are most closely related to man in their response should be determined. In addition, more experimental data is needed for the occupational PCBs settings; i.e., via the respiratory or dermal route. Additional toxicological evaluations should be made of fish and other foods which contain high levels of environmentally accumulated PCBs and other contaminents. Appropriate toxicological studies should be undertaken with PBBs in order to be able to make some predictions on possible human effects.

The quantitative evaluation of halogenated biphenyl exposure to man with respect to blood and body fat levels, and any possible health effects, needs further study. With respect to PCBs, this may be accomplished by identifying a population of individuals consuming large amounts of fish contaminated with high levels of



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PCBs. In addition, those exposed industrially should be included in these efforts. With respect to PBBs, appropriate studies should be initiated with Michigan farm families exposed through consumption of contaminated meat and dairy products.

Individual chemical components of PCB residues should be qualitatively identified and procedures for improving quantitation of these residues should be investigated. Commercial PCB (and PBB) mixtures need to be analyzed further to determine which chlorinated dibenzofurans (or, in the case of PCBs, whether any brominated dibenzofurans) are present as contaminents. Specific chlorinated dibenzofuran compounds must be synthesized for use in development of analytical procedures and to aid in identification of contaminants. Analytical procedures are needed to permit examination of foods for presence of chlorinated dibenzofurans. Toxicological studies should be carried out on chlorinated dibenzofurans and brominated dibenzofurans.



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## DETAILED RECOMMENDATIONS

### CHEMISTRY

Investigate procedures (such as Webb and McCall's, J. Chromatog. Sci. 11, 366-373 (1973)) for possible improvement of the quantitation step in the analysis of PCB residues.

Present procedures for analysis of PCBs yield a "PCB residue" that is commonly examined using electron capture GC. Individual components of this residue should be identified and it should be determined whether any of the electron capture GC peaks can be attributed to Cl-DBFs or Cl-naphthalenes.

Analyze commercial PCB mixture (e.g. Aroclors) to determine which Cl-DBFs are present as contaminants. Qualitative and where possible, quantitative studies should be carried out.

Synthesized, individual chlorobiphenyls should be examined for presence of chlorinated dibenzofurans before metabolism or toxicity studies are carried out with the biphenyl. (The importance of this is shown by the reported formation of 2,3,7,8-$Cl_4$ DBF as a side-product in the Ullmann coupling reaction used to synthesize the symmetrical 2,2',4,4',5,5'-hexachlorobiphenyl.)

Synthesize, purify and characterize by physical and chemical means a range of Cl-DBFs to be used for analytical procedure development as reference standards for confirmation purposes and for suitable toxicological studies.



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Develop a suitable procedure for analysis of edible fish for Cl-DBFs, emphasizing recovery and quantitation of 2,3,7,8-tetrachlorodibenzofuran.

Analyze a commercial mixture of brominated biphenyls (Firemaster BP-6) for brominated dibenzofuran content.

Analyze long-used heat exchange, transformer and capicitor fluids for chlorinated dibenzofurans.

Synthesize specific $^{14}C$ or tritium radiolabeled chlorinated biphenyls that may be needed for toxicity and metabolism studies.

Study the reported conversion of certain chlorinated biphenyls to Cl-DBFs under sunlight or sunlight-simulating conditions. If feasible, extend the work to additional Cl-biphenyls.

Simulating conditions applicable to heat exchange units and/or transformers, determine whether Cl-DBF content of PCB mixtures increases when the PCBs are heated and/or exposed to air.

Carry out laboratory studies, simulating environmental conditions, to determine alteration products from Cl-DBFs when irradiated or heated.

METABOLISM AND BIOCHEMICAL TOXICITY

Long-term feeding studies approaching life-time should be conducted with the less chlorinated PCB formulations. These



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studies should include at least two of the less chlorinated PCB formulations and a series of carefully selected individual PCBs.

Further and more complete metabolism studies should be done with the more highly chlorinated PCBs, for comparison with those done with PCBs having five or less chlorine atoms.

The effect of chlorine position on PCB metabolism and arene oxide formation should be further elucidated.

The formation of specific chlorinated dibenzofurans as possible PCB metabolites should be studied, their biochemical toxicity in mammalian species determined and the magnitude of their toxicity ascertained. These studies will also include those specific chlorinated dibenzofurans identified in new and used PCBs.

If possible, arene oxide metabolites of several PCBs should be synthesized and their toxicology investigated.

Within an isomeric series of PCBs, (e.g., tetra or hexa), metabolism of the more toxic members should be compared to those which are less toxic.

Metabolism of individual PBBs, comparable to those for PCBs, should be studied. These data should be related to known toxicities.



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The toxicology of known PCB metabolites should be studied.

Porphyrin excretion by the people and animals exposed in the Michigan PBB incident should be checked.

People who are suspected to have received high exposures of PCBs or PBBs should be checked for any changes in enzyme levels—e.g., protein-bound iodine, antipyrene metabolism, serum cholesterol, etc.

The adsorption, distribution and excretion studies of halogenated dibenzofurans and PBBs should be studied.

Further studies of possible PCB metabolism by fish.

Study the degree of chlorination and length of exposure on the estrogenic effects and reproductive organs with particular emphasis on the lower chlorinated PCBs (e.g., 1016).

ANIMAL TOXICOLOGY

Inhalation studies with "heated" PCB mixtures simulating occupational exposure and dermal toxicity studies should be conducted.

Studies as to why some species seem to be more susceptible to the toxic effects of PCBs than others and of these species, which are most closely related in their response to humans should be undertaken.



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Subacute and chronic toxicity studies should be conducted with hexabromobiphenyl (Firemaster FF-1) in order to be able to make some predictions on human PBB toxicity.

The toxicity of fish containing high levels of PCBs which have been accumulated by environmental exposure should be evaluated. It should be established whether and which toxic impurities are present in used PCBs, such as transformer oil and capacitor fluid.

Long-term feeding studies, approaching life-time, should be conducted with laboratory animals and selected commercial mixtures of PCBs and PBBs as well as selected individual PCBs and PBBs.

Further and more complete metabolism studies should be done with individual PCBs and PBBs. The formation of toxic metabolites of these compounds should be considered and carefully investigated.

Efforts to detect and quantitate specific biochemical mechanisms which are affected or altered by exposure to PCBs or PBBs should be continued.

## HUMAN EXPOSURE

### Polychlorinated biphenyls

Recommend that a population of fish eaters be identified who are consuming substantial amounts of fresh water fish with high levels of PCBs. This study population should be followed prospectively with adequate dietary histories, blood and body fat levels



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of PCBs, a health questionnaire, liver function tests and other biochemical studies.

Recommend a similar study be carried out in an industrial population exposed to PCBs.

Systematic studies to identify sub-groups of the total U.S. population who consume high levels of fish with more accurate measures of PCB exposure.

Polybrominated biphenyls

Recommend that a large scale epidemiological study be carried out to (1) identify all farm family members from quarantined farms in Michigan, (2) identify those individuals with secondary exposure to PBB contamination through the purchase of dairy products from quarantined farms on a regular basis, and (3) a suitable control group of non-exposed farm families. These groups should be followed to assess any short and long term adverse health effects related to PBB exposure.

Subgroups of these farm families from quarantined farms should be studied by:

1. Repeat PBB blood levels.
2. Matched blood-body fat PBB levels.
3. Pregnant female-infant studies.
4. A battery of biochemical studies.



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Ancillary Studies

Toxicology studies in non-human primates and rodents with hexabrominated biphenyl and any brominated contaminants of hexabrominated biphenyls.

Toxicology studies in several animal species (mice, rats, dogs, non-human primates using hexabrominated biphenyl and several PCBs to determine species differences in response to the effects of these compounds.

Toxicology studies feeding hexabrominated biphenyls and selected PCBs singly and in combination to determine if there are different or additive effects when fed in combination.



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# CHEMISTRY OF PCBs AND PBBs

## I Introduction

A recent book by Hutzinger et al. (1974) reviews the chemistry of chlorinated biphenyls through the end of 1973. This wide-ranging review also includes discussions of metabolism and determination of chlorinated biphenyls.

The present report reviews areas of chlorobiphenyl and bromobiphenyl chemistry held to be pertinent to the Subcommittee's concern for possible health effects of these types of compounds. No attempt at complete coverage of these subject areas is claimed. Findings reported since 1973 are particularly stressed.

Non-metabolic alteration of the halogenated biphenyls is considered in terms of agents most available to induce chemical change in the environment. Expected differences between chlorinated biphenyls and brominated biphenyls in their environmental chemistry are commented on.

It is possible that certain of the chlorinated biphenyl compounds will be shown to have relatively greater potential hazard for health. Therefore, occurrence and fate in the environment and metabolism and toxic effects in animals are related to chemical structure of the biphenyls.

For the general public, food is presumed to be the major



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source of chlorinated biphenyl compounds. The important concepts related to analytical methodology for PCB (and PBB) residues, particularly in foods, are reviewed. Alternative means of quantitation of these residues and problems in the quantitation techniques are discussed.

The pertinent aspects of chlorinated dibenzofurans, as known contaminants in commercial PCBs and as potentially significant environmental contaminants, are reviewed.

In reviewing the chemistry related to health effects of the PCBs and PBBs, it must be remembered that environmental contamination by these two commercial chemical mixtures is vastly different in magnitude. The former have been steadily released into the environment, in many countries, presumably over decades, and are now found to be a pervasive, world-wide contaminant. The number of chlorinated biphenyls reaching the environment probably number nearly 100 different compounds. The PBBs, encompassing a small number of chemical structures to begin with, are of concern due to a single, fairly recent contamination incident, apparently limited to the State of Michigan.

## II. Chemistry of Chlorinated Biphenyls

### A. Synthesis and analysis of Commercial Mixtures

Commercial preparation of mixtures of chlorinated biphenyls by reaction of biphenyl with chlorine has been described by Hubbard (1964). A more recent discussion of this subject,



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including the newer preparation designated as Aroclor 1016, is now available (Mieure et al., 1976). Analysis of the chlorinated biphenyl content of various American commercial PCB mixtures (Aroclor) has been reviewed by Hutzinger et al. (1974). An estimated 40 to 60 different chlorinated biphenyl compounds are present in each of the higher chlorinated commercial mixture. (There are 209 possible compounds obtainable by substituting chlorine for hydrogen on from one to ten different positions on the biphenyl ring system; see Appendix.) PCB commercial mixtures produced in the U.S. and elsewhere have been shown to contain classes of compounds other than the chlorinated biphenyls: chlorinated napthalenes and chlorinated dibenzofurans (Cl-DBFs), for example (Vos et al., 1970; Roach and Pomerantz, 1974 a,b; Bowes et al., 1975a). The possibility that naphthalene and dibenzofuran contaminate the technical biphenyl feedstock used in preparation of the commercial PCB mixtures has not been excluded.

Available toxicity information indicates that, of the identified types of contaminants in commercial PCB mixtures, Cl-DBFs pose the greatest potential hazard (See Section II.E on Chlorinated Dibenzofurans). As a result, the Cl-DBFs have become the focus of studies on contaminants in these mixtures. No report has appeared of an attempted complete content analysis of the trace impurities in a commercial PCB mixture.



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B. Non-metabolic Alteration of Chlorinated Biphenyls

Chlorinated biphenyls, as is typical of aryl chlorides generally, are quite stable to chemical alteration. Consideration of possible non-metabolic alteration routes for these compounds in the environment suggests air oxidation, aqueous hydrolysis, and photoalteration in sunlight as potential reactions to be investigated. (Thermal decomposition would be of concern in connection with incineration conditions meant to destroy waste chlorinated biphenyls). Study of these reactions under the complex and variable sets of conditions existing in the environment is difficult. Chemical reactivity of chlorinated biphenyls has therefore been studied almost entirely under the more controlled conditions of the laboratory.

Oxidation

PCBs are fairly stable to oxidation under moderate conditions. PCBs are stable under conditions which easily oxidize DDE, and can be separated from DDE by oxidizing DDE to the more polar dichlorobenzophenone prior to column chromatographic separation (Trotter, 1975). To effect the oxidation of DDE, the PCB-DDE solution is refluxed in 2.5% chromic acid-acetic acid on a steam bath. Some of the lower chlorinated biphenyls, however, are not recovered after the oxidation. It is reported (Weingarten, 1961) that mono-, di-, and trichlorobiphenyls are oxidized in 7.5%



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chromic acid-acetic acid to their respective (chloro-) benzoic acids. With vigorous oxidizing conditions in the environment, some PCB oxidation, especially of the lower chlorinated biphenyls may occur. It is difficult, however, to assess the total extent of possible environmental oxidation of PCBs.

Hydrolysis and Alcoholysis

PCBs are fairly stable to hydrolysis under moderate conditions. When refluxed with 2% KOH in ethanol, PCBs are stable (Trotter, 1975; Young and Burke, 1972). A halogen on a chlorobiphenyl molecule is not easily displaced in a nucleophilic substitution reaction. Under vigorous conditions the 4 and the 4,4' positions of decachlorobiphenyl are found to be the most susceptible to chlorine displacement by hydroxide and methoxide ions. Decachlorobiphenyl can be hydrolyzed to octachloro-4,4'-biphenylol when treated with aqueous alkali at high temperatures in an autoclave (Smith, 1948; Societe d'Electro Chimie, 1962). Heating 2,5-dichlorobiphenyl with sodium methoxide produces 2-chloro-5-biphenylol (deCrauw, 1931). Decachlorobiphenyl when treated with sodium methoxide in pyridine yields the 4-methoxy- and 4,4'-dimethoxy-chlorobiphenyl (Binns and Suschitzky, 1971). It is probable that the non-metabolic hydrolysis or alcoholysis of PCBs in the environment is limited.

Photochemistry

Exact environmental conditions for the photochemistry of



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environmental contaminants, including PCBs, are sometimes difficult to simulate in the laboratory. Limited understanding exists concerning the parallel of laboratory photoreactions and reaction rates under non-environmental conditions and possible environmental photolyses. The exact environmental conditions under which residues may photoreact can be diverse and difficult to evaluate. Unknown or discounted factors in the environment, such as possible sensiters or quenchers, may be important. The frequency (energy) and intensity of a laboratory photolysis light may not be comparable with environmental conditions. 3000 Å probably represents the practical lower limit of the ultraviolet (UV) portion of sunlight (Crosby, 1969). The residue must be exposed to sufficient light to react appreciably. It is necessary to consider the physical state of the residue exposed to light in the environment. The residue may exist as a solid or liquid (e.g., a film), a solution or a vapor. In photolyzing solutions the solvent can have an important effect. Photoreactions which proceed in one solvent may not proceed or may proceed at a different rate in another solvent. A muddy river containing a residue may yield different results compared to a laboratory photolysis in hexane. Laboratory photolyses under a variety of conditions which simulate most, if not all, of the environmental conditions of photolysis could be extremely significant and relevant. In the absence of these, we must rely on the questionable extrapolation of laboratory photolyses under non-environmental conditions.



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The photochemistry of PCBs has been described by Hutzinger, et al.,(1972; 1974). In contrast to oxidation and hydrolysis, PCBs are fairly easily photoreacted under certain laboratory conditions. PCB photochemistry has been studied in various solutions, as a thin film, and as a vapor. In hydrocarbon solvents progressive reductive dechlorination of the PCB is the predominant photochemical reaction (Hutzinger et al., 1974). Irradiation of solutions of chlorobiphenyls (Hutzinger et al., 1972) and trapped gas liquid chromatographic (GLC) effluents (Hannan, et al.,1973) have shown that PCBs with higher chlorine content dechlorinate more readily. 0.1% hexane solutions of 2,2',5,5'-tetrachlorobiphenyl and 2,2',4,4',5,5'-hexachlorobiphenyl were each irradiated with 3100 Å light in a Rayonet reactor (Hutzinger et al., 1972). After 24 hours of irradiation, 33% of the 2,2',5.5'-tetrachlorobiphenyl was unreacted while < 1% of the 2,2'4,4',5,5'-hexachlorobiphenyl remained. In dechlorinating PCBs by photoexcitation, ortho chlorines preferentially cleave (Ruzo et al., 1974). With PCBs containing only meta and para chlorines, meta chlorines are lost preferentially. The rate of dechlorination is faster in alcohol solvents, such as methanol (Hustert and Korte, 1972) than in hydrocarbon solvents. Irradiation of 2,2',4,4',5,5'-hexachlorobiphenyl in methanol yields, in addition to reductive dechlorination, ring methoxylation (Ruzo et al., 1974). Photochemical studies of PCBs in aqueous solutions may offer a significant relevance to environmental PCB photochemistry. A 0.4% Aroclor 1254 suspension in water-dioxane (7+3) in the presence of sodium bicarbonate with



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air bubbling through the mixture was irradiated with 3100 Å light. Two thin layer chromatographic fractions of the Aroclor 1254 after irradiation corresponding to the "addition of water to chlorobiphenyls" and a "carboxy fraction" were found (Hutzinger et al., 1972). Presumably PCBs formed by reductive dechlorination were also produced. Black light irradiation of Aroclor 1254 as a thin layer film in the presence of water yields a "carboxy fractic and a fraction whose mass spectrum indicates hydroxychlorobiphenyls (Hutzinger et al., 1972). Sunlight irradiation of certain chlorobiphenyls as a thin film without the presence of water gives reductive dechlorinated products (Hutzinger et al., 1974). Chlorinated terphenyls and quaterphenyls are also produced by the black light and sunlight irradiation of certain chlorobiphenyls as a thin film (Hutzinger et al., 1974). Vapor phase photolysis of PCBs may be extremely relevant especially for the more volatile components of Aroclors with low chlorine content. Sunlamp irradiation of the vapor phase of a refluxing suspension of 2 2',5,5'-tetrachlorobiphenyl and water yields "carboxy" products (Hutzinger et al., 1972).

These laboratory photoreactions represer . chemical conversions which may proceed in the environment. The photoreaction rates under diverse e vironmental conditions and the extent of PCB degradation or reaction in the environment are extremely difficult to assess.



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The reported low-yield conversion of certain chlorinated biphenyls to Cl-DBF is discussed under Chlorinated Dibenzofuran, Section II.E.

C. Chemical Structure Related to Occurrence, Fate and Effects of Chlorinated Biphenyls

The physical and chemical properties of chlorinated biphenyl compounds and commercial mixtures of PCBs vary greatly depending on the degree and position of chlorine substitution on the biphenyl ring system. Of particular importance to their environmental occurrence and fate are the properties of volatility, water solubility, bioaccumulation, biodegradability and photostability. Volatility, water solubility and bioaccumulation are of more importance as mechanisms of introduction into and transport within the environment. Biodegradation and photodegradation are more important as mechanisms of removal.

There is data (Mieure et al., 1975) which indicates that the higher chlorinated biphenyls are less volatile and less water soluble than the lower chlorinated biphenyls. These two factors would tend to enhance the ratio of lower to higher chlorinated biphenyls in the environment. There is no clear data to assess the involvement of chlorine position in volatility and water solubility, but is likely to be of less importance. Therefore, the lower chlorinated biphenyls are more likely to volatilize into and be selectively transported in the aqueous environment.



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The relationships between structure and bioaccumulation factors for PCB isomers are only beginning to be assessed (McKinney, 1975; Sugiura et al, 1975) with the availability of purified specific isomers for study. However, it appears (Zitko et al., 1972; Veith, 1975) that the peak patterns from gas chromatography found in environmental samples representing the biosphere most closely resemble the higher chlorinated commercial mixture, Aroclor 1254. This suggests a selective accumulation of the more highly chlorinated components of the commercial mixtures in biological material. This also indicates that the higher chlorinated biphenyls are able to find their way into the environment in spite of their poorer volatility and water solubility. Pharmacokinetic studies (Matthews, 1975) with selected radiolabeled chlorinated biphenyl compounds have confirmed the trend toward increasing biological half-life with increasing chlorine number. However, these findings may be the result of both higher accumulation rates and lower elimination rates for the higher chlorinated biphenyls. It may be of interest to note that fish from lower and intermediate levels of the food web have been found (Zitko et al., 1972) to contain lower amounts of hexa and higher amounts of tetra and penta than higher trophic level white and silky sharks and aquatic birds This suggests that selectivity in bioaccumulation is also a function of biospecies.

Again the effects of varying chlorine position are less



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clearly understood, and the best assessment of this comes from study of two or more members of an isomeric series of the chlorinated biphenyls. Recent work with five symmetrical hexachlorobiphenyl isomers in chicks (McKinney et al., in press; Goldstein et al., in press) and subsequently in mice (Biocca, 1975) has demonstrated that separate and distinct differences in isomer toxicity are possible and that the differences are related to chemical structure via effects of varying chlorine substitution on compound lipophilicity and metabolism. The hexachloro isomers studied were 3,4,5,3',4',5'-; 2,4,6,2',4',6'-; 2,3,4,2',3',4'-; 2,4,5,2',4',5'-; 2,3,6 2',3',6'- and 2.3,5 2', 3',5'-. Those hexa-isomers with 4,4'-substitution appear to be more rapidly accumulated and may be more slowly metabolized, i.e., these isomers showed greater accumulation in adipose tissue and increased activity in the liver and overall greater toxicity. These differences are believed to be associated with differences in molecular polarizability and to be measureable by spectroscopic and chromatographic techniques.

Other workers (Ax et al., 1976; Ax and Hansen, 1975) have compared the toxicity of two pentachlorobiphenyls in laying chicks. The isomer with 4,4'-substitution (2,4,5,3',4') showed higher average embryonic mortality and teratogenicity in unhatched eggs but lower decreased fertility in the chick than 2,3,6,2',3' the isomer not chlorinated in the 4-position. Although it is difficult to assess the role of metabolism here, the data would suggest a



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more rapid accumulation of the 2,4,5,3'4'-isomer in the egg. Other workers (Bush et al., 1974) have already observed a correlation between PCB content of eggs and embryo mortality and teratogenicity.

Various other workers (Ecobichon and Comeau, 1975; Johnstone et al., 1974; Hill et al., 1974) studying a range of chlorinated biphenyls have generally observed the importance of degree of chlorination as well as position of chlorine substitution (especially 4,4'-substitution) in overall biological effectiveness. Although some of the compounds studied are unrealistic as components of commercial PCB mixtures, they have served as models to demonstrate the importance of chlorine number and position. Nevertheless, of 50 possible components of Aroclor 1254 identifed by Sissons and Welti (1971) as many as 19 (38%) could have 4,4'-substitution. It is of interest to note here that one of the possible persistent isomers found in Yusho patient tissue has been tentatively identified (Kuratsune, 1975) as the 2,3,4,5,3',4'-heptachlorobiphenyl. This isomer has been prepared (McKinney, personal communication) and its toxicty will be determined especially in relation to the highly toxic 3,4,5,3',4',5'-hexa isomer previously tested.

It has been only recently that hydroxylated metabolites of PCBs have been isolated and identified (Jansson et al., 1975) in environmental samples. The study of the metabolism of the commercial mixtures (deFreitas and Noretrom, 1974) themselves has



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received little attention for obvious reasons. Recently, the metabolism of purified chlorinated biphneyls, some radiolabeled for quantitation purposes, has been studied in a number of biological systems. Although there are a large number of publications in the literature on this subject, this report will concern itself with those studies which seem to support generalities in terms of the effects of varying chlorine number and position on metabolism with particular reference to mammalian systems and to realistic components of Aroclors.

The distribution and excretion of a series of four radiolabeled ($^{14}C$)chlorobiphenyls which have degrees of chlorination similar to and are themselves constituents of Aroclors 1221, 1242, 1254, and 1260 have been studied (Matthews, 1975) in the male rat. These studies have clearly shown an increasing biological half-life with increasing chlorine number for the series (4-; 4,4'-, 2,4,5,2',5'- and 2,4,5,2',4',5'-) which is believed to be related to the rate of metabolism and the ease of formation of the arene oxide intermediate. These compounds were also dosed at realistic exposure levels (0.06 to 6.0 mg/kg) which showed that little if any of the chlorobiphenyl is excreted in unchanged form. Most other metabolic studies (Hutzinger et al., 1974) with selected chlorobiphenyls have dealt with much higher dose levels (generally resulting in excretion of much unchanged biphenyl) and unlabeled compounds which are of little quantitative value in assessing the effects of chlorine degree and position.



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There has been no similar study with selected chlorinated biphenyls within an isomeric series to assess the effects of chlorine position on metabolism with the possible exception of an incomplete study (Hass et al., unpublished) on the identification of metabolites from the excreta of chicks (and mice) fed symmetrical hexachlorobiphenyl isomers. It is essential that this be done with symmetrical isomers (in order to simplify the problem of interpretation) and preferably in two or more isomeric series (tetras and hexas).

The various metabolism studies have generally shown the occurrence of polar hydroxylated compounds as major components of the metabolite mixture. As the degree of chlorination increases, hydroxylation can be concurrent or concomitant with dechlorination (Hutzinger et al., 1974). There is also evidence for the formation of methyl ethers (Safe, 1975) and methylchlorobiphenyls (Hass et al., unpublished) as metabolites.

There is an increasing body of evidence (Safe et al., 1975) to support the arene oxide intermediate in hydroxylation, especially (Chen et al., Division of Nutrition and Disposition, in press) in cases where vicinal (1-2,-adjacent) unsubstituted carbons are found in the molecule. The corresponding dihydrodiol and/or catechol generally occur along with the phenolic metabolites.

Although it has not been studied in mammalian systems, there is at least one report (Baxter et al., 1975) that studies of



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individual chlorinated biphenyls do not accurately predict the rates of metabolism of the same compound in simple mixtures. Therefore, the overall problem of metabolic degradation of the commercial mixtures may be complicated by the possibility that certain PCBs are potent enzyme inducers but poor substrates and vice versa with various degrees in between. This is further complicated by the fact that certain chlorinated biphenyls occur in optically active forms and only one enantiomer may be biologically active and undergo enzyme interactions. The existence of nine of the major, and ten of the minor constituents of Aroclors 1242, 1254, and 1260 in optically active forms has been predicted (Kaiser, 1974).

D. Determination of PCB Residues

The determination of PCB residues has been reviewed in detail by the WHO Task Group report on "Environmental Health Criteria for Polychlorinated Biphenyls and Terphenyls" (WHO Task Group, 1975) and in "Chemistry of PCBs" (Hutzinger et al., 1974).

PCBs are lipophilic, quite similar in this respect to DDE, the metabolite of DDT. An analyses for PCB residues follow procedures the same as or similar to those used for multiple residues of organochlorine pesticides (WHO Task Group, 1975; Hutzinger et al., 1974; Food and Drug Administration, 1968-1975, Horwitz, 1975; Hearing Clerk, Dept. HEW, 1973). The individual steps that follow sampling are extraction of residues from the sample, isolation of residues from coextractives (cleanup),



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separation of PCBs from interfering chlorinated hydrocarbon pesticides, quantitation, and confirmation of identity. In general analytical methods capable of completely extracting residues of organochlorine pesticides from sample substrates and of quantitatively recovering the pesticides through subsequent cleanup procedures are also capable of achieving quantitative analysis for PCBs. The lipophilicity of chlorobiphenyls which increases with increasing chlorine content may influence their recovery through some analytical methods (Stalling et al., 1972). For example, recoveries through the frequently used cleanup step, partitioning of PCB from a petroleum ether or hexane solution of fat or oil to acetonitrile, exceed 95 percent for Aroclor 1242 but drop to 75-80 percent for Aroclor 1260 (Hearing Clerk, Dept. HEW, 1973). Overall ability of typical analytical methodology to recover PCBs added in vivo to food samples is shown by two interlaboratory studies conducted within the Food and Drug Administration (FDA) and one study conducted for the Association of Official Analytical Chemists (AOAC) (Food and Drug Administration, 1971; Burke, 1972; Food and Drug Administration, 1973; Sawyer, 1973). Using methodology described in the FDA Pesticide Analytical Manual and in the Book of Official Methods of the AOAC, average recoveries and coefficients of variation for determination of Aroclors added as unknowns to different foods were: Aroclor 1254-fish, 74 $\pm$ 9 percent; Aroclor 1254-infant chicken, 89 $\pm$ 22 percent; Aroclor 1242-chicken fat, 101 $\pm$ 13 percent; Aroclor 1248-chicken fat, 96 $\pm$ 9



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percent;Aroclor 1254-fish, 75 + 14 percent; Aroclor 1260-fish, 75 + 15 percent. The three studies involving fish required separation of the PCBs from the DDT group before quantitation.

Gas chromatography with electron capture detection is the most widely used procedure for determination of PCB residues. Gas chromatographic columns with methyl silicone liquid phases are widely used under conditions that typically separate the Aroclors and PCB residues into about 15 peaks (Fishbein, 1972). The 10 percent DC-200 (or OV-101) column described in the FDA Pesticide Analytical Manual separates Aroclor 1254 into 14 peaks (Armour, 1972). These analytical columns do not give a highly defined representation of the Aroclor or PCB residue; several of the peaks in Aroclors result from mixtures of more than one chlorobiphenyl (Sissons and Welti, 1971; Stalling and Huckens, 1971; Webb and McCall, 1972; Webb and McCall, 1973). A column prepared from purified Apiezon L has been shown to separate a 54 percent chlorine PCB mixture into over 40 peaks (Jensen and Sundström, 1974). Separation of the PCB residue into three fractions by chromatography on charcoal prior to examination on the Apiezon L column has made it possible to characterize and quantitate nearly 60 technical PCB components. This method provides a way to get much needed, more detailed information on the compositon of PCB residues and might be used in special studies. The increased time and complexity of this approach would probably limit its application in regular monitoring analysis.



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The need to confirm residue identity and the procedures available are similar for PCBs and pesticides. The multipeak gas chromatographic pattern of a PCB residue may be very similar to that of a commercial Aroclor or, as in most biological samples, the residue peak pattern may be changed to varying degrees from that of a given Aroclor. The use of column chromatographic or chemical reaction procedures to separate PCBs from organochlorine pesticides increases the certainty of the gas chromatographic identification. Procedures readily available to the residue laboratory for confirming the identity of PCBs include: halogen specific gas chromatographic detectors (Hearing Clerk, Dept. HEW, 1973), stability of the residue peak pattern after refluxing the extract with alcoholic alkaline solution (Young and Burke, 1972), perchlorination of the residue to the decachlorobiphenyl derivative (Berg et al., 1971; Armour, 1973), and thin layer chromatography (Fehringer and Westfall, 1971; deVoe and Peet, 1971). Mass spectrometry is not readily available to many laboratories conducting analyses for PCBs nor would the expense justify regular use in monitoring programs. However, the use of mass spectrometry is encouraged for residues and samples which are unusual or significant and for the occasional examination of a so-called routine sample.

Two especially critical considerations are associated with the determination of PCB residues: (1) their separation from potentially interfering organochlorine pesticides, particularly



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DDT, TDE, and DDL or the multicomponent chlordane or toxaphene, and (2) quantitative measurement of the multicomponent PCB residue which, in biological organisms, is usually changed in relative amounts of chlorobiphenyl components from commercial Aroclors or which may result from mixture of PCB from different sources. To achieve reliable residue results it is essential that the analyst correctly make critical judgements and interpretations in dealing with mixed residues of PCBs and organochlorine pesticides and in quantitation of the multicomponent PCB residue. There is no substitute for analyst experience in this analysis.

PCBs are separated from certain pesticides in the usual cleanup procedures, e.g., adsorption chromatography on Florisil or alumina or by gel permeation chromatography (Food and Drug Administration, 1968-1975; Stalling et al., 1972; Holden and Marsden, 1969). When DDT, TDE, DDE, chlordane, or toxaphene are present, ancillary procedures designed to separate these chemicals from PCBs must be used. The DDT analogs, particularly DDE, are the pesticide residues most frequently encountered in samples and DDE is the most difficult to separate from the PCBs. The effective electron capture gas chromatographic response for DDE is 20-30 times greater than for an equal weight of Aroclor 1254 and the response for DDT is only slightly less than that for DDE (Food and Drug Administration, 1968-1975). Without proper treatment these pesticides can readily interfere in the determination of PCBs. Some procedures utilizing column chromatography



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on alumina or Florisil separate DDT and TDE and intentionally collect DDE and PCBs in the same fraction (Holden and Marsden, 1969; Reynolds, 1969), which is analyzed. In these cases the analyst must exclude the DDE region of the gas chromatogram from the quantitative measurement of PCBs. Column chromatography on charcoal also has been proposed for separation of PCBs from DDE, DDT, and other organochlorine pesticides (Berg et al., 1971). Column chromatography on silicic acid is probably the most widely used procedure for separating PCBs from DDT and analogs (Food and Drug Administration, 1968-1975; Armour and Burke, 1970). The technique is fairly lengthy and difficult to reproduce, requiring empirical standardization in each laboratory (Sawyer, 1973; Masumoto, 1972; Edwards, 1974). The separation of DDE is probably not 100 percent effective but under ideal conditions practically all DDE can be separated; the usual inconsistency is for a portion of the DDE to be eluted from the column with the PCBs, requiring allowances to be made in PCB quantitation. The lower chlorinated PCBs present the greatest difficulty in separation from the DDT group by column chromatographic procedures. DDE can also be separated from PCBs by oxidation to the dichlorobenzophenone, followed by a chromatographic separation of this more polar derivative from PCBs. DDT and TDE may be dehydrochlorinated to their respective olefins and similarly separated along with DDE (Mulhern et al., 1971; Collins et al., 1972; Trotter, 1974). This technique has not received as much application as the chromatographic procedures, probably because pesticides and lower



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chlorinated biphenyls are destroyed or changed, preventing their determination. A variation on this approach used sodium dichromate plus a minute amount of sulfuric acid, rather than dehydrochlorination followed by oxidation with chromium trioxide in acetic acid. The dichromate reagent is reported to convert DDE quantitatively to the dichlorobensophenone without affecting DDT, TDE, or any of the chlorobiphenyls (Jensen and Sundström, 1974).

Quantitation of PCB residues is done in most laboratories by comparison of measurements made on the multicomponent electron capture gas chromatograms of the residue and a known quantity of reference material. The PCB residue in biological samples, as mentioned earlier, is very likely to be changed in the relative amounts of chlorobiphenyl components from any one Aroclor or the residue may be a mixture of PCBs from different sources (Food and Drug Administration, 1968-1975; Jensen and Sundström, 1974; Cook, 1972). The response of the electron capture detector varies with the number and location of chlorine atoms in the biphenyl molecule (Gregory, 1968; Zitko et al., 1971). With the electron capture system described in the FDA Pesticide Analytical Manual, the response/unit/weight increases about 6 fold from Aroclor 1242 to Aroclor 1260 (Hearing Clerk, Dept. HEW, 1973). The halogen specific microcoulometric and electrolytic conductivity detectors respond proportionally to the weight of chlorine present and in theory could provide a more accurate measurement of a PCB residue that is not exactly the



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same composition as the reference (Hearing Clerk, Dept. HEW, 1973). However, lower sensitivity, difficulty in maintaining optimum performance, and limited availability in residue laboratories has limited the use of these detectors. Unless the PCB residue and reference Aroclors are exactly the same in chlorobiphenyl composition, the gas chromatographic determination cannot be considered accurate; the greater the difference between residue and reference composition, the greater the deviation between determined and the true residue level (Hearing Clerk, Dept. HEW, 1973; Beezhoold and Stout, 1973). The most frequently used approaches to quantitation have been selected for the reference the Aroclor with the most similar gas chromatographic pattern to the residue and: (1) compared the response of a single peak in the residue with the response of the counterpart peak in the Aroclor reference; or (2) compared the total response for several peaks from the residue to the total response for the counterpart peaks in the reference; or (3) compared the total response for all peaks in the residue with the total response for all peaks in the reference; or (4) in a greater effort to duplicate the residue peak pattern, prepared a reference made up of one or more Aroclors to simulate the gas chromatographic pattern of the residue, and compared the total response for all peaks of the residue to the total response for all peaks in the reference (Sawyer, 1973; Beezhold and Stout, 1973). Response has been measured in terms of both peak height and area (Sawyer, 1973). The latter approach which utilizes an Aroclor or a mixture



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of Aroclors to duplicate the residue peak pattern is recommended by the AOAC for quantitation of PCB residues in certain foods (Horwitz, 1975). This is a currently accepted and practical way to quantitate PCB residues. Perchlorination of PCBs to decachlorobiphenyl has been suggested as a means to improve precision and consistency in PCB determination (Berg et al., 1971; Armour, 1973). This approach will not improve the accuracy of quantitation, however, because proportions of individual chlorobiphenyls in the PCB residue remain unknown; equal weights of individual chlorobiphenyls of different chlorine content result in different weights of decachlorobiphenyl. Contaminations found in antimony pentachloride, the perchlorination reagent, also detract from this procedure for quantitation purposes (Trotter and Young, 1975). An approach to quantitation which appears to have practical merit as well as offering improved accuracy in quantitation has been advanced by Webb and McCall (1973). The PCB residue is quantitated peak by peak in comparison to reference Aroclors which have been characterized as to the number of chlorines and the fraction of total Aroclor weight represented by each electron capture peak after separation on a widely used (methyl silicone liquid phase) gas chromatographic column. The availability of carefully characterized Aroclors and evaluation in practice are required to fully evaluate the merits of this procedure.



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Precision in the interlaboratory determination of PCBs is slightly less than with the common organochlorine pesticides. In several interlaboratory studies involving biological samples and paperboard containing either added Aroclors or actual residues of PCB the coefficients of variation are about ± 20 percent. The results of studies conducted by the AOAC and the FDA involving samples containing added Aroclors have been mentioned above (Food and Drug Administration, 1971; Burke, 1971; Food and Drug Administration, 1973; Sawyer, 1973); the recoveries of added Aroclors ranged from 74 to 101 percent and coefficients of variation from 9 to 15 percent. The levels of Aroclors added in these studies ranged from about 2 to 8 ppm with the exception of 0.2 ppm Aroclor 1254 added to chicken infant food. With samples containing actual residues, 9 laboratories in the AOAC study reported 9.2 ppm ± 8 percent for a residue in chicken fat and 4.5 ppm ± 20 percent for a residue in Lake Michigan chubs. The residue in the chubs was determined after separation of DDE, DDT, and TDE by column chromatography on silicic acid. In a study by 8 laboratories in cooperation with the International Council for the Exploration of the Sea, PCB residues determined in a fish oil averaged 1.97 ppm ± 47 percent. A much better coefficient of variation, about ± 11 percent, was obtained when the same fish oil was fortified with an additional 10 ppm PCB (International Council for Exploratoin of the Sea, 1974). In an AOAC study of the method for PCB in paperboard 11 laboratories analyzed a paperboard sample manufactured to contain Aroclor 1242 and reported 5.6 ppm ±



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16 percent (Finsterwalder, 1974). The inter- or intra-laboratory precision of the determination of PCB in any sample type is improved by use of the same analytical procedures, especially in quantitation of the residue.

The lower limit of quantitation for PCB residues will vary among laboratories depending upon the objectives of their analyses and upon the particular details of the analytical methods, especially the sensitivity of gas chromatographic detection and the sample size. The quantitation limit achieved will be higher than for organochlorine pesticides because of lower effective detector response to the technical PCB mixtures; 20-30 times more Aroclor 1254 or 30-50 times more Aroclor 1242 than DDE is required for the major peaks to produce the same peak height as DDE in electron capture gas chromatography (Food and Drug Administration, 1968-1975). Other factors which may restrict the attainment of low limits of quantitation and affect analytical reliability are interfering residues in the sample and laboratory contamination. Fish from some locations contain residues of organochlorine pesticides and possibly other contaminants at levels that would cause increasing difficulty as the PCB residue decreases below about one ppm. Contamination of laboratory equipment, reagents, and samples with PCB from containers, previous samples, control runs, and unknown sources, must be carefully guarded against in analyses where a low level of quantitation is necessary, for example with human blood or certain



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environmental samples (Trotter, 1975; Jensen et al., 1972; Giam and Wong, 1972). Perchlorination of the PCB residue to decachlorobiphenyl offers the possibility of about 25 fold increase in sensitivity in the gas chromatographic determination (Armour, 1973). However, contaminants present in antimony pentachloride severely restrict or prohibit application of this procedure to determination of low levels of PCBs (Trottar and Young, 1975). Analytical methods used by the FDA for PCB residues also recover chlorinated napthalenes (Armour and Burke, 1971). If present alone, chlorinated naphthalenes would be recognized by the analyst. However, their presence in admixture with PCBs would probably go unrecognized unless amounts were greater than PCBs or the residue was examined by mass spectrometry. Chlorinated naphthalenes are oxidized by procedures used to differentiate DDE in the presence of PCBs and could be removed from interfering in the PCB determination (Holmes and Walden, 1972).

From examination of procedures used to analyze for chlorinated dibenzofurans in commercial PCB formulations and from analytical studies with chlorinated dibenzodioxins it can be inferred that some chlorinated dibenzofurans may be recovered along with PCBs through analytical methods using Florisil and column chromatography (Vos, et al., 1970; Roach and Pomerantz, 1974; Bowes, et al., 1975; Porter and Burke, 1971). The chlorinated dibenzofurans have gas chromatographic properties similar to the PCBs (Bowes, 1975).



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It is very doubtful, however, that a relatively small amount of chlorinated dibenzofuran in the presence of PCBs would be recognized in the usual analysis for PCBs.

Little study has been made of residue analytical methodology for chlorobiphenylols (hydroxylated chlorinated biphenyls). Procedures used for their determination differ substantially from those used for PCBs (Bache and Lisk, 1973; Zitko et al., 1974). It is not likely that chlorobiphenylols would be recovered through column chromatographic and separation procedures used for PCBs.

In the discussion of PCB residues in foods given in the report of the Human Exposure Group a large share of the information comes from the FDA's surveillance programs. The analytical methodology used in the analyses is described in the FDA Pesticide Analytical Manual, Vol. I, and in Official Methods of Analysis of the AOAC. Quantitation of PCB residues is by gas chromatography with electron capture or halogen specific electrochemical detectors. The total response (area or peak height) for the PCB residue is compared to the total response for the Aroclor reference (or mixture of Aroclors) having the most similar gas chromatographic pattern. Aroclor reference materials are each from a single master lot. Results are reported as ppm of the Aroclor(s) used for the quantitation reference. With certain exceptions the limit of quantitation for PCBs (based on the electron capture detector response to Aroclor 1254) is



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about 0.2 ppm for individual foods and about 0.05 ppm for Total Diet composites. Fish from certain fresh water locations are the food most frequently and consistently found to contain PCB residues. The residues in fish are most often similar in gas chromatographic peak pattern to Aroclor 1254 but usually with higher concentrations of the late eluting (higher chlorination) chlorobiphenyl components.

E. Chlorinated Dibenzofurans

(See Appendix for structural information and definitions)

In considering the potential hazard to humans of the commercial chemical mixtures of chlorinated biphenyls called PCBs, one must consider which if any identified trace contaminant in these complex mixtures might contribute significantly to the overall hazard potential of the mixture. Although there is evidence for the presence of chlorinated napthalenes and possibly chlorineted terphenyls in PCBs, the chlorinated dibenzofuran (Cl-DBF) contaminants are regarded as a greater potential danger for several reasons. In using a chick bioassay to monitor fractionation of commercial PCB mixtrues, it was found (Vos et al., 1970) that the fraction most toxic to chicks, and far more toxic than the other fractions, contained Cl-DBFs and chlorinated naphthalenes. What limited toxicological information there is available on these halogenated naphthalenes, terphenyls and dibenzofurans suggests that only the Cl-DBFs may be more toxic than chlorinated biphenyls by orders of magnitude (Bauer et al., 1961; Moore et al., 1976).



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The extremely high toxicity of some of the chlorinated-dibenzo-p-dioxins (NIEHS Conference 1973; Schwetz, et al., 1973) a class of organic chemicals very similar in structure to the Cl-DBFs, also suggests the latter are likely to be highly toxic. Knowledge of the chemistry and toxicity of various chlorinated dibenzo-p-dioxin structures and the relationship between toxicity and structure, although recently developed, is more complete than for Cl-DBFs. Therefore, findings in the chlorodioxin field provide indicators to assess the difficulties likely to be encountered in the study of Cl-DBFs.

Chlorinated dibenzofuran contaminants have been reported and confirmed in PCB mixtures manufactured in Germany and France (Vos et al., 1970; Bowes, et al., 1975a) in Japan (Roach and Pomerantz, 1974a; Nagayama, et al.[1] 1975) and in the United States (Aroclors ) (Roach and Pomerantz, 1974b; Bowes, et al., 1975a). An additional claim that a cl-DBF contaminated an Aroclor was not sufficiently supported by the evidence presented (Curley et al., 1975). In the PCBs of U.S. manufacture, the range of Cl-DBFs found was from dichloro through hexachloro (see references cited above) and sufficient evidence was obtained to show that 2,3.7,8-tetrachloro and 2,3,4,7,8 -pentachlorodibenzofuran are present in Aroclors (Bowes et al., 1975b). That the Cl-DBFs reported were actual contaminants and not artifacts resulting from the experimental procedures used, was considered in two instances (Roach and Pomerantz, 1974b; Nagayama et al., 1975).



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Proper quantitation of the individual Cl-DBF contaminants is very difficult. It requires full structural identification of the Cl-DBF contaminant, availability of the contaminant as a highly purified reference standard for quantitation purposes and evaluation of the procedure used to concentrate and separate the Cl-DBFs, (e.g. from chlorobiphenyls) to determine the capability of the procedure to recover the specific Cl-DBFs in the PCB mixture. Reported attempts to quantitate the Cl-DBFs in PCB suggest contamination levels in the low parts per million range for total chlorinated dibenzofurans (Bowes et al., 1975a; Nagayama et al., 1975). Many of the reported values probably can be taken as minimal in the absence of recovery data.

Although the source of the Cl-DBF contaminants in commercial PCB mixtures has not been determined, several possibilities may be considered. The simplest explanation would be the likely presence of dibenzofuran (parent compound) in the technical grade biphenyl subjected to the chlorination process. From a consideration of the procedures used in the commercial synthesis of PCBs (Hubbard, 1964) various chlorinated biphenyls, if substituted in the ortho and ortho-prime positions with hydroxy groups and/or chlorine atoms, might ring close to form the furanoid ring by dehydration or dehydrochlorination.

It is important to recall that Cl-DBFs have been reported as contaminants also in other chemicals of commercial importance. Samples of pentachlorophenol (Schwetz et al., 1974) as well as



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lower chlorinated phenols (Firestone et al., 1972) and hexachlorobenzene, (Villanueva et al., 1974) have been found to contain a range of Cl-DBFs. The ppm level of Cl-DBFs in certain pentachlorophenols examined by Schwetz and co-workers (1974) was considerably higher than the highest level thus far reported in commercial PCB mixtures; the chlorine level in the reported dibenzofuran contaminants of pentachlorophenol ranged from hexa to octa.

It is now known that "PCB residues" obtained from environmental samples usually do not simply represent an easily identified original commercial PCB mixture. For example, there is evidence from the study of electron capture gas chromatograms that certain chlorinated biphenyl compounds are preferentially lost or concentrated in passing through environmental media. This has led to recent studies of the toxicity, metabolism and physiological effects of specific, individual chlorinated biphenyl compounds, synthesized by routes intended to yield a single compound of known structure. In addition to permitting the development of structure-activity correlations, these studies presumably were intended to avoid complications in interpreting results from the testing of the commercial chlorinated biphenyl mixtures (Vos and Notenboom-Ram, 1972), now known to contain toxic Cl-DBFs as contaminants. Interpretaion of results from the testing of individual chlorinated biphenyl compounds, however, may not be entirely straight forward. In the Ullmann coupling reaction of



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2,4,5-trichlorobenzene. It has been reported that the highly toxic 2,3,7,8 tetrachlorodibenzofuran is formed in 3% yield in addition to the expected 2,2'4,4',5,5'-hexachlorobiphenyl product (Moron, et al., 1973). This hexachlorobiphenyl is a significant constituent of PCB mixtures (Simons and Elti, 1971; Tas and Kleipool, 1972; Jensen and Sundström, 1974) and has been singled out for biological studies by several workers (Vos and Notenboom-Ram, 1972; Johnstone et al., 1974; Hansell and Ecobichon, 1974). Careful purification and, if necessary, chemical analysis of symmetrical chlorobiphenyls prepared by the Ullmann coupling reaction (for dibenzofuran content) is suggested as a prerequisite to biological testing of the biphenyl.

There are two other aspects of the chemical relationship between chlorinated biphenyls and Cl-DBFs that need to be mentioned. Both deal with possible environmental alteration. Evidence has been obtained for photochemical conversion of certain chlorinated biphenyls to Cl-DBFs in very low yields (Crosby and Moilanen, 1973). Although the scope of this study was limited, formation of Cl-DBF was shown to occur under conditions of sunlight irradiation and in the laboratory using sunlight-simulating conditions. Two different ortho-chlorinated biphenyls (the 2,5-dichloro and 2,2',5,5'-tetrachloro) were claimed to produce approximately 0.2% steady-state yields of a monochloro DBF. The limited reports regarding the photochemical decomposition of Cl-DBFs (Crosby and Moilanen, 1973; Hutzinger et al., 1973) indicate relatively



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rapid destruction of the compounds may take place in the environment. Since reductive dechlorination is a major route of photochemical alteration of Cl-DBFs, the possibility of dechlorination of highly chlorinated DBFs to more toxic DBFs of lower chlorine content must be considered. The extent to which specific chlorinated biphenyls, found as major constituents of commercial PCB mixtures, can be photochemically converted to Cl-DBFs can only be determined by further experimental studies.

Analysis of the Yusho oil (the rice oil contaminated by PCB-containing heat exchange fluid implicated in the Japanese "Yusho" poisoning incident) by Nagayama et al., (1975) for Cl-DBF content, led to a value of 5 ppm Cl-DBFs. This value is about 300 times the Cl-DBF level expected in the Yusho oil if one simply assumes contamination of the oil by Cl-DBFs to be proportional to the Cl-DBF level found in unused Kanechlor 400 (the Japanese PCB mixture claimed to have been used as the heat exchange fluid) and to the level of PCB in the Yusho oil. This led Kuratsune et al., (1975) to suggest that Cl-DBF levels increased in the heat exchange fluid through use. Going a step further, these questions may be raised: Is Cl-DBF concentration increased, generally, in PCB-containing heat exchange fluids through use? Does such increase occur in PCB-containing transformer fluids or electrical capacitors through use?

A critical review is needed of the published data upon which Kuratsune's suggestion is based (Nagayama et al., 1975), particularly



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since the results from a recent analysis of a portion of Yusho oil (Trotter, 1976) raises questions about the concentration of chlorinated biphenyls in the Yusho oil. Further laboratory studies suggested by such a review and analyses of selected, used PCB-containing industrial fluids might help to answer the above questions.

Considering the Cl-DBFs as the toxicologically most significant class of chemical impurities in the PCB mixtures, the questions may be asked: what portion of the observed effects from animal and human exposure to PCB mixtures can be attributed to the Cl-DBFs? It is doubtful this question can be answered in any quantitatively explicit way. There are analytical problems in identifying and measuring accurately the amounts of chlorinated dibenzofurans in various PCB mixtures. Even though only a relatively few of the 135 possible chlorinated dibenzofurans may be identified in PCB mixtures, the toxicity of these compounds can be expected to very with both number and ring position of chlorine atoms in the Cl-DBF molecule. Toxicity data on specific, high purity Cl-DBFs is very limited because the compounds themselves are not readily available. In addition to these, essentially chemical, difficulties, toxicological judgements would be difficult to make. To begin to evaluate the toxicological "burden" to be placed on Cl-DBF contaminants in commercial PCB mixtures, a practical approach would be to use analytical procedures that can recover and quantitate specific Cl-DBF compounds known to be highly toxic. For example, 2,3,7,8-tetrachlorodibenzofuran, already reported by Bowes (1975b) as



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present in PCBs, can serve as a measure of potential hazard until further toxicity data for Cl-DBFs become available.

To date, there have been no published, positive findings of Cl-DBFs in environmental samples or in foods. Analytical procedures to detect Cl-DBFs in such samples should be developed, tested and applied to appropriate samples. If chlorinated dibenzo-p-dioxin residue analytical findings can be taken as an indicator, procedures suitable for parts per trillion detection of Cl-DBFs will be required and quantitation will be difficult.

No clear evidence for the presence of chlorinated dibenzo-p-dioxins in commercial PCB mixtures has been reported. A recent publication suggesting the presence of such contaminants in some PCBs and in a synthesized chlorinated biphenyl (Ax and Hansen, 1975) almost certainly misinterpreted the significance of the experimental finding related to the dioxins.

## III. Chemistry of Brominated Biphenyls

### A. Comparison of Brominated Biphenyls with Chlorinated Biphenyls

Unlike PCBs, the chemistry and stability of PBBs have not been well studied and documented in the literature. It is difficult to assess the stability and the extent of possible chemical conversion of PBBs in the environment. PBBs can be compared chemically to the PCBs. Since both bromine and chlorine are



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halogens, PBB and PCB chemistry should be similar in some respects. Bromine, however, is a better leaving group in chemicel reactions than chlorine. The described laborstory experiments concerning PBB alkaline hydrolysis and photolysis show that bromine is more labile then chlorine under comparable reaction parameters. PBBs may, therefore, be less stable in the environment if PBBs were under the same physical reaction parameters in the environment as PCBs. The physicel state and other physicsl reaction parameters of PBB and PCB residues in the environment can determine reactivity. Liquids and vapors often react more readily than solids. Aroclors are liquids, though some Aroclors are extremely viscous at room temperature. Some PCBs, especially the lower chlorinated components, are volatile. FireMaster BP-6, however, is a solid and has an extremely low vapor pressure. The production, distribution, and usage of PBBs has not been as wide spread as PCBs. PBBs, unlike PCBs, may not be physically located in a position for chemical reaction. FireMaster BP-6 has been extensively used, for example, in thermoplastics, such as typewriter and business machine housings (Kerst, 1974). FireMaster BP-6 has little tendency to migrate from the thermoplastic into which it is incorporated. The major portion of the products into which FireMaster BP-6 is incorporated is assumed to be buried eventually in refuse dumps. PCBs, on the other hand, are often used in transformers and capacitors and can be exposed to high temperetures which can accelerate possible chemical reactions. PCB residues are found in many diverse locations



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throughout most of the world. Many of these locations may offer conducive settings for chemical and metabolic conversions. PCB residues in a body of water may react photolytically, whereas PBB residues buried in a refuse dump would not be in a position to absorb light.

B. Non-metabolic Alteration of Brominated Biphenyls

Oxidation and Hydrolysis

The stability of PBBs to oxidation has not been studied and documented in the literature. FireMaster BP-6 is unstable to alkaline hydrolysis. After refluxing FireMaster BP-6 with 2% KOH in ethanol, GLC chromatograms show erratic degredation of hexabromobiphenyl (DET,DO,DCT,DCH,1976), the major component of FireMaster BP-6. The possible rate of PBB hydrolysis in the environment under milder conditions is not known.

Photochemistry

Recent analysis of FireMaster BP-6 by GCMS using an OV 101 column demonstrated 12 peaks whose mass spectrum corresponds to 2 pentas, 4 hexas, 4 heptas and 2 octas. However, about 50 percent of the material is 1 hexa isomer with the next most abundant component being the hepta isomer at about 20-25 percent. Nuclear magnetic resonance and chemical studies have identified the principal component of Firemaster BP-6 as 2,2'4,4'5,5'-hexa-bromobiphenyl (Anderson et al., 1974; Sundström et al., 1976).



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The photolytic rates of reactivity of FireMaster BP-6 and 2,2'4,4'5,5'-hexachlorobiphenyl in methanol solutions irradiated with 3000 Å light were compared (Ruzo and Zabik, 1975). The rate of reaction of FireMaster BP-6 was determined presumably by the decrease in the GLC peak of hexabromobipenyl. Hexabromobiphenyl was found to be seven times as reactive as the corresponding PCB. Hexabromobiphenyl was found to undergo photolytic reductive debromination in methanol yielding penta- and tetrabromobiphenyls and also > 1% methoxylated product. Dimethoxy tetrabromobiphenyl was identified as a product by mass spectrometry. The highly efficient photoreactivity of PBB may be due to: 1. enhanced intersystem crossing to a triplet state due to vibronic coupling with the brominee, 2. steric interference due to the bromines, and 3. the relatively low carbon-bromine bond energy. The aromatic carbon-bromine bond energy is 71 kcal/mole vs the aromatic carbon-chlorine bond energy of 86 kcal/mole (Kerst, 1974). As with PCBs (Ruzo, et al., 1974), presumably ortho halogens preferentially cleave in PBBs upon photoexcitation.

The possible rates and extent of photolytic reactivity of PBBs in the environment are not known. The photochemistry of PBBs has not been studied in the vapor or solid states. Since FireMaster BP-6 has an extremely low vapor pressure, vapor state photochemistry in the environment would be extremely limited. The extent of thin film PBB photochemistry in the environment may also be limited. PBB contamination and resulting



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photochemistry in river waters, except in the vicinity of PBB production facilities, may be limited. PBBs incorporated into thermoplastics, which are presumably buried eventually in a refuse dump, are not likely to absorb much light for photolytic reaction.

C. Brominated Biphenyls Determination of Residues

Polybrominated biphenyl (PBB) residues are analyzed for in foods by methods quite similar to those used for organochlorine pesticides and polychlorinated biphenyls (PCBs) (Food and Drug Administration, 1968-1975; Fehringer, 1975; Food and Drug Administration, 1976). Recoveries of PBB added to samples in vivo through these procedures is over 80 percent. A method to improve extraction efficiency of PBB residue from dry, high fat animal feeds has been reported (Fehringer, 1975).

The major difference from methods for organochlorine pesticides is in the gas chromatographic determination which is done at higher temperatures or on columns with low liquid phase loads because of the lower volatility of PBBs. Methyl silicones or Silar-10C, the liquid phases most frequently used, separate the technical PBB, FireMaster BP-6, into 1-9 peaks depending on the chromatographic conditions selected. PCBs and organochlorine pesticides which would be recovered through extraction and cleanup procedures together with PBBs have earlier retention times under these conditions and are separated from the



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major PBB constituent, hexabromobiphenyl. The retention times of hexabromobiphenyl and decachlorobiphenyl are not greatly different on methyl silicone liquid phases but are widely separated on Silar-10C. Elution of the Florisil column used for cleanup of extracts with petroleum ether rather than a mixture of petroleum ether and ethyl ether separates PBBs from most organochlorine pesticides prior to gas chromatography and improves cleanup for the PBB determination.

PBB residues are detected and quantitated with the electron capture gas chromatography detector. Quantitation is based on comparison of the size of the hexabromobiphenyl peak in the residue to the size of the hexabromobiphenyl peak in a known weight of the reference material, FireMaster BP-6. FireMaster BP-6 is a technical mixture containing 60-70 percent hexabromobiphenyls (Sundström and Hutzinger, 1976). The accuracy of residue quantitation is affected by any change in the composition of the residue from that of the reference technical PBB mixture. The identity of PBB residues can be confirmed by thin layer chromatography, photochemical alteration, halogen-specific gas chromatographic detection, and by different retention times on Silar-10C and methyl silicon gas chromatographic columns (Food and Drug Administration, 1976; Fehringer, 1974; Fehringer, 1975; Erney, 1975). Mass spectrometry has also been used to confirm the identity of the PBB residue (Ayers, 1975).



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The limit of quantitation for PBB residues, including capability for residue identity confirmation, is about 0.05 ppm in fats and 0.01 ppm in non-fats (Food and Drug Administration, 1976). Formalized interlaboratory studies have not been reported for analytical methods for determination of PBB residues.

D. Possibility of Brominated Dibenzofurans in Commercial PBB Mixtures

There has been no report, so far, of the finding of brominated dibenzofuran (Br-DBF) compounds in commercial PBB mixtures. Examination of FireMaster BP-6 for possible contamination by Br-DBF is in progress in several laboratories.



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METABOLISM AND BIOCHEMICAL TOXICITY OF PCBs and PBBs

I. Effects of PCBs on Biochemical Functions

One of the major biochemical effects of PCBs is the induction of microsomal enzymes in the liver. Risebrough et al., (1968) suggested that PCBs had the capability to induce the activities of microsomal enzymes. Subsequently, Street et al., (1969) demonstrated the induction of liver enzymes in rats by PCB's. Since then, a great number of articles have been published on this subject.

The enzyme systems studied have included mainly hydroxylases, N- and O-demethylases and nitroreductases, and to a lesser extent nonspecific carboxylesterase, bromosulphophthalein-glutathione conjugating enzyme, p-nitrophenol UDP-glucuronyl transferase and EPN-detoxification systems.

The induction of microsomal enzymes by commercial PCBs has been demonstrated perorally in rabbits (Villeneuve et al., (1971), rats (Litterst et al., 1972a) and primates Allen et al., 1974) and via intraperitoneal injection (Bickers et al., 1972) and skin application in rats (Bickers et al., 1975). Values reported for the threshold of enzyme induction by PCB's vary between 0.5-25 ppm. (Litterst et al., 1972b; Villeneuve et al., 1971; Turner et al., 1974).

The time-course of microsomal syzyme induction was studied by Litterst et al., (1974) in rats. They found that significant levels of enzyme induction occurred after 7 days of feeding PCBs



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in the diet. A single oral dose in rats resulted in maximum enzyme activities at 24 hours. Bickets et al., (1974); 1975) have shown that maximum induction occurred within 2-6 days in rats after cutaneous exposure to Aroclor 1254 or microscope immersion oils containing 30-45% chlorine.

In order to study the effect of chlorine content of PCB's on enzyme induction, different commercial PCBs were investigated. Litterst et al., (1972b) studied PCBs with chlorine contents from 42 to 60%. In rats the maximum activity of demethylase was observed with the 54% chlorine PCB's; maximum activity for nitroreductase was obtained with the 60% chlorine PCBs. Chen et al., (1973) reported that 54% chlorinated PCBs gave maximum response of demethylase in rats. Similar results were reported by intraperitoneal injection of various PCBs in rats (Ecobichon et al., 1974). Goldstein et al., (1975) and Iverson et al., (1975) studied the effects of Aroclor 1016 and Aroclor 1242 on enzyme induction in rats. Both PCBs have approximately 42% chlorine, but the content of the penta-, hexa-, and hepta-chlorine isomers of Aroclor 1016 is reduced to 10% of that in Aroclor 1242. Their results showed that Aroclor 1242 was a much more potent inducer than Aroclor 1016 in female rats. In comparing Aroclor 1232, Aroclor 1248 and Aroclor 1260, Schmoldt et al., (1974) found that Aroclor 1260 was the most potent enzyme inducer in rats.

The synthetic and isomerically pure PCBs have been investigated for enzyme induction, Chen et al., 91973b) found that both



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2,3,4,5- and 3,5,3',5'-tetrachlorobiphenyls induced enzyme systems in male rats, but no significant differences were found between the two isomers. Johnstone et al., (1974); Ecobichon et al., (1975). Ecobichon (1975) compared the effects of position of chlorine atoms on the ring. They found that enhanced induction of mono-oxygenases was observed for PCBs having chlorine atoms substituted at the 4- and 4'- positions irrespective of chlorination at other positions. Substitution at the 2-position was next in importance followed by substitution at the 3-position. They concluded that the position of chlorination was as important as the degree of chlorination.

There are two types of enzyme inducers being reported in the literature. One group to which phenobarbital belongs resulted in increased cytochrome P-450 content, as well as increased benzpyrene hydroxylase and ethylmorphine demethylase activities in the liver. The second group includes polycyclic hydrocarbons. This group stimulates the formation of cytochrome P-448 and an increase in hydroxylation but not demethylation. Alvares et al., (1973) reported that rats treated with PCBs produced an increase in cytochrome P-448, hydroxylase, as well as demethylase. Therefore, it appears that PCBs display induction behavior typical of both groups.

Most of the enzyme induction studies were evaluated in vitro. Several workers indeed have observed enzyme induction effects in vivo by demonstrating shortened barbiturate sleeping times (Bickers et al., 1972; Johnstone et al., 1974; Villeneuve et al., 1972):



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Commercial PCBs are known to contain small amounts of dibenzofurans (DBF). DBFs were found to be 170 times as potent as PCBs in inducing enzymes.

Alvales & Kappas (1975) reported the induction of hydroxylase and demethylase in placenta, as well as in the fetus, when pregnant rats were treated with PCBs Vainio (1974) reported that Clophen A50 enhanced hydroxylase 5-fold in the lung and 8-fold in the kidney microsomes, whereas the microsomes from duodenal mucosa exhibited no enhancement. Benzpyrene hydroxylase activity in skin was also increased by PCBs (Bickers et al., 1975).

Vos and Koeman (1970) found that several tissues of PCB-treated chickens were strongly fluorescent under ultra violet light. They suggest that PCBs could induce chemical porphyria in chickens. Later, they demonstrated that mitochondrial ALA Synthetase activity increased 20-fold and fecal porphyrin levels were significantly increased in Japanese quail treated with PCBs (Vos, 1971; Vos et al., (1972). They suggested that the porphyria caused by PCBs is due to the increase of ALA synthetase, followed by overproduction of porphyrins. Similar results in rats were obtained by Goldstein et al., (1974), and Bruckner et al., (1974). Goldstein et al., (1974) suggested that the induction of ALA synthetase is probably not the primary cause of PCB-induced porphyria because of the delayed onset of porphyria as compared with ALA synthetase induction. They suggest that the ALA synthetase in the liver may be related to the increase of cytochrome P-450 content. This



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idea was supported by Grote et al., (1975). Aroclor 1242 was found to be a more potent inducer of liver porphyria than Aroclor 1016 (Goldstain et al., 1975). Both Aroclors have similar chlorine content, but Aroclor 1242 contains 9% homologs with five or more chlorines while Aroclor 1016 containe only 1% of such homologs. Sinclair and Granick (1974) showed that cultured liver cells respond rapidly to PCBs in accumulation of uroporphyrin. Simon et al., (1974) suggested that porphyria may be due to the destruction of the phospholipid structure of cell membrane, resulting in increased permeability of the substrate. Hirayama et al., (1974) observed hypo-bilirubinaemia in Yusho patients and suggested that the induction of bilirubin UDP-glucuronyltransferase may be responsible for the low plasma bilirubin content. However, recently Bastomsky et al., (1975) showed that the lowering of serum bilirubin concentration by PCBs is due to the reduced binding of bilirubin to plasma protein rther than to the enzyme induction. Wit (1972) has shown that PBBs which are similar to PCBs are potent hepto-porphy-rinogenic chemicals.

The chemical and structural similarity of DDT to PCBs led Jefferies et al., (1972) to study the effects of PCBs on thyroid functions. They found an increase in thyroid weight in PCB-treated black-backed gulls as compared to controls. Hurst et al., (1974) found no clear-cut effect of PCBs on thyroid gland size in quails. Byrne et al., (1975) reported that PCBs increased



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thyroxine levels in the blood and also promoted peripheral degradation of thyroxine in female minks. Bastomsky (1974) found a 4 to 5-fold increase of biliary excretion of thyroxine in rats treated with PCBs. PCBs also elevated iodine uptake by the thyroid and reduced serum PBI concentration. Recently Bastomsky and Wise, (1975) showed similar results in rats due to either intraperitoneal injection or skin application of microscope immersion oil. Walker (1975) found that inclusion of Aroclor 1254 in fish food resulted in an increase in fish thyroid activity.

The effects of PCBs on ATPase and oxidative phosphorylation have been examined by a number of investigators. $Na^{+}K^{+}$-ATPase and $Mg^{++}$ ATPase from several fish tissues were found to be inhibited by PCBs. (Yap et al., 1971; Cutkomp et al., 1972; Desaiah et al., 1972; Koch et al., (1972). Further study showed that in vitro data were not in agreement with the in vivo data (Yap et al., (197à). The inhibition of ATPase in fishes by PCBs was also reported by Kinter et al., (1972) and by Davis et al., (1972).

Sivalingan et al., (1973) investigated the effects of PCB on the oxidative phosphorylation of rat liver mitochondria. They showed that the mode of inhibition appears to be different with PCBs of different chlorine content. PCBs with low chlorine content inhibited energy and electron transfer but PCBs with high chlorine content not only inhibited energy and electron transfer but also had an uncoupling effect. LaRocca et al.,



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(1975) showed that the administration of commercial PCBs, as well as purified isomers to rats inhibited the total ATPase activity in liver, kidney and brain tissues.

Pardini (1971) has demonstrated that a marked inhibition of respiratory enzyme systems occurred when heavy beef heart mitochondria were exposed in vitro to numerous PCBs. Chesney and Allen (1974) showed that the addition of PCBs to rat liver mitochondria in vitro caused an inhibition of oxidative phosphorylation and respiration. Feeding rats with PCBs at the 1000 ppm level increased the oxidative phosphorylation in liver mitochondria, however no effect was seen at the 100 ppm level.

Sharp et al., (1974) reported that the inhibition of beef brain and rabbit kidney $Na^+K^+$-ATPases can be reversed or prevented by phosphatidylserine or phosphatidylinositol, but not by phosphatidychloline and phosphatidylethanolamine. They suggested that acidic phospholipids are required to stabilize the enzyme and thereby overcome the effect of PCBs.

Risebrough et al., (1968) suggested that PCBs had the capacity to enhance steroid-hydroxylating enzyme activity, thus affecting estradiol metabolism. In addition, certain low chlorine containing PCBs have estrogenic activity which is reflected by an increase in glycogen content of the uterus of immature rats (Bitman and Cecil, 1970). Similar results were obtained by Fumiko (1972) in rats (Bitman and Cecil, 1970). Similar results were obtained by



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Fumiko (1972) in rats and by Orberg, and Kihlstrom (197 ) in mice. Orberg et al., (1972) found that the estrous cycle of the mouse increased in length after a single injection of PCBs. Linder and Peakall, (1973) reported an increased rate of metabolism of estradiol in vitro in the liver of American kestrels after the feeding of PCBs. Similar results were also reported in white leghorn cockerels and pullets (Nowicki and Norman 1972). Platonow and Funnell (1971) demonstrated that feeding PCBs to cockerels resulted in a decrease in testicular and comb growth. Decreased urinary excretions of estrogen and dehydroepiandrosterone were also observed in cockerels when PCBs were administered in high doses (Platonow et al., 1972). Ecobichon and Mackenzie, (1974) studied the uterotropic activity of commercial and isomerically pure PCBs in rats. They found significant changes with a number of PCB mixtures but experiments with the isomerically pure PCBs were inconclusive. Changes were observed with 2-chloro and 2,2-chlorobiphenyls. Nelson (1974) reported that PCBs are effective inhibitors of the binding of $^{3}H$-estradiol to the rat uterus cytosol fraction in vitro. Recent date showed that PCB feeding caused neither significant chromosomal damage nor arrest in the rate of spermatogenesis in male rats (Dikshith et al., (1975).

Flick et al., (1965) noted enlarged adrenals and small spleens in PCB-treated white leghorn cockerels. Wassermann and Wassermann (1972) and Wessermann et al., (1973) reported that rats receiving 200 ppm PCBs showed an increase in plasma corticosterone levels.



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These changes were concimitant with morphological changes in the zona fasciculata of the adrenal gland which indicated increased activity. Sanders, et al., (1974) have shown similar increases in serum corticosterone in mice fed 60 to 1000 ppm PCBs.

It is known that DDT decreases the utilization of dietary carotene and liver storage of vitamin A in rats and cattle (Phillips, 1963; Phillips, and Hidiroglou, 1965). Villeneuve et al., 1971) reported that Aroclor 1254 but not Aroclor 1221 reduced the liver concentration of vitamin A in pregnant rabbits. Decreased liver vitamin A concentrations were found in male and female rats and also Japanese quail after PCB feeding (Cecil et al., 1973). However, the laying female quail did not appear to be affected.

The effects of PCBs on vitamin E have been studied by Combs et al., (1975). Their results indicated that dietary PCBs increased the incidence of exudative diathesis in chicks and that this increase can be overcome by increasing dietary vitamin E or selenium. The vitamin D mediated calcium metabolism was found to be altered by the oral administration of PCBs to chickens (Wong et al., 1974).

## II. The Comparative Biochemical Toxicity of Aroclor 1254 and Firemaster BP-6

PCB's have been reported to affect a large number of biochemical systems. To the extent that it has been reported, PBBs



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affect most of these same systems in a similar fashion. However, quantitative and possibly qualitative differences do not appear to exist in the response of some biological parameters. These include thyroid hyperplasia (Norris et al., 1973) and microsomal enzyme induction (Farber and Baker, 1970).

A preliminary study was conducted to compare the relative effect of Aroclor 1254 (Aro) with Firemaster BP-6 (FM) (Garthoff et al., 1975). This was set up as a broad survey to give an estimate of the relative biological activity of these chemicals in may different systems and pin point areas where further more definitive work should be done.

Table 1 summarizes the time sequence of the various parameters studied. Adult male rats were fed either 0, 5, 50, or 500 ppm Aroclor 1254 or Firemaster BP-6, mixed in the diet, for 2, 3, or 5 weeks.

Due to the large number of parameters examined and the time restraints placed on the study, the experimental design did not permit a totally unambiguous comparison between all of the effects caused by Aro and FM. Some apparent differences in biochemical responses determined at sacrifice may have been partly due to the fact that, for the three-week study, rats were exposed to FM for 48 hours longer than rats were exposed to Aro. Although it is felt that under the experimental conditions, this effect should be small, critical comparison of Aro and FM toxicity should be



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considered tentative until smaller experiments are conducted which can eliminate this difference in exposure time. The following evaluations were made based on the assumption that this two day difference had no effect.

Some parameters showed no consistent or dose related change with either chemical at any level. These included relative kidney weight and testes weight, hematology, SGPT, SGOT, plasma BUN, plasma corticosterone, the rate of liver protein synthesis per g tissue, liver dry weight, lyophilizable kidney, liver, or testes, the incidence of chromosome abnormalities and the number of cells in mitosis from bone marrow of spermatogenial cells.

Growth and adipose tissue weight were equally depressed by Aro and FM when fed over a three week period at 500 ppm. The growth effect appeared to be due primarily to decreased food efficiency.

Three parameters were more sensitive to Aro than to Fm. These included the decreased liver protein and liver RNA per g tissue and the decreased plasma glucose.

On the other hand, six parameters were more sensitive to FM. These included liver growth, increased liver lipid (total lipid, cholesterol, phospholipid, and neutral lipid), elevated plasma cholesterol, increased microsomal enzyme activities, impaired mitochondrial function, and decreased liver RNA synthesis.



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Disruptions of the redox and energy states of the cell were produced by dietary exposure to Aro or FM. These effects were indicated by studies wit isolated mitochondria and frozen clamped liver. In the latter case the concentrations of adenine nucleotides, pyridine nucleotides, and glycolytic intermediates were determined. On a molar basis hexabromobiphenyl was reported to be about five times more potent than Arochlor 1254 in causing some increased microsomal enzyme activities (Farber and Baker, 1974). A three-fold difference was observed between Firemaster BP-6 and Arochlor 1254 (Table 3).

A pathology team autopsied rats after both three and five weeks exposure to Aro and FM (Kasza et al., 1975). Pathological analysis of the liver of rats fed Aro or FM for three weeks showed no abnormalities on gross observation. No histopathological lesions were seen in controls, Aro groups, or the 5 and 50 ppm FM groups, after three weeks. In the 500 ppm FM group, minimal vacuolation and focal hepatitis were each seen in one rat. After five weeks exposure, gross observation of the liver indicated an increasing incidence of enlargement, friability, and prominent lobular pattern with increasing dose of both chemicals. Microscopically the 5 ppb Aro group appeared normal whi'e 2/7 rats fed 5 ppm FM had minimal cytoplasmic degeneration. All rats had this lesion at 50 and 500 ppm Aro or FM, with increasing cellular lipid. This lesion was recent, centrilobular and coalescing with no change in fibrous tissue or glycogen. The histopathology was qualitatively similar but slightly more severe in rats fed FM



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than in those fed Aro and is consistent with biochemical findings. The biochemical findings were all made after three weeks exposure at which time histopathological changes were not evident. Even at five weeks the histopathology was not severe enough to alter most of the clinical chemistry in the rat.

In most parameters evaluated, FM either caused an effect at a lower dose level or caused an effect in a shorter time or caused a greater effect at the same dose level, expressed as ppm in the diet, as Aro. When expressed on a molar basis FM becomes even more effective than PCB. Exceptions to this observation were noted at the 500 ppm dose level where differential effects of Aro and Fm were often eliminated.

A summary of the major effects seen are found in Tables 2, 3, and 4. In all but one case the maximum response of a parameter occurred at the highest dose level. Microsomal enzyme activities were, in general, greater at the 50 ppm level and the decline at 500 ppm relative to the 50 ppm level was more marked with FM.

III. Metabolism of PCB and PBB Mixture

Generally, the tissues from animals and man containing PCB from environmental exposure have GLC patterns resembling those of PCB mixtures with more than 50% chlorination. This is in marked contrast to the major manufactured products that generally contain 42% or less chlorine. While one cannot rule out the possiblity that differential uses favored introduction of more highly



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TABLE 1

PARAMETERS EVALUATED IN STUDY

EXPOSURE TIME

(WEEKS)

| PARAMETERS | TWO | THREE | FIVE |
|---|---|---|---|
| Body weight | x | x | x |
| Liver weight | x | x | x |
| Other tissue weights | | x | |
| Mitochondrial respiration | x | | |
| Liver composition | | x | |
| Liver protein synthesis | | x | |
| Liver RNA synthesis | | x | |
| Liver Microsomal enzyme activities | | x | |
| Liver intermediary metabolites | | x | |
| Serum cholesterol | | x | |
| Serum enzymes | | x | x |
| Plasma corticosterone | | x | |
| Plasma protein | | x | |
| BUN | | x | x |
| Gross pathology | | x | x |
| Histopathology | | x | x |
| Cytogenetic analysis | | | x |



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TABLE 2

RELATIVE BIOCHEMICAL RESPONSE TO Aro AND FM AFTER TWO WEEKS DIETARY EXPOSURE

| | LEL[1] ppm | | Maximum Effect % Control | |
|---|---|---|---|---|
| | Aro | FM | Aro | FM |
| **PARAMETERS** | | | | |
| Liver weight / Body weight | 500 | 50 | 170 | 183 |
| Mitochondrial respiration | 500 | 500 | 152 | 192 |

[1]LEL = lowest effective level that showed a dose-response relationship.

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TABLE 3

RELATIVE BIOCHEMICAL RESPONSE TO Aro AND FM
AFTER THREE WEEKS DIETARY EXPOSURE

| | LEL[1] ppm | | Maximum Effect % Control | |
|---|---|---|---|---|
| | Aro | FM | Aro | FM |
| **PARAMETERS** | | | | |
| Body weight gain | 50 | 500 | 52 | 52 |
| Liver weight / Body weight | 50 | 50 | 203 | 214 |
| Liver dry wt. | NE[3] | 50 | NE[3] | 107 |
| Liver lipid | 50 | 5 | 138 | 156 |
| Liver cholesterol | 500 | 50 | 226 | 232 |
| Liver protein | 500 | NE | 84 | NE |
| Liver RNA | 5 | 50 | 79 | 84 |
| Liver DNA | 5 | 5 | 58 | 60 |
| Liver RNA anythesia | 500 | 5 | 78 | 86 |



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TABLE 3 (Cont'd)

| PARAMETERS | LEL[1] ppm Aro | LEL[1] ppm FM | Maximum Effect % Control Aro | Maximum Effect % Control FM |
|---|---|---|---|---|
| para-Nitrobenzoate reductase activity | 50 | 5 | 288[2] | 520[2] |
| Plasma glucose | 50 | NE[3] | 79 | NE[3] |
| Plasma cholesterol | 500 | 5 | 178 | 256 |

[1]Lowest effective level that shows a dose response relationship.

[2]Maximum effect occurred at 50 ppm dose.

[3]NE = No effect

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TABLE 4

RESPONSE OF RELATIVE LIVER WEIGHT TO Aro AND FM AFTER FIVE WEEKS DIETARY EXPOSURE*

| PARAMETER | LEL ppm | | Maximum Effect % Control | |
|---|---|---|---|---|
| | Aro | FM | Aro | FM |
| $\frac{\text{Liver weight}}{\text{body weight}}$ | 500 | 50 | 171 | 216 |

* Legend - see Table 3.



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chlorinated materials in the environment, the observation has led to the general belief that the less chlorinated components are more readily metabolized.

The environmental observations have been confirmed in a large number of experimental feeding studies in a number of mammalian and avian species. The absence or diminished concentration of the early eluting peaks have been raported in rats (Grant et al., 1971a, Curley at al., 1971), rabbit (Grant et al., 1971b), cow (Friee at al., 1973a), quail (Bailey and Bunyan, 1972), and hens (Fries at al., 1973b).

The peaks not present in tissues and products generally are the early eluting peaks that correspond to the PCBs with lower degree of chlorination. The observations are consistent with the belief that the rate of metabolic attack on PCBs decreases with increasing chlorination. Studies of the single PCB homologs of various degrees of chlorination have shown that those with five or less chlorine atoms are more radily metabolized and excretad than the PCBs with higher chlorination (Berlin et al., 1975; Hutzinger et al., 1972; Melves and Brandt, 1973). The position of chlorine substitution also affecte t..e retention and elimination of single homologs (de Freites and Norstrom, 1974). Metabolism of individual PCBs will be discussed more thoroughly in another section.



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Among the various food species there appears to be one exception to the general rule that the less chlorinated PCBs are hydroxylated. The PCB residues in fish closely resemble the PCB to which the fish was exposed. There was little change in the relative concentrations of the various homologs (Stalling and Mayer, 1972). Consistant with this observation it has been found that mono-, di-, and tetrachlorobiphenyls are not metabolized by trout (Hutzinger et al, 1972).

Among the food producing animals (cattle, chickens) concentrations of PCB residues in milk and eggs reach a "steady state" after approximately 8 to 12 weeks of continuous intake (Fries et al., 1973a; Fries et al., 1973b). The ratio of residue levels in food products to levels in the animal's diet are greatest with PCBs of 54% chlorination. The residue levels in milk are approximately 4 times the residue level in the cow's diet. In hens, residue levels in eggs are approximately equal to the diet while residue level in the body tissue (fat basis) is approximately 6 times the dietary level. The ratio of residue level in product and tissues to intake is lower with PCBs of greater than 54% chlorination. Since it is unlikely that there is greater metabolism of the most highly chlorinated PCBs, the results suggest that the absorption for the GI tract is lower for the more highly chlorinated PCBs (Smith et al., 1976). PCBs of four or fewer chlorines rarely occur in animal products when the animals are fed environmentally realistic levels.



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Much less is known about the metabolism of PBB than PCBs. However, the observations that have been reported tend to be consistent with the reports on PCB metabolism.

While the PBB of most immediate concern for human health (BP-6, Michigan Chemical Co.) is a multicomponent mixture like the PCBs, it is much simpler than the ordinary commercial PCB mixtures. The major component is a single hexabromobiphenyl accounting for over 60% of the total material. The only other component occurring in significant amounts is a single heptabromobiphenyl.

In general, the measures of retention in edible tissues and excretion in edible products are similar for the hexabromobiphenyl and the hexachlorobiphenyls (Fries et al., 1976; Fries and Marrow, 1975). Heptabromobiphenyl retention and excretion is only 1/10 of that of the hexabromobiphenyl. In cattle that have been heavily contaminated with BP-6, the heptabromobiphenyl was barely detectable (milk or tissues) 6 to 8 months after the time of ingestion even though the hexabromobiphenyl level was over 1000 ppm (Fries et al., 1975). Since the heptabromobiphenyl is not more likely to be metabolized by the animal than the hexabromobiphenyl, it is reasonable to assume that its absorption is lower or its elimination through bile is greater than the hexabromobiphenyl component. Over 60% of octabromobiphenyl is excreted in the feces as the parent compound when it is fed to rats (Norris et al., 1974). This observation supports the conclusion that there is less absorption with increasing halogenation.



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## IV. The Metabolism of Individual PCBs

Quantitative studies of the distribution of PCBs indicate that most, if not all, PCBs which contain six or fewer chlorine atoms are efficiently absorbed from the gut of higher animals (Albro and Fishbein, 1972; Matthews and Anderson, 1975a; Van Miller et al., 1975). Furthermore, only about one-tenth of the absorbed dose is excreted prior to metabolism to more polar compounds (Matthews and Anderson, 1975a; Van Miller et al., 1975; Balin et al., 1975). A number of the less chlorinated PCBs (i.e., mono-, di-, tri-, and tetrachlorobiphenyls) are readily metabolized and excreted by birds and mammals. There is little evidence that fish can metabolize any PCB.

It has been adequately demonstrated that a number of mammalian species and the pigeon can readily degrade and excrete monochlorobiphenyls, primarily 4-chlorobiphenyl (Matthews and Anderson, 1975a; Block and Cornish, 1959; Hutzinger et al., 1972; Safe et al., 1975a,b,c). However, the rate of metabolism drops sharply as the number of chlorines per molecule increases. For example, the rate of metabolism and excretion of several PCB isomers has been shown to decrease as the number of chlorines increases in the following order: 4-chlorobiphenyl > 4,4'-dichlorobiphenyl > 4,4'-dichlorobiphenyl > 2,5,2',5'-tetrachlorobiphenyl > 2,4,5,2',5'-pentachlorobiphenyl (Matthews and Anderson, 1975a; Van Miller et al., 1975; Matthews and Anderson, 1975b). The rate



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of metabolism and excretion of 4,4'-dichlorobiphenyl is approximately one-half that of 4-chlorobiphenyl. In each case the PCBs are metabolized primarily by hydroxylation and conjugation with glucluronic acid (Matthews and Anderson, 1975a; Balin et al., 1975). The importance of the route of excretion, i.e., urine or feces, is determined primarily by the degree of chlorination. Approximately 60, 30, and 7 percent of a single dose of a mono-, a di-, and a pentachlorobiphenyl, respectively, were excreted in the urine of rats. The remainder of the dose was excreted in the feces (Matthews and Anderson, 1975a).

Once the number of chlorine atoms on the biphenyl molecule reaches four, the position of the chlorine atoms becomes a more important factor in determining the rate of metabolism and excretion (Matthews and Anderson, 1975a; De Freitas and Noratrom, 1974). As was observed with the chlorinated benzenes (Jondorf et al., 1955; Jondorf et al., 1958) those PCBs which have two adjacent unsubatituted carbon atoms are metabolized and excreted more rapidly than PCBs having the same degree of chlorination, but which do not have two adjacent unsubstituted carbon atoms (Matthews and Anderson, 1975a). It is also likely that, as with the chlorinated benzenes, the absence of two adjacent unsubstituted carbon atoms has a greater effect as the degree of chlorination increases. For example, the initial half-life of 3,5,5',5'-tetrachlorobiphenyl is only about two-fold greater



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in the rat (Matthews and Tuey, 1976) than that of 2,5,2',5'-tetrachlorobiphenyl in the same species (Van Miller et al., 1975). On the other hand, the initial half-life of 2,4,5,2',4',5'-hexachlorobiphenyl appears to be infinite, whereas the initial half-life of 2,4,5,2',5'-pentachlorobiphenyl is only about two days (Matthews and Anderson, 1975a). This particular hexachlorobiphenyl has been shown to be in the highest concentration of any PCB found in the adipose tissue of the Swedish population (Jensen et al., 1974).

The importance of two adjacent unsubstituted carbon atoms to the metabolism of PCBs may be derived from their effect on the formation of arene oxide intermediates during the metabolism of these compounds by the hepatic mixed-function oxidases. Evidence for the formation of arene oxide intermediates in PCB metabolism was first provided by Gardner et al., (Gardner et al., 1973) in their study of the metabolism of 2,5,2',5'-tetrachlorobiphenyl and in a later study of 4-chlorobiphenyl metabolism by Safe et al., (1975c) which was specifically designed to provide evidence of an arene oxide intermediate. Metabolism of PCBs which do not have two adjacent unsubstituted carbon atoms by hepatic mixed-function oxidases, via direct hydroxylation of the biphenyl molecule and/or the formation of an arene oxide between a chlorinated and an unsubstituted carbon atom could occur, but each of these mechanisms would be expected to be slower than arene oxide formation between two unsubstituted carbon atoms (Daly et al., 1972; Jerina and Daly, 1974). The latter mechanism would be expected



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to result in dechlorination and/or shifting of a chlorine atom (Daly et al., 1972). The metabolism of selected hexachlorobiphenyls has been reported (Jensen and Sundstrom, 1974; Hutzinger et al., 1974; Hass, unpublished), and in two incidents (Hutzinger et al., 1974; McKinney et al., unpublished) dechlorination and/or chlorine shifting was observed, but in all cases the rate of metabolism was quite slow.

Thus, those PCBs which are more readily metabolized and excreted may also more readily form arene oxides. Certain arene oxides have been implicated as potential carcinogens (Daly et al., 1972; Jerine and Daly, 1974) as have certain PCB formulations (Ito et al., 1973; Kimbrough and Linder, 1973; Kimbrough et al., 1975), but there is no published evidence which indicates that the less chlorinated PCB formulations have been so rigorously tested.



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## Introduction

A number of reviews on different aspects of the toxicology of the PCBs have been written recently and no effort has been made in this report to give a complete outline of the animal toxicology (Kimbrough, 1974, Peakall and Risebrough, 1975; Nelson et al., 1972; Fishbein, 1974).

The material summarized in this report consists predominantly of more recently published information on toxic effects of PCBs in mammals and experimental studies in birds. Since Nelson et al. (1972) discussed the toxic effects produced in fish by PCBs extensively, the reader is referred to this and the other reviews cited above for information on toxicity of PCBs in this species.

The emphasis of the present review was placed on chronic toxic effects wherever possible. Unfortunately, very little information was available for halogenated dibenzofurans and the brominated biphenyl mixtures.

## Acute Toxicity

### Polychlorinated Biphenyls (PCB)

The acute oral $LD_{50}$ of the PCBs in rats, rabbits and mice ranges from 1 to 10 g/kg body weight (Tables 5,6). There is some indication that young animals may be more sensitive than adults,



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and females more sensitive than males (Table 5). According to the classification used by the American Industrial Hygiene Association (Hodge & Sterner, 1949), the acute toxicity of the PCBs can be classed as slightly toxic (0.5 to 5 g/kg body weight) to practically non-toxic (5-15 g/kg body weight).

The symptoms associated with administration of a toxic dose of PCB (Aroclor 1242) in rats consist of diarrhea, diminished exploratory behavior, decreased response to pain stimuli, chromodacryarrhea, adipsia, oliguria, anorexia, erythema of limbs, followed by ataxia, coma and death. Death occurred up to 14 days following administration of the toxic dose. 24 hours after administration of a single toxic dose of Aroclor 1242, gross pathology indicated all organs appeared normal except the liver and kidneys. No inflammation was observed in the abdominal and intestinal mucosa. Histopathological findings showed large discrete sudanophilic vacuoles in the hepatocytes. Widely scattered foci of tubular epithelial cells were present in the kidneys (Bruckner, et al., 1973). However, Kimbrough (1974) reported ulceration of the gastric and duodenal mucosa in rats after single oral doses of 3,000 mg/kg Aroclor 1254 or 1260.

Only one metabolite of PCB is known to have been tested for the acute $LD_{50}$. The 5-OH metabolite of 2,4,3'4'-tetrachlorobiphenyl was known to be more toxic than the parent compound (0.43 g/kgBW/2.15 g/kgBW (Table 5). Dermal toxicity of PCB is summarized in Table 7.



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Polybrominated Biphenyls (PBBs)

Polybrominated biphenyls show a low order of acute toxicity to the rat (Table 8). In the case of chicks, a 37% mortality was reported when PBB was administered in the diet at a level of 400 ppm, for 15 days. Seven out of eight guinea pigs fed 50 ppm dietary PBB died (Vos and VanGerderen, 1973). Effects noted included cortical atrophy of the thymus, adrenal enlargement, marked depletion of the follicles and periarteriolar lymphocytes sheath of the spleen.

Chlorinated Dibenzofurans

See subacute toxicity.

Chlorinated Napthalenes

See end of chapter.



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TABLE 5

| Compound Tested | Species | Route | g/kg body wt $LD_{50}$ | Reference |
|---|---|---|---|---|
| Aroclor 1254 | Rat(Adult-Sherman Strain) | oral | 4-10 | Linder, et al. (1974) |
| " 1260 | Rat(Adult-Sherman Strain) | oral | 4-10 | " |
| " 1254 | Rat(weanling " " ) | oral | 1.295 | " |
| " 1260 | Rat(weanling " " ) | oral | 1.315 | " |
| " 1254 | Rat(female " " ) | intravenous | 0 358 | " |
| " 1221 | Rat(female " " ) | oral | 4.00 | Panel on Hazardous Trace Substances (1972) |
| " 1262 | Rat(female " " ) | oral | 11.3 | |
| " 1240 | Rat | oral | 4.25 | Bruckner, et al (1973) |
| " 1254 | Rat(Wister-30-day-old M-F) | oral | 1.3 | Grant & Phillips (1974) |
| " 1254 | Rat(Wister-60-day-old M-F) | oral | 1.4 | " |
| " 1254 | Rat(Wister-120-day-old M) | oral | 2.0 | " |
| " 1254 | Rat(Wistar-120-day-old F) | oral | 2.5 | " |
| Kaneclor-400 | Rat(Wistar Strain - Male) | oral | 1.30 (ml/Kg) | Ikeda, et al. (1970) |
| Kaneclor-400 | Rat(Wistar Strain - Female) | oral | 1.14 (ml/Kg) | |
| Kaneclor-400 | Mice(CFI Strain - Male) | oral | 1.875 (ml/Kg) | |
| Kaneclor-400 | Mice(CFI Strain - Female) | oral | 1.57 (ml/Kg | |
| Kaneclor-300 | Rat(Wistar Strain - Male) | oral | 1.15 | |
| Kaneclor-300 | Rat(Wistar Strain - Female) | oral | 1.05 | |
| BP-200 (biphenyls of dicloride and below | Mice(dd-strain-female) | oral | 6.36 | Kitamura (1972) |
| 2,4'-dichlorobiphenyl | Mice(dd-strain-female) | oral | 7.86 | " |
| trichlorobiphenyl | Mice(dd-strain-female) | oral | 3.06 - 4.25 | |
| biphenyls of trichloride and below | Mice(dd-strain-female) | oral | 9 27 | " |
| 2,4,3',4'-tetrachlorobiphenyl | Mice(CFI strain) | intraperitoneal | 2 15 | Yamamoto & Yoshimura (1973) |
| 5-OH derivative of 2,4,3'4'-tetrachlorobiphenyl | Mice(CFI strain) | " | 0.43 | " |
| 2,3,4,3'4'-pentachlorobiphenyl | Mice(CFI strain) | " | 0.65 | " |



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TABLE 6

Oral $LD_{50}$ (Rat) (Panel on Hazardous Substances, 1972)

| Compound Tested | | g/Kg body wt. |
|---|---|---|
| Aroclor 1221 | (undiluted) | 3.98 |
| Aroclor 1232 | (undiluted) | 4.47 |
| Aroclor 1242 | (undiluted) | 8.65 |
| Aroclor 1248 | (undiluted) | 11.0 |
| Aroclor 1260 | (50% soln in corn oil) | 10.0 |
| Aroclor 1262 | (50% soln in corn oil) | 11.3 |
| Aroclor 1268 | (50% soln in corn oil) | 10.9 |



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TABLE 7

Skin $MLD_{50}$ Rabbits (Panel on Hazardous Substances, 1972)

| Compound tested | | g/Kg body wt. | |
|---|---|---|---|
| Aroclor 1221 | (undiluted) | >2.000 | <3.169 |
| Aroclor 1232 | (undiluted) | >1.26 | <2.0 |
| Aroclor 1242 | (undiluted) | > .794 | <1.269 |
| Aroclor 1248 | (undiluted) | > .794 | <1.269 |
| Aroclor 1260 | (50% soln in corn oil) | >1.26 | <2.0 |
| Aroclor 1262 | (50% soln in corn oil) | >1.26 | <3.16 |
| Aroclor 1268 | (33.3% soln in corn oil) | >2.5 | |

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TABLE 8

$LD_{50}$ Polybrominated Biphenyls

| Compound | Species | Route | $LD_{50}$ | Reference |
|---|---|---|---|---|
| Firemaster BP6 | Rat | Oral | 21.5 g/Kg | Hill Top Research, Inc. (1970) to Michigan Chemical Company |
| Octabromobiphenyl | Rat | Oral | More than 2 g/Kg | Aftosmis, et al., 1972 |
| Octabromobiphenyl | Bobwhite Quail | Oral | More than 12.5 g/Kg | " |
| Hexabromobiphenyl | Rabbit | Dermal | More than 5 g/Kg | " |
| Octabromobiphenyl | Rabbit | Dermal | More than 10 g/Kg | " |



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## SUB-ACUTE AND REPRODUCTIVE EFFECTS OF PCB'S, PBB'S AND CHLORINATED DIBENZOFURANS

### A. Mammalian Sub-acute Effects of PCBs

In contrast to the acute toxicity (oral and dermal) of the PCBs which is of relatively low order when the substances are administered as a single dose, aspects of sub-acute toxicity of both the PCBs and individual chlorinated biphenyls appear to be of far greater concern exhibiting species sensitivity and cumulative toxic effects following continuous exposure at low levels.

When groups of mice were given toxic rice bran oil containing 1,600 ppm PCBs or 5000 ppm of PCB mixture with 48% chlorine, the toxicity response wee similar in the two groups (Nishizumi 1970).

In a group of rats fed 2000 ppm of Phenochlor DP-6, deaths occurred between the 12th and 26th days with enlarged livers, atrophy of the spleene and a progressively induced hepatic porphyria wae noted (Voe and Koeman, 1970). Administration of 1000 ppm of Arochlor 1254 in the diet of rats resulted in deaths between the 28th day and 53rd day (100% mortality) of feeding (Tucker and Crabtree 1970), 50% mortality et the same doeege in an 8-month study was reported by Kimbrough et al. (1972).



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Two groups of 52 male Sherman strain weanling rats were fed 100 ppm Aroclor 1242 (6.6-3.89 mg Aroclor/kg body weight) and 100 ppm Aroclor 1016 (6.9-3.5 mg Aroclor/kg body weight) respectively for 6 months and 54 rats were fed ground chow. Groups of 4 rats were sacrificed at intervals. Microscopic examination of the liver showed evidence of lipid accumulation, enlarged liver cells, inclusions in a number of livers and hemorrhage and necrosis in one animal per experimental group each No difference in effect between the two compounds was noted (Burse et al., 1974).

Rats given 100 mg Aroclor 1242/kg body weight by stomach tube every other day for 3 weeks showed histopathological changes only in the liver and kidneys as foci of sudanophilic vacuolation but no overt sign of toxicity (Bruckner et al., 1973).

Rats (Sprague-Dawley) given 100 ppm of 2,5,2', 5'-tetrachlorobiphenyl in the diet for 3 weeks showed less liver hypertrophy and fewer biochemical changes than rats given a similar dose of Aroclor 1248 (Allen et al., 1975).

Administration of 100 mg/kg body weight of Aroclors 1242, or 1254 orally to rabbits once a week for 14 weeks resulted in enlarged livers with apparent destruction of the rough endoplasmic reticulum and atrophy of the uteri, while Aroclor 1221 did not cause this effect (Koller and Zinkl, 1973).



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The unusual sensitivity of mink to PCBs was reported by Aulerich et al., (1973) who noted 100% mortality within 6 months following administration of 30 ppm PCB (10 ppm each of Aroclor 1242, 1248 and 1254) to adult mink. Feeding of Aroclor 1254 at levels of 5 and 10 ppm in the diet for 4 months led to a dose-dependent retardation of weight gain of growing female mink. Platonow and Karstad (1973) produced 100% mortality in mink when they fed this species 3.6 ppm Aroclor 1254 in the diet for 105 days.

Rhesus monkeys are also very sensitive to PCBs (McNulty, 1975). For example, adult female rhesus monkeys (Macaca Mulatta) have been fed diets containing 2.5, 5.0 and 25.0 ppm of Aroclor 1248 over periods ranging from 2 months for the higher PCB levels to 1 year for the two lower doses (Allen et al., 1974; Barsotti and Allen, 1975). Animals on the 25 ppm dose developed facial edema, alopecia and acne within 1 month, and 1 of 6 animals had expired as a result of PCB intoxication 2 months after removal from the diet. The total intake of PCBs ranged from 250 to 400 mg/animal. As was the case with the higher doses, these animals developed anemia, hypoproteinemia, bone marrow atrophy and severe hypertrophic hyperplastic gastritis. PCB concentrations in subcutaneous adipose tissue averaged 127 ppm at termination of experimental diet and 34 ppm 8 months later. Surviving animals continued to show clinical signs and lesions of PCB intoxication 2 years following PCB exposure (Allen, 1975).



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When adult rhesus monkeys were fed diets containing 2.5 and 5.0 ppm Aroclor 1248 for 1 year, the females of the group developed periorbital edema, alopecia, erythema and acne from lesions that involved the face and neck within 1 to 2 months (Allen et al., 1974; Allen, 1975). Although males consumed more PCB, they exhibited only moderate periorbital edema and erythema. An abnormal dysplastic growth pattern of the gastric mucosa was also noted in male rhesus monkeys fed 300 ppm Aroclor 1248 for 3 months (Allen and Norback, 1973).

Cynomegalous or squirrel monkeys fed different dietary levels of Kanechlor 400 showed weight loss, palpebral edema, enlargement of the liver and pneumonia. The lowest effective total dose was 300 mg/kg bw given over 20-32 weeks (Nishizumi, 1970).

A rhesus monkey fed 3 ppm Aroclor 1242 for 8 months died spontaneously and on autopsy was found to possess a gastric lesion manifested by extensive down growth of the epithelium deep into the subserosa and into the muscular wall of the stomach itself (McNulty, 1975).

One monkey fed 2,4,4'-trichlorobiphenyl and another fed 2,4,5,4'5'-pentachlorobiphenyl at 10 ppm for 90 days were reported clinically well and autopsies showed no positive findings (Oregon State University, 1975; McNulty, 1975).



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Third-litter sows received feed containing Aroclor 1242 at a concentration of 20 ppm throughout gestation and nursing. Treated sows differed significantly from control sows in the number of live pigs farrowed. Treated sows also had more mummified fetuses. Performance of live-born pigs was not affected by feeding PCB to the sows during nursing. Pathologic changes in the sows included hypertrophy of the liver and erosion of the stomach. Slight atrophy of the spleen and thyroid gland were observed in pigs from treated sows (Hansen et al., 1975).

Dermal toxicity studies in rabbits of technical PCB samples that contain an average of 60% chlorine (Phenoclor DP6, Clophen A60, and Aroclor 1260) as well as fractions containing tetra and pentachlorodibenzofuran have been recently described by Vos and Beems (1971). PCB-induced skin lesions were hyperplasia and hyperkeratosis of the epidermal and follicular epithelium following application of 118 mg of the three PCBs (5 times per week for 38 days) in the back skin of adult female New Zealand rabbits. Histopathology of the liver included centrolobular degeneration, centrolobular liver cell atrophy, focal necrosis, and cytoplasmic hyalin degeneration. PCB-induced kidney lesions were hydropic degeneration of the convoluted tubules and tubular dilation with the presence of casts. Definitive hyperplasia and hyperkeratosis of the follicular epithelium of the ear skin were seen after the topical application of fractions of Phenoclor and Clophen, while the fraction from Aroclor caused a minimal hyperplasia and hyper-



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keratosis of the follicular epithelium. Other effects elicited by the dermal application of the PCBs including thymus atrophy and lymphopenia as well as elevated excretion of fecal coproporphyria and protoporphyria.

From the response of the back skin and the liver of the rabbit to the three PCB mixtures, and from the response of the ear to the 25% diethyl ether-hexane fractions it was concluded that there were definite quantitative differences in toxicity, at lesst between the samples used in the above study and prior studiee. The extent to which these samples are representative of the normal commercial output has not been established and emphesizes the difficulty in the evaluation of toxicity data of PCBs in which the samples may differ in the amount and nature of toxic impurities.

Vos & Notenboom-Ram (1972) compared the toxicity of Aroclor 1260 with a single isomer 2,4,5,2',4',5'-hexechlorobiphenyl in New Zealand rabbits. Dermal applications of a total 120 mg Aroclor 1260 (5 times/week for 28 daye) resulted in early macroscopic skin leeione. The lesions in a 2,4,5,2',4',5'-hexachlorobiphanyl group of rabbits treeted similarly appeared later and were leee severe.

When groupe of 5 male mice were fed dietary levels of 10, 30, 100 and 300 ppm of the PCB isomers 3,4,5,3',4',5'-hexachlorobiphenyl, 2,4,5,2',4',5'-hexachlorobiphenyl and 2,4,6,2',4',6'-hexachlorobiphenyl



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for 4 weeks it was noted that the 3,4,5,3',4',5', isomer of hexachlorobiphenyl caused death in mice at a dietery intake of 10 ppm (2.1 mg/kg bw) within 36-47 days of exposure while consumption of the other two isomers resulted only in death at a dietary level of 300 ppm (64.3 mg/kg bw). Body weight was reduced at the lowest dose for the 3,4,5,3',4',5', isomer and only at the highest dose for the other two isomers. Atrophy of the thymus and hepatomegaly was noted at the lowest dose in the mice receiving the 3,4,5,3',4',5', isomer and only thie isomer produced experimental porphyris.

B. Avian subacute toxic effect of PCBs.

Chickens fed 10 ppm Aroclor 1242, or 100 ppm Aroclor 1254 showed diminished growth but a dietary level of 100 ppm Aroclor 1260 did not produce thie effect (Keplinger, et al., 1971).

Symptoms of PCB poisoning in birds consist of tremor, ataxia, ruffling, loss of feathers. Edema of the subcutaneous tissues, and fluid accumulation of the abdominal and thoraxic cavities are characteristic signs at autopsy. These findings are responsible for this disease being designated chick edema disease. Several outbreaks of chick edema disease caused by PCBs have been reported (Firestone 1973, Kohanowa 1969a, 1969b).

Chicks (one day old white leghorn cockerels) were fed the following PCB isomers: 2,3,6,2',3',6'- hexachlorobiphenyl 2,4,6,2',4',6', hexachlorobiphenyl, 2,3,4,2'3',4'-hexachlorobiphenyl,



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2,3,6,2',3',6', hexachlorobiphanyl and 2,4,5,2',4',5' hexachlorobiphenyl for 21 days at a level of 400 ppm. The isomer 2,4,5,2',4',5' hexachlorobiphenyl was also fed at the dietary level of 100 ppm. The 3,4,5,3',4',5' isomer was the most toxic causing death of all chicks within 11 days after onset of exposure. These chicks had chick edema disease, atrophy of the thymus and loss of subcutaneous and visceral adipose tissue. The other isomers also caused weight loss the 2,4,6,2',4',6' isomer being the least effective. Liver enlargement was noted for all groups the 2,3,6,2',3',6' being the least effective and the 2,4,6,2',4',6' the most effective. The microscopic findings made in the liver were most pronounced in the group fed the 2,4,6,2', 4',6' isomer. (Since the chicks fed the 3,4,5,3',4',5' isomer died relatively early in the experiment pathological findings in the livers of these chicks are not comperable to those made for the other isomers.)

When chicks were fed 5% rice oil which had produced Yusho they developed chick edema in 17 days. Similar symptoms were produced when chicks were fed 400 ppm Kaneclor 400 (Goto et al., 1969).

C. Mammalian Reproductive Effects of PCBs

Oral administration of 0.025 mg/day of Clophen A60 in peanut oil to NMRI-mice for 10 weeks lengthened the estrus cycle from 6.6 $\pm$ 2.5 days to 8.7 $\pm$ 4.3 days (Orberg and Kihlstrom, 1973: Kihlstrom et al., 1973). Females treated analogously for 62 days



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and mated with untreated males exhibited a significant decrease in the implantation rate from 87.0% in the controls to 79.5%. Male castrated mice were given 0.25 gm PCB (Clophen A 60) in peanut oil/day by stomach tube for 28 days and 70 µg testosterone propionate daily on days 19-28. The dry weight of the seminal vesicles after this treatment period was compared to castrated male mice given testosterone but not PCB and was found to be significanlty reduced.

Groups of 10 NMR I female mice were injected subcutaneously with 50 mg/kg body weight of Clophen A 60 in peanut oil or only the vehicle on the day of parturition and thereafter once weekly for 3 weeks. The 209 offspring were bred when reaching sexual maturity by mating them in pairs as follows: 17 pairs of controls, 19 pairs where the males had received Clophen A 60 as a suckling, 23 pairs where the female had received Clophen A 60 as sucklings and 23 pairs were both sexes had received Clophen A 60 as sucklings. The controls had an average of 11.5 offspring per litter while the parents where both mates received PCBs as sucklings had 8.8 offspring per litter. The resorption rate for both groups was the same but the number of implantations were reduced. Reproduction was not affected where only one mate received the PCB (Kihlstrom et al.,1975).

Keplinger et al.,(1972) reported low mating indices and decreased survival of pups for rats receiving Aroclor 1242 at 100 ppm, and decreased survival of pups receiving Aroclor 1254



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at 100 ppm. No reproductive effects were found with Aroclor 1260 at 1, 10, or 100 ppm, or with Aroclor 1242 or 1254 at 1 or 10 ppm. These studies suggest that in mammals reproductive effects decrease with increasing chlorination.

Sherman rats were exposed to polychlorinated biphenyls, Aroclor 1254 and Aroclor 1260. Rats exposed to Aroclor 1254 at dietary levels of 20 ppm or more had fewer pups per litter than the controls in the F1b and F2 generations (Linder et al., 1974). The 100 ppm exposure level of Aroclor 1254 increased mortality in the F1b offspring and markedly decreased mating performance of the F1b adults. The 500 ppm dietary level of Aroclor 1260 reduced litter size and decreased survival in the F1 litters. Dietary levels of 5 ppm Aroclor 1254 and 100 ppm Aroclor 1260 had no effect on reproduction in rats exposed through two generations. Liver weights were increased in 21-day-old F1 male weanlings at the 1 ppm level of Aroclor 1254 and in either sex of F1 and F2 weanlings at 5 ppm or higher levels of both Aroclor 1254 and 1260. Histological changes in the liver and increased liver weights were observed in adult rats exposed to the higher levels. Pregnant rats given Aroclor 1254 at the rate of 100 mg/kg/day on days 7-15 of gestation produced grossly normal litters, but only 30.1% of the pups survived to weaning. Reproduction and pup survival were not affected at dosage rates of 50 mg Aroclor 1254/kg bw/day or 100 mg Aroclor 1260/kg bw/day.



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Rabbits have been found to be more sensitive to PCBs than rats in regard to fetotoxic and reproductive affects (Villeneuve et al., 1971). In rabbits, no effects were reported at dosages of 6.25 or 10 mg Aroclor 1254 /kg/day, but at dosages of 12.5 to 50 mg/kg/day, fatotoxic effects were noted. Abortions, maternal death and stillborns occurred, but no consistent skeletal abnormalities were found.

Ringar et al., (1972) found that 1 ppm of Aroclor 1254 caused a slight reduction in reproductive success in mink, but at 5 ppm Aroclor 1254 complete reproductive failure was noted. In the same study 12 mink were fed a diet containing 30% Coho Salmon from Lake Michigan. None of the mink whelpedwhile 11 of 12 controls whelped.

Female adult rheaus monkeys fed Aroclor 1248 at a level of 25 ppm for 2 months noted at beginning of the 5th month experienced either fetal resorption or abortion during their second month of pregnancy (evidenced by regression in uterine size and reestablishment of menstruation (Allen et al., 1974). A third animal carried her fetus to term. This infant's weight was considerably lower than the average rhesus infant (375g ve 544 $\pm$ 101g) and had 25 $\mu$g PCB/g/(ppm) of adrenal end adipose tissue. Adult female rhesus monkeys fed 2.5 and 5.0 ppm Aroclor 1248 were bred to control males following the establishment of a relatively steady tissue level of PCB at 6 months. Following 3 mating, 12.5% of the 5 ppm group and 37.5% of the 2.5 ppm group was pregnant



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compared to 90% in the control group (Allen, 1975). The conception rate of females bred to PCB-fed males was equally as great as that in females bred to control males.

D. Mammalian Sub-Acute and Reproductive Effects of PBBs

Male Sprague Dawley rats maintained on diets containing 0, 0.1, 0.01 or 1% octobromobiphenyl for 30 days did not exhibit overt toxicity during this period. Gross pathological studies revealed enlarged livers and some kidney changes (petechial hemorrhage, enlargement), while histopathological changes in the 1% group included liver lesions consisting of centrilobular cytoplasmic enlargement and vaculation and kidney lesions consisting of hyaline degenerative cytoplasmic changes (Aftomis et al., 1972). Thyroid hyperplasia was observed in all rats in all test groups.

Rats were fed 100 and 1000 ppm octabromobiphenyl in their diets and then a number were sacrificed after 2 and 4 weeks of feeding and after 2,6 and 18 weeks of recovery on an octabromobiphenyl-free diet. The hepatocytes were markedly enlarged in rats fed 100 ppm and 1000 ppm of the PBB after 2 weeks of treatment. Histopathologic changes were characterized by hepatocellular hypertrophy with the pathologic changes localized mostly in the centralobular zone (Aftomis et al., 1972b). The cytoplasmic inclusions were more pronounced in the livers of rats fed 1000 ppm than 100 ppm of the PBB. There was some evidence of recovery since the livers of the 100 ppm level animals returned almost to normal 2 weeks after



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withdrawal of the compound.

Preliminary 28-day feeding studies in rats with octabromobiphenyl at 100 and 1000 ppm demonstrated liver enlargement, hepatocellular alterations with cytoplasmic inclusions and accumulative effects of bromine in the fat, liver and muscle (Aftosmis et al., 1972).

Norris et al., (1974) demonstrated hematological changes, liver enlargement and liver and kidney lesions at all levels of octabromobiphenyl (10, 000, 1000 and 100 ppm) in the diet of rats. When three groups of primagravid rats fed diets containing 100, 1000 or 10,000 ppm of octabromobiphenyl (BB-8) from day 6 through 15 of pregnancy and the pups delivered on day 20, anasarca was observed in one fetus at each of the two highest levels and gastroschisis was observed in another fetus at each of these levels. No other gross effects were observed in either the mothers or the pups and dose-related levels of bromine were found in the fetuses (Aftomis et al., 1972a).

When 1 g/kg of octobromobiphenyl was applied in corn oil under occuluded conditions to rabbits for 5 hrs/day for 10 days over a 2-week period, liver enlargement was found while at 0.1g/kg no enlargment was found (Aftosmis, et al., 1972b).

Hexabromobiphenyl (Firemaster BP-6) fed to pregnant Sprague-Dawley rats and Swiss/ICR mice (100 and 1000 ppm) resulted in a dose dependent decrease in mean fetal weight. (Number of pregnant



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mice: 16 controls, 9 received 100 ppm and 12 received 1000 ppm. Number of pregnant rats: 12 controls, 7 received 100 ppm and 6 received 1000 ppm). Exencephaly was produced in the offspring of the mice that received both 100 (3/121)and 1,000 ppm (2/174 dosages while cleft palate (4/87) and defective kidneys (2/87) were noted in offspring at the 1,000 ppm level. Nonpregnant mice fed 1,000 ppm PBBs for 11 days showed marked increase in liver size and weight (Corbett et al., 1975).

Cows from the Halbert herd received feed containing PBB at a level of 2914 ppm at a consumption rate of approximately 15 lbs/head/day (200 cows) or 8 lbs/head/day (200 cows) for a period of approximately 16 days. Early signs of toxicity included anorexia, decreased milk production, increased frequency of urination, lacrimation, some lameness, and shrinking of the udder of recently freshened cows. Upon removal of the contaminated feed appetite improved. Cows bred 4-6 weeks prior to onset came back in heat suggesting early resorptions.

Later signs exhibited by the cows included hematomas, abscesses, weight loss particularly in high producing cows, abnormal hoof growth, rough hair coat, alopecia, thickening of skin, dystocia, lack of udder development at freshening, metritis, and negligible milk production in cows freshening 3-4 months post-exposure. Cows also exhibited a lack of wound healing, general weakness and were highly susceptible to stress. Six months post-exposure non-lactating cows showed depression of



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appetite, prepartum weakness, failed to develop signs of labor and died without calving. Gross pathology included liver and renal degeneration, and hematomas and abscesses in the peritoneal and thoracic cavities.

Twelve 6 to 18 month-old heifers and bulls were offered the contaminated feed ad libitum. Five of the animals were dead within 6 weeks. Younger animals became prostrate, went through a short period of coma and died. Signs and lesions included testicular atrophy, abdominal adhesions, and massive liver abscesses. At 5 months only 2 animals remained alive. These became anorexic, would consume only milk and developed signs of hyperkeratosis over the entire body (Jackson and Halbert, 1974; Welborn, 1975).

The health status of 16 herds of dairy cattle exposed to low levels of polybrominated biphenyl (PBB) was compared with that of 15 control herds. Milk production of the contaminated herds was not significangly changed in 1972, 1973, and 1974, and was not significantly different from that of control herds in the same years. Mortality of adult cows and calves, the percentages of cows culled from the herds because of old age and low production, disease, or sterility, and the general health conditions were similar in the two groups. Serum concentrations of calcium, glucose, and cholesterol in contaminated herds were significantly different from those of the control herds, but the relationship to PBB exposure needs further investigation (Mercer et al., 1976, In Press).



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A study consisting of an investigation of records and herd history of herds accidentally exposed to PBBs was also done. Significant findings included a decrease in production of 20 to 50%; severe weight loss at calving time; sterility; decreased growth rate in young animals; evidence of soreness and stiffness; poor response to therapy of common diseases; prolonged wound healing; abnormal hoof growth; calf losses; and malformed calves. (Deming, 1975).

Polybrominated biphenyl was administered to four cows at a dose rate of 10 mg/head/day for 60 days. Although the study was not designed as a toxicology study, retrospective evaluation of the data provided no evidence of clinical problems among test animals during the period of exposure to PBB nor for a year thereafter (Frias and Marrow, 1975).

E. Chlorinated Dibenzofurans - Mammalian Toxicity

It is presently assumed because of our experience with the chlorinated dibenzodioxins, that the 2,3,7,8-tetrachlorodibenzofuran is the most toxic compound of this group of chemicals, while the dibenzofuran itself or the octadibenzofuran have little inherent toxicity on an acute basis (Kimbrough, 1974).

Baue et al, in 1961 demonstrated the toxicity of mixtures of tri and tetrachlorodibenzofurans. A single dose of 1 mg or 0.5 mg/kg body weight given orally to rabbits caused severe and often fatal liver necrosis. Application of this material to the rabbit



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ear caused severe hyperkeratosis at the application site.

The single oral $LD_{50}$ for tetrachlorodibenzofuran for guinea pigs is between 5 and 10 μg/kg. Symptoms of toxicity were severe weight loss, atrophy of the thymus and spleen. Hemorrhages were observed in the adrenals, urinary bladder and single cell necrosis was noted in the liver. C. D. Strain rats given as much as 1000 μg/kg bw of the TCDF failed to show any symptoms of toxicity and microscopic examination of their tissues did not reveal any abnormalities. Similarly oral single doses of up to 6000 μg/kg body weight of TCDF failed to elicit any toxic effect in C57Bl mice. A subcutaneous dose of 6000 μg/kg bw given to these mice produced weight loss, hepatomegaly and atrophy of the thymus but no fatalities (Moore et al., 1976).

While these findings indicate that in mice TCDF is 30 fold less toxic than TCDD, the overall estimated difference in toxicity between TCDD and TCDF is about tenfold (Bauer et al., 1961; Kimbrough, 1974; J. G. Vos, personal communication).

F. Chlorinated Dibenzofurans - Avian Toxicity

When 1 day old chicks were started on daily oral doses of 5 μg/kg TCDF daily most of them died within 11.5 days. One of 6 chicks died given daily oral doses of 1 μg/kg bw. Symptoms of toxicity at the low as well as the high doses consisted primarily of weight loss, decreased food consumption and general unthriftyness. At autopsy the most striking findings were subcutaneous edema, ascites and hydropericardium and atrophy of the thymus.



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An inflammatory exudate was noticed on microscopic examination in the lung and the pericardial surface of the heart showed inflammatory changes. Congestion and hemorrhage into the gastrointestinal tract was also observed (Moore et al., 1976).

G. Brominated Dibenzofurans - Mammalian Toxicity

When daily doses of 4 μg/rabbit of 2,3,7,8-tetrabromodibenzofurans were applied to the rabbit ear for 5 days giving a total dose of 20 μg/rabbit or about 5-6 μg/kg bw, the rabbits (4) developed hyperkeratosis of the treated ear and areas of liver cell necrosis (Kimbrough, personal communication). No other information is presently available on the toxicity of brominated dibenzofurans.

H. Longterm Toxicity Including Tumorogenesis

Chlorinated biphenyls

Mice

When male BALB c/J mice were fed Aroclor 1254 for 11 months at a dietary level of 0 or 300 ppm (49.8 mg/kg body weight) 22 of 50 mice in the experimental group and 24 of 50 in the control group survived. The livers of the test animals were markedly increased in size, and showed a qualitative increase in porphyrin. A total of 9 experimental mice showed 10 neoplastic nodules (hematomas, hyperplastic nodules) in their liver ranging in size from 0.1 - 0.5 cm in diameter. Other morphological changes observed in the livers of the experimental group including pleomorphism,



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areas of necrosis, and foci of adenofibrosis. (Kimbrough and Linder, 1974). Neither tumors nor the other morphological changes were noted in the controls. Only one of 24 surviving male BALB c/J mice fed Aroclor 1254 for 6 months followed by a control diet for 5 months had a hepatoma (neoplastic nodule) measuring 0.3 cm in diameter.

Ito et al., (1973) fed male dd mice 500, 250, 100 and 0 ppm Kanechlor 500, 400 and 300 in the diet respectively. After one year, hepatocellular carcinomas were observed in 5/12 mice fed Kanechlor 500 at a dietary level of 500 ppm, the other 7 mice in this group had nodular hyperplasia (neoplastic nodules?). No metastases were observed. Feeding $\alpha$, $\beta$, or $\gamma$ hexachlorocyclohexane (BHC) isomers with or without Kanechlor 500 at a dietary level of 250 ppm to mice for 24 weeks did not produce any tumors in the liver of animals receiving only the PCB (Ito et al., 1973). Liver tumors were observed in mice receiving 250 ppm $\alpha$ BHC alone or 250 ppm PCB and either 100 or 50 ppm $\alpha$ BHC or 250 or 100 ppm $\beta$ BHC. Thus PCBs enhanced the tumor development of liver tumors of $\alpha$ or $\beta$ BHC isomers.

Rats

Dietary exposure of male Sherman strain rats to levels of 500 ppm Aroclor 1254 (36.4 mg/kg body weight) for 6 months resulted in pronounced lipid accumulation in the liver which persisted for a 10 months observation period following the discontinuation of



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the dietary exposure to PCBs. Adenofibrosis (cholangiofibrosis) which was observed in the livers of rats ingesting the PCB for 6 months was still present 10 months later. Whether this lesion is persistent could not definitely be determined since high levels of the higher chlorinated biphenyl isomers were still present in adipose tissue and liver (Kimbrough, et al., 1973). Adenofibrosis is a focal proliferation of glandular epithelium forming ducts often surrounded by extensive fibrosis. This lesion usually occurs concomitantly in rat livers with hepatocellular carcinomas and has been produced by many known hepatocarcinogens. It was first described by Edwards and White (1941). The extremely long retention of some PCB isomers in rats was again noted when male Sherman strain rats were fed 100 ppm Aroclor 1254 for 6 months and then allowed to recover for 16 months. 4.4 ppm of PCB derived material was present in the liver and 152 ppm in adipose tissue on a wet weight basis (Kimbrough, 1976).

Continued dietary exposure to commercial PCB mixtures in mammals results in their storage in adipose tissue and over an extended period high levels may be attained (Curley, et al., 1971; Burse, et al., 1974). Blood levels are more a function of the dose, the rate of metabolism, availability of binding sites in blood, chemical characteristics of the compound given, rather than the amount stored in adipose tissue (Smrek, et al., in preparation).



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In another study, 200 experimental female Sherman strain rats were fed 100 mg/kg Aroclor 1260 in the diet (11.6 - 4.3 mg/kg body weight) for approximately 21 months and 200 female rats were kept as controls. A total of 184 experimental rats and 173 controls survived to sacrifice at 23 months. Hepatocellular carcinomas were observed in 26/184 experimental rats and in 1/173 controls. None of the controls but 146/184 experimental rats had neoplastic nodules within the liver (hyperplastic nodules). Areas of hepatocellular alteration were noted in 28/173 controls and 182/184 experimental rats. The incidence of tumors in other organs was not altered by PCB ingestion and metatases from the liver tumors were not observed (Kimbrough et al., 1975). Adenofibrosis of the liver was also noted in this study. In reporting these liver tumors the classification developed in a rat liver tumor workshop was followed (Squire and LeVitt, 1975).

Liver tumors (multiple adenomatous nodules) were also induced in 6/10 female Donryu rats with Kanechlor 400 but not in male rats. The dietary exposure varied throughout the study from 38-616 ppm (Kimura and Baba, 1973). The induction of liver tumors by known carcinogens in Sprague Dawley rats such as 3'methyl-4-dimethylamino-azobenzene, N-2-fluorenyl acetamide and diethylnitrosamine was inhibited by Kanechlor 500 (Makiura et al.,1974).

Groups of 50 male and female Charles River rats were fed dietary levels of 0, 1, 10 and 100 ppm of Aroclor 1242, 1254, and 1260 respectively (Calandra, 1976). At 3, 6, and 12 months after



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onset of the experiment, 5 rats per each group and sex were sacrificed. The surviving rats were sacrificied 24 months after onset of the experiment. The only body weight depression noted was observed in the female rats fed Aroclor 1254. The author reported that important histopathologic changes were only noted in the rats fed the different Aroclors for 24 months.

The liver weights were increased in the male and female rats fed 100 ppm of Aroclor 1260 and 1254 and only in females fed Aroclor 1242 at a dietary level of 100 ppm. It was noted that of a total of 20 surviving male and female rats (10 male and 10 female rats) fed 100 ppm Aroclor 1242 in the diet for 24 months 8/20 rats showed liver lesions called nodular hyperplasia, 2/20 had hepatomas and 1/20 had a cholangiohepatoma.

A total of 27 male and female rats (13 male and 14 female rats) survived to 24 months in the groups of rats fed 100 ppm Aroclor 1254 of these 13/27 had nodular hyperplasia of the liver 4/27 showed hepatomas and 2/27 showed cholangiohepatomas.

Similar findings were made in the total 27 surviving male and female rats fed 100 ppm Aroclor 1260 for 24 months. In these groups 7/27 male and female rats had nodular hyperplasia of the liver. 5/27 had hepatomas, and 2/27 had cholangiohepatomas.

Groups of 4 male and 4 female dogs were fed dietary levels of 0, 1, 10, and 100 ppm Aroclor 1242, 1254, and 1260 respectively for 2 years, 1 female fed 100 ppm Aroclor 1242, 1 male fed 10 ppm Aroclor 1242, 1 female dog fed 100 ppm Aroclor 1254, 1 female



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fed 10 ppm and 1 male and 1 female fed 100 ppm Aroclor 1260 died. At the dietary level of 100 ppm of either Aroclor 1254 or Aroclor 1260 the body weight gain was slightly decreased for both sexes. Aroclor 1260 also reduced the body weight gain in the females at the dietary level of 10 ppm. In the beagles fed Aroclor 1260 at a dietary level of 100 ppm, the liver weight was increased and the serum alkaline phosphatase was elevated. All but 1 dog that died had either chronic peritionitis or chronic pneumonitis. Many of the dogs fed 10 or 100 ppm of different PCBs showed chronic inflammation of a variety of organs. Male dogs fed Aroclor 1254 showed diffuse hyperplasia of the interstitial cells of the testes and aspermatogenesis (Monsanto, 1971).

Decachlorobiphenyl

No long term studies are available.

Polybrominated Biphenyls

No long term studies are available.

Chlorinated Dibenzofurans and Chlorinated Naphthalenes

No long term studies are available.

Immunosuppressive Effects of PCB and PBB

Polychlorinated biphenyls (PCB) have been implicated as immunosuppressants. Friend and Trainer (1970) found a decreased resistance to duck hepatitis virus in PCB exposed ducklings.

PCB has been shown to produce lymphoid atrophy in rabbits (Street and Sherma, 1975; Vos and Beems, 1971), chickens (Flick et al., 1975; Vos and Koeman, 1970) and guinea pigs (Vos and DeRoij, 1972; Vos and van Genderen, 1973). In addition, PCBs were found to suppress humoral immune responses to several antigens in rabbits (Koller and Thigpen, 1973, Street and Sharma, 1975) and guinea pigs (Vos and DeRoij, 1972, Vos and van Genderen, 1973) and to suppress cell mediated immune reactions in guinea pigs (Vos and van Genderen, 1973). In all of these studies the PCBs studied have been commercial mixtures; the role of contaminants such as chlorinated dibenzofurans needs to be considered.

The effect of HBB on the immune system was studied in chickens and guinea pigs (Vos and van Genderen). In chickens the atrophy of the lymphoid tissue was observed in bursal follicles and in the spleen. It suppressed the humoral immune response in guinea pigs.

## MUTAGENICITY AND TERATOLOGY

### Polychlorinated Biphenyls

The mutegenicity and cytotoxicity of PCBs, Aroclor 1242 and 1254, were studied by Green et al., 1975, a, b. The lack of mutagenic activity as measured by the dominant lethal test was reported for male Osborne-Mendel rats subjected to four different dosage regiment: (1) single oral intubations of 625, 1250 or 2500 mg/kg (for Aroclor 1242); (2) five daily doses of 125 or 250 mg/kg (for Aroclor 1242), 75, 150 or 300 mg/kg (for Aroclor



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1254); (3) five daily doses of Aroclor 1254 at 150 mg/kg, then starved overnight prior to mating; (4) chronic feeding for 70 days at a dietary level of 25 or 100 ppm prior to mating (Green et al., 1975a). In a second study, bone marrow and spermatogonial cells were cytogeneticelly investigated from Aroclor-treated Osborne-Mendel rats. Aroclor 1242 was given orally at single doses of 1250, 2500, or 5000 mg/kg or as four daily doses of 500 mg/kg/day; Aroclor 1254 was administered orally as five daily doses of 75, 150, or 300 mg/kg/day. No statistically significant increases in chromosomal aberations were found for the Aroclor-treated rats. As indicated by Green et al, (1975 b), while these studies were negative in regard to chromosomal mutations, they do not entirely rule out the possibility that PCBs may induce point mutations. To date, there are no known reports in the literature concerning the induction of point mutations by PCBs in laboratory model systems. The observations of Nilsson and Ravel, 1974, in Drosophila where no chromosomal breakage was found among Clophen A30- or A50-treated flies, are fully consistent with those of Green et al., (1975, a, b).

TERATOLOGY

Female NMRI mice, orally administered 0.025 mg/day of Clophen 60, showed evidence of an increased estrus cycle from 6.6 ± 2.5 days to 8.7 ± 4.3 days and a decreased ova implantation rate from 87.0 to 79.5% (Orberg and Kihlström, 1973). Kihlström et al., (1974 a), also demonstrated a decrease in the frequency of implanted ova



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among young animals nursed on milk containing PCB (Clophen 60). Both the male and the female contributed to the magnitude of the reduction of implantations. The mechanism by which exposure of the developing male to milk containing PCB leads to a reduction in the precent of ova implanted is unknown but need not necessarily involve dominant lethal events.

No reports of frank terata were found, but a number of studies indicated the facility of PCBs for placental transport. Grant et al., (1971 b), showed that the concentration of PCB in feral tissue from oral administration of Aroclor 1254 was dose-dependent and that accumulation occurred in fetal liver where higher concentrations were found than in maternal livar. Placental transport of PCB has been reported for the rabbit, rat, mouse, and cow (Grant et al., 1971; Platonow and Chen, 1972; Berlin et al., 1975; Melvas and Brandt, 1973; Brandt and Ullberg, 1973). Evidence of placental transfer was also observed among Yusho patients. While PCB has no known or clearly defined teratogenic effect in mammals, their easy passage across the placenta suggests the possibility of potential for some form of fetal toxicity.

Chlorinated Dibenzofurans

No information available.

Brominated Biphenyls

15 pregnant rats were force fed 100 mg hexabromobiphenyl/kg



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body weight on days 6, 8, 10, 12, 14 and 16 of pregnancy in corn oil. 14 control rats were given corn oil. The pups were obtained by cesarian on day 19 of pregnancy. No terata were observed and no difference between control and experimental rats was noted in any of the other parameters studied.

5 control and 6 rBB treated animals were treated with colchicine and bone marrow was studied cytogenetically. Animals treated with PBB and colchicine had higher metaphase and mitotic indices than non-treated animals. Chromosome aberrations were not noted (Fiscor and Warts, 1976).

Toxicity of Chlorinated Naphthalenes

The following information is derived from a review published by the EPA, namely Environmental Hazard Assessment Report 'Chlorinated Naphthalenes', EPA-560/8-75-001*. For literature references please refer to this report.

Animal Studies

The following tables summarize the available studies in laboratory animals. Three routes of exposure are reported.

*This document is available through the National Technical Information Service, Springfield, VA 22151. 'Preliminary Environmental Hazard Assessment of Chlorinated Naphthalenes, Siliconee, Fluorocarbons, Benzenepolycarboxylates and Chlorophenols'. (NTIS accession number PB-238-074/AS)



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TABLE 9

(a) Dietary Exposure

| Compound | Species | Dose Level | Days Exposed | Effect Reported |
|---|---|---|---|---|
| Mono- | | | None reported | |
| Di- | Rat | 5g/Kg feed | 15 | Increase in liver weight, growth impaired, rough coat. |
| Tri- | Mouse | 2.5 mg/day | 20 | None. |
| Tri- | Rat | 300 mg/day | 9-136 | Fatty infiltration of liver. |
| Tri/Tetra | Rabbit | 15 mg/Kg body wt. | 60 | None. |
| Penta | Rat | 50 mg day | 63 | Jaundice/fatty degeneration of liver. |
| Penta/Hexa- | Rat | 300 mg/day | 33 | Death with yellow livers and extreme fatty degeneration. |
| Penta/Hexa | Rat | 100 mg/day | 55 | Death with yellow livers and extreme fatty degeneration. |
| Penta- | Guinea pig | 2.5 mg/Kg in oil | 48 | Death-Severe weight loss fatty degeneration of liver. |
| Hexa- | Rat | 20 mg/Kg | 84 | Weight loss. |
| Hexa- | Rat | 63 mg/Kg | 84 | Weight loss. |
| Hexa- | Rat | 200 mg/Kg | 84 | Death. |
| Heptachloro- | | | None reported | |
| Octa- | Rabbit | 1 g | Single dose | Death in seven days. |
| Octa | Rat | 0.5, 2 or 5g/Kg | 22 | Decrease in liver but not plasma Vit. A. |

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TABLE 10

(b) Inhalation Studies

| Compound | Species | Dose Level | Days Exposed | Effect Reported |
|---|---|---|---|---|
| Tri- | Rat | 0.05-0.2 mg/L | 2 hr/day for 20 days | No toxic effect. |
| Tri- | Rat | 1.31 mg/L | 16hr/day for 134 days | No toxic effect. |
| Tri- | Rat | 10.98 mg/L | 16hr/day for 102 days | Slight liver discoloration, 5% rats showed increased fatty degeneration |
| Penta- | Rat | 1.16 mg/L | 16hr/day for 52 days | Jaundice, enlarged liver and 69% fatality. |

No studies have been reported for the other chlorinated naphthalenes.

(c) Dermal Studies

| Compound | Species | Conc Applied | Days of Application | Effect |
|---|---|---|---|---|
| Mono- | Rabbit(ear) | 90 mg/g Acetone | 5-7 | Mild reddening. |
| Mono- | Rabbit(ear) | 500 mg/g Acetone | 5-7 | Severe reddening, no decrease of sebaceous glands. |
| Tri- | Rat(skin) | Not given | 2 hr/day/40-60 days | None reported |
| Tri- | Mice (skin) | Not given | 2 hr/day/40-60 days | None reported |
| Penta- | Swine (skin) | 60 mg/Lx3L | 6 days/week-4 weeks | Slight hyperkeratosis. |
| Penta- | Rabbit(ear) | 30 mg/g acetone/day | 5 | Mild dermatitis, hair follicular attenuation. |
| Hexa- | Rabbit(ear) | 30 mg/g acetone | 5 | Decrease in sebaceous glands |



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## TOXICITY IN BIRDS

### Turkey Poults

Penta/Hexachloronaphthalenes in the diet.

(a) 5 ppm for 40 days caused 23% reduction in weight gain with 6.5% mortality.

(b) 100 ppm - all poults died within 33 days. Histopathologic examination showed enlarged and darkened livers.

The $LC_{50}$ was 20 ppm with an average decrease in weight of 51%.

### Chickens

Penta/Hexachloronapthalene in the diet.

(a) No effects reported at dietary levels of 4 and 20 ppm (35 days).

(b) 100 ppm prevented egg production (35 days).

(c) 2550 ppm caused 100% mortality (14 days).

Effects reported included enlarged fibrous livers, lack of feather pigmentation and pericardial and peritoneal edema. A fourfold increase in Vitamin A in the diet decreased toxic effects.



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TOXICITY IN DOMESTIC ANIMALS

Ingestion is the most important route for exposure of domestic animals.

Cattle

Highly chlorinated naphthalene poisoning in cattle is known as bovine hyperkeratosis or X-disease. Detailed information on toxic doses is lacking. Penta/hexa mixture at a dose level of 5.5 mg/Kg body wt (orally) for 5 days caused a sharp drop in plasma Vit. A by the end of the third day and depressed plasma Vit. A for over 30 days. A single oral dose of 1 mg/Kg body wt of hexachloronaphthalene caused mortality within two weeks. The first effect of chloronaphthalene poisoning is an interference with the biotransformation of carotene to Vit. A. The Vit. A. deficiency is quickly followed by inflammation of the oral mucosa, lacrimation, excessive salivation and irregular food consumption. Gross physical effects that develop during the course of disease include a general thickening of the skin caused by over development of the skin's hairy layer with loss of hair (hyperkeratosis). There may be degeneration or irregular growth of the horne. Continued exposure results in anemia, dehydration, loss of weight, fever and death. There may be severe liver damage.

Swine

No toxic effects were observed at levels of 100 mg/Kg



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body wt of the chlorinated naphthalenes (degree of chlorination was not stated). 150 mg/kg body wt caused a marked depression of plasma Vit. A, and death occurred at a dose level of 198 mg/Kg body wt.

Sheep

Toxic effects are observed at doses of chlorinated napthalene 10 X that required to produce toxicity in cattle. Sheep do not show cutaneous hyperkeratosis or an excessive drop in plasma Vit. A as observed in cattle.

Effects reported include nasal discharge, weakness, loss of weight, loss of appetite, ascites, necrosis and cirrhosis of the liver and cardiovascular effects.

Toxicity in Man

The most important route for mans occupational exposure to chlorinated naphthalene is via the inhalation route. Dermal absorption route may also occur.

Penta/hexachloronaphthalene at a concentration of 30 mg/g acetone, applied to the backs of human volunteers for a period of six weeks caused typical chloracne.

Daily topical application of mono-dichlorinated naphthalene to the human ear in mineral oil suspension for 30 days caused no observable effect. Two clinically distinct but oftern concurrent syndromes have been described, namely, liver necrosis, and chloracne



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with itching. A number of deaths involving acute atrophy of the liver have been reported; with the course of the disease similar to other forms of severe liver damage. Autopsy of fatally exposed workers show severe yellow atrophy of the liver. Definite data relating to dose response effects are in general lacking. However, two dermal studies with man have been reported.

SUMMARY

The degree of toxicity of chlorinated napthalenes increases with the degree of chlorination. The higher chlorinated naphthalenes (especially hexa and penta) have been associated with chlorecne and liver damage in man. Dose response effects have not been estsblished.

Cattle are more susceptible to the effects of chlorinated nephthalene than other domestic animals. The effects observed include development of Vit. A deficiency, hyperkeratosis and liver damage.



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# HUMAN EXPOSURE TO POLYCHLORINATED BIPHENYLS

## I. Introduction

Although the polychlorinated biphenyls (PCBs) have been used in a wide variety of industrial applications in the U.S. from 1930 until 1970, when their distribution was voluntarily restricted to closed systems, there is scant information on human exposure to adverse human health effects in the U.S. In the U.S., minimal human exposure of the population to PCBs is limited to food, air and water, while significant human exposure appears to be limited to sports fishermen consuming fresh water fish from contaminated streams and lakes; and to occupational exposure in industrial workers.

## II. Dietary Exposure to PCBs

The May 1972 Report of the Interdepartmental Task Force on PCBs summarized the findings in food sampling programs conducted by the Food and Drug Administration and the Department of Agriculture. Jelinek et al., (1975) have provided an update to that summary and evaluated trends which is presented below.

The 1972 Task Force report outlined three broad sources of food contamination. These are restated here in order to show which sources have been effectively controlled and which sources apparently persist, as reflected by results obtained in continued food sampling.



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1. Environmental contamination - background levels of PCBs in fish from lakes and streams.

2. Industrial accidents - isolated incidents involving direct leakage and spillage or contact of PCB fluids and other PCB containing materials on animal feeds, feed ingredients or food.

3. Food packaging materials - PCB migration to foods packaged in PCB contaminated paper products.

As discussed in the 1972 Task Force report, there were a number of other positive findings in foods which could not be traced to one of these broad sources prior to 1972 and the source of such occasional findings remain speculative. However, in more recent years, essentially all positive findings of PCBs in the food supply have been attributable to one or more of these three sources.

Jelinek and Corneliussen (1975) reported that for the 1969-1975 period, there has been a significant decrease in PCB levels in all foods with the exception of fish, where no particular trend has been noted. From that observation they concluded that procedures instituted to exclude the use of PCBs in "open" applications have been effective, but that more needs to be done to prevent their entry into the acquatic environment.

Jelinek and Corneliussen (1975) presented data on FDA's sampling program showing a very conclusive decline in the FYs 1973-1975



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period in terms of occurrence rate of PCBs and maximum levels found in milk, eggs, cheese, animal feeds and components, processed fruits, and baby foods. Except for fish, occurrence rates in FY'75 had dropped to zero or less than 1% for all categories, and there were no findings in excess of the existing temporary tolerances. Similar data were presented covering USDA's sampling of meat and poultry which also showed a decline, to 0.3% positive and only 0.6% above the temporary tolerance.

Jelinek and Corneliussen (1975) also reviewed FDA Total Diet Study findings (FY'71-75) which showed that all food classes of the total diet have declined to no PCB occurrence and no calculable daily intake of PCBs, except in the meat-fish-poultry composites. About 40% of these composites continue to contain detectable PCBs, although only traces have been detected in the latter years. The limit of quantitative detection in the Total Diet Study is about 0.05 ppm; positive findings below that level are generally reported as traces since they are not reliably quantitated.

Occurrence of PCBs ended in 1973 in all other categories, notably in dairy product composites and the composites of grain and cereal products. This probably reflects the control measures effected on animal feeds and on the control of ingredients in recycled paper used for food packaging. What remains, based on the total diet study, is a continued low level occurrence (about 40% positive and only at trace levels) in the meat-fish-poultry



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composites. The fact that levels in these composites have declined to only traces further supports the inference that the meat and poultry and eggs no longer contain detectable PCBs and that the low level findings are primarily due to the fish in those composites. This would imply that the PCB levels in the diet may have "bottomed out" and may remain static until such time as there is a change in the levels for fish.

Estimates of dietary intake of PCBs based on the Total Diet Study must be taken as gross estimates because only traces are currently found and some type of estimated value must be assigned to those traces to calculate microgram per day intakes of PCBs. Jelinek and Corneliussen used a set of consistent assumptions for those traces in computing the daily intakes for the FY-1971-75 period. Their calculations showed a steady decline from 15 mcg per day down to 8.7 mcg/day, with all of this coming from the meat-fish-poultry group in FY'74-75 (minor and declining portions attributable to other food classes in earlier years.)

Review of this information implies that, although the various assumptions for trace residue reportings and varying food consumption date have a significant effect on daily intake calculations, an overall estimate in the order of 5-10 mcg/day PCB dietary intake of PCBs for th general public seems reasonable. This figure would not be applicable to diets which may include regular consumption of fish from certain locations with high PCB levels.



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## III. PCB Residues in Fish

Review of the data gathered by various agencies on fish residues indicates a serious incompatability of sampling programs for obtaining data which reliably define trends in PCB levels. The Food and Drug Administration's primary concern is the levels which exist in the edible portion of commercially imported fish which are marketed interstate. Therefore, in most cases, the heads, entrails, skin and fins are excluded. Furthermore, FDA sampling has not included fish which are caught and consumed locally (freshwater fish) and the sampling has dealt with the most important consumption species which are primarily of marine origin and which generally do not contain PCB levels as high as freshwater species from certain locations.

In order to determine the human exposure to PCBs through dietary fish, one must know the levels of PCBs residues in the edible portions of fish, and the amounts of the various kinds of fish consumed by the population at large, and by special subpopulations. During the past few weeks, information has been compiled by National Marine Fisheries Service-NOAA on the PCBs levels of these residues in marine and freshwater fish and shellfish, and on dietary habits.

Table 11 lists the 20 most important types of fish eaten in the U.S. today, and the mean daily amount of each type eaten by



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TABLE 11

Fish and Shellfish Consumption in the United States
(September 1973-August 1974)

| | Rank | Amount (mill.) lbs/yr) | Percent of total by wt. | Number of Actual Users (millions) | Mean amount per user (grams/day |
|---|---|---|---|---|---|
| Total | | 2957. | 100. | 197. | 18.7 |
| Tuna (mainly canned) | 1 | 634 | 21.4 | 130. | 6.1 |
| Unclassified* (mainly breaded, incl. fish sticks) | 2 | 542. | 18.4 | 68. | 10.0 |
| Shrimp | 3 | 301. | 10.2 | 45. | 8.3 |
| Ocean Perch* | 4 | 149. | 5.0 | 19. | 9.7 |
| Flounder | 5 | 144. | 4.9 | 21. | 8.6 |
| Clams | 6 | 113. | 3.8 | 18. | 7.6 |
| Crabs/Lobsters | 7 | 110. | 3.7 | 13. | 10.6 |
| Salmon | 8 | 101. | 3.4 | 19. | 6.7 |
| Oysters/Scallops | 9 | 88. | 3.0 | 14. | 7.8 |
| Trout** | 9 | 88. | 3.0 | 9. | 12.3 |
| Cod* | 11 | 78. | 2.7 | 12. | 8.1 |
| Bass** | 12 | 73. | 2.5 | 7.6 | 12.0 |
| Catfish** | 12 | 73. | 2.5 | 7.5 | 12.1 |
| Haddock* | 12 | 73. | 2.5 | 11. | 8.6 |
| Pollock* | 15 | 60. | 2.0 | 11. | 6.8 |
| Herring/Smelt/ Sardines | 16 | 54. | 1.8 | 10. | 6.7 |
| Pike** | 17 | 35. | 1.2 | 2.5 | 17.4 |
| Halibut* | 18 | 32. | 1.1 | 5.0 | 8.0 |
| Snapper | 18 | 32. | 1.1 | 4.3 | 9.3 |
| Whiting | 20 | 25. | 0.9 | 3.2 | 9.7 |
| All other classified | | 152. | 5.1 | n.a. | n.a. |

* Mainly imports
** Fresh water



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the subpopulations of actual users. The Table includes all dietary fish, from sport fishing and from domestic and imported commercial fishing. These 20 items comprise 95% of the fish products eaten. The results are taken from information in a recent Seafood Consumption Study by the National Purchase Diary of Schaumburg, Illinois, sponsored by the Tuna Research Foundation, and National Marine Fisheries Service data on the amount of fish and shellfish processed and distributed in the U.S. Several items in the Table are worth noting for purposes of the present analysis. (1) Although ninety-three percent of the U.S. population (197 million) eat fish, the average annual per capital consumption of fish is small: 15.0 lbs/year--fish eater; (2) a large "Unclassified" fish fraction exists in the U.S. diet, ranking just below tuna in importance; (3) the major portion of many of our most familiar types of seafood is imported; and (4) freshwater species, led by trout, bass, and catfish, comprise about 9% of our total fish diet.

A compilation of PCB Data, representing the results of all the measurements known to NMFS of PCBs in fish used in the U.S. diet is presented in the Compendium of PCB data. It includes data from government reports, and state and private publications. An effort was made to include current, unpublished data from marine laboratories. The compilation includes the results of a comprehensive literature search by the NOAA-Environmental Data Service (EDS). An EDS-ENDEX data search for unpublished data



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and a EDS-OASIS search were made, crossing marine food species with PCBs.

Two significant sources of data--the 1975 FDA-Market Basket results, and the results of the extensive Fall 1975 New York State Department of Environment analyses were not available to NMFS at the time Compendium was published. However, FDA has indicated that the 1975 Market Basket results do not differ dramatically from 1973/4 data. Preliminary results of the New York Study are included in Table 13.

Table 12 summarizes the PCB information assembled in the Compendium relating to the 20 most important kinds of dietary fish identified in Table 11. Although at one time or another, some PCB measurements have been made on many fish, Table 11 points out the inadequacy of information on PCB residues in the most important items in the fish diet. Sampling and analysis have been sporadic and not designed to monitor trends in human exposure. Systematic surveys of neither the important items nor the species most likely to be contaminated have been undertaken. Very few recent (1975) measurements have been reported.

However fragmentary the data on PCB levels may be, they do not include a single measurement exceeding 1 ppm in any of the ten most important fish foods, except for Dover sole and crab taken in the immediate vicinity of Los Angeles sewerage outfalls, and crab living on contaminated sediments in Upper Chesapeake Bay.



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TABLE 12

Maximum Reported PCB Residue Levels in
Fish Muscle (White Meat)

| Rank# | Kind | Range and Max. Level Reported (ppm) (see Compendium) | Mean Level in the U.S. Diet |
|---|---|---|---|
| 1. | Tuna | 0-0.49 (1973) | Unknown* |
| 2. | Unclassified (mainly breaded) | no data | Unknown |
| 3. | Shrimp | 0-0.167 (1971) | Unknown* |
| 4. | Ocean Perch | 0-0.25 (1974) | Unknown* |
| 5. | Flounder/Sole | 0.6.3[a] (1972) | Unknown* |
| 6. | Clams | 0-0.819 (1974) | Unknown* |
| 7. | Crabs/Lobster | 0-4.9[b] (1972) | Unknown* |
| 8. | Salmon | 0-0.5 (1974) | Unknown* |
| 9. | Oysters, Scallops | 0-0.43 (1974) | Unknown* |
| 10. | Trout (excl. Lake Trout) | 0-0.56 (1972) | Unknown* |
| 11. | Cod | 0-0.4 (1974) | Unknown* |
| 12. | Bass | 0.3-8.4[c] (1974) | Unknown* |
| 13. | Catfish | 0-4.4[d] (1974) | Unknown* |
| 14. | Haddock | 0-1.16 (1974) | Unknown* |
| 15. | Pollock | 0.0 (1974) | Unknown* |
| 16. | Herring/Smelt/ Sardines (marina | 0-1.6 (1974) | Unknown* |
| 17. | Pike | 0-0.65 (1974) | Unknown* |
| 18. | Halibut | 0-0.149 (1974) | Unknown* |
| 19. | Snapper | 0-0.1 (1974) | Unknown* |
| 20. | Whiting | 0.0 (1974) | Unknown* |

# These 20 items compose 95% of the fish in the U.S. diet.

* Probably well below 1 ppm

[a]Single Dover Sole taken near Los Angeles municipal sewerage outfall. The mean value of Sole samples taken in this vicinity was 1.3 ppm in 1975. Mean values of other samples taken away from outfalls were less than 1 ppm in 1975. In any case, although Dover sole is an important food fish in Northern California, it is not taken commercially in Southern California waters.

[b]Single yellowneck crab taken near Los Angeles outfall--mean of group including this crab was less than 1 ppm. A composite of Upper Chesapeake Bay crab contained 1.2 ppm in 1974.

[c]Taken in 1974 from St. Croix River, Wisconsin--mean of 4 White Bass was 6.2 ppm.

[d]Taken from Mississippi River, Minnesota--mean of a sample of 5 catfish was 2.5 ppm.



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No more than 1.6 ppm has been reported in items 11 through 20 in importance, with the exception of freshwater bass and catfish living in river and lake areas known to be contaminated with PCBs. Nonetheless, as emphasized in the Table, there have been no PCB measurements on the important unclassified dietary fish items, and the analytical information is in no instance systematic enough to permit an estimate of the actual mean PCB values, hence human exposure. The strongest statement that can be made, based on the incomplete data summarized in the Compendium, is that most of the important fish meat items eaten in the U.S. probably contain mean PCB residue levels well below 1 ppm.

Fragmentary evidence also leads to the conclusion that the PCB levels in fish liver, roe and oil are approximately ten times that in the edible muscle and the PCB concentrations are greatest in the older larger oily fish, especially those that spend part or all of their lives in contaminated lakes, rivers or estuaries.

Table 13 lists the location of the species where high PCB levels, exceeding the FDA 5 ppm guideline, have been reported. Although the trout, salmon and chub in Lakes Michigan and Ontario and the Hudson River bass, perch and eel constitute only a few percent at most of the national fish diet, in some areas they are a significant part of the fish diets.

In order to discover what is presently known about U.S. fish consumption habits, a wide range of government and private sources



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TABLE 13

Areas Where at Least One Example of High* PCB Residue Levels Have Been Reported in the Listed Species

| Species | Location |
|---|---|
| Striped Bass | Hudson River, New Jersey |
| Chub | Lake Michigan |
| Carp | Mississippi River, Minn., Lake Onandaga, NY |
| Chain Pickerel, Alewife | Hudson River |
| Coho Salmon | Lake Michigan, Lake Ontario, Hudson River |
| Chinook Salmon | Lake Michigan, Lake Ontario, Hudson River |
| Steelhead Trout | Lake Michigan, Lake Ontario |
| Lake Trout | Lake Michigan, Lake Ontario, Lake George, NY |
| Bass, Smallmouth | Lake Onandaga, Genessee River, NY Hudson River, Mohawk River, St. Lawrence River |
| White Perch | Lake Onandaga, Hudson River |
| Alewife | Hudson River |
| White Sucker | Hudson River, Mohawk River |
| Walleye | Hudson River, Mohawk River, Black River, NY |
| Bass, Largemouth | Hudson River, Mohawk River |
| Yellow Perch | Hudson River |
| American Eel, Crabs, Sturgeon | Hudson River |
| Rock Bass, Catfish | Hudson River |
| Bluefish | Long Island Sound (muscle from one large oily fish contained 8 ppm) |

This Table includes recent data from New York State Department of Environmental Conservation not included in the NMFS Compilation.

The Hudson River above Fort Edward, New York, is not contaminated with PCBs and the residue levels in fish are quite low.

*Above the current 5 ppm FDA Guideline.



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were consulted. A number of pre-1970 studies are available, including a 1969 NMFS Survey of Fish Purchases and a 1968-70 HEW Ten State Nutrition Survey. In addition, a few specialized reports have been published, including annual FDA Regional Basket Surveys that do not include fish eaten in restaurants and institutions or by sport fishermen, and surveys by the Sport Fishing Institute and a 1974 Texas A&M Analysis of Seafood Consumption Patterns in Texas. However, the most up to date and comprehensive data are available in the previously mentioned 1973-74 Seafood Consumption Study by the National Purchase Diary.

Table 14 summarizes results of fish consumption demographic information that defines the number of U.S. consumers who would exceed an arbitrarily set dietary micro-constituent level. To date, this kind of information is available only for total dietary fish and one subcomponent, tuna. Table 14 indicates that 75 million U.S. consumers eat more than the national average of 18.7 grams of fish per day and 29 million would exceed the present dietary intake limit of PCB (175 mcg/day) if the 35 or more g/day of fish eaten was contaminated on average to 5 ppm, the current FDA guideline level. There is, of course, no evidence that the mean PCB level in fish is anywhere near 5 ppm. Nonatheless, should it in the future reach 1 ppm, then by an extrapolation from Table 14, an estimate can be made that about 200,000 U.S. consumers would be exceeding the 200 mcg/day FDA limit. An



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TABLE 14

Number of U.S. Consumers Who Eat More Than X

| X (g/day) | Number (thousands of consumers) Total Fish | Tuna |
|---|---|---|
| 0 | 197,000 | 130,000 |
| 18.7 (national avg.) | 75,000 | 5,700 |
| 35 | 29,000 | 1,300 |
| 50 | 14,000 | 400 |
| 100 | 2,000 | 43 |
| 150 | 400 | 10 |

* Derived from graphs on pp. 21 and 23 of the Seafood Consumption Study, September 1973--August 1974 by the National Purchase Diary, Shaumburg, Illinois, 1975. Sponsored by the Tuna Research Foundation.



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another hypothetical case, if the PCB level in tuna should ever reach 1 ppm, then about 5,000 tuna fish consumers, those who eat in excess of 175 g/day, would exceed the 200mcg/day limit from tuna alone.

Survey data do show that in general U.S. fish eaters include a wide variety of fish items in their diet. Thus, even if certain items contain toxic micro-constituents at or near FDA guideline level, the great majority of consumers would receive only a small fraction of their average dietary intake from these sources.

Clearly, there is a need for demographic information, similar to that available on total fish and tuna, on the other principal foodfish items in the U.S. diet; and a need to extend and better measure the high-consumption tail of the distributions, to get a correct picture of the number and type of consumers eating in excess of 100g/day of any single fish product.

IV. Conclusions

At present, it is difficult to estimate human exposure to PCB from eating fish, either for the population as a whole or for subgroups at higher risk. Fragmentary evidence suggests that the exposure to the population as a whole from PCB residues in ingested fish are probably well below 19 mcg/day-consumer, based on PCB levels which are probably well below 1 ppm and 19g fish consumed/day.



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Although it is important to know the total population exposure, the first priority should be given to defining the dietary intake of PCBs in special subpopulations at high risk by reason of high consumption of fatty fish, or high sensitivity to the toxic action of PCBs. Such populations might include pregnant women and young children, dieters, low income groups and sport fishermen living in coastal and lake regions where there is known PCB contamination. Primate studies suggest that pregnant women and young children will be particularly sensitive to PCBs, and dieters, and in some areas low income groups may rely heavily on fish for food. For each subpopulation, the amount and kind of fish in the diet and the PCB residue levels in the fish eaten must be obtained in order to determine exposure. None of the necessary information to define the human PCB exposure in subpopulations is presently known.

The first priority in systematically monitoring and analyzing for PCB residues in fish should therefore be among the oily fish such as salmon, trout, mackerel, shad, sardines, herring, bluefin tuna, sable fish, white fish, striped bass, and bluefish, that have been identified as significant food items in the diets of special subpopulations of consumers.

Although only minimal dietary exposure to PCBs may occur through food, there is evidence of potentially significant exposure



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in those sub-groups of the population who regularly consume freshwater fish from waters which are contaminated with PCBs at levels which exceed the FDA temporary tolerances. These sub-groups of the population would include sports fishermen and others who consume locally caught freshwater fish.

In surveys conducted by the State of New York and FDA, samples of 17 species of fish were collected from 10 points on the Hudson River between Glens Falls and Alpine, 13 samples were collected at Glens Falls and 68 samples below Glens Falls. The data from these surveys indicate that 53 out of 68 samples of fish collected at or below Fort Edward (approximately 7 miles below Glens Falls) contained PCBs in excess of 5 ppm. The average level of PCB contamination in these 68 samples was 31.3 ppm. In contrast, only trace levels were found in the 13 samples obtained at Glens Falls. A high proportion of finfish and eels from all sampling points below Glens Falls contained PCB levels in excess of 5 ppm. The Fort Edward sampling point had the highest levels for the species examined, with individual samples ranging from 20.1 to 178 ppm.

Twenty-eight of the 33 samples of coho and chinook salmon from Lake Ontario exceeded 5 ppm, with a range from 21 ppm to 24.6 ppm. Of the 4 samples of lake trout tested, results were in the range of 10 to 15 ppm. All other species of fish sampled from Lake Ontario were at 5 ppm or less PCBs.

The Great Lakes represent another area of the nation contaminated with PCBs. The Michigan Water Resources Commission has reported



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that many surface water samples contained PCBs at concentrations above the detection limit of 10 ppt. (10 significant levels of PCBs have been found in rivers and streams discharging into the Great Lakes. Effluents from wastewater treatment plants servicing industrialized communities have been found to be highly contaminated.

Sampling surveys of Great Lakes fish have shown that most species tested contained detectable levels of PCBs and that residue levels were generally proportional to fish size (age) and highest in the predator species. Except for whitefish, the species of commercial or sport interest (trouts and salmons) from Lake Michigan were found to be highly contaminated with PCBs. Data obtained from lake trout collected from various areas of Lake Michigan show mean PCB levels ranging from 3.06 ppm to 11.93 ppm.

The Michigan Department of Public Health has recently completed a study (Humphrey et al., 1976) which attempted to assess some of the consequences of human exposure to PCBs from the consumption of sportsfish caught in different areas of Lake Michigan. The study included exposed and control subjects from five areas of Michigan bordering on Lake Michigan. Exposed study subjects were those individuals who consumed at least 24 to 26 pounds of Great Lakes fish per year. Control subjects were those individuals who consumed less than six pounds of Great Lakes fish per year.

A preliminary assessment of the findings in the study indicate that the most frequently recorded quantity of fish consumed by the



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study participants was in the 24-25 pounds per year range. The highest recorded fish consumption over the two year period of the study was 180 pounds per year and the highest single season consumption was 260 pounds. Mean PCB levels in whole Lake Trout are reported as 18.93 ppm in 1973 and 22.91 ppm in 1974, and 12.17 ppm in Coho Salmon in 1973 and 10.45 ppm in 1974.

However, comparisons of PCB levels in raw vs. cooked fish indicate that actual human exposure to PCB from fish consumption is less than might be expected from the raw fish data. This is not unexpected since preparation (trimming away fatty tissue) and cooking have been shown to decrease the concentration of PCBs. For example, the PCB level in cooked Lake Trout consumed by the study participants ranged from 1.03 ppm to 4.67 ppm; in cooked salmon from 0.48 ppm to 5.38 ppm; and in other cooked fish from 0.36 ppm to 2.06 ppm. These levels are decidedly lower than the level of PCB contamination reported in raw trout and salmon.

PCBs were found in 501 blood and breast milk specimens collected from study participants during the study. The values ranged from a low of 0.007 ppm in blood in the control group to a high of 0.366 ppm in the exposed group. Although there was a wide range of blood values for each quantity of fish consumed, there was a highly significant correlation between the reported quantity of Lake Michigan fish consumed and the concentration of PCB in the blood of study participants. Using a natural log transformation



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of PCB values, the correlation between amount of fish consumed and PCB blood levels was significant ($t=6.24$, $p= 0.0001$), with higher reported fish consumption being associated with higher blood PCB levels.

No annual variation in PCB blood levels in humans could be demonstrated. The mean PCB blood values for the control and exposed groups did not appear to change markedly from 1973 to 1974. In addition, abstinence from Lake Michigan fish consumption for a period of 90 days or more did not change the PCB blood levels significantly. PCB blood levels over the abstaining period shows variation but no steady decline in PCB. In fact, more subjects showed no change or a rise than showed a decline in PCB blood levels over time.

The calculated quantity of PCB ingested by eating Lake Michigan fish averaged 46.5 mg/year and ranged from 14.17 to 114.31 mg/year. PCB ingestion for each individual was determined by proportioning the reported annual fish consumption by frequency of species eaten and the cooked fish PCB levels for those fish. The community average for cooked fish was used in instances were cooked fish determination were not available for a study participant. Because fish consumption was found to vary from year to year the average annual consumption for each individual for the two baseline years of study was used in each case.



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Results from this study indicate the strong likelihood that the maximum allowable ingestion of 1 mcg/kg body weight/day recommended for protracted exposure to PCB is being exceeded by the majority of the exposed group participants, and by inference, by a substantial number of individuals participating in sport fishing and who consume quantities of contaminated fish. The calculated mean daily dose received by the exposed group is 1.7 mcg/kg/day and ranges from 0.49 mcg to 3.94 mcg/kg/day. If the average annual rate of PCB ingestion from fish indicated by these study results were continued over the years, and if net accumulation occurs, the average sports fisherman consuming contaminated fish could receive a total PCB dose equal to the 200 mg safety limit in approximately 4.3 years. Under the same set of assumptions, individuals consuming greater than average amounts of contaminated fish would reach the total dose level sooner.

No adverse health affects or group of symptoms could be identified in the exposed group that were clearly related to PCB exposure. This implies that exposure to PCBs from eating contaminated fish at the levels observed and the presence of PCBs in these exposed persons has not caused any observable adverse health effects at the time of the study. However this does not exclude the possibility that effects too subtle for detection are occurring or the possibility of long-term adverse health effects.



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Considerable scientific interest has centered on the Yusho incident in Japan where in 1968 human intoxication with Kanechlor 400, a PCB manufactured in Japan, was noted when a heat exchange leaked this PCB into rice oil ("Yusho" oil) which was consumed by Japanese families.

The typical clinical findings included chloracne and increased pigmentation of the skin, increased eye discharge, transient visual disturbances, feeling of weakness, numbness in limbs, headaches and disturbances in liver function. Most of the babies born to mothers with Yusho had skin discoloration which slowly regressed as the children grew in size. Adult "Yusho" patients had protracted clinical disease with a slow regression of symptoms and signs, suggesting a slow metabolism and excretion of the PCB in humans, probably resulting from a long biological half-life.

A review of the literature extent in 1972 led to the following facts with respect to this tragic incident in Japan:

1. The average PCB content of the rice oil in the dose-response epidemiologic study was 2,500 ppm.

2. In this dose-response epidemiologic study, the avarage cumulative intake of PCBs, leading to overt symptomatology, was 2000 g. In this same study, the lowest dose leading to overt symptomatology was 500 mg.



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3. The toxicity seen was the result of the ingestion of the PCBs contaminant in the rice oil.

A summary updating of the "Yusho" incident was presented by Professor Kuratsume at the National Conference on Polychlorinated Biphenyls, Chicago, Illinois, November 19, 1975 (Kuratsume et al., 1976). This report affects the original conclusions enumerated above.

1. PCBs Content of "Yusho" Oil.

The first reported PCB determination was made on a canned contaminated rice oil produced on February 5, 1968, and whose consumption was associated with overt symptomology. One group of Yusho patients was associated with ingestion of this contaminated canned rice oil produced or shipped on February 5 and 6, 1968 (Kuratsume et al., 1972).

The detailed epidemiologic studies are given in the papers by Kuratsume et al., (1969) and Yoshimura (1971) and summarized in English by Kuratsume (1972), Kuratsume et al., (1971) and Kuratsume et al., (1972). Of 325 patients queried, 166 of 170 used canned rice oil produced or shipped on February 5 and 6, 1968, and 143 of 155 used bottled rice oil shipped from February 5 to 15, 1968. Tsukamoto et al., (1969) determined that the organic chlorine content of canned rice oils produced or shipped on February 5, 1968, was 1000 to 1500 ppm, by chemical and activation analysis. Since Kanechlor 400 had a 48 per



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cent chlorine content, if one attributed all the organic chlorine to this specific chlorinated biphenyl, then the canned rice oil contained, as an average, 2500 ppm PCBs Kuratsune et al., (1969) reported that a qualitative GLC survey of 109 samples of bottled rice oil shipped between October, 1967, and October, 1968, showed "significant" PCBs content only in those samples between February 7 to 10, 1968. The largest amount was reported in the sample of February 7, 1968, but the quantity was not given. No bottled rice oil samples were available for February 5 and 6, 1968. However, in the paper by Kuratsune et al., (1971), the most marked contamination of bottled rice oil was for that shipped on February 7, 1968; and the maximum chlorine content was 462 ppm. One may assume that this number applies to the sample of bottled rice oil of February 7, 1968, giving this sample a PCBs content of 924 ppm. Kuratsune et al., (1976), reporting the work of Nagayama et al., (1975b), gives the PCBs analyses for three samples of toxic "Yusho" oil, used by three independent families with "Yusho", as approximately 800 to 1000 ppm. These were GLC/MS measurements. Nagayama, et al., (1975b), described these samples as follows:

"Rice oil used by patients with Yuso ("Yusho oil"): 3 samples of rice oil used by 3 independent families with Yusho in 1968. These samples had been kept in glass bottles with glass stoppers at room temperature in our laboratory until 1974 when the analysis was made."



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Kuratsume (1976) pointed out that these three samples were canned oil and considered to have been produced and shipped on February 5 or 6, 1968. This is also suggested in footnote a, Table 5, in the paper by Kuratsume et al., (1976). It was reported previously that, based on organic chlorine determinations, Tsukamoto et al., (1969) reported an average PCB content of the canned oil of February 5, 1968, to be 2500 ppm. Now, based on GLC/MS, we have a PCB content of ca 1000 ppm for canned oil sample of February 5 or 6, 1968.

What are the possible explanations for this discrepancy in values? One possibility is based on the assumptions associated with the method of measurement by Tsukamoto et al., (1969); namely, determination of total organic chlorine. Thus, any orgnic chlorine present in the rice oil, which was not a chlorinated biphenyl, would be included in the PCB analysis. Some possibilities include: chlorinated dibenzofurans, chlorinated naphthalenes, chlorinated paraffins, other chlorinated aliphatics and non-biphenyl aromatics.

Another possibility is related to the distribution, with time, of the contaminated rice oil samples. February 5, 1968, occurred on a Monday. The first rice oil samples reported to contain PCBs were produced on this date (Tsukamoto et al.; 1969). If the leak first occurred sometime over the weekend (February 3 and 4, 1968), and



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production was resumed Monday morning, February 5, 1968, then one would expect the samples of this date to be higher in PCB content.

Therefore, one would expect the first batches Monday morning to be highest with a dimunution of PCB content with time. Tsukamoto et al., (1969) reported the organic chlorine content of six canned oils produced Monday, February 5, 1968, as follows: 1020, 1170, 1070, 1080, 1030, and 1500 ppm. In the survey of bottled oils, Kuratsune et al., (1969) reported that for those samples where PCBs were found, the greatest amount was in the samples of February 7 (Wednesday); and after February 11. It was inferred from the paper by Kuratsune et al., (1971) that the PCB content of the bottled rice oil samples of February 7, 1968, was 924 ppm. In addition, in Table 5, Kuratsune et al., (1976) gave a value of 134 ppm PCB for a rice oil produced on February 10, 1968. Thus, the available data does confirm a diminution of PCB contamination of the rice oil with time. This is summarized below:

| Date | Source | PCB content (ppm) | Method |
|---|---|---|---|
| Feb. 5, 1968 | canned | 2000-3000 | organic Cl |
| Feb. 5 or 6, 1968 | canned (?) | ca 1000 | GLC/MS |
| Feb. 7, 1968 | bottled | 924 | organic Cl |
| Feb. 10, 1968 | bottled | 134 | " |
| After Feb. 11 1968 | bottled | just detectable | GLC (?) |



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The resolution of the varying PCB values for the canned oil cannot be made. However, Kuratsune (1976) confirmed that the production of rice oil had been suspended for an unknown period of time prior to February 6, 1968, when it resumed.

2. Dose-Response

There were 1291 "Yusho" patients as of April 30, 1975 Oema, 1975). Of this total group, the dose-response relationship was developed from a detailed analysis of 146 patients (Yoshimura 1971; Kuratsune et al., 1972). These 146 patients all consumed contaminated canned rice oil produced on February 5, 1968 contained 2000 to 3000 ppm PCBs as Kanechlor 400 (Tsukamoto et al., (1972) estimated that the lowest dose producing symptoms was 0.5 gm.

If the PCB content of this canned oil was ca 1000 ppm, as discussed in the previous section, then the average dose producing an overt effect was 0.8 gm. and the lowest dose producing an overt effect was 0.2 gm.

3. Causative Factors.

Originally, rice oil contaminated with a heat exchanger, Kanechlor 400, a polychlorinated biphenyl, was associated with "Yusho" symptomatology (see discussion in section



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1 above). PCBs were identified in the contaminated rice oil consumed and in the blood and tissues of Yusho patients. Therefore, the effects seen were attributed to PCBs.

In the review by Kuratsune et al., (1976), a new factor was introduced into the system; namely, the canned rice oil was also contaminated with chlorinated dibenzofurans to the extent of 5 ppm. In addition, in this same paper by Kuratsune et al., (1976), they presented data of Nagayama et al., showing polychlorinated dibenzofurans in the liver and adipose tissue of Yusho patients, while none was found in that of a control group. The ratio of PCBs to PCDFs in Kanechlor 400, a Yusho oil of February 5 or 6, 1968, in adipose tissue and liver from a Yusho patient were 50,000; 200; 144; and 4 respectively. Thus, relative to Yusho oil, the liver with a PCB/PCDF ratio of 4 to 1, appears to selectively concentrate PCDFs relative to PCBs.

Relative to PCBs, PCDFs are about 250 times higher in the rice oil than in Kanechlor 400 (Nagayama et al, 1975b; Kuratsune et al., 1976).[1] They also stated that PCDFs may be a factor in Yusho disease. In another

[1] It should be noted that we are comparing chlorinated dibenzofuran content of an unused Kanechlor 400 to that after prolonged use in a heat exchanger.



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paper Nagayama et al., (1975a) reported that it "is said that the toxicity of PCDF is said to be from 200 to 500 times that of PCB". At the lower factor (200), contaminated rice oil would be expected to be twice as toxic as Kanechlor 400. Also, in this same paper they stated that the symptoms seen in Yusho patients seemed to be more severe than would be expected just from the PCB intake associated with the oil.

If PCDF is 200 to 500 times more toxic than PCB (Nagayama et al., 1975a), then contaminated rice oil would be 2 to 3.5 times more toxic than that expected from its PCB content, alone. Kuratsume et al., (1972), estimated that if the average amount of contaminated rice oil ingested by a "Yusho" exposed person, is related solely in terms of its PCB equivalent, then the amount of oil needed to be ingested, on that basis, would be 1600 ml to 2800 ml. At a PCB content of 2500 ppm, this is an ingestion of 4 to 7 gm of PCBs equivalent; whereas at a PCB content of 1000 ppm, this is an ingestion of 1.6 to 2.8 gm of PCBs equivalent.

## Conclusion

The complexities and uncertainties associated with the most recent reports of the "Yusho" incident in Japan, make it extremely difficult to quantify possible human health effects resulting from exposure to PCBs alone.



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## V. Occupational Exposure to PCBs

The earliest reports of adverse health effects due to exposure of workers to PCBs in this country are probably those of Schwartz (1936) who described skin lesions and symptoms of systemic poisoning among workers who were said to have inhaled chloro diphenyls; their complaints included digestive disturbances, burning of the eyes, impotence, and hematuria. Patch tests with the chlorodiphenyls were negative, and Schwartz speculated that mechanical plugging of the follicles of the skin as the fumes solidified on it were responsible for the skin lesions; the chlorine present in the products were thought to then exert an irritating effect on the plugged follicles, and to thus cause suppuration. No quantitative data were reported, but a number of preventive practices were recommended.

There have been numerous reports over the ensuing years (Schwartz and Bartow, 1942), (Greenburg et al., 1939), (Drinker et al., 1937), (Good et al., 1943), (Schwartz, 1943) and (Meigs, et al., 1954) of cutaneous eruptions and of systemic manifestations (sometimes fatal) as well, among marine electricians, machinists, capacitor and transformer manufacturing workers, and others occupationally exposed to PCBs. However, in many of these reports the exposures are described as having been to mixtures of chlorinated hydrocarbons, quite often of chlorinated naphthalenes and PCBs.



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The skin lesions described by Schwartz (1936) have come to be designated generally as "chloracne." Chloracne can be produced by a number of chemical compounds, including chlorinated dibenzofurans and certain isomers of the chlorinated dibenzodioxins (Kimbrough, 1972). Oily skin and large pores seem to predispose to the disease, while the opposite is the case for smooth, tender skin (Kimbrough, 1974). Chloracne also has occurred among workers engaged in the production of 2,4,5-T (Poland et al., 1971).

Part of the chloracne lesion resembles adolescent acne, but it is generally more severe, and lesion distribution is inconsistent with, although it may be superimposed upon, adolescent acne. It is known that chloracne can be produced by either the systemic absorption of chlorinated biphenyls or the direct application of chloracnegenic compounds to the skin. Systemic effects sometimes result after occupational chloracne has manifested itself; these may include loss of appetite, nausea, edema of the face and hands, abdominal pain, vomiting, and burning and soreness of the eyes. No fully satisfactory explanations have been made of the development of chloracne. Chloracne is generally very persistent, and there is the preferred control measure. An excellent review of this subject is that by Crow (1970).

The U.S. occupational standards for PCBs are: 8-hour time-weighted average exposure limits of 1 $mg/m^3$ (skin) for the 42% chlorinated product, and 0.5 $mg/m^3$ (skin) for the 54% chlorinated product (CFR, Title 29, Part 1910.93). The National Institute for Occupational Safety and Health has under way the preparation of



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criteria for a recommended standard for occupational exposure to polychlorinated biphenyls. In addition to an environmental exposure limit, this document will provide comprehensive recommendations for medical surveillance, safe work practices, and engineering controls to ensure employee protection during occupational exposure over a working lifetime; it is scheduled for transmittal to the Department of Labor in late 1976 or early 1977. NIOSH also is undertaking certain studies of employee populations (mainly in U.S. capacitor factories) in order to better define the nature and extent of any chronic and/or life-shortening effects of exposure to PCB.

Practically all of the currently available information on worker health and PCB exposure is found in foreign literature. For example, Karppahen and Kolho (1974) reported on relationships between the concentration of PCB's in the blood of all, and in the adipose tissues of some, of 29 persons in "good health"* from

*Kolho has stated that the capacitor plant workers received quarterly health examinations, and, in addition to the clinical examination made at the time of the investigation, serum alkaline phosphatase, GOT, and GPT activities were determined; the 6 employees with the highest blood concentrations of PCBs also had BSP excretion tests performed. All results were normal.

In view of claims based on animal experiments that PCBs can induce liver microsomal enzyme activity, the authors also determined the half-time of antipyrin before and after phenobarbital induction (1 mg phenobarbital/kg body weight/day/3 days) for 6 capacitor plant employees and 6 controls; no enzyme induction was observed. Further, because of claims that PCB's have effects on steroid metabolism, 4 capacitor plant workers were tested for ACTH (serum, presumably); all results were reported to be within the normal range (letter from S. Hernberg to A. C. Kolbye, Jr., 12/23/75.



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three different employee groups in Finland. None of the employees had no history of occupational exposure to PCBs, six of them had handled PCB samples in an analytical laboratory, and the other eleven of them had been employed for 6 years in a capacitor factory where Aroclor 1242 was used as the impregnating fluid. It was stated that average PCB concentrations in the air of the capacitor factory had not exceeded "internationally accepted limits**," and that special attention had been given to skin protection.

Table 15 shows the observed tissue concentratins of PCB's; the authors were unable to detect any biological effects of the approximately 50-fold larger PCB concentrations in the blood of the capacitor plant employees, compared to the "unexposed" control group.

Ouw et al., (1974) conducted a survey to determine the degree of absorption and the health effects of exposure to "electrical grade" Aroclor 1242 (that "did not contain any impurities") for varying periods of time. PCB concentrations in the air of a capacitor factory in New South Wales, Australia*, were measured, and 34 occupationally exposed employees (15 males, 19 females, ranging in age from 33 to 55 years) were examined and compared with volunteer controls (23 males, 7 females ranging in age from 20 to 50

*The Joint ILO/WHO Committee on Occupational Health, in its sixth report (World Health Organization Technical Report Series No. 415, 1969), recommended for international adoption a "safe concentration zone" of 1 mg/m$^3$ for chlorinated derivatives of diphenyl.

**Aroclor 1242 is not manufactured in Australia (Ouw et al., 1974).



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TABLE 15

Concentration of PCB's

| Subjects | In blood Fat basis mg/kg | | In adipose tissue Fat basis mg/kg | |
|---|---|---|---|---|
| | Average | Range | Sample no | |
| Workers in capacitor factory | 313 | 100-700 | 1 | 200 |
| | | | 2 | 11 |
| | | | 3 | 160 |
| | | | 4 | 285 |
| | | | 5 | 635 |
| Persons handling PCB's in analytical laboratory | 53 | 33-71 | not analyzed | |
| Persons without any special exposure to PCB's | 5.4 | 3.6-9.9 | 1 | 2.3 |
| | | | 2 | 1.5 |

From Karppanen and Kolho (1974)



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years) having no history of occupational exposure to PCB's. Study parameters included occupational and medical histories, PCB-in-blood concentration estimates, and liver function (serum bilirubin, alkaline phosphatase, total protein, and GPT, and BSP excretion) tests.

The authors noted that exposed workers tended to complain of eye, face, and skin "burning," that the PCB "'fume' ... has a pungent smell which often causes persistent body odour," and that the employees with higher blood concentrations of PCBs complained most often of the skin lesions (one case of chloracne, five cases of "an eczematous rash on" the legs and hands), although there apparently was poor correlation of blood PCB's with the severity of the complaints. No significant health effects were observed among those of the 34 workers whose blood PCB concentrations were below 200 ppb.

There was a statistically significant difference between the blood PCB concentrations of the exposed group and the control group (P less than 0.01). Table 16 shows the concentrations of Aroclor 1242 in the capacitor plant air prior to and after exhaust ventilation system "improvements" had been made. (The Australian National Health and Medical Research Council recommended (Atmospheric Contaminants, 1970) exposure limit values of 1.00 $mg/m^3$ for PCBs of 42% and 54% chlorine content, respectively). Table 17 shows that there was no lowering of blood PCB concentration among those workers tested two months after the installation of a more



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TABLE 16

Arocloro 1242 concentrations in the air inside capacitor plant before and after improvement of exhaust ventilation system

| No. | Areas in the Impregnation Room | Aroclor concentration in $mg/m^3$ | |
|---|---|---|---|
| | | Before | After |
| 1 | Area in the unloading tank in front of exhaust register from operator's breathing zone | 1.44 | 0.75 |
| 2 | Area in the unloading tank not in front of exhaust register | 2.22 | 0.7 |
| 3 | General atmosphere near tank | 1.08 | 0.18 |
| 4 | Soldering area | 0.32 | 0.08 |

From Ouw et al (1974)



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Mean blood Aroclor 1242 levels before and after improvement of exhaust ventilation and the recommendation to use "suitable impervious" gloves.

| Group | Mean blood Aroclor levels in ppb. Retention times relative to Aldrin I. | | |
|---|---|---|---|
| | 0.69 | 1.31 | 1.41 |
| Before | 281.6 | 135.1 | 58.41 |
| After | 477.2 | 225.4 | 524.7 |
| Statistical differences | Not significant difference (P 0.01) | Not significant difference () 0.01) | Significant difference (P 0.05) |

From Ouw et al (1974)



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efficient exhaust ventilation system and the concomitant recommendation to wear "suitable impervious gloves" in order to reduce PCB absorption through the skin. The authors suggested that a failure to adhere strictly to the glove recommendation might explain the continued elevation blood PCB levels. Another possibility might be that PCBs were continually mobilized from storage in adipose tissues.

The Japanese Ministry of Labor (Regulation for the Prevention of Disturbances due to Specified Chemical Substances, under the Act for Safety and Health of Workers, of April 28, 1971) established an occupational environmental exposure limit for PCBs of 0.5 $mg/m^3$ at 25° C, one atm (Ad Hoc Committee, 1974). Japanese importation of PCBs commenced around 1950, and early uses were as dielectrics in capacitors and transformers. In 1954, Kanegafuchi Chemical Industry Co., Ltd. started PCB production in Japan and it was at about this time that chloracne eruptions among workers were first reported. Slightly earlier (1953) incidents, however, were reported by Hara (1969), of changes attributable to PCBs in blood and urine findings among capacitor factory workers.

According to an Ad Hoc Committee sponsored by the Japanese Environmental Agency (1974), PCB concentrations in the air of a Japanese capacitor factory (where the major PCB constituent was "diphenyl pentachloride") were found to range from 0.37 to 6.75 $mg/m^3$; this was between the years 1953 and 1957, and prior to the



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dets when the aforementioned occupational exposure limit was promulgeted. The highest concentration was measured where PCBs were heated to drive out entrained/dissolved air. Admittedly, the analytical methodology for PCBs in those years was not as refined as that now in use, but it is thought likely (authors) that some employees were exposed continually et concentrations of several milligrams per cubic meter.

The Janepese literature is said to have conteined no further reports on PCB-in-air measurements until the spring of 1972, just prior to the suspension of their usage. The production of PCBs was diacontinued in Jepan in June 1972, and their importetion was diecontinued in the following September. At thet time Hasegawe et al., (1973) reported PCB measurements in the air of one PCB (Kanechlors 200-600) production plant (0.005-0.02 ppm), four cepacitor manufecturing plants (Kanechlor 300 used) (0.01-0.05 ppm), and one plant where PCBs (Kanechlor 300) were used as e heat exchange medium (0.002 ppm). Peak values of 0.17 ppm and 0.67 ppm, respectively, were found in a cepacitor plant "air-riddance" procese area and in e capecitor impregnating area where tank leakage had occurred. Estimated (presumably 8-hour workday) time-weighted average exposures of effected employees were 0.02-0.03 ppm. Airborne PCBs in the capacitor plants were reported to consist of 70-80% vapor for material corresponding to Kanechlor 200, and of perticulates lerger tha.. 0.1 micron for Kanechlor 300. The authors ettributed the 3/1 vapor/perticulate mix to eslective evaporation of low boiling components of Kanechlor 300.



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Although no measurements were available, Hasegawa et al., (1973) estimated that environmental conditions in carbonless copy paper manufacturing facilities (using Kanechlor 300) were roughly similar to those found in capacitor factories. Beginning in February 1971, PCBs were phased out as the microcapsular solvent in this process, being replaced by "SAS" or KMC-oil;" however, during the period from November 1972 to January 1973, when occupational health surveys were being performed relative to SAS or KMC-oil toxicity, PCB concentrations of 0.013-0.4 ppb were measured in the environments of these facilities. General environmental concentrations of PCBs at that time were less than 0.0005 ppb, and outdoor PCB concentrations around the facilities were 0.0043 ppb (Hasegawa et al., 1973) and 0.009 ppb (Sato and Hasegawa, 1974). In an office room where carbonless copy paper was used, an air concentration 1.1 mcg PCB/$m^3$ was measured, and in the carbonless copy paper storage area of a post office, PCB concentrations of 8.7-21.1 mcg/$m^3$ were detected (Nishiyama et al., 1973).

The Japanese Ministry, in its "Regulations for Physical Examinations for the Prevention of Disturbances due to Specific Chemical Substances," promulgated on October 14, 1971, advised PCB users to examine employees for dermal andhepatic "symptoms" and their anamnesis, "subjective symptoms" such as anorexia and asthenia, and urine urobilinogen, in the first (pre-employment?) physical examination, and to "survey" working conditions, blood tests and liver function tests in the "secondary" (periodic re-?)



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examination. A 1973 notice adivsed those users of PCB's as heat exchange fluids to perform comprehensive health surveillance (Environmental Agency of Japan 1974). Yamamoto (1974) reported "positive" findings in 37 employees of 323 (total employment = 706 in 51 establishments) who were given "special" physical examinations in 1971.

Hara (1969) reported a 20-30% incidence of dermal "symptoms," consisting of "distinctive hair follicles" in exposed areas such as the face, neck, and forearm, and pimple-like skin eruptions of the face and neck, among employees in a capacitor factory that had been in operation between 1953 and 1963 (pentechlorobiphenyls 1953-1957; Kanechlor 300, 1958-onward). No other, e.g., hepatic, dysfunction was observed, and no quantitative information was reported in the reference from which the preceding report was obtained; however, in a follow-up survey of workers in capacitor factories, Hara et al (1974) reported that by one year after the suspension of PCB usage skin findings had become milder.

Hasegawa et al., (1973) reported on a 1972 survey of capacitor factories, in which they noted that persons working in environments that contained 0.2-0.3 mg PCB's/$m^3$ (0.02-0.03 ppm) showed dermatolpgic ailments that included "brown chromodermatosis" of the dorsal joints of the hands and fingers and nail bed, and "acneform exanthema." Several cases of comedo or acneform exanthema of the jaw, back, and thighs were seen also. These signs were no longer observed one month after the cessation of the handling of PCB's.



MONS 219401

In a factory where Kanechlor 300 was employed as a heat exchange medium, the environmental concentration of PCB's was reported to be 0.02/mg/m$^3$, and no dermal manifestations were observed among its employees (Hasegawa et al., 1973). Nor did Hashimoto (1974) observe any abnormalities among 236 workers in such a facility, to whom he administered "examination centered around liver function tests."

In 1972, when Hasegawa et al., (1973) performed a health survey of workers in carbonless copy paper factories, i.e., 2 years after the use of PCBs in such processes had ceased, they noted no dermal effects, and no liver function, blood, or urine test abnormalities were seen, with the exception of "slight abnormalities in lipid metabolism". PCB levels in the blood of these workers were reported as "approximately 0.01-0.02 ppm," or what amounted to a decrease to 10% of the levels during the period of PCB usage. The authors' data indicated that PCB's collected from the work atmosphere had apparently degraded to a product containing one less chlorine atom than the average number of chlorine atoms in the PCB's handled in the factory, and that, conversely, the PCB's found in the workers' blood contained one more chlorine atom than did the PCB's handled in the factory, i.e., if the PCB's found in the workers' blood were inhaled at the workplace, it can be surmised that the di- and trichloro- biphenyls disappeared rapidly from the body, whereas the tetrachlorobiphenyl was metabolized slowly.



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Haeegawa et al., (1973) concluded that blood analyeis ie a feasible method for determining body burdens of PCBs, and "probably more ueaful than the technique presently used by extracting adipose tissue and using it for PCB;" urinalysis for "PCBs did not appeal to them as a viable monitoring method. They noted, however, that there was no correlation between length of employment (and presumably, duretion of expoeura) and the amount of PCBs accumulated in the blood, i.e., with continued exposure, PCB levels in the blood did not increase linearly; in light of the demonstrated slow excretion rate of PCBs from the body, the authors conjectured that some different biotransformative mechanism comes into play after blood PCB levels exceed approximately 1 ppm (perhaps fat etorage). Thie would seem to belie their confidence in blood analyeis as a reliable index of expoeure.

Kitamure et al., (1973) reported that the mean PCB level in the blood of 10 capacitor factory workare was 0.82 ppm (0.32-2.1 ppm) immediately after the cessation of PCB ueage, and 0.31 ppm 3 monthe later, when almoet all of the eubject were obeerved to have ekin aigns. It seems significant that the blood PCB concentrations obeerved in thie atudy were eaveral fold higher than those seen by Haeegawa et al., (1973). Kitamura et al., (1973) attributed the skin effects to PCB'e accumulated in the workare' bodies during tha period of PCB usage. No other abnormalities were reported.



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Sagami et al., (1973) reported the case of a housewife in whom they observed skin effects several years after she had left 6 years of employment in a capacitor factory. She was found to have PCB concentrations of up to 0.13 ppm in her blood and of 42 ppm in samples her subcutaneous adipose tissue; this contrasted sharply with the findings of Hara et al (1974) that the blood PCB levels of 3 workers who remained at the same factory had decreased to 0.05 ppm and below, one year after the cessation of PCB usage (froe levels of 0.05-0.3 ppm during the time of PCB usage (Hasegawa et al., 1973).

N.B. Some capacitor manufacturers in the U.S. use epoxide-type alkylating agents as additives to prolong the service life of PCBs. EPA sec. 308 responses from the General Electric Co., probably contain identifying data.

## VI. Air and Water Exposure to PCBs

As indicated previously, nominal human exposure to PCB's in the U.S. population may occur from air and water. Samples of ambient air were collected in suburban areas of Miami, Florida; Jackson, Mississippi; and Fort Collins, Colorado. Preliminary results (Kutz and Yang, 1975) for samples taken in April, May and June of 1975 show that PCBs were present at all locations. Although the data varied, the average concentrations at each of the three locations was approximately 100 nanograms per cubic meter. Initial



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identification of the PCBs indicated that they were most comparable to the Aroclor 1254 standard.

Dennie (1975) has reported that data gathered from monitoring activities of surface waters and bottom sediments of the major drainage basins of the United States indicate the widespread occurrence of PCBs in both surface water and bottom deposits. A preliminary assessment of PCB levels shows mean residue levels in water ranged from 0.01 to 0.05 mcg/liter. The 0.05 mcg/liter were found in the South Atlantic Slope and Eastern Gulf of Mexico drainage basin. In general, the lowest PCB residue levels were found in drainage basins west of the Mississippi.

Kleinert (1975) has summarized the work completed by the Wisconsin Department of Natural Resources to identify some PCB sources in the environment in Wisconsin. Effluents from cooling water in aluminum foundries contained PCBs ranging from 11.5 to 335 ppb. Investigations revealed the common source to be leaking hydraulic fluids containing PCBs which were used in die cast machines. Effluents from paper mills that recycle wastepapers has measureable discharges ranging from 0.01 to 25 ppb.

Snow samples (Kleinert, 1975) were collected early in 1975. Analysis of the snow melt water from Racine, Kenosha, Madison, and Milwaukee revealed concentrations from 0.17 to 0.24 ppb. The author concludes that these values suggest that fallout of PCBs from the air may be a principal source of PCBs entering the waters



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of the State.

Hesse (1975) indicates that similar to the studies in Wisconsin, not all industrial effluents tested in Michigan contain measureable PCB levels. In testing over 900 industrial samples, approximately 40 percent contained PCBs above the 0.1 mcg/liter laboratory sensitivity limit. Twenty-three percent of the industries tested had greater than 0.5 mcg/liter, 18 parcant greater than 1 mcg/liter, 6 percent greater than 10 mcg/liter and 2 percent greater than 100 mcg/liter.

Although much of the PCBs entering municipal waste treatment facilities are removed and become incorporated into the waste sludge, a sampling of 58 municipal wastewater treatment plant effluents throughout Michigan in 1973 showed an average concentration of 0.52 mcg/liter. Concentrations of PCBs are much higher in the sludges. The average for all plants was 15.6 mg/kg with individual values as high as 350 mg/kg. Since sewage sludges are commonly disposed of by incineration, spreading on agricultural land, or placing in landfills, the addition of PCBs to the environment is obvious.

## VII. Residues of PCB's in Human Tissue and Milk

Yobs (1972) has reported that 31.1 percent of 637 samples of human adipose tissue collected from the general population as part of the Human Monitoring Survey during 1971 were positive for PCBs in measurable amounts. These samples were collected in 18 states



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and the District of Columbia and positive samples were obtained from each of the sampling states and District. The percentage of PCB levels ranged from 34.2 percent non-detected, 33.3 percent less than 1 ppm, 27.3 percent 1-2 ppm and 5.2 percent greater than 5.2 ppm.

Kutz and Strassman (1975) have described the results of PCB monitoring during fiscal years 1973 and 1974 in which 35.1 percent and 40.3 percent, respectively, of the tissues collected contained levels of 1 ppm or more of PCBs on a net-weight basis. Analysis of the tissue revealed that the compounds found in adipose tissue were most comparable to those prevalent in Aroclor 1254 and Aroclor 1260. Additional analysis indicated that the most frequently encountered PCB residues were penta-, hexa-, and heptachlorobiphenyl compounds.

Residues of PCBs have also been detected in a study of human milk collected in Colorado where 8 or 40 samples contained residues ranging from 40 to 100 ppb (Savage et al., 1973).

A study of adipose tissue samples collected at autopsy from Canadians (Grant et al., 1975) indicates that the majority of Canadians have adipose tissue residues of 1 to 2 mg/kg of PCB. All adipose tissues had detectable levels of PCBs and 30 percent of the samples had PCB residues greater than 1 mg/kg with a range



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of 0.11 to 6.60 mg/kg. PCB residues in human milk from Ontario residents were found to be approximately 1 mg/kg on a fat basis.



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## HUMAN EXPOSURE TO THE POLYBROMINATED BIPHENYLS

### I. Introduction and Background

The polybrominated biphenyls (PBBs) in this report refers to either Firemaster BP-6 or hexabromobiphenyl manufactured by Michigan Chemical Corporation for use as a flame retardant for thermoplastics. This product is a mixture of brominated biphenyls with an average bromine content equivalent to about six bromine atoms per biphenyl molecule. Firemaster BP-6 is a mixture of the following brominated biphenyls[1]:

Tetrabromobiphenyl 2.0%

Pentabromobiphenyl 10.6%

Hexabromobiphenyl 62.8%

Heptabromobiphenyl 13.8%

Other bromobiphenyls 11.4%

The Michigan Chemical Corporation has stated that to their knowledge Firemaster BP-6 is the only polybrominated biphenyl produced in commercial quantity in the U.S. Their production estimates are:

| Year | Pounds |
|---|---|
| 1970 | 20,000 |
| 1971 | 200,000 |
| 1972 | 2,300,000 |
| 1973 | 3,900,000 |
| 1974 (Projected) | 4,800,000 |



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Firemaster BP-6 has been used as a flame retardant in the manufacture of typewriter, calculator and microfilm reader housings, radio and TV parts, miscellaneous small automotive parts and small parts for electrical applications. The use of Firemaster BP-6 has been restricted to those applications where the end-use product is not exposed to either animal feed or food and there is no known use in flame retarding fabrics where human exposure would occur (1).

The ultimate disposition of Firemaster BP-6 upon burial is uncertain. The Michigan Chemical Corporation has stated that in their opinion this material will eventually undergo oxidative/ biological degradation forming carbon dioxide, water and bromide ion (1).

II. Human Exposure

In October of 1973, adverse health effects were observed in cattle in several dairy herds in the State of Michigan. At that time, the cattle refused to eat manufactured feed; milk production decreased; there was a loss in body weight and the cattle developed abnormal hoof growth with lameness; cattle and swine aborted; and farmers reported the inability to breed heifers after they consumed feed manufactured by Farm Bureau Services. A herd of some 100 head of cattle sent to slaughter during this time period. exhibited enlarged livers.



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Analysis of samples of the suspected feed by laboratories of the U.S. Department of Agriculture at Beltsville, Maryland, revealed that the feed was contaminated with a flame retardant chemical, hexabrominated biphenyl. Subsequent investigation revealed that the Michigan Chemical Corporation manufactured magnesium oxide, a dairy feed supplement sold under the tradename Nutrimaster, and a flame retardant, hexabrominated biphenyl, sold under the tradename Firemaster BP-6. Both of these products were distributed in brown paper bags with either the name Nutrimaster or Firemaster stenciled across the top of the bag. When the top of the bag was torn off and discarded, identification was essentially lost.

As the result of a mix-up in bags, Firemaster BP-6 was mixed with animal feed in place of the Nutrimaster, apparently in the same proportion of use for the Nutrimaster. It appears that three kinds of feed were initially involved in this episode with PBB levels as follows:

| Feed No. | PBB (ppm) |
|---|---|
| 405 | 2.4 |
| 410 | 1,790 |
| 407 | 4,300 |

Samples of milk collected from individual farms soon after the PBB was identified as the contaminant ranged from 2.8 ppm on



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a fat basis to 270.5 ppm on a fat basis. Other products seized and destroyed included:

| | |
|---|---|
| Butter | Range 1-2 ppm |
| Cheese | 1.4 - 15.0 ppm |
| Canned milk | 1.15 - 1.62 ppm |

It has been estimated that between the onset of contamination in the fall of 1973 and the establishment of the quarantine of affected herds and flocks in the spring of 1974 over 10,000 Michigan residents have been exposed to PBB through the consumption of contaminated milk, meat and other dairy products. A considerable amount of variation in exposure has proabably occurred in both length of exposure and levels of exposure. As a group, the farm family members have been at greatest risk followed by those individuals who purchased dairy products from contaminated farms on a regular basis.

In order to determine whether or not persons exposed to PBB-contaminated products had suffered any acute adverse health effects, the Michigan Department of Public Health undertook a series of studies in the summer and fall of 1974. Study participants for the exposed group were dairy farm residents from farms which had been quarantined by the Michigan Department of Agriculture. The exposed subjects were limited to those who had lived or worked on the quarantined dairy farms for more than six months since May



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of 1973. Non-exposed subjects were randomly selected from a list of dairy producers in the same geographical area where farms had not been quarantined.

A total of 298 persons were interviewed in the study and physical examinations and/or blood samples were obtained for 110 persons in the exposed group and 104 persons from the control group.

The Michigan Department of Public Health has reported (1975) that responses to a set of 24 specific medical conditions revealed that none of the health complaints occurred consistently in either of the study groups. Statistical analysis showed that none of the listed complaints was significantly more frequent in those persons with the highest PBB levels when compared to other study subjects. Physical examinations of adults and children showed no unusual abnormalities of the heart, liver, spleen or nervous system. Urinalyses and complete blood counts did not reveal a significant excess of unusual abnormalities related to exposure or PBB levels.

These studies showed that blood levels of PBB were significantly higher in the study subjects from quarantined farms as compared to those from the non-quarantined farms; although some subjects from the farms showed low PBB blood levels (Table 18).

Several exposed females delivered normal babies without complication. Tests showed concentrations of PBB in breast milk to be considerably higher than that found in paired blood plasma (Table 19).



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Table 18

Distribution of PBB Blood Levels, Michigan, 1974

| PBB Blood Levels (ppm) | Quarantined Farms | | | | Nonquarantined Farms | | | |
|---|---|---|---|---|---|---|---|---|
| | Adults | | Children | | Adults | | Children | |
| | No. | % | No. | % | No. | % | No. | % |
| 0 | 3 | 3.7 | - | - | 21 | 28.4 | - | - |
| 0.002 - 0.019 | 43 | 52.4 | 8 | 28.6 | 52 | 70.3 | 29 | 96.7 |
| 0.020 - 0.090 | 19 | 23.2 | 10 | 35.7 | 1 | 1.4 | 1 | 3.3 |
| 0.100 - 0.490 | 11 | 13.4 | 3 | 10.7 | 0 | 0 | 0 | 0 |
| 0.500 - 2.260 | 6 | 7.3 | 7 | 25.0 | 0 | 0 | 0 | 0 |
| TOTAL | 82 | 100.0 | 28 | 100.0 | 74 | 100.1 | 30 | 100.0 |

Table 19

Comparison of PBB Concentrations In Human Breast Milk and Blood Plasma

| Date | PBB Levels (ppm) | |
|---|---|---|
| | Breast Milk | Blood Plasma |
| 12/74 | 10.800 | .082 |
| 6/74 | 22.700 | .252 |
| 10/74 | 1.800 | .014 |
| 3/75 | .210 | .003 |
| 3/75 | 92.660 | 1.068 |

Paired samples of adipose tissue and blood were collected in a group of 13 individuals entering the hospital for surgical procedures. The concentration of PBB in adipose tissue ranged from 61 to 370 times the PBB value found in blood plasma with an average ratio of 175 to 1 (Table 20).



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TABLE 20

Comparison of PBB Concentrations in
Human Adipose Tissue and Blood Plasma

| Date | PBB Levels (ppm) Adipose Tissue | Blood Plasma |
|---|---|---|
| 11/4 | .410 | .002 |
| 6/74 - 1/75 | 1.400 | .005 |
| 3/75 | 3.000 | .012 |
| 3/75 | 174.000 | 1.068 |
| 3/75 | .248 | .002 |
| 3/75 | .274 | .003 |
| 4/75 | .177 | .003 |
| 4/75 | .152 | .003 |
| 4/75 | 1.140 | .004 |
| 4/75 | .808 | .004 |
| 4/75 | .530 | .007 |
| 4/75 | .210 | .003 |
| 5/75 | 1.110 | .003 |

Reports of health complaints such as numbness, stomach pain, headache, fatigue and anxiety continue to be reported in the various newspapers in Michigan. Reports have also appeared in the press that several physicians in Michigan have reported abnormal liver function tests in patients exposed to PBB. Attempts to verify these reports with physicians have been unsuccessful.



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Bache, C. A. and Lisk, D. J. (1973), A Qualitative Method for Detecting Hydroxylated Metabolites of Polychlorinated Biphenyls, Bull Environ Contam Toxicol 9:315-317.

Bailey, S., and Bunyan, P. J. (1972), Interpretation of Persistence and Effects of Polychlorinated Biphenyls in Birds, Nature 236:34-36.

Barsotti, D. A., and Allen, J. R. (1975), Effects of Polychlorinated Biphenyls on Reproduction in the Primate, Fed Proc 34:338.

Bastomsky, C. H. (1974), Effects of a Polychlorinated Biphenyl Mixture (Aroclor 1254) and DDT on Biliary Thyroxin Excretion in Rats, Endocrinol 95:1150-1155.

Bastomsky, C. H., Solymoss, B., Zsigmond, G., and Wyse, J. M. (1975), Mechanism of Polychlorinated Biphenyl-Induced Hypobilirubinaemia, Clin Chim Acta 61:171-174.

Bastomsky, C. H., and Wyse, J. M. (1975), Enhanced Thyroxine Metabolism Following Cutaneous Application of Microscope Immersion Oil, Res Commun Chem Pathol Pharmacol 10:725-733.

Bauer, H., Schulz, K. H., and Spiegelberg, U. (1961), Berufliche Vergiftungen bei der Herstellung von Chlorophenol Verbindungen, Arch Gewerbepath Gewerbehyg 18:538-555.

Bauer, H., Schulz, K. H., and Spiegelberg, U. (1961), Occupational Intoxications in Manufacturing Chlorophenol Compounds, Arch Gewerbepath Gewerbehyg 18:538-555.

Baxter, R. A., Gilbert, P. E., Lidgett, R. A., Meinprize, J. H., and Vodden, H. A. (1975), The Degradation of PCBs by Microorganisms, Sci of the Total Environ 4:53-61.

Beazhold, F. L. and Stout, V. F. (1973), The Use and Effect of Mixed Standards on the Quantitation of Polychlorinated Biphenyls. Bull Environ Contam and Toxicol 10:10-15.

Berg, O. W., Diosady, P. L., and Rees, G. A. V. (1971), Column Chromatographic Separation of Polychlorinated Biphenyls from Chlorinated Hydrocarbon Pesticides and Subsequent Gas Chromatographic Quantitation in Terms of Derivatives, Report of Ontario Water Resources Commission.

Berlin, M., Gage, J., and Holm, S. (1975), The Distribution and Metabolism of 2,4,5,2',5'-Pentachlorobiphenyl, Arch Environ Health 30:141-147.

Bickers, D. R., Eiseman, J., Kappas, A., and Alvarez, A. P. (1975), Microscope Immersion Oils. Effects of Skin Application on Cutaneous and Hepatic Drug Metabolizing Enzymes, Biochem Pharmacol 24:779-783.

Bickers, D. R., Harber, L. C., Kappas, A., and Alvarez, A. P. (1972), Polychlorinated biphenyls. Comparative Effects of High and Low Chlorine Containing Aroclors on Hepatic Mixed Function Oxidase, Res Comm Chem Pathol Pharm 3:505-512.

Bickers, D. R., Kappas, A., and Alvarez, A. P. (1974), Differences in Inducibility of Cutaneous and Hepatic Drug Metabolizing Enzymes and Cytochrome P 450 by Polychlorinated Biphenyls and 1,1,1-Trichloro-2,-bis (p-Chlorophenyl) Ethane (DDT), J Pharmacol Exp Ther 188:300-309.

Binns, F. and Suschitzky, H. (1971), Polyhalogenoaromatic Compounds. Part XX. Some Reactions of Decachlorobiphenyl, J Chem Soc (C) 1913-1917.

Biocca, M. (1975), Toxicology of Selected Symmetrical Hexachlorobiphenyls. Biological Responses in Chickens and Mice, National Conference on Polychlorinated Biphenyls, Chicago.

Biocca, M., Moore, J. A., Gupta, B. N., McKinney, J. D. (1975), Toxicology of Selected Symmetrical Hexachlorobiphenyl Isomers. Biological Responses in Chicks and Mice, Nat Conf on Polychlorinated Biphenyls, Chicago.

Bitman, J., and Cecil, H. C. (1970), Estrogenic Activity of DDT Analogs and Polychlorinated Biphenyls, J Agric Food Chem 18:1108-1112.

Block, W. D. and Cornish, H. (1959), Metabolism of Biphenyl and Chlorobiphenyl in the Rabbit, J Biol Chem 234:3301-3302.

Bowes, G. W., Mulvihill, M. J., Simoneit, B. R. T., Burlingame, A. L., and Risebrough, R. W. (1975a), Identification of Chlorinated Dibenzofurans in American Polychlorinated Biphenyls, Nature 256:305-307.

Bowes, G. W., Mulvihill, M. J., DeCamp, M. R., and Kende, A. S. (1975b), Gas Chromatographic Characteristics of Authentic Chlorinated Dibenzofurans; Identification of Two Isomers in American Japanese Polychlorinated Biphenyls, J Agr Food Chem 23:1222-1223.

Bowes, G. W., et al. (1975), Identification of Chlorinated Dibenzofurans in American Polychlorinated Biphenyls, Nature 256:305-307.

Brandt, I. and Ullberg, S. (1973), Distribution of Hexa- and Octachloro Biphenyl in Mice and Quails, Report to the National Swedish Environment Protection Board.

Bruckner, J. V., Khanna, K. L., and Cornish, H. H. (1973), Biological Responses of the Rat to Polychlorinated Biphenyls. Toxicol Appl Pharmacol 24:434-438.

Bruckner, J. V., Khanna, K. L., and Cornish, H. H. (1974), Effect of Prolonged Ingestion of Polychlorinated Biphenyls on the Rat, Food Cosmet Toxicol 12:323-330.

Burse, V. W., Kimbrough, R. D., Villanueva, E. C., Jennings, R. W., Linder, R. E., Sovocool, G. W. (1974), Polychlorinated Biphenyls, Storage, Distribution, Excretion, and Recovery: Liver Morphology After Prolonged Dietary Ingestion. Arch Environ Health 29:301-307.

Burke, J. A. (1972), Report on Chlorinated Insecticides, JAOAC 55:284-287.

Bush, B., Tumasonis, C. F., Baker, F. D. (1974), Toxicity and Persistence of PCB Homologs and Isomers in the Avian System, Arch Environ Contam and Toxicol 2(3):195-212.

Byrne, J. J., Rainaka, P., Ringer, R. K., and Aulerich, R. (1975), Influences of Polychlorinated Biphenyl Mixture Aroclor 1254 Administration on Reproduction and Thyroid Function in Mink Mustela-vison, Fed Proc 34:321.

Calandra, J. C.: Summary of Toxicological Studies on Commercial PCBs. Two-year Chronicoral Toxicity Study of Aroclor 1242, 1254, 1260 in beagle dogs. Monsanto Company, St. Louis, MO. 63116.

Cecil, H. C., Harris, S. J., Bitman, J. and Fries, G. F. (1973), Polychlorinated Biphenyl-Induced Decrease in Liver Vitamin A in Japanese Quail and Rats, Bull Environ Contam Toxicol 9:179-185.

Chem, P. R., Mehendale, H. M., and Fishbein, L. (1973b), Effect of Two Isomeric Tetrachlorobiphenyls on Rats and Their Hepatic Enzymes, Arch Environ Contam Toxicol 1:36-47.

Chem, P. R., McKinney, J. D., and Matthews, H. B. (1975), Metabolism of 2,4,5,2'5'-Pentachlorobiphenyl in the Rat: Qualitative and Quantitative Aspects, in preperation.

Chen, T. S., and Dubois, K. P. (1973a), Enzyme Inducing Effect of Polychlorinated Biphenyls, Toxicol Appl Pharmacol 26:504-512.

Chesney, C. F., and Allen, J. R. (1974), Oxidative Phosphorylation and Respiration by Liver Mitochondria from Polychlorinated Biphenyl Intoxicated Rats, Biochem Pharmacol 23:1577-1582.

Collins, G. B., Holmes, D. C., and Jackson, F. J. (1972), The Estimation of Chlorobiphenyls, J Chromatogr 71:443-449.

Combs, G. F., Jr., Cantor, A. H., and Scott, M. L. (1975a), Effects of Dietary Polychlorinated Biphenyls on Vitamin E and Selenium Nutrition in the Chick, Poult Sci 54:1143-1152.

Cook, J. W. (1972), Some Chemical Aspects of Polychlorinated Biphenyls (PCBs), Environ Health Perspec 1:3-13.

Corbett, T. H., Beaudoin, A. R., Cornell, R. G., Anver, M. R., Schumacher, R., Endres, J. and Szwabowska, M. (1975), A Toxicity of Polybrominated Biphenyls (Firemaster BP-6) in Rodents, Environ Research 10:390.

Crosby, D. (1969), Experimental Approaches to Pesticide Photodecompositon, Residue Rev 25:1-12.

Crosby, D. B. and Moilanen, K. W. (1973), Photodecomposition of Chlorinated Biphenyls and Dibensofurans, Bull Envir Contam and Tox 10:372-377.

Crow, K. D. (1970), Chloracne, Trans St Johns Hosp Dermat Soc 56:79-99.

Curley, A., Burse, V. W., Jennings, R. W., Villanueva, E. C., and Kimbrough, R. D. (1975), Evidence of Tetrachlorodibenzofuran (TCDF) in Aroclor 1254$^{R}$ and the Urine of Rats Following Dietary Exposure to Aroclor$^{R}$, Bull Envir Contam and Tox 14:153-158.

Curley, A., Burse, V. W., Grim, N. E., Jennings, R. W., and Linder, R. E., (1971), Polychlorinated Biphenyls: Distribution and Storage in Body Fluids and Tissues of Sherman Rats, Environ Res 4:481-495.

Cutkomp, L. K., Yap, H. H., Desaiah, H. H., and Koch, R. B. (1972), The Sensitivity of ATP-ases to Polychlorinated Biphenyls, Environ Health Perspect 1:165-168.

Daly, J. W., Jerina, D. M., and Witkop, B. (1972), Arene Oxides and the NIH Shift: The Metabolism, Toxicity, and Carcinogenicity of Aromatic Compounds, Experiente 28:1129-1149.

Davis, P. W., Friedhoff, J. M., and Wedemeyer, G. A. (1972), Organochlorine Insecticide, Herbicide, and Polychlorinated Biphenyl (PCB) Inhibition of $Na^{+}K^{+}$ -ATPase in Rainbow Trout, Bull Environ Contam Toxicol 8:69-72.

deCrauw, R. (1931), The Principle of Induced Alternating Polarity in Connection with the Reactions of Derivatives of p-Dichlorobenzene and Other Compounds with Sodium Methylate, Rec Trav Chem 50, 753-792.

deFreitas, A. S. and Norstrom, R. J. (1974), Turnover and Metabolism of PCBs in Relation to their Chemical Structure and the Movement of Lipids in the Pigeon, Can J Physiol 52:1080-1094.

Deming, D. V. M. (1975), Summary of Team Evaluation Effort in Herds Quarantined for Polybrominated Biphenyls (PBB) in a Report from the State Senate Special Investigating Committee (Senator John A. Welborn, Chairman). The Contamination Crisis in Michigan, July.

Dennie, D. S. (1975), Polychlorinated Biphenyls in the Surface Waters and Bottom Sediments of the Major Drainage Basins of the United States, Nat Conference on Polychlorinated Biphenyls, Chicago.

Desaiah, D., Cutkomp, L. K., Yap, H. H., and Koch, R. B. (1972), Inhibition of Oligomycin-sensitive and Insensitive Magnesium Adenosine Triphosphatase Activity in Fish by Polychlorinated Biphenyls, Biochem Pharmacol 21:857-865.

DET-DO, DCT, DCH (Feb. 9, 1976), Polybrominated Biphenyl Methodology, Food and Drug Administration Laboratory Information Bulletin 1705D.

deVos, R. H. and Peet, E. W. (1971), Thin Layer Chromatography Polychlorinated Biphenyls, Bull Environ Contam Toxicol 6:164-170.

Dikshith, T. S. S., Rockwood, W., Abraham, R., and Coulston, F. (1975), Effects of a Polychlorinated Biphenyl (Aroclor 1254) on Rat Testis, Exp Mol Pathol 22:376-385.

Drinker, C. K., Warren, M. F., and Bennett, G. A. (1937), The Problem of Possible Systemic Effects from Certain Chlorinated Hydrocarbons, J Ind Hyg Toxicol 19(7):282-311.

Ecobichon, D. J., and Comeau, A. M. (1975), Isomerically Pure Chlorobiphenyl Congeners and Hepatic Function in the Rat: Influence of Position and Degree of Chlorination, Toxicol Appl Pharmacol 33:94-105.

Ecobichon, D. J., and Mackenzie, D. O., (1974a), Uterotropic Activity of Commercial and Isomerically Pure Chlorobiphenyls in the Rat, Res Commun Chem Pathol Pharmacol 9, 85-95.

Edwards, J. E. and White, J. (1951), Pathology Changes with Special Reference to Pigmentation and Classification of Hepatic Tumors in Rats Fed P-dimethylamine Acobenzene (butter yellow), J Natl Cancer Institute 2:157-183.

Edwards, R. (1974), Some Factors in the Separation of Polychlorobiphenyls (PCBs) from Organochlorine Pesticides by Column Chromatography Combined with Gas-Liquid Chromatography, Pestic Sci 5:293-304.

Erney, R. D. (1975), Confirmation of Polybrominated Biphenyl Residues in Animal Feeds and Dairy Products Using an Ultraviolet Irradiation Gas-Liquid Chromatographic Technique, JAOAC 58:1202-1205.

Farber, T. M. and Baker, A. (1974), Microsomal Enzyme Induction by Hexabromobiphenyl, Soc Toxicol Meeting, Washington, DC, March 10-14.

Fehringer, N. V. and Westfall, J. E. (1971), Separation and Identification of DDT Analogs in the Presence of Polychlorinated Biphenyl Compounds by Two-Dimensional Thin Layer Chromatography, J Chromatogr 57:397-405.

Fehringer, N. V. (1974), TLC of Polybrominated Biphenyls in Animal Feeds and Dairy Products, Food and Drug Administration Laboratory Information Bulletin, No. 1705.

Fehringer, N. V. (1975), Determination of Polybrominated Biphenyl Residues in Dry Animal Feeds, JAOAC 58:1206-1210.

Ficsor, G. and Wertz, G. F. (1976), Polybrominated Biphenyl Nonteratogenic C-mitosis Synergist in the Rat, Pres. at the 7th Annual meeting of the Environmental Mutogen Society, Atlanta, GA.

Finsterwalder, C. E. (1974), Collaborative Study of the Determination of Polychlorinated Biphenyls in Paperboard, JAOAC 57:518-521.

Firestone, D., Ress, J., Brown, N. L., Barron, R. P., and Damico, J. N. (1972), Determination of Polychlorodibenzo-p-dioxins and Related Compounds in Commercial Chlorophenols, JAOAC 55:85-92.

BIBLIOGRAPHY


AD HOC COMMITTEE OF JAPAN PUBLIC HEALTH ASSOCIATION. (1974), Environmental Health Criteria for Polychlorinated Bi- and Terphenyls. Environmental Agency of Japan/WHO Environmental Health Criteria Programme.

Aftosmis, J. G., Culik, R., Lee, K. P. (1972), Toxicology of Brominated Biphenyls: I. Oral Toxicity and Embryotoxicity, Toxicol Appl Pharmacol 22:316.

Aftosmis, J. G., Culik, R., Lee, K. P., Sherman, H., and Waritz, R. S. (1972a), The Toxicology of Brominated Biphenyls: I. Oral Toxicity and Embryotoxicity, Presented at the Society of Toxicology Meeting in Williamsburg, VA. Toxicol Appl Pharmacol 22:316.

Aftosmis, J. G., Dashiell, O. C., Griffith, F. D., Hornberger, C. S., McDonnell, M. E., Sherman, H., Tayfun, F. O., and Waritz, R. S. (1972b), Toxicology of Brominated Biphenyls: II. Skin, Eye, and Inhalation Toxicity and an Acute Test for Evaluating Hepatotoxicity and Accumulation in Body Fat, Toxicol Appl Pharmacol 22:316-317.

Albro, P. W. and Fishbein, L. (1972), Intestinal Absorption of Polychlorinated Biphenyls in Rats, Bull Envir Contam Toxicol 8:26-31.

Allen, J. R. and Norback, D. H. (1973), Polychlorinated Biphenyl and Triphenyl Induced Gastric Mucosal Hyperplasia in Primates, Science 179:498-499.

Allen, J. R., Carstens, L. A., and Barsotti, D. A. (1974), Residual Effects of Short-term, Low-level Exposure of Nonhuman Primates to Polychlorinated Biphenyls, Toxicol Appl Pharmacol 30:440-451.

Allen, J. R., Norback, D. H., and Hsu, I. C. (1974), Tissue Modifications in Monkeys as Related to Absorption, Distribution, and Excretion of Polychlorinated Biphenyls, Arch Environ Contam Toxicol 2:86-95.

Allen, J. R. (1975), Responses of the Nonhuman Primate to Polychlorinated Biphenyl Exposure, Fed Proc 34:1675-1679.

Allen, J. R., Carstens, L. A., Abrahamson, L. J., and Marler, R. J. (1975), Responses of Rats and Nonhuman Primates to 2,5,2',5'-Tetrachlorobiphenyl, Env Res 9:265-273.


MONS 219422

Alvares, A. P., Bickers, D. R., and Kappas, A. (1973), Polychlorinated Biphenyls. New Type of Inducer of Cytochrome P 448 in the Liver, Proc Nat Acad Sci 70:1321-1325.

Alvares, A. P., and Kappas, A. (1975), Induction of Aryl Hydrocarbon Hydroxylase by Polychlorinated Biphenyls in the Retroplacental Unit and Neonatal Livers During Lactation, FEBS Lett 50:172-174.

Andersson, K., Norstrom, A., Rappe, C., Rasmuson, B., and Swahlen, H. (1974), Photochemical Degradation of PCB, PBB and Other Flame Retardants, 34th International Congress of Pesticide Chemistry, Helsinki, Finland.

Araki, Y. (1974), Comparison of Hepatic Enzyme-Inducing Activities of Chlorinated Dibenzofuran and Chlorinated Dibenzodioxin, Fukuoka-Igaku-Zasshi 65:61-64.

Armour, J. A. and Burke, J. A. (1970), Method for Separating Polychlorinated Biphenyls from DDT and its Analogs, JAOAC 53:761-768.

Armour, J. A. (1972), Gas Chromatographic Data for Polychlorinated Biphenyl Components in Six Aroclors, J Chromatogr 72:275-282.

Armour, J. A. (1973), Quantitative Perchlorination of Polychlorinated Biphenyls as a Method for Confirmatory Residue Measurement and Identification, JAOAC 56:987-993.

Armour, J. A. and Burke, J. A. (1971), Behavior of Chlorinated Naphthalenes in Analytical Methods for Organochlorine Pesticides and Polychlorinated Biphenyls, JAOAC 54:175-177.

Aulerich, R. J., Ringer, R. K., and Iwamoto, S. (1973), Reproductive Failure and Mortality in Mink Fed on Great Lakes Fish, J Reprod Fert Suppl 19:365-376.

AUSTRALIAN NATIONAL HEALTH AND MEDICAL RESEARCH COUNCIL. (1970), Atmospheric Contaminants NHJMRC, Canberra.

Ax, R. L. and Hansen, L. G. (1975), Effects of Purified PCB Analogs on Chicken Reproduction, Poultry Science 54(3):895-900.

Ax, R. L., Miller, W. L., and Hansen, L. G. (1976), Toxicity Comparisons of Two Pentachlorobiphenyls. Abstract for presentation at American Society of Animal Science - Midwestern Section.

Fishbein, L. (1974), Toxicity of Chlorinated Biphenyls, Ann Rev of Pharm 14:139-156.

Fishbein, L. (1972), Chromatographic and Biological Aspects of Polychlorinated Biphenyls, J Chromatogr 68, 345-426.

Flick, D. F., O'Dell, R. S., and Childs, V. A. (1975), Studies of the Chick Edema Disease. Similarity of Symptoms Produced by Feeding Chlorinated Biphenyl, Poultry Sci 44, 1460-1465.

Food and Drug Administration Pesticide Analytical Manual, Vol. I Ed. 1968; rev.: July 1969, July 1970, April 1971, Jan 1972, Sept 1972, June 1973, March 1975, July 1975.

Food and Drug Administration, Private Communication (June 8, 1971), Report of Sample No. 4 1970 Interlaboratory Quality Assurance Program (Pesticides in Fish).

Food and Drug Administration, Private Communication (Jan 12, 1973), Sample No. 1 FY/73 Interlaboratory Performance Assurance Program (PCB and Dieldrin in Baby Food).

Friend, M. and Trainer, D. O. (1970), Polychlorinated Biphenyl: Interaction with Duck Hepatitis Virus, Science 170:1314-1316.

Fries, G. F., Marrow, G. S., and Gordon, C. H. (1973a), Long-term Studies of Residue Retention and Excretion by Cows Fed a Polychlorinated Biphenyl (Aroclor 1254), J. Ag Fd Chem 21:117-121.

Fries, G. F., Lillie, R. J., Cecil, H. C., and Bitman, J. (1973b), Retention of Excretion of Polychlorinated Biphenyl Residues by Laying Hens, 165th Meeting of American Chemical Society (Abstract).

Fries, G. F., Marrow, G. S. (1975), Excretion of Polybrominated Biphenyls into the Milk of Cows, Journal of Dairy Science 58:947-951.

Fries, G. S., Cecil, H. C., Bitman, J., and Lillie, R. J. (1976), Retention and Excretion of Polybrominated Biphenyls by Hens, Bull Environ Contam Toxicol, in press.

Fries, G. F., Marrow, G. S., Dotering, C. N., Frewitt, L. R., and Cook, R. M. (1975), Distribution of Polybrominated Biphenyl Residues in Tissues of Dairy Cattle, J Diary Sci 58:764 (Abstract).

Fumiko, K. (1972), Estradiol-potentiating Action of PCB (Polychloro-Biphenyl) of PCB (Polychlorobiphenyl), Fukucku-Igaku-Zasshi 63:374-377.

MONS 219424

Gardner, A. M., Chen, J. T., Roach, A. C., and Ragelis, E. P. (1973), Polychlorinated Biphenyls: Hydroxylated Urinary Metabolites of 2,5,2',5'-Tetrachlorobiphenyl Identified in Rabbits, Biochem Biophys Res Comm 55:1377-1384.

Garthoff, L. H., Friedman, L., Hurley, N., Farber, T. M., Patara, E. L., Locke, K. K., Green, S., Sobotka, T. J., Moreland, F., Keys, J., Scalera, J., Rothlein, J., Taylor, M. J., Story, G., Graham, C. H., Marks, E., Cerra, F., and Sporn, E. M. (1975), Biochemical and Cytogenetic Effects Caused by Ingestion of Aroclor 1254 or Firemaster BP-6. Testimony of Dr. A. C. Kolbye at Polybrominated Biphenyl Conference, Michigan Dept. Agric., Lansing, Michigan, May 29.

Glam, C. S. and Wong, M. K. (1972), Problems of Background Contamination in the Analysis of Open Ocean Biota for Chlorinated Hydrocarbons, J. Chromatogr 72:283-292.

Goldstein, J. A., Hickman, P., and Jue, D. L. (1974), Experimental Hepatic Porphyria Induced by Polychlorinated Biphenyls, Toxicol Appl Pharmacol 27:437-448.

Goldstein, J. A., Hickman, P., and Jue, D. L. (1975), Experimental Hepatic Porphyria Induced by Polychlorinated Biphenyls, Toxicol Appl Pharmacol 27:437-448.

Goldstein, J. A., Hickman, P., Bursa, V. W., Bergman, J. (1975), Comparative Study of Two Polychlorinated Biphenyl Mixtures (Aroclor 1242 and 1016) Containing 42% Chlorine on Induction of Hepatic Porphyria and Drug Metabolizing Enzymes, Toxicol Appl Pharmacol 32:461-473.

Goldstein, J. A., McKinney, J. D., Lucier, G. W., Hickman, P., Bergman, H., and Moore, J. A., Toxicological Assessment of Hexachlorobiphenyl Isomers and 2,3,7,8-Tetrachlorodibenzofuran in Chicks. II. Effects on Drug Metabolism and Porphyrin Accumulation, Toxicol Appl Pharmacol, in press.

Good, C. M. and Pansky, N. (1943), Halowax Acne ("Cable Rash"), Arch Dermatol Syphilol 48(3):251-257.

Goto, M., Sakaguchi, K. and Ogawa K. (1969), Hydropericardium Assay of Rice Oil which Caused Yusho and of Kanechlor 400 in Chickens, Fukuoka Acta Medica 60:131-136.

Grant, D. L., Phillips, W. E. J., and Villeneuve, D. C. (1971a), Metabolism of a Polychlorinated Biphenyl (Aroclor 1254) Mixture in the Rat, Bull Environ Contam Toxicol 6:102-112.

Grant, D. L., Villeneuva, D. C., McCully, K. A., and Phillips, W. E. J. (1971b), Placental Transfer of Polychlorinated Biphenyls in the Rabbit, Environ Physiol 1:61-66.

Grant, D. L., and Phillips, W. E. J. (1974), The Effect of Age and Sex on the Toxicity of Aroclor 1254, Bull Environ Contam Toxicol 12:145-152.

Grant, D. L., Mes, J., and Frank, R. (1975), PCB Residues in Human Adipose Tissue and Milk, Nat Conference on Polychlorinated Biphenyls, Chicago.

Green, S., Sauro, F. M., and Friedman, L. (1975), Lack of Dominant Lethality in Rats Treated with Polychlorinated Biphenyls, Fd Cosmet Toxicol 13:507-510.

Green, S., Carr, J. V., Fahner, K. A., and Oswald, E. J. (1975), Lack of Cytogenetic Effects in Bone Marrow Spermatogonial Cells in Rats Treated with Polychlorinated Biphenyls, Bull Environ Contam Toxicol 13:14-22.

Greenberg, L., Mayers, M. R., and Smith, A. R. (1939), The Systemic Effects Resulting from Exposure to Certain Chlorinated Hydrocarbons, J Ind Hyg Toxicol 21(2):29-38.

Grote, W., Schmoldt, A., Benthe, H. F. (1975), Hepatic Porphyrin Synthesis in Rats after Pretreatment with Polychlorinated Biphenyls, Acta Pharmacol Toxicol 36, 215-224.

Gregory, N. L. (1968), J Chem Soc 13:295.

Hannan, E., Bills, D., and Herring, J. (1973), Analysis of Polychlorinated Biphenyls by Gas Chromatography and Ultraviolet Irradiation, J Agric Food Chem 21:87-90.

Hansell, M. M. and Ecobichon, D. J. (1974), Effects of Chemically Pure Chlorobiphenyls on the Morphology of Rat Liver, Tox and Appl Pharm 28:418-427.

Hansen, L. G., Byerly, C. S., Metcalf, R. L., Bevill, R. F. (1975), Effect of a Polychlorinated Biphenyl Mixture on Swine Reproduction and Tissue Residues, Am J Vet Res 36(1):23-26.

Hara, I. (1969), Health Supervision of Workers Exposed to Chlorobiphenyl in an Electric Condenser Factory, Proc Osaka Prefect Inst Pub Health Ed., Industry Health 7:26-31. In Japanese.

Hara, I., Harada, A., Kumura, S., Endo, T. and Kawano, K. (1974), Follow-up Health Examination in an Electric Condenser Factory after Cessation of PCBs Usage (1st Report), Japan J Ind Health 16:365-366. In Japanese.

Hasegawa, J., Sato, M., and Tsuruta, H. (1973), Report on Survey of Work Area Environmental where PCB is Handled and of the Health Workers Handling PCB. Ministry of Labor, Bureau of Labor Standards and Institute of Industrial Health, Republic of Japan. In Japanese.

Hasegawa, H., Sato, M., and Tsuruta, H. (1973), An Investigation on the Toxicity of Some New Substances Used as PCB Replacement, Concentrations in Air of SAS, KMC-oil and PCBs in Carbonless Paper Producing Plants and Health Examination of Workers, Special Research Report 141-211, Research Coordination Bureau, Science and Technology Agency, Republic of Japan. In Japanese.

Hashimoto, I. (1974), Liver Function Test Results of PCB Influence Checkup, Jap J Ind Health 47:264-269. In Japanese.

Hass, J. R., Chae, K., and McKinney, J. D., unpublished observations.

Hearing Clerk, Dept. HEW (Feb. 1973), Analytical Methodology for Polychlorinated Biphenyls.

Hesse, J. L. (1975), Polychlorinated Biphenyl Usage and Sources of Loss to the Environment in Michigan, Nat Conference on Polychlorinated Biphenyls, Chicago.

Hill, E. F., Heath, R. G., Spann, J. W., and Williams, J. D. (1974), Polychlorinated Biphenyl Toxicity to Japanese Quail as Related to Degree of Chlorination, Poultry Science 53(2):597-604.

Hirayama, C., Okumura, M., Nagi, J., Masuda, Y. (1974), Hypobilirubinaemia in Patients with Polychlorinated Biphenyls Poisoning. Clin Chem Acta 55:97-100.

Hodge, H. C., and Sterner, J. H. (1949), Tabulation, Toxicity Classes, Am Indust Hygiene Assoc Quarterly 10:93.

Holden, A. V. and Marsden, K. (1969), Single Stage Cleanup of Animal Tissue Extracts for Organochlorine Residue Analysis, J. Chromatogr 44:481-492.

Holmes, D. C. and Waldan, M. (1972), A Simple Differentiation of Polychlorobiphenyls from Chlorinated Naphthalenes, J Chromatogr 71, 562-563.

Horwitz, W., Ed, (1975), Official Methods of Analysis of the Association of Official Analytical Chemists, 12th Ed, Section 29.001-29.018.

Hubbard, H. L. (1964), Chlorinated Biphenyl and Related Compounds, in "Encyclopedia of Chemical Technology," 2nd Ed, 5:289-298.

Humphrey, H. E. B., and Hayner, N. S. (1975), Polybrominated Biphenyls. An Agricultural Incidence and Its Consequences. An Epidemiological Investigation of Human Exposure, Trace Metals Conference, University of Missouri, Summer.

Humphrey, H. E. B., Price, H. A., and Budd, M. L. (1976), Evaluation of Changes of the Level of Polychlorinated Biphenyls (PCBs) in Human Tissue, Final Report of FDA Contract No. 223-73-2209.

Hurst, J. G., Newcomer, W. S., and Morrison, J. A. (1974), Effects of DDT, Toxaphene, and Polychlorinated Biphenyl on Thyroid Function in Bobwhite Quail, Poult Sci 53:125-133.

Huetert, K. and Korte, F (1972), Ecological Chemistry. XXXVIII. Synthesis of Polychlorinated Biphenyls and Their Reactions under UV Irradiation, Chemosphere 1:7-10.

Hutzinger, O., Nash, D. M., Safe, S., De Freitas, R. S. W., Norstrom, R. J., Wildish, D. J., and Zitko, V. (1972a), Polychlorinated Biphenyls: Metabolic Behaviour of Pure Isomers in Pigeons, Rats and Brook Trout, Science 178:312-313.

Hutzinger, O., Safe, S., and Zitko, V. (1972), Photochemical Degradation of Chlorobiphenyls (PCBs) Environ Health Perspect 1:15-20.

Hutzinger, O., Safe, S., Wentzell, B. R., and Zitko, V. (1973), Photochemical Degradation of di- and octachlorodibenzofuran, Environ Health Perspectives (exptl issue No. 5), 267-271.

Hutzinger, O., Safe, S., and Zitko, V. (1974a) The Chemistry of PCBs, CRC Press, Cleveland, Ohio 44128.

Hutzinger, O., Jamieson, W. D., Safe, S., Paulmann, L., and Ammon, R. (1974), Identification of Metabolic Dechlorination of Highly Chlorinated Biphenyl in Rabbit Nature 252:698-699.

Ikeda, Y., Horiuchi, S., Yoshimoto, B., Falaye, T., Kawamata, K., Kaneko, T. (1970), Studies on the Toxicity of Hazardous Substances, Special Research Report on Prevention of Outbreak of Yusho and its Diagnosis and Treatment. 74-12T.

INTERDEPARTMENT TASK FORCE ON PCBs. (1972), Washington, DC. Polychlorinated Biphenyls and the Environment.

International Council for the Exploration of the Sea. (1974), Report of Working Group for the International Study of the Pollution of the North Sea and its Effects on Living Resources and their Exploration., Charlottenlund Slot, DK-2920, Denmark, Cooperative Research Report No. 30.

Ito, N., Nagasaki, H., Arai, M., et al. (1973), Histopathologic Studies on Liver Tumorigeneais Induced in Mice by Technical Polychlorinated Biphenyls and its Promoting Effect on Liver Tumors Induced by Benzene Hexachloride. J Natl Cancer Inst 51:1637-1646.

Ito, N., Nagasaki, H., Arai, M., Makiura, S., Sugihara, S., and Hirao, K. (1973), Histopathologic Studies on Liver Tumorigenesis Induced in Mice by Technical Polychlorinated Biphenyls and its Promoting Effect on Liver Tumors Induced by Benzene Hexachloride, Journal of the Natl Cancer Institute 51:1637-1642.

Iveraon, F., Villeneuve, D. C., Grant, D. L., and Hatina, G. V. (1975), Effect of Aroclor 1016 and 1242 on Selected Enzyme Systems in the Rat, Bull Environ Contam Toxicol 13:456-463.

Jackson, T. F., Halbert, M. S. (1974), A Toxic Syndrome Associated with the Feeding of Polybrominated Biphenyl Contaminated Protein Concentrate to Dairy Cattle, JAVMA 65(5):437-439.

Jansson, B., Jensen, S., Olsson, M., Renberg, L., Sundstrom, G., and Vaz, R. (1975), Identification by GC-MS of Phenolic Metabolites of PCB and p,p'-DDE Isolated From Baltic Guillemot and Seal Ambio 4(2):93-97.

Jefferies, D. J., and Faralow, J. L. F. (1972), Effect of One Polychlorinated Biphenyl on Size and Activity of the Gull Thyroid, Bull Environ Contam Toxicol 8:306-310.

Jelinek, C. F. and Corneluissen, P. E. (1975), Levels of PCBs in the U.S. Food Supply. Nat Conference on Polychlorinated Biphenyls, Chicago.

Jensen, S., Renberg, L., and Olsson, M. (1972), PCB Contamination from Boat Bottom Paint and Levels of PCB in Plankton Outside a Polluted Area, Nature 240:358-360.

MONS 219429

Jensen, S. and Sundstrom, G. (1974), Structures and Levels of Most Chlorobiphenyls in Two Technical PCB Products and in Human Adipose Tissue, Ambio 3:70-76.

Jensen, S. and Sundstrom, G. (1974), Metabolic Hydroxylation of a Chlorobiphenyl Containing only Isolated Unsubstituted Positions-2,2',4,4',5,5'-hexachlorobiphenyl, Nature 251:219-220.

Jerina, D. M. and Daly, J. W. (1974), Arene Oxides: A New Aspect of Drug Metabolism, Science 185:573-582.

Johnstone, G. J., Ecobichon, D. J., and Hutzinger, O. (1974), Effect of Pure Chlorinated Biphenyl Compounds on Hepatic Function in the Rat, Toxicol Appl Pharmacol 28:66-81.

Johnstone, G. J., Ecobichon. J., and Hutzinger, O. (1974), The Influence of Pure Polychlorinated Biphenyl Compounds on Hepatic Function in the Rat, Tox and Appl Pharm 28:66-81.

Jondorf, W. R., Parke, D. V., and Williams, R. T. (1955), Studies in Detoxification. The Metabolism of Halogenobenzenes:1:2:3-, 1:2:4- and 1:3:5-trichlorobenzenes, Biochem J 61:512-521.

Jondorf, W. R., Parke, D. V., and Williams, R. T. (1958), Studies in Detoxification. The Metabolism of Halogenobenzenes:1:2:3:4-, 1:2:3:5- and 1:2:4:5-tetrachlorobenzenes, Biochem J 69:181-189.

Kaiser, K., "On the Optical Activity of PCBs." (1974), Environ Pollut 7:93-101.

Karppanan, E. and Kolho. (1974), The Concentration of PCB in Human Blood and Adipose Tissue in Three Different Research Groups. National Swedish Environmental Protection Board Publications 1974:4E: PCB Conference II. Vallingby, Sweden.

Kasza, L., Haberman, B., Brown, C., Capen, C., Trump, B., Hayee, E., Gilmore, C., and Weinberger, M. (1975), FDA Study F-84, PCB and PBB in Rats, (pathology report), October 1.

Kaplinger, M. L., et al. (1971), Toxicological Studies with the Chlorinated Biphenyls, Proceedings of the NIEHS Polychlorinated Biphenyl Meeting, Dec. 20-21.

Kaplinger, M. L., Francher, O. E., and Calandra, J. C. (1972), Toxicologic Studies with Polychlorinated Biphenyls, Toxicol Appl Pharmacol 19:402-403.

Kerst, A. (Sept 23, 1974), A Report to the Michigan Environmental Review Board, Michigan Chemical Corp.

Kihlstrom, J. E., Orberg, J., Lundberg, C., Danielsson, P. O., and Sydhoff, J. (1973), Effects of PCB on Mammalian Reproduction. PCB Conference II, National Swedish Environmental Protection Board Publications 4E:109-112.

Kihlstrom, J. E., Lundberg, C., Orberg, J., Danielsson, P. O., and Sydhoff, J. (1975), Sexual Functions of Mice Neonatally Exposed to DDT or PCB, Env Physiol Biochem 5:54-57.

Kimbrough, R. D. (personal communication, CDC, HEW, Atlanta, GA 30333).

Kimbrough, R. D., Linder, R. E., and Gaines, T. B. (1972), Morphological Changes in Livers of Rats Fed Polychlorinated Biphenyls, Arch Env Hlth 25:354-364.

Kinter, W. B., Markens, L. S., Janicki, R. H., and Guarino, A. M. (1972), Studies on the Mechanism of Toxicity of DDT and Polychlorinated biphenyl (PCBs) Disruption of Osmoregulation in Marine Fish, Environ Health Perspect 1:169-173.

Kimbrough, R. D., Linder, R. E., Burse, V. W., and Jennings, R. W. (1973), Adenofibrosis in the Rat Liver, Arch Environ Health 27:390-395.

Kimbrough, R. D. (1974), The Toxicity of Polychlorinated Polycyclic Compounds and Related Chemicals, Critical Reviews Toxicol 2(4):445-498.

Kimbrough, R. D. and Linder, R. E. (1974), Induction of Adenofibrosis and Hepatomas of the Liver in balb/cJ mice by Polychlorinated Biphenyls (Aroclor 1254), J Natl Cancer Institute 53:547-552.

Kimbrough, R. D., Squire, R. A., Linder, R. E., Strandberg, J. D., Montali, R. J., and Burse, V. W. (1975), Induction of Liver Tumors in Sherman Strain Female Rats by Polychlorinated Biphenyl Aroclor 1260, J. Natl Cancer Inst 55:1453-1459.

Kimbrough, R. D. (1974). The Toxicity of Polychlorinated, Polytyclic Compounds and Related Chemicals. Critical Reviews in Toxicology 2:445-498.

Kimbrough, R. (1975). Pathological Findings Associated with Chronic Experimental Exposure to PCBs. Nat Conf on Polychlorinated Biphenyls. Chicago, Illinois.

Kimura, N. T. and Baba, T. (1974). Neoplastic Changes in the Rat Liver Induced by Polychlorinated Biphenyl. Gann 64:105-108.

Kitamura, M. (1972), Studies on the Chronic Toxicity. Spacial Research Report on Prevention of Environmental Pollution by PCB-like Substances. Environ Health Criteria for Polychlorinated bi and terphenyls. Japanese Public Health
Kohanawa, M., Shoya, S., Ogura, Y., Moriwaki, M., and Kawasaki, M. (1969a), Poisoning Due to an Oily By-product of Rice-bran Similar to Chick Edema Disease. I. Occurrence and Toxicity Test. Nat Inst Anim Health Quart 9:213-219.

Kitamura, M., Tsukamoto, T., Sumino, K. Hayakawa, K., Shibata, T., and Hirano, I. (1973), The PCB Levels in the Blood of Workers Employed in a Condenser Factory, Jap J Ind Health 47:354-355. In Japanese.

Kleinert, S. J. (1975), Sources of Polychlorinated Biphenyls in Wisconsin. In press. Presented at the National Conference on Polychlorinated Biphenyls, Chicago.

Koch, R. B., Desaiah, D., Yap, H. H., and Cutkomp, L. K. (1972), Polychlorinated Biphenyls. Effect of Long-term Exposure on ATPase Activity in Fish, Pimephales Promelas. Bull Environ Contam Toxicol 7:87-92.

Kohanawa, M., Shoya, S., Yonemura, T., Nishimura, K., and Tsushio, Y. (1969b), Poisoning Due to an Oily By-product of Rice-bran Similar to Chick Edema Disease II. Tetrachlorodiphenyl as Toxic Substance, Nat Inst Anim Health Quart 9:220-228.

Koller, L. D. and Thigpen, J. E. (1973), Reduction of Antibody to Virus in Polychlorinated Biphenyl-Exposed Rabbits, Am J Vet Res 34:1605-1606.

Koller, L. D. and Zinkl, J. G. (1973), Pathology of Polychlorinated Biphenyls in Rabbits, Am J Pathol 70:363-373.

Kuratsune, M., Yoshimura, T., Matsuzaka, J., and Yamaguchi, A. (1971), Yuso, a Poisoning Caused by Rice Oil Contaminated with Polychlorinated Biphenyls, HSMHA Health Reports 86:1083-1091.

Kuratsune, M., Yoshimura, T., Matsuzaka, J., and Yamaguchi, A. (1972). Epidemiologic Study on Yusho, a Poisoning Caused by Ingestion of Rice Oil Contaminated with a Commercial Bi and Polychlorinated Biphenyls. Environmental Health Perspectives, (Exptl. Issue, No. 1), 119-128.

Kuratsune, M. (1972), An Abstract of Results of Laboratory Examinations of Patients with Yusho and of Animal Experiments. Environmental Health Perspectives, (Exptl. Issue No. 1), 129-136.

Kuratsune, M., Masuda, Y., and Nagayama, J. (1975), Some of the Recent Findings Concerning Yusho, Nat Conference on Polychlorinated Biphenyls, Chicago.

Kuratsune, M. (1976), Personal Communication.

Kutz, F. W. and Strassman, S. C. (1975), Residues of Polychlorinated Biphenyls in the General Population of the United States, Nat Conference on Polychlorinated Biphenyls, Chicago.

Kutz, F. W. and Yang, H. S. C. (1975), A Note on Polychlorinated Biphenyls in Air, Nat Conference on Polychlorinated Biphenyls, Chicago.

La Rocca, P. T. and Carlson, G. P. (1975), The Inhibitory Activity of Polychlorinated Biphenyls on ATPase Activity, Fed Proc 34:246.

Lincer, J. I. and Peakall, D. B. (1973), PCB (Polychlorinated Biphenyls), Pharmacodynamics in the Ring Dove and Early Gas Chromatographic Peak Diminution, Environ Pollut 4:59-68.

Linder, R. E., Gaines, T. B., and Kimbrough, R. D. (1974), The Effect of PCB on Rat Reproduction, Food Cosmet Toxicol 12:63-77.

Litterst, C. L. and Van Loon, E. J. (1972a), Enzyme Induction by Polychlorinated Biphenyls Relative to Known Inducing Agents, Proc Soc Exp Biol Med 141:765-768.

Litterst, C. L., Farber, T. M., Baker, A. M., and Van Loon, E. J. (1972b), Effect of Polychlorinated Biphenyls on Hepatic Microsomal Enzymes in the Rat, Toxicol Appl Pharmacol 23:112-122.

Litterst, C. L., and Van Loon, E. J. (1974), Time-course of Induction of Microsomal Enzymes Following Treatment with Polychlorinated Biphenyl, Bull Environ Contam Toxicol 11:206-212.

Makiura, S., Ace, H., Sugihara, S., eta al. (1974), Inhibitory Effect of Polychlorinated Biphenyls on Liver Tumorigenesis in Rats Treated with 3'-methyl -4 dimethyl aminoazobenzene, N-2-fluorenyl-acetamide and diethyl nitrosamine, J Natl Cancer Inst 53:1253-1257.

Masumoto, H. T. (1972), Study of the Silicic Acid Procedure of Armour and Burke for the Separation of Polychlorinated Biphenyls from DDT and Analogs, JAOAC 55:1092-1100.

Matthews, H. B. and Anderson, M. W. (1975), The Distribution and Excretion of 2,4,5,2',5'-pentachlorobiphenyl in the Rat, Drug Metab Disp 3:211-219.

Matthews, H. B. and Anderson, M. W. (1975), Effect of Chlorination on the Distribution and Excretion of Polychlorinated Biphenyls, Drug Metab Disp 3:371-380.

Matthew, H. B. (1975), PCB Chlorination vs. PCB Distribution and Excretion, Nat Conference on Polychlorinated Biphenyls, Chicago.

Matthews, H. B. and Tuey, D. B. (1976), The distribution and Excretion of 3,5,3',5'-tetrachlorobiphenyl in the Male Rat. Paper No. 133, Fifteenth Annual Meeting of the Society of Toxicology Atlanta, GA.

McKinnie, J. D., Haas, J. R., and Chae, K. (1975), Metabolism of Pure Hexachlorobiphenyl Isomers in Chicks, Personal Communication.

McKinney, J. D., Chae, K., Gupta, B. N., Moore, J. A., and Goldstein, J. A. (1975), Toxicological Assessment of Hexachlorobiphenyl Isomers and 2,3,7,8-Tetrachlorodibenzofuran in Chicks. I. Relationship of Chemical Parameters, Toxicol Appl Pharmacol in press.

McKinney, J. D. (1975), Toxicology of Selected Symmetrical Hexachlorobiphenyl Isomers: Correlating Biological Effects with Chemical Structure, Nat Conference on Polychlorinated Biphenyls, Chicago.

McNulty, W. P. (1975), Primate Study, Nat Conference on Polychlorinated Biphenyls, Chicago.

Meigs, J. W., Albom, J. J., and Kartin, B. L. (1954), Chloracne from an Unusual Exposure to Arochlor, JAMA 154:1417-1418.

MONS 219434

Melves, B. and Brandt, I. (1973), The Distribution and Metabolise of Labelled Polychlorinated Biphenyls in Mice and Quails. PCB Conference II. National Swedish Environment Board, Publications 4E:87-90.

MICHIGAN CHEMICAL COMPANY. (1974), Review of Polybrominated Biphenyls, Presented to the Michigan Environmental Review Board, September.

Mieure, J. P., Hicks, O., Kaley, R. G. and Saeger, V. W. (1976), Characterization of Polychlorinated Biphenyls, Nat Conference on Polychlorinated Biphenyls, Chicago.

Moore, J. A., Gupta, B. N., and Vos, J. G. (1976), Toxicity of 2,3,7,8-tetrachlorodibenzofuran, Nat Conference on Polychlorinated Biphenyls, Chicago.

Moron, M., Sundstrom, G. and Wachtmeister, C. A. (1973), Polychlorinated Biphenyls VI. 2,3,7,8-Tetrachlorodibenzofuran, a Critical Byproduct in the Synthesis of 2,2',4,4',5,5'-Hexachlorobiphenyl by the Ullmann Reaction, Acta Chem Scand 27:3121-3122.

Motovchi, M. (1956), Toxicity of Chlorinated Diphenyl, J Sci Labor 32:320. In Japanese.

Mulhern, B., Cromare, E., Reichel, W. L., (1971), Semiquantitative Determination of Polychlorinated Biphenyls in Tissue Samples by Thin Layer Chromatography JAOAC 54:548-550.

Nagayama, J., Masuda, Y., and Kuratsune, M (1975a), Dibenzofurans in Kanechlors, Jap J of Hygiene 30:126-129.

Nagayama, J., Masuda, Y., and Kuratsune, M. (1975b), Chlorinated Dibenzofurans in Kanechlors and Rice Oils Used by Patients with Yusho, Fukuoka, Acta Medica 66:593-599.

Nelson, J. A. (1974), Effects of dichlorodiphenyltrichloroethane (DDT) Analogs and Polychlorinated Biphenyls (PCB) Mixtures on 17 Beta-estradiol-3H to Rat Uterine Receptor, Biochem Pharmacol 23, 447-451.

NIEHS Conference on the Toxicity of the Chlorinated Dibenzo-p-dioxins and Dibenzofurans (1973) Research Triange Park, N.C., April 2-3; Environmental Health Perspective 5.

Nishiyama, K., Yano, H., and Kamano, M. (1973), Determination Method for Polychlorobiphenyl in Air and Its Vaporization from Non-carbon Copy Paper, Shikoku Acta Med 29(4):305-310. In Japanese.

Nishizumi, M. (1970), Light and Electron Microscope Study of Chlorobiphenyl Poisoning, Arch Environ Health 21:620-632.

Norris, J. M., Ehrmantraut, J. W., Gibbons, C. L., Kociba, R. J., Schwetz, B. A., Rose, J. Q., Humiston, C. G., Jewett, G. L., Crummett, W. B., Gehring, P. J., Tirsell, J. B., and Brosier, J. S. (1973), Toxicological Environmental Factors Involved in the Selection of Decabromodiphenyl Oxide as a Fire Retardant Chemical, Applied Polymer Symposium No. 22:195-219.

Norris, J. M., Ehrmanteant, J. W., Gibbons, G. L., Kociba, R. J., Schwertz. B. B., Rose, J. Q., Humiston, G. G., Jewett, G. L., Crummette, W. B., Gehring, P. J., Tirsell, J. R., Brosier, J. S., (1974), Toxicoloigical and Environmental Factors Involved in the Selection of Decabromodiphenyl Oxide as a Fire Retardant Chemical, J Fire Flammability/Combustion Toxicology 1:42.

Nowicki, H. G. and Norman, A. W. (1972), Enhanced Hepatic Metabolism of Testosterone, 4-Androstene-3, 17 dione, and 17 Beta-Estradiol in Chickens Pretreated with DDT or Polychlorinated Biphenyls, Steroids 19:85-99.

Oeme, T. (1975), Foreward. The Fifth Reports of the Study on "Yusho" and PCB (Polychlorinated Biphenyls), Fukuoka Acta Medica 55:547.

Orberg, J., Johansson, N., Kilstrom, J. E., and Lundberg, C. (1972), Administration of DDT and PCB Prolongs Estrous Cycle in Mice, AMBIO 1:148-149.

Orberg, J. and Kihlstrom, J. E. (1973), Effects of Long-term Feeding of Polychlorinated Biphenyls (PCB, Clophen A60) on the Length of the Oestrous Cycle and on the Frequency of Implanted Ova in the Mouse, Env Res 6:176-179.

Orberg, J. and Lundberg, D. H. (1974), Some Effects of DDT and PCB on the Hormonal System in the Male Mouse, Environ Physiol Biochem 4:116-120.

Oregon State University (1975), Summary of Research on PCBs, Experimental Health Sciences Center, Oregon State University (NIEHS Grant ES 00210, November, Page 10.)

Ouw, K. H., Simpson, G. R., and Siyali, D. S. (1974), The Use and Health Effects of Aroclor 1242, a Polychlorinated Biphenyl in an Electrical Industry in N.S.W., Australia. Report. Div. of Occup. Health and Rad. Control Health commission of New South Wales, Australia.

Panel on Hazardous Substances (1972), Polychlorinated Biphenyls, Environmental Impact, Environ Research 5:249-362.

Pardini, R. S. (1971), Polychlorinated Biphenyls (PCB), Effect on Mitochondrial Enzyme Systems, Bull Environ Contam Toxicol 6:539-545.

PCB's. (1976), Polychlorinated Biphenyls (PCBs) in Certain Freshwater Fish, Federal Register 1(39):8409-8410.

Peakall, D. B. and Risebrough, R. W. (1975), PCBs and Their Environmental Effects, CRC Critical Reviews in Environmental Control 5:469-508.

Phillips, W. E. J. (1963), DDT and the Metabolism of Vitamin A and Carotene in the Rat, Can J Biochem Physiol 41:1973-1802.

Phillips, W. E. J. and Hidiroglou, M. J. (1965), Carotenoid and Vitamin A Concentrations in Serum Liver of steer Fed Forages Treated with DDT or MCPA, J Agric Food Chem 13:254-256.

Platonow, N. S., Liptrap, R. M., and Geissinger, H. D. (1972), Distribution and Excretion of Polychlorinated Biphenyls (Aroclor 1254) and their Effect on Urinary Gonadal Steroid Levels in the Boar, Bull Environ Contam Toxicol 7:358-365.

Platonow, N. and Chen, N. Y. (1973), Transplacental Transfer of Polychlorinated Biphenyls (Aroclor (1254) in a Cow, Vet Record 92:69-70.

Platonow, N. S., and Funnell, H. S. (1971), Anti-androgenic-like Effect of Polychlorinated Biphenyls in Cockerels, Vet Rec 88:109-110.

Platonow, N. S. and Karstad, L. H. (1973), Dietary Effects of Polychlorinated Biphenyls on Mink, Can J Comp Med 37:90-95.

Poland, A. P., Smith, D., Metter, G., and Possick, P. (1971), A Health Survey of Workers in a 2,4-D and 2,3,5-T Plant. Arch Environ Health 22:316-327.

Porter, M. L. and Burke, J. A. (1971), Separation of Three Chlorodibenzo-p-dioxins from Some Polychlorinated Biphenyls by Chromatography on an Aluminum Oxide Column, JAOAC 54:1426-1428.

Reynolds, L. M. (1969), Polychlorobiphenyl (PCB's) and Their Interference with Pesticide Residue Analysis, Bull Environ Contam Toxicol 4:128-143.

MONS 219437

Ringer, R. K., Aulerich, R. J., and Zabik, M. (1972), Effects of Dietary Polychlorinated Biphenyls on Growth and Reproduction of Mink, Am Chem Soc Meeting, 12:149.

Risenbrough, R. W., Reiche, P., Herman, S. G., Peakall, D. B., and Kirven, M. N. (1968), Polychlorinated Biphenyls in the Global Ecosystem, Nature (London) 220:1098-1102.

Roach, J. A. G. and Pomerantz, I. H. (1974a), The Findings of Chlorinated Dibenzofurane in a Japanese Polychlorinated Biphenyls Sample, Bull Envir Contam and Tox 12:338-342.

Roach, J. A. G. and Pomerantz, I. H. (1974b), The Finding of Chlorinated Dibenzofurans in Aroclor PCBs of Recent Manufacture, Paper No. 53 presented at 88th Annual Meeting of Assn. of Official Anal. Chemists, Washington, DC, Oct 14-17.

Ruzo, L., Zakik, M., and Schuetz, R. (1974), Photochemistry of Bioactive Compounds: Photoproducts and Kinetics of Polychlorinated Biphenyls, J Agric Food Chem 22:199-202.

Ruzo, L. and Zabik, M. (1975), Polyhalogenated Biphenyls: Photolysis of Hexabromo and Hexachlorobiphenyls in Methanol Solution, Bull Environ Contam Toxicol 13:181-182.

Safe, S., Hutzinger, O., and Jones, D. (1975), The Mechanism of Chlorobiphenyl Metabolism, J Agric Food Chem 23(5):851-853.

Safe, S., Ruzo, L. O., Jones, D., Platonow, N. W., and Hutzinger, O., (1975), The Metabolism of 4-Chlorobiphenyl in the Pig, Can J Physiol Pharmacol 53:392-396.

Safe, S. Platonow, N., and Hutzinger, O. (1975b), The Metabolism of Chlorobiphenyls in the Goat and Cow, J Agr Food Chem 23:259-261.

Sagami, S., Kamata, S., and Yoshida, M. (1973), A Case of Chronic PCB-Intoxication, Nishinihon Hifuka 36(6):694-701. In Japanese.

Sanders, O. T., Zepp, R. L., and Kirkpatrick, R. L. (1974), Effect of PCB (Polychlorinated Biphenyls) Ingestion on Sleeping Times, Organ Weights, Food Consumption, Serum Corticosterone, and Survival of Albino Mice, Bull Environ Contam Toxicol 12:394-399.

Sato, M. and Hasegawa, H. (1974), PCB Concentrations in the Blood of Workers Employed in Carbonless Copying Paper Production, Jap J Ind Health 16:365. In Japanese.

Sawyer, L. D. (1973), Collaborative Study of the Recovery and Gas Chromatographic Quantitation of Polychlorinated Biphenyls in Chicken Fat and Polychlorinated Biphenyl - DDT Combinations in Fish, JAOAC 56:1015-1023.

Schmoldt, A., Benthe, H. F., and Fruehling, R. (1974), Induction of Rat Liver Enzymes by Polychlorinated Biphenyls (PCBs) in Dependence on the Dose and Chlorine Content, Arch Toxicol 32:69-81.

Schulte, E. and Acker, L. (1974), Identifizierung und Metabolisierbarkeit von Polychlorierten Biphenylen, Naturwissenschaften 61:79-80.

Schwartz, L. (1943), An Outbreak of Halowax Acne ("Cable Rash") Among Electricians, JAMA 122(3):158-161.

Schwartz, L., and Barlow, F. A. (1942), Chloracne from Cutting Oils, Pub Health Rpts 57(47):1747-1750.

Schwartz, L. (1936), Dermatitis from Synthetic Resins and Waxes, Am J Pub Health 26:586-592.

Schwetz, B. A., Norris, J. M., Sparschu, G. L., Rowe, V. K., and Gehring, P. J. (1973), Toxicology of Chlorinated Dibenzo-p-dioxins, Adv Chem Series 120:55-69.

Schwetz, B. a., Keller, P. A., and Gehring, P. J. (1974), The Effect of Purified and Commercial Grade Pentachlorophenol on Rat Embryonal and Fetal Development, Tox and Appl Pharm 28:151-161.

Sharp, C. W., Hunt, D. G., Clements, S. T., and Wilson, W. E. (1974), Influence of Dichlorodiphenyltrichloroethane, Polychlorinated Biphenyls and Anionic Amphiphilic Compounds on Stabilization of Sodium and Potassium Activated Adenosine Triphosphatases by Acidic Phospholipids, Mol Pharmacol 10:119-129.

Simon, N. and Siklosi, C. (1974), Influence of Environmental Factors on Porphyrin Metabolism, Berufs-Dermatosen 22:237-259.

Sinclair, P. R., Granick, S. (1974), Uroporphyrin Formation Induced by Chlorinated Hydrocarbons (Lindane, Polychlorinated Biphenyls, Tetrachlorodibenzo-p-dioxin). Requirements for Endogenous Iron, Proten Synthesis, and Drug Metabolizing Activity, Biochem Biophys Res Commun 61:124-133.

Sissons, D. and Welti, D. (1971), Structural Identification of Polychlorinated Biphenyls in Commercial Mixtures by Gas-Liquid Chromatography, Nuclear Magnetic Resonance and Mass Spectrometry. J. Chromatog. 60:15-32.

Sivalingam, P. M., Yoshida, T., and Inada, Y. (1973), Modes of Inhibitory Effects of Polychlorinated Biphenyls on Oxidative Phosphorylation of Mitochondria, Bull Environ Contam Toxicol 10:242-247.

Smith, F. (1948), U. S. Patent 2,449,088; (1949), Chem Abstr 43: 813.

Smith, L. W., Friea, G. F., and Weinland, B. T. (1973), Utilization of Contaminated Dehydrated Poultry Excreta as Feed by Cows and Polychlorinated Biphenyl Residues in Milk, J Dairy Sci, in press.

Smrek, A., Adams, S., Liddle, J., and Kimbrough, R.: Pharmacodynamics of Mirex in Goats: Effect of Reproduction and Lactation. Submitted to Arch Environ Health.

Societe d'Dlectrochimie, d'Electrometallurgie E+ des Acieries Electriques d'Ugine (1962) Belgian Patent 613,066; (1962) Chem Abstr 57:16492.

Squire, R. A. and Levitt, H. (1975), Report of Workshop on Classification of Specific Neoptocellular Lesions in Rats, Cancer Research 35:3214-3223.

Stalling, D. L. and Huckens, J. N. (1971), Gas-Liquid Chromatography - Mass Spectrometry Characterization of Polychlorinated Biphenyl (Aroclors) and $^{36}Cl$-Labeling of Aroclors 1248 and 1254, JAOAC 54:801-807.

Stalling, D. L. and Mayer, F. L. Jr. (1972), Toxicities of PCBs to Fish and Environmental Residues, Environmental Health Perspectives 1:159-164.

Stalling, D. L., Tindle, R. C., Johnson, J. L. (1972), Cleanup of Pesticide and Polychlorinated Biphenyl Residues in Fish Extracts by Gel Permeation Chromatography, JAOAC 55:32-38.

Street, J. C. and Sharma, R. P. (1975), Alteraction of Induced Cellular and Humoral Immune Responses by Pesticides and Chemicals of Environmental Concern: Quantitative Studies of Immunosuppression by DDT, Aroclor 1254, Carbaryl, Carbofuran, and Methylparathion, Tox Appl Pharm 32:587-602

Street, J. C., Urry, F. M., Wagstaff, D. J., and Blau, A. D. (1969), Comparative Effects of Polychlorinated Biphenyls and Organochlorine Pesticides in Induction of Hepatic Microsomal Enzymes, 158th National Meeting, American Chemical Society, New York, NY.

Sugiura, K., Haltori, M., Baba, M., and Goto, M. (1975), Accumulation and Excretion of PCBs in the Mouse, Chemosphere 4(3):181-188.

Sundstrom, G., Hutzinger, O., and Safe, S. (1976), Identification of 2,2',4,4',5,5'-Hexabromobiphenyl as the Major Component of Flame Retardant FireMaster$^{R}$ BP-6, Chemosphere 5:11-14.

Tee, A. C. and Klaipool, R. J. C. (1972), Characterization of the Components of Technically Polychlorinated Biphenyl Mixtures - II, Bull Envir Contam and Tox 8:32-37.

Trotter, W. (1975), Removing the Interference of DDT and its Analogs in the Analysis for Residues of Polychlorinated Biphenyls, JAOAC 58:461-465.

Trotter, W. J. and Young, S. J. V. (1975), Limitation on the Use of Antimony Pentechloride for Perchlorination of Polychlorinated Biphenyls, JAOAC 58:466-468.

Trotter, W. J. (1976), PCB Analysis of Rice Oil Associated with Yusho, Memo to Frank Cordle, FDA, March 11.

Tsukamoto, H., et al. (1969), The Chemical Studies on Detection of Toxic Compounds in the Rice Bran Oils Used by the Patients of Yusho, Fukuoka, Acta Medica 60:496-512.

Tucker, R. K., and Crabtree, D. G. (1970), Handbook of Toxicity of Pesticides to Wildlife, Bureau fo Sport, Fisheries, and Wildlife, Resources Publ. 84, U.S. Dept. of Interior, Washington D. C.

Turner, J. C., and Green, R. S. (1974), Effect of a Polychlorinated Biphenyl (Aroclor 1254) on Liver Microsomal Enzymes in the Male Rat, Bull Environ Contam Toxicol 12:669-671.

Vainio, H. (1974), Enhancement of Microsomal Drug Oxidation and Glucuronidation in Rat Liver by an Environmental Chemical, Polychlorinated Biphenyl, Chem - Biol Interact 9:379-387.

Van Miller, J. P., Hsu, I. C., and Allen, J. R. (1975), Distribution and Metabolism of $^{3}$H-2,5,2',5-tetra-chlorobiphenyl in Rats, Proc Soc Exp Biol Med 148:682-687.

Veith, G. D. (1975), Baseline Concentrations of Polychlorinated Biphenyls and DDT in Lake Michigan Fish, 1971, Pest Monitoring Journal 9:21-29.

Villeneuve, D. C., Grant, D. L., Phillips, W. E. J., Clark, M. L., and Clegg, D. J. (1971a), Effects of PCB (Polychlorinated Biphenyl: Aroclors 1221 and 1254), Administration on Microsomal Enzyme Activity in Pregnant Rabbits, Bull Environ Contam Toxicol 6:120-128.

Villeneuve, D. C., Grant, D. L., and Phillips, W. E. J. (1972), Modification of Pentoberbital Sleeping Times in Rats Following Chronic PCB Ingestion, Bull Environ Contam Toxicol 7:264-269.

Villanueva, E. C., Jennings, R. W., Burse, V. W., and Kimbrough, R. D. (1974), Evidence of Chlorodibenzo-p-dioxin and Chlorodibenzofuran in Hexachlorobenzene, J Agr Food Chem 22:916-917.

Villeneuve, D. C., Grant, D. L., Khera, K., Clegg, D. J., Baer, H., and Phillips, W. E. J. (1971b), The Fetotoxicity of a Polychlorinated Biphenyl Mixture (Aroclor 1254) in the Rabbit and in the Rat, Env Physiol 1:67-71.

Vos, J. G. and Koeman, J. H. (1970), Comparative Toxicologic Study with Polychlorinated Biphenyls in Chicken with Special Reference to Porphyria, Edema Formation, Liver Necrosis, and Tissue Residues, Tox Appl Pharm 17:656-668.

Vos, J. G., Koeman, J. J., Van der Maas, H. L., Ten Noever de Brauw, M. C., and De Vos, R. H. (1970), Identification and Toxicological Evaluation of Chlorinated Dibenzofuran and Chlorinated Naphthalene in Two Commercial Polychlorinated Biphenyls, Fd Cosmet Toxicol 8:625-633.

Vos, J. G. and Beems, R. B. (1971), Dermal Toxicity Studies of Technical Polychlorinated Biphenyls and Fractions Thereof in Rabbits, Tox Appl Pharm 19:617-633.

Vos, J. G. (1971), Polychlorinated Biphenyls as Inducers of Hepatic Porphyria in Japanese Quail with Special Reference to -Aminolevulinic Acid Synthesase Activity, Fluorescence, Acid Residues in Liver, Toxicol Appl Pharmacol 20:232-240.

Vos, J. G. and Notenboom-Ram, E. (1972), Comparative Toxicity Study of 2,4,5,2',4',5'-Hexachlorobiphenyl and a Polychlorinated Biphenyl Mixture in Rabbits, Tox and Appl Pharm 23:563-578.

Vos, J. G. (1972), Toxicology of PCB's for Mammals and for Birds, Env Hlth Persp 1:105-117.

Vos, J. G. and VanGenderen, H. (1973), Toxicological Aspects of Immuno-suppression. Aspects of Immunosuppression Published in Pesticides in the Environment, a Continuing Controversy, Ed. by W. B. Deichman, (8th International Conference on Toxicology and Occupational Medicine, Miami), Published by Intercontinental Med. Book Co., New York, NY.

Vos, J. G. and DeRoij, T. L. (1972), Immunosuppressive Activity of a Polychlorinated Biphenyl Preparation on the Humoral Immune Response in Guinea Pigs, Tox Appl Pharm 21:549-555.

Walker, C. R. (1975), Pre-1972 Knowledge of Non Human Effects of Polychlorinated Biphenyls, Nat Conf on Polychlorinated Biphenyls, Chicago, Illinois.

Wasserman, D., Wasserman, M. (1972), Ultrastructure of Adrenal Zona Fasciculata in Polychlorinated Biphenyls Receiving Rats, 17th Intern Congress Occ Health, Buenos Aires.

Wasserman, D., Wasserman, M., Cucos, S., and Djavaherian, M. (1973), Function of Adrenal Glad-zone Fasciculata in Rats Receiving Polychlorinated Biphenyls, Environ Res 6:334-338.

Webb, R. G. and McCall A. C. (1972), Identities of Polychlorinated Biphenyl Isomers in Aroclors, JAOAC 55:746-752.

Webb, R. G. and McCall, A. C. (1973), Quantitative PCB Standards for Electron Capture Gas Chromatography, J Chromatogr Sci 11:366-373.

Weingarten, H. (1961), Steric Effects in the Gomberg Reaction, J Org Chem 26730-733.

Welborn, J. A., Senator and Chairman, A Report from the State Senate Special Investigating Committee, The Contamination Crisis in Michigan, July, 1975.

WHO Task Group (1975), Environmental Health Criter for Polychlorinated Biphenyls and Terphenyls.

Wit, J. G. (1972), Enzyme Induction with Specific Reference to Chemical Porphyria, Third Technical Meeting on Occurrence and Significance of Chemicals in the Environment, Berlin, Germany.

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Wong, R. G., Nowicki, H. G., and Norman, A. W. (1974), Effects of Polychlorinated Biphenyls on Calciferol (Vitamin D) Mediated Calcium Metabolism, Pestic Biochem Physiol 4:170-177.

Yamamoto, H., and Yoshimura, H. (1973), Metabolic Studies on PCBs. III. Complete Structure and Acute Toxicity of the Metabolites of 2,4,3'4'-tetrachlorobiphenyl, Chem Pharm Bull 21:2237-2242.

Yamamoto, H. (1974), Statistics of Occupational Diseases (Result of Special Medical Examination), New Occupational Health Handbook 1010-1013, The Institute for Science of Labour, In Japanese.

Yap, H. H., Desaiah, D., Cutkomp, L. K., and Koch, R. B. (1971), Sensitivity of Fish atpase to Polychlorinated Biphenyls, Nature (London) 233:61-62.

Yoba, A. R. (1972), Levels of Polychlorinated Biphenyls in Adipose Tissue of the General Population of the Nation, Environ Health Perspectives 1:79-81.

Yushimura, T. (1971), Epidemiological Analysis of "Yusho" Patients with Special Reference to Sex, Age, Clinical Grades and Oil Consumption, Fukuoka, Acta Medica 62:104-108.

Young, S. and Burke, J. (1972), Micro Scale Alkali Treatment for Use in Pesticide Residue Confirmation and Sample Cleanup, Bull Environ Contam Toxicol 7:160-167.

Zapp, R. L. Jr., Sanders, O. T., and Kirkpatrick, R. L. (1974), Reduction of Pentobarbital-Induced Sleeping Times in PCB (Polychlorinated Biphenyl)-Treated Cottontail Rabbits, Bull Environ Contam Toxicol 12:518-521.

Zitko, V., Hutzinger, O., and Choi, P. M. K. (1972), Contamination of the Bay of Fundy - Gulf of Maine Area with PCBs, PCTs, Chlorinated DBF and DBD, Environ Health Perspect 1:47-50.

Zitko, V., Hutzinger, O., and Safe, S. (1971), Retention Times and Electron Capture Detector Responses of Some Individual Chlorobiphenyls, Bull Environ Contam and Toxicol 6:160-163.

Zitko, V., Hutzinger, O., and Safe, S. (1971), Retention Times and Electron-capture Detector Responses of Some Individual Chlorobiphenyls, Bull Environ Contam Toxicol 6:160-163.

Zitko, V., Hutzinger, O., and Choi, P. M. K. (1974), Determination of Pentachlorophenol and Chlorobiphenylols in Biological Samples, Bull Environ Contam Toxicol 12:649-653.

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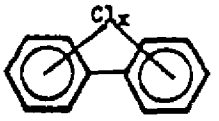
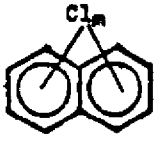
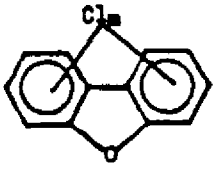
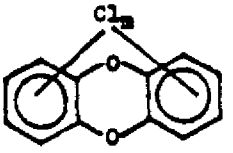
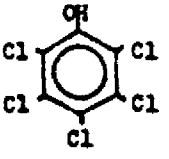
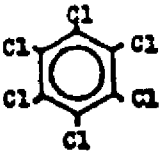
Appendix A
CHLORINATED AROMATIC COMPOUNDS
REFERRED TO IN CHEMISTRY REPORT

| NAME | STRUCTURE | EXTENT OF CHLORINATION POSSIBLE | NUMBER OF CHLORINATED DERIVATIVES POSSIBLE |
|---|---|---|---|
| Chlorinated biphenyls |  | x= 1-10 | 209 |
| Chlorinated naphthalenes |  | n=1-8 | 75 |
| Chlorinated dibenzofurans |  | n=1-8 | 135 |
| Chlorinated dibenzo-p-dioxins |  | n=1-8 | 75 |
| pentachlorophenol |  | - | - |
| hexachlorobenzene |  | - | - |

MONS 219445

Calculation of dosages received by monkeys in reference 16

The amount of food eaten in relation to body weight varies between different animal species, including man. To enable comparisons, food or food component intake frequently is standardized by expressing the amount eaten as a function of body weight. The latter is known as the dosage, while the amount eaten would be the dose.

Table 1 lists mean body weights for all experimental animals. These range from 5.57 kg pretest to 4.84 kg at the end of 5 months on test. An overall mean body weight of 5.32 kg. was estimated by averaging the six figures listed.

Table 3 shows total 6-month polychlorinated biphenyl intake as 85.29 mg for 2.5 ppm group and 175.36 mg for the 5.0 ppm group.

Average daily intake was calculated by use of the following formula:

$$\mu g/kg/day = \frac{\text{total mg intake}}{\text{body wt. (kg)}} \times \frac{1}{180 \text{ days}} \times \frac{1000\ \mu g}{mg}$$

At 2.5 ppm, the dosage was approximately 89 µg/kg/day

At 5.0 ppm, the dosage was approximately 183 µg/kg/day

For comparison, FDA estimated safe intake as 1 µg/kg/day (14)

Thus, even though the monkey dietary levels of 5.0 ppm and 2.5 ppm were described as levels "equal to, and 50% of, the concentrations allowed in certain foods destined for human consumption" (16), it appears that the monkeys actually received about one hundred and two hundred times the dosages which FDA estimated as safe from the human experience at Yusho.

Monsanto

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INDUSTRIAL BIO-TEST LABORATORIES, INC. REPORTS

Two-Year Chronic Oral Toxicity
Study With
Aroclor 1242
In Albino Rats

Histopathological Evaluation
of Additional Liver Sections

March 24, 1975

.....

Two-Year Chronic Oral Toxicity
Study With
Aroclor 1254
In Albino Rats

Histopathological Evaluation
of Additional Liver Sections

March 24, 1975

.....

Two-Year Chronic Oral Toxicity
Study with
Aroclor 1260
in Albino Rats

Histopathological Evaluation
of Additional Liver Sections

March 24, 1975

THE EPPLEY INSTITUTE REPORTS

Report on
Histopathological Re-evaluation of
Tissues from Female Sherman Rats
Fed Aroclor 1260

.....

Report on
Histopathological Re-evaluation of
Livers from Rats Treated with Aroclor

Attachment 2-2

Submitted by

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C+S. 341

REPORT TO

MONSANTO COMPANY

# TWO - YEAR CHRONIC ORAL TOXICITY STUDY WITH AROCLOR 1242 IN ALBINO RATS

## HISTOPATHOLOGICAL EVALUATION OF ADDITIONAL LIVER SECTIONS

MARCH 24, 1975

IBT NO. 641 - 06672

Industrial BIO-TEST Laboratories, Inc.

REPORT TO

MONSANTO COMPANY

# TWO - YEAR CHRONIC ORAL TOXICITY STUDY WITH AROCLOR 1242 IN ALBINO RATS

## HISTOPATHOLOGICAL EVALUATION OF ADDITIONAL LIVER SECTIONS

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## I. Introduction

At the request of Dr. Levinskas of the Monsanto Company, additional sections of liver from a two - year chronic oral toxicity study of Aroclor 1242 in rats (IBT No. 622-07298) were processed into H & E stained sections and evaluated by light microscopy. The following report presents the results of this study.

II. Summary

In most instances, the spectrum of treatment-related histopathological findings in the liver from this re-evaluation did not differ significantly from that previously reported in our original report dated November 12, 1971. There were three hepatomas detected among three of the animals at the highest treatment level (100 ppm) of the 24-Month Sacrifice. No hepatocellular carcinomas were observed. The other treatment-related lesions reported are regarded as degenerative or hyperplastic in nature and they are morphologic manifestations of an adaptive response of the liver associated with biotransformation of the test material. In general, the latter findings were confined primarily to test animals of the final sacrifice and they were dose-related in incidence and severity.

In conclusion, Aroclor 1242 does not appear to be carcinogenic in rats fed for two years at levels up to and including 100 ppm.

Respectfully submitted,

INDUSTRIAL BIO-TEST LABORATORIES,

Report Prepared and Reviewed by:

D. E. Gordon, D.V.M., Ph.D.
Section Head, Pathology

Report Approved by:

M. L. Kepinger, Ph.D.
Manager, Toxicology

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IBT No. 622-07298
Monsanto

I have examined additional sections of liver from rats of IBT No. 622-07298 and have re-evaluated them for evidence of carcinogenesis. I have tabulated my findings for Aroclor 1242 on the following pages:

There is evidence of a chemical effect on the liver which is most pronounced at time of the terminal (24-Month) sacrifice. This consists of a hepatocellular alteration beginning as focal hypertrophy which progresses to nodular hyperplasia, and in a few animals to hepatoma or cholangiohepatoma. The hypertrophic cells contain large volumes of light staining cytoplasm which is probably rich in glycogen and endoplasmic reticulum. In some sections, ring-shaped structures, probably representing whorls of proliferative endoplasmic reticulum, were seen in the cytoplasm of these cells.

The hyperplastic nodules occupied larger areas and often compressed surrounding cells of the normal liver parenchyma. These nodules also contained the same hypertrophic cells.

Those lesions classified as hepatomas or cholangiohepatomas were larger nodules which showed evidence of confluence or some variation in cell size, shape, staining or a ductular or adenomatous pattern of growth. In the absence of metastasis, invasiveness, severe basophilia, mitoses, or other evidence of anaplasia, these are benign tumors rather than malignant tumors (carcinomas). There was no evidence of metastasis or invasiveness of these tumors in this study.

With exception of the vacuolar changes in the cytoplasm of hepatocytes, the more severe hepatocellular alterations were confined to animals of the 24-Month Sacrifice.

Ward R Richter

Ward R. Richter, D.V.M., M.S.
Dipl., Am. College Vet. Path.

Primary Liver Lesions - Aroclor 1242

3 Month Sacrifice - Rats

| Lesion | Group | | | |
|---|---|---|---|---|
| | TI | TII | THI | Control |
| Vacuolar Chang- | 0/8 | 2/10 | 1/9 | 2/10 |
| Focal Hypertrophy | 0/8 | 0/10 | 0/9 | 0/10 |
| Nodular Hyperplasia | 0/8 | 0/10 | 0/9 | 0/10 |
| Hepatoma | 0/8 | 0/10 | 0/9 | 0/10 |
| Ductular Hyperplasia | 0/8 | 0/10 | 1/9 | 0/10 |
| Cholangio- Hepatoma | 0/8 | 0/10 | 0/9 | 0/10 |
| Hepatocellular Necrosis | 1/8 | 0/10 | 0/9 | 0/10 |

Primary Liver Lesions - Aroclor 1242

6 Month Sacrifice - Rats

| Lesion | Group | | | |
|---|---|---|---|---|
| | TI | TII | TIII | Control |
| Vacuolar Change | 0/10 | 0/8 | 0/9 | 1/10 |
| Focal Hypertrophy | 0/10 | 0/8 | 0/9 | 0/10 |
| Nodular Hyperplasia | 0/10 | 0/8 | 0/9 | 0/10 |
| Hepatoma | 0/10 | 0/8 | 0/9 | 0/10 |
| Ductular Hyperplasia | 0/10 | 0/8 | 0/9 | 0/10 |
| Cholangio-Hepatoma | 0/10 | 0/8 | 0/9 | 0/10 |
| Hepatocellular Necrosis | 1/10 | 0/8 | 0/9 | 1/10 |

Primary Liver Lesions - Aroclor 1242

12 Month Sacrifice - Rats

| Lesion | Group | | | |
|---|---|---|---|---|
| | TI | TII | T III | Control |
| Vacuolar Change | 2/10 | 4/10 | 0/9 | 1/10 |
| Focal Hypertrophy | 0/10 | 0/10 | 1/9 | 0/10 |
| Nodular Hyperplasia | 0/10 | 0/10 | 0/9 | 0/10 |
| Hepatoma | 0/10 | 0/10 | 0/9 | 0/10 |
| Ductular Hyperplasia | 0/10 | 1/10 | 0/9 | 1/10 |
| Cholangio-Hepatoma | 0/10 | 0/10 | 0/9 | 0/10 |
| Hepatocellular Necrosis | 0/10 | 0/10 | 0/10 | 0/10 |

Primary Liver Lesions - Aroclor 1242

Terminal Sacrifice - Rats

| Lesion | Group | | | |
|---|---|---|---|---|
| | TI | TII | TIII | Control |
| Vacuolar Change | 7/31 | 8/30 | 9/20 | 1/23 |
| Focal Hypertrophy | 2/31 | 3/30 | 8/20 | 0/23 |
| Nodular Hyperplasia | 0/31 | 2/30 | 8/20 | 1/23 |
| Hepatoma | 0/31 | 0/30 | 2/20 | 0/23 |
| Ductular Hyperplasia | 3/31 | 3/30 | 3/20 | 5/23 |
| Cholangio-Hepatoma | 0/31 | 0/30 | 1/20 | 0/23 |
| Hepatocellular Necrosis | 1/31 | 1/30 | 0/20 | 1/23 |

AROCLOR - 1242

Tabulation of Individual Liver Lesions in Rats

| Sacrifice Interval | Dose Level | Animal No. and Sex | Liver Lesions | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Reticulum Cell Sarcoma metastatic | Mammary tumor, metastatic | Telangiectasis, focal |
| Months | C-I 1ppm | 596M | | | | | | | | | | | |
| | | 598" | | | | | | | | | | | |
| | | 599" | | | | | | | | | | | |
| | | 636F | | | | | | | | | | | |
| | | 637" | | | | | | | | | | | |
| | | 638" | | | | | | | | | | | |
| | | 639" | | + | | | | | | | | | |
| | | 640" | | | | | | | | | | | |
| | C-II 10ppm | 676M | | | | | | | | | | | |
| | | 677" | + | | | | | | | | | | |
| | | 678" | + | | | | | | | | | | |
| | | 679" | | | | | | | | | | | |
| | | 680" | | | | | | | | | | | |
| | | 716F | | | | | | | | | | | |
| | | 717" | | | | | | | | | | | |
| | | 718" | | | | | | | | | | | |
| | | 719" | | | | | | | | | | | |
| | | 720" | | | | | | | | | | | |
| | C-III 100ppm | 756M | | | | | | | | | | | |
| | | 757" | | | | | | | | | | | |
| | | 758" | ++ | | | | | | | | | | |
| | | 759" | | | | | | | | | | | |
| | | 760" | | | | | | + | | | | | |
| | | 796F | | | | | | | | | | | |
| | | 797" | | | | | | | | | | | |
| | | 798" | | | | | | | | | | | |
| | | 800" | | | | | | | | | | | |
| | Control | 36M | + | | | | | | | | | | |
| | | 37" | | | | | | | | | | | |
| | | 38" | | | | | | | | | | | |
| | | 39" | + | | | | | | | | | | |
| | | 40" | | | | | | | | | | | |
| | | 76F | | | | | | | | | | | |
| | | 77" | | | | | | | | | | | |
| | | 78" | | | | | | | | | | | |
| | | 79" | | | | | | | | | | | |
| | | 80" | | | | | | | | | | | |

MONS 219457

AROCLOR - 1242

Tabulation of Individual Liver Lesions in Rats



| crifice erval | Dose Level | Animal No. and Sex | Liver Lesions: Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Reticulum Cell Sarcoma metastatic | Mammary tumor, metastatic | Telangiectasia, focal |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Month | C-I | 1011M | | + | + | | | | | | | | |
| | 1ppm | 1012" | | | | | | | | | | | |
| | | 1013" | | | + | | | | | | | | |
| | | 1014" | | | | | | | | | | | |
| | | 1015" | | | | | | | | | | | |
| | | 1026F | | | | | | | | | | | |
| | | 1027" | | | | | | | | | | | |
| | | 1028" | | | | | | | | | | | |
| | | 1029" | | | | | | | | | | | |
| | | 1030" | | | + | | | | | | | | |
| | C-II | 1041M | | | | | | | | | | | |
| | 10ppm | 1042" | | | | | | | | | | | |
| | | 1043" | | | | | | | | | | | |
| | | 1045" | | | | | | | | | | | |
| | | 1056F | | | | | | | | | | | |
| | | 1057" | | | | | | | | | | | |
| | | 1058" | | | | | | | | | | | |
| | | 1059" | | | | | | | | | | | |
| | C-III | 1071M | | | | | | | | | | | |
| | 100ppm | 1072" | | | | | | | | | | | |
| | | 1073" | | | | | | | | | | | |
| | | 1075" | | | | | | | | | | | |
| | | 1086F | | | | | | | | | | | |
| | | 1087" | | | | | | | | | | | |
| | | 1088" | | | | | | | | | | | |
| | | 1089" | | | | | | | | | | | |
| | | 1090" | | | | | | | | | | | |
| | Control | 801M | + | + | | | | | | | | | |
| | | 802" | | | | | | | | | | | |
| | | 803" | | | | | | | | | | | |
| | | 804" | | | | | | | | | | | |
| | | 805" | | | | | | | | | | | |
| | | 816F | | | | | | | | | | | |
| | | 817" | | | | | | | | | | | |
| | | 818" | | | | | | | | | | | |
| | | 819" | | | | | | | | | | | |
| | | 820" | | | | | | | | | | | |

MONS 219458

AROCLOR - 1242

Tabulation of Individual Liver Lesions in Rats



| Sacrifice Interval | Dose Level | Animal No. and Sex | Liver Lesions | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Reticulum Cell Sarcoma metastatic | Mammary tumor, metastatic | Telangiectasis, focal |
| 2 Month | C-I 1ppm | 71M | | | | | | | | | | | |
| | | 72" | + | | | | | | | | | | |
| | | 73" | | | | | | | | | | | |
| | | 74" | | | | | | | | | | | |
| | | 75" | | | | | | | | | | | |
| | | 76F | + | | | | | | | | | | |
| | | 77" | | | | | | | | | | | |
| | | 78" | | | | | | | | | | | |
| | | 79" | | | | | | | | | | | |
| | | 80" | | | | | | | | | | | |
| | C-II 10ppm | 81M | + | | | | | | | | | | |
| | | 82" | + | | | | | | | | | | |
| | | 83" | | | | | | | | | | | |
| | | 84" | + | | | | | + | | | | | |
| | | 85" | | | | | | | | | | | |
| | | 86F | | | | | | | | | | | |
| | | 87" | | | + | | | | | | | | |
| | | 88" | | | | | | | | | | | |
| | | 89" | | | | | | | | | | | |
| | | 90" | + | | | | | | | | | | |
| | C-III | 91M | | | | | | | | | | | |
| | | 92" | | | | | | | | | | | |
| | | 93" | | | | | | | | | | | |
| | | 94" | | | | | | | | | | | |
| | | 95" | | | | | | | | | | | |
| | | 96F | | | | | | | | | | | |
| | | 97" | | | | + | | | | | | | |
| | | 98" | | | | | | | | | | | |
| | | 99" | | | | | | | | | | | |
| | Control | 1M | | | | | | | | | | | |
| | | 2" | + | | | | | | | | | | |
| | | 3" | | | | | | | | | | | |
| | | 4" | | | | | | | | | | | |
| | | 5" | | | | | | | | | | | |
| | | 6F | | | | | | | | | | | |
| | | 7" | | | | | | | | | | | |
| | | 8" | | | | | | | | | | | |

MONS 219459

Tabulation of Individual Liver Lesions in Rats

| Sacrifice Interval | Dose Level | Animal No. and Sex | Liver Lesions | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Reticulum Cell Sarcoma metastatic | Mammary tumor, metastatic |
| 4 Month | C-1 1ppm | 564M | | | | | | | | | | |
| | | 566" | | | | | | | | | | |
| | | 591" | | | | | | | | | | |
| | | 567" | | | | + | | | | | | |
| | | 569" | | | | | | | | | P | |
| | | 572" | | | | | | + | | | | |
| | | 576" | | | | | | | | | | |
| | | 580" | | | | | | | | | | |
| | | 588" | | | | | | | | | | |
| | | 589" | | | | | | | | | P | |
| | | 595" | + | | | | | | | | | |
| | | 601F | | | | | | + | | | | |
| | | 604" | | | | | | | | | | |
| | | 605" | | | | | | | | | | |
| | | 582" | + | | | | | | | | | |
| | | 583" | | | | | | | | | | |
| | | 607" | | | | + | | + | | | | |
| | | 608" | + | | | | | | | | | |
| | | 609" | | | | | | | | | | |
| | | 613" | + | | | | | | | | | |
| | | 621" | | | | | | | | | | |
| | | 627" | + | | | | | | | | | |
| | | 628" | + | | | | | | | | | |
| | | 631" | | | | | | | | | | |
| | | 634" | | | | | | | | | | |
| | | 635" | | + | | | | | | | | |
| | | 606" | | | | | | | | | | |
| | | 610" | + | | | | | | | | | |
| | | 620" | | | | | | | | | | |
| | | 626" | | | | | | | | | | |
| | | 633" | | | | | | | | | | |

MONS 219460

AROCLOR - 1242

Tabulation of Individual Liver Lesions in Rats

| Sacrifice Interval | Dose Level | Animal No. and Sex | Liver Lesions | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Reticulum Cell Sarcoma metastatic | Mammary tumor, metastatic |
| 24 Month | C-II | 655M | | | | | | | | | | |
| | 10ppm | 657" | + | | | | | | | | | |
| | | 662" | ++ | | | | | | | | | |
| | | 667" | | | | | | | | | | |
| | | 640" | | | | | P | | | | | |
| | | 641" | | | | | | | | | | |
| | | 643" | | | | | | | | | | |
| | | 644" | | | | | | +++ | | | | |
| | | 645" | | | | | | | | | | |
| | | 646" | | | | | | | | | | |
| | | 649" | | | | | | | | | | |
| | | 651" | + | | | | | | | | | |
| | | 673" | + | | | + | | | | | | |
| | | 686F | | | | | | | | | | |
| | | 687" | | | | | | + | | | | |
| | | 689" | | | | | | | | | | |
| | | 685" | | | | | | | | | | |
| | | 690" | | | | | | | | | | |
| | | 695" | | | | | | | | | | P |
| | | 697" | | | | | | | | | | |
| | | 700" | | | | + | | | | | | |
| | | 703" | +- | | | | | | | | | |
| | | 707" | | | | | P | | | | | |
| | | 708" | +- | + | | | | +++ | | | | |
| | | 711" | | | | | | | | | | |
| | | 712" | + | | | | | | | | | |
| | | 682" | + | | | | | | | | | |
| | | 683" | | | | | | | | | | |
| | | 691" | | | | | | | | | | |
| | | 702" | | | | + | | | | | | |

MONS 219461

AROCLOR - 1242

Tabulation of Individual Liver Lesions in Rats

| Sacrifice Interval | Dose Level | Animal No. and Sex | Liver Lesions | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Reticulum Cell Sarcoma metastatic | Mammary tumor, metastatic |
| 24 Month | C-III 100ppm | 740M | | | | | | | | | | |
| | | 722" | | | | | | | | | | |
| | | 724" | +++ | | | | P | | | | | |
| | | 744" | | | | | | | | | | |
| | | 746" | | | | + | | | | | | |
| | | 748" | ++ | | | | P | | | | | |
| | | 768F | | | | + | P | | | | | |
| | | 771" | + | | | ++ | P | | | P | | |
| | | 775" | | | | +++ | | | | | | |
| | | Ext" | +++ | | | | | | | | | |
| | | 733" | | | | | | | | | | |
| | | 765" | + | | | | | + | | | | |
| | | 766" | ++ | | | + | P | + | | | | |
| | | 767" | | | | | | | P | | | |
| | | 776" | | | | + | P | + | | | | |
| | | 773" | | | | + | P | | | | | |
| | | 786" | +++ | | | | | | | | | |
| | | 789" | | | | | | | P | | | |
| | | 793" | ++++ | | | | | | | | | |
| | | Ext" | ++ | | | + | P | | | | | |
| | Control | 8M | | | | | | + | | | | |
| | | 11" | | | | | | | | | | |
| | | 12" | | | | | | | | | | |
| | | 21" | | | | | | | | | | |
| | | 23" | | | | | | | | | | |
| | | 33" | | | | | | + | | | | |
| | | 34" | | | | | | | | | | |
| | | 35" | | | | | | | | | | |
| | | Ext" | | | | | | | | | | |
| | | 46F | | | + | | | | | | | |
| | | 47" | | | | | | | | | | |
| | | 49" | | | | | | + | | | | |
| | | 51" | | | | | | ++ | | | | |
| | | 52" | | | | | | | | | | |
| | | 56" | | | | | | | | | | |
| | | 59" | | | | | | | | | | |
| | | 62" | | | | | | | | | | |

MONS 219462

AROCLOR - 1242

Tabulation of Individual Liver Lesions in Rats

| Sacrifice Interval | Dose Level | Animal No. and Sex | Liver Lesions | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Reticulum Cell Sarcoma metastatic | Mammary tumor, metastatic | Telangiectasis, focal |
| 4 Month | Control | 63F | | | | | | | | | | | |
| | | 64" | | + | | | P | | | | | | |
| | | 66" | | | | | | | | | | | |
| | | 70" | | | | | | | | | | | |
| | | 71" | | | | | | | | | | | |
| | | Ext" | + | | | | | + | | | | | |

**Grading System**

\+ = Minimal in severity
++ = Mild in severity
+++ = Moderate in severity
++++ = Marked in severity
P = Present, no grade

Industrial BIO-TEST Laboratories, Inc
1810 FRONTAGE ROAD
NORTHBROOK, ILLINOIS 60062

REPORT TO

MONSANTO COMPANY

# TWO - YEAR CHRONIC ORAL TOXICITY STUDY WITH AROCLOR 1254 IN ALBINO RATS

## HISTOPATHOLOGICAL EVALUATION OF ADDITIONAL LIVER SECTIONS

MARCH 24, 1975

IBT NO. 641 - 06672

Industrial BIO-TEST Laboratories, Inc

REPORT TO

MONSANTO COMPANY

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## HISTOPATHOLOGICAL EVALUATION OF ADDITIONAL LIVER SECTIONS

MARCH 24, 1975

IBT NO. 641 - 06672

## I. Introduction

At the request of Dr. Levinskas of the Monsanto Company, additional sections of liver from a two - year chronic oral toxicity study of Aroclor 1254 in rats (IBT No. 622-07298) were processed into H & E stained sections and evaluated by light microscopy. The following report presents the results of this study.

II. Summary

In most instances, the spectrum of treatment-related histopathological findings in the liver from this re-evaluation did not differ significantly from that previously reported in our original report dated November 12, 1971. There were six hepatomas detected among six of the animals at the highest treatment level (100 ppm) of the 24-Month Sacrifice. No hepatocellular carcinomas were observed. One liver tumor (a hepatoma) previously reported in a T-II animal (No. 445) was re-classified as nodular hyperplasia. The other treatment-related lesions reported are regarded as degenerative or hyperplastic in nature and they are morphologic manifestations of an adaptive response of the liver ascribed to biotransformation of the test material. The latter lesions were confined primarily to test animals of the 12 and 24 month sacrifices and they were dose-related in incidence and severity.

In conclusion, Aroclor 1254 does not appear to be carcinogenic in rats fed for two years at levels up to and including 100 ppm.

Respectfully submitted,

INDUSTRIAL BIO-TEST LABORATORIES, INC.

Report Prepared and Reviewed by:

D. E. Gordon, D.V.M., Ph.D.
Section Head, Pathology

Report Approved by:

M. L. Kepinger, Ph.D.
Manager, Toxicology

aew

IBT No. 622-07298
Monsanto

I have examined additional sections of liver from rats of IBT No. 622-07298 and have re-evaluated them for evidence of carcinogenesis. I have tabulated my findings for Aroclor 1254 on the following pages:

There is evidence of a chemical effect on the liver which is most pronounced at time of the 12-Month and 24-Month sacrifice. This consists of a hepatocellular alteration beginning as focal hypertrophy which progresses to nodular hyperplasia, and in a few animals to hepatoma or cholangiohepatoma. The hypertrophic cells contain large amounts of light staining cytoplasm which is probably rich in glycogen and endoplasmic reticulum. In some cases ring shaped structures, probably representing whorls of proliferative endoplasmic reticulum, were seen in the cytoplasm of these cells.

The hyperplastic nodules occupied larger areas and often compressed surrounding cells of the normal liver parenchyma. They also contained the same hypertrophic cells.

Those lesions classified as hepatomas or cholangiohepatomas were larger nodules which showed evidence of confluence or some variation in cell size, shape, staining or a ductular or adenomatous pattern of growth. In the absence of metastasis, invasiveness, severe basophilia, mitoses, or other evidence of anaplasia; these are benign tumors rather than malignant carcinomas. There was no evidence of metastasis or invasiveness of these tumors in this study.

With exception of the vacuolar changes in the cytoplasm of hepatocytes, the more severe hepatocellular alterations were confined to animals of the 24-Month Sacrifice.

Ward R. Richter

Ward R. Richter, D.V.M., M.S.
Dipl., Am. College Vet. Path.

Primary Liver Lesions - Aroclor 1254

3 Month Sacrifice - Rats

| Lesion | Group | | | |
|---|---|---|---|---|
| | TI | TII | TIII | Control |
| Vacuolar Change | 3/10 | 3/10 | 1/10 | 2/10 |
| Focal Hypertrophy | 0/10 | 0/10 | 1/10 | 0/10 |
| Nodular Hyperplasia | 0/10 | 0/10 | 0/10 | 0/10 |
| Hepatoma | 0/10 | 0/10 | 0/10 | 0/10 |
| Ductular Hyperplasia | 0/10 | 0/10 | 0/10 | 0/10 |
| Cholangio-Hepatoma | 0/10 | 0/10 | 0/10 | 0/10 |
| Hepatocellular Necrosis | 1/10 | 1/10 | 0/10 | 0/10 |

Primary Liver Lesions - Aroclor 1254

6 Month Sacrifice - Rats

| Lesion | Group | | | |
|---|---|---|---|---|
| | TI | TII | TIII | Control |
| Vacuolar Change | 0/10 | 0/10 | 1/10 | 1/10 |
| Focal Hypertrophy | 0/10 | 0/10 | 4/10 | 0/10 |
| Nodular Hyperplasia | 0/10 | 0/10 | 0/10 | 0/10 |
| Hepatoma | 0/10 | 0/10 | 0/10 | 0/10 |
| Ductular Hyperplasia | 0/10 | 0/10 | 0/10 | 0/10 |
| Cholangio-Hepatoma | 0/10 | 0/10 | 0/10 | 0/10 |
| Hepatocellular Necrosis | 2/10 | 0/10 | 1/10 | 1/10 |

Primary Liver Lesions - Aroclor 1254

12 Month Sacrifice - Rats

| Lesion | Group | | | |
|---|---|---|---|---|
| | TI | TII | TIII | Control |
| Vacuolar Change | 1/10 | 4/10 | 1/10 | 1/10 |
| Focal Hypertrophy | 0/10 | 4/10 | 2/10 | 0/10 |
| Nodular Hyperplasia | 0/10 | 0/10 | 0/10 | 0/10 |
| Hepatoma | 0/10 | 0/10 | 0/10 | 0/10 |
| Ductular Hyperplasia | 1/10 | 1/10 | 0/10 | 1/10 |
| Cholangio-Hepatoma | 0/10 | 0/10 | 0/10 | 0/10 |
| Hepatocellular Necrosis | 0/10 | 0/10 | 1/10 | 0/10 |

Primary Liver Lesions - Aroclor 1254

Terminal Sacrifice - Rats

| Lesion | Group | | | |
|---|---|---|---|---|
| | TI | TII | TIII | Control |
| Vacuolar Change | 8/31 | 10/26 | 13/27 | 1/23 |
| Focal Hypertrophy | 3/31 | 5/26 | 13/27 | 0/23 |
| Nodular Hyperplasia | 0/31 | 3/26 | 13/27 | 1/23 |
| Hepatoma | 0/31 | 0/26 | 4/27 | 0/23 |
| Ductular Hyperplasia | 6/31 | 3/26 | 4/27 | 5/23 |
| Cholangio-Hepatoma | 0/31 | 0/26 | 2/27 | 0/23 |
| Hepatocellular Necrosis | 3/31 | 1/26 | 2/27 | 1/23 |

Tabulation of Individual Liver Lesions in Rats

| Sacrifice Interval | Dose Level | Animal No. and Sex | Liver Lesions | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Squamous Cell Carcinoma, metastatic | Mammary tumor, metastatic | Hepatitis/Cirrhosis |
| Month | B-I 1ppm | 356M | ++ | | | | | | | | | | |
| | | 357" | + | | | | | | | | | | |
| | | 358" | | | | | | | | | | | |
| | | 359" | | + | | | | | | | | | |
| | | 360" | + | | | | | | | | | | |
| | | 396F | | | | | | | | | | | |
| | | 397" | | | | | | | | | | | |
| | | 398" | | | | | | | | | | | |
| | | 399" | | | | | | | | | | | |
| | | 400" | | | | | | | | | | | |
| | B-II 10ppm | 436M | + | | | | | | | | | | |
| | | 437" | | | | | | | | | | | |
| | | 438" | + | | | | | | | | | | |
| | | 439" | | | | | | | | | | | |
| | | 440" | + | | | | | | | | | | |
| | | 476F | | + | | | | | | | | | |
| | | 477" | | | | | | | | | | | |
| | | 478" | | | | | | | | | | | |
| | | 479" | | | | | | | | | | | |
| | | 480" | | | | | | | | | | | |
| | B-III 100ppm | 516M | | | | | | | | | | | |
| | | 517" | | | | | | | | | | | |
| | | 518" | + | + | + | | | | | | | | |
| | | 519" | | | | | | | | | | | |
| | | 520" | | | | | | | | | | | |
| | | 555F | | | | | | | | | | | |
| | | 557" | | | | | | | | | | | |
| | | 558" | | | | | | | | | | | |
| | | 559" | | | | | | | | | | | |
| | | 560" | | | | | | | | | | | |

MONS 219473

Tabulation of Individual Liver Lesions in Rats

| rifice erval | Dose Level | Animal No. and Sex | Liver Lesions | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Squamous Cell Carcinoma, metastatic | Mammary tumor, metastatic |
| Month | B-I | 921M | | + | | | | | | | | |
| | 1ppm | 922" | | | | | | | | | | |
| | | 923" | | + | | | | | | | | |
| | | 924" | | | | | | | | | | |
| | | 925" | | | | | | | | | | |
| | | 936F | | | | | | | | | | |
| | | 937" | | | | | | | | | | |
| | | 938" | | | | | | | | | | |
| | | 939" | | | | | | | | | | |
| | | 940" | | | | | | | | | | |
| | B-II | 950M | | | | | | | | | | |
| | 10ppm | 951" | | | | | | | | | | |
| | | 952" | | | | | | | | | | |
| | | 953" | | | | | | | | | | |
| | | 954" | | | | | | | | | | |
| | | 966F | | | | | | | | | | |
| | | 967" | | | | | | | | | | |
| | | 968" | | | | | | | | | | |
| | | 969" | | | | | | | | | | |
| | | 970" | | | | | | | | | | |
| | B-III | 981M | | + | + | | | | | | | |
| | 100ppm | 982" | | | + | | | | | | | |
| | | 983" | | | | | | | | | | |
| | | 984" | | | | | | | | | | |
| | | 985" | | | | | | | | | | |
| | | 996F | | | | + | | | | | | |
| | | 997" | | | | + | | | | | | |
| | | 998" | | | | | | | | | | |
| | | 999" | | + | | + | | | | | | |
| | | 1000" | | | | + | | | | | | |

MONS 219474

AROCLOR - 1254

Tabulation of Individual Liver Lesions in Rats

| Sacrifice Interval | Dose Level | Animal No. and Sex | Liver Lesions | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Squamous Cell Carcinoma, undifferentiated | Mammary tumor, metastatic | [illegible] |
| 2 Month | B-I 1ppm | 41M | + | | | | | | | | | | |
| | | 42" | | | | | | | | | | | |
| | | 43" | | | | | | | | | | | |
| | | 44" | | | | | | | | | | | |
| | | 45" | | | | | | | | | | | |
| | | 46F | | | | | | | | | | | |
| | | 47" | | | | | | | | | | | |
| | | 48" | | | | | | + | | | | | |
| | | 49" | | | | | | | | | | | |
| | | 50" | | | | | | | | | | | |
| | B-II 10ppm | 51M | | | | | | | | | | | |
| | | 52" | ++ | | | | | | | | | | |
| | | 53" | ++ | | | | | | | | | | |
| | | 54" | | | + | | | | | | | | |
| | | 55" | + | | | | | | | | | | |
| | | 56F | | | + | | | | | | | | |
| | | 57" | + | | | + | | | | | | | |
| | | 58" | | | | + | | | | | | | |
| | | 59" | | | | + | | | | | | | |
| | | 60" | | | | + | | + | | | | | |
| | B-III 100ppm | 61M | | | | + | | | | | | | |
| | | 62" | | | | | | | | | | | |
| | | 63" | | | | | | | | | | | |
| | | 64" | | | | | | | | | | | |
| | | 65" | ++ | | | + | | | | | | | |
| | | 66F | | | | | | | | | | | |
| | | 67" | | | + | | | | | | | | |
| | | 68" | | | | | | | | | | | |
| | | 69" | | | | | | | | | | | |
| | | 70" | | + | | | | | | | | | |

MONS 219475

AROCLOR - 1254

Tabulation of Individual Liver Lesions in Rats

| rifice erval | Dose Level | Animal No. and Sex | Liver Lesions | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Squatious Cell Carcinoma, metastatic | Pulmonary tumor, metastatic |
| Month | B-I 1ppm | 326M | | | | | | | | | | |
| | | 330" | | | | | | | | | | |
| | | 335" | | | | | | | | | | |
| | | 338" | | | | | | | | | | |
| | | 351" | ++ | + | | | | | | | | |
| | | 323" | | + | | | | | | | | |
| | | 325" | | | | | | | | | | |
| | | 343" | + | | | | | | | | | |
| | | 346" | | | + | + | | | | | | |
| | | 342F | | | | | | | | | | |
| | | 362" | | | | | | | | | | |
| | | 368" | | | | | | + | | | | |
| | | 374" | | | | | | | | | | |
| | | 375" | | | | | | + | | | | |
| | | 376" | | | | + | | | | | | |
| | | 475" | +++ | | | | | | | | | |
| | | 366" | | | | | | + | | | | |
| | | 367" | | | | | | | | | | |
| | | 372" | | | | | | + | | | | |
| | | 377" | ++ | ++ | | + | | + | | | | |
| | | 378" | + | | | | | | | | | |
| | | 379" | + | | | | | | | | | |
| | | 380" | | | | | | | | | | |
| | | 381" | | | | | | | | | | |
| | | 384" | | | | | | | | | | |
| | | 387" | | | | | | | | | | |
| | | 388" | | | | | | | | | | |
| | | 389" | | | | | | | | | | |
| | | 391" | + | | | | | + | | | | |
| | | 394" | +++ | | | | | | | | | |
| | | Ext" | | | ++ | | | | | | | |

MONS 219476

Tabulation of Individual Liver Lesions in Rats

| rifice erval | Dose Level | Anima No. and Sex | Liver Lesions | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocytes | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Squamous Cell Carcinoma, metastatic | Mammary tumor, metastatic |
| Month | B-II | 415M | | | | | | | | | | |
| | 10ppm | 417" | | | | ++ | | + | | | | |
| | | 418" | | | | | | | | | | |
| | | 419" | | | | | | | | | | |
| | | 423" | | | | | | | | | | |
| | | 426" | ++ | | | | | | | | | |
| | | 427" | + | | | | | | | | | |
| | | 491" | | | | | | | | | P | |
| | | 402" | + | | | | | | | | | |
| | | 403" | ++ | | | | | | | | | |
| | | 405" | | | | | | | | | | |
| | | 410" | | | | | | | | | | |
| | | 434" | + | | | | | | | | | |
| | | 450F | | | | | | | | | | |
| | | 451" | + | | | | P | | | | | |
| | | 452" | + | + | | | P | | | | | |
| | | 456" | | | | + | | | | | | |
| | | 459" | | | | ++ | | ++ | | | | |
| | | 463" | ++ | | | | | | | | | |
| | | 469" | | | | + | | | | | | |
| | | 470" | | | | | | | | | | |
| | | 471" | | | | | | | | | | |
| | | 478" | | | | | | + | | | | |
| | | 443" | | | | | | | | | | |
| | | 445" | | | | | P | | | | | |
| | | 448" | | | | | | | | | | |

MONS 219477

AROCLOR - 1254

Tabulation of Individual Liver Lesions in Rats

| Office Interval | Dose Level | Animal No. and Sex | Liver Lesions: Cytoplasmic vacuolation of hepatocytes | Focal necrosis of hepatocytes | Focal lymphoid infiltrations | Focal hypertrophy of hepatocyles | Nodular hyperplasia of hepatocytes | Focal bile duct hyperplasia | Hepatoma | Cholangio-hepatoma | Squamous Cell Carcinoma, metastatic | Mammary tumor, metastatic | [illegible] |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Month | B-III 100ppm | 484M | ++ | | | | | | | | | | +/ |
| | | 491" | | | | + | P | | P | | | | |
| | | 497" | | | | ++ | P | | | | | | |
| | | 499" | | | | + | P | | | | | | |
| | | 507" | | | | | P | | | | | | +/ |
| | | 509" | | | | | | | | | | | |
| | | 510" | + | | | | | | | | | | |
| | | 512" | | | | | | | | | | | |
| | | 483" | ++ | | | | | + | | | | | |
| | | 489" | + | | | + | | | | | | | |
| | | 522F | ++ | | | ++ | | | | | | | |
| | | 524" | ++ | | | + | P | | | | | | |
| | | 527" | +++ | | | + | P | | | | | P | |
| | | 541" | | | | + | | | P | | | | |
| | | 547" | ++ | | | | P | | | | | | |
| | | Ext1" | ++ | | | ++++ | P | | | | | | |
| | | Ext2" | | | | | P | + | | P | | | |
| | | Ext3" | ++ | | | +++ | | | | | | | |
| | | Ext4" | | | | | P | | P | | | | |
| | | Ext5" | | + | | | P | | | | | | |
| | | 536" | ++ | | | +++ | P | | | | | | |
| | | 537" | | | | | | | | | | | |
| | | 539" | ++ | | | ++ | | + | | | | | |
| | | 441" | | | | | | | | | | | |
| | | Ex1" | | | | | | | P | | | | |
| | | Ex2" | ++ | | | | | | | P | | | |
| | | Ex3" | | | | + | P | + | | | | | |

Grading System

+ = minimal in severity
++ = mild in severity
+++ = moderate in severity
++++ = marked in severity
P = Present, no grade
- = Absent

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