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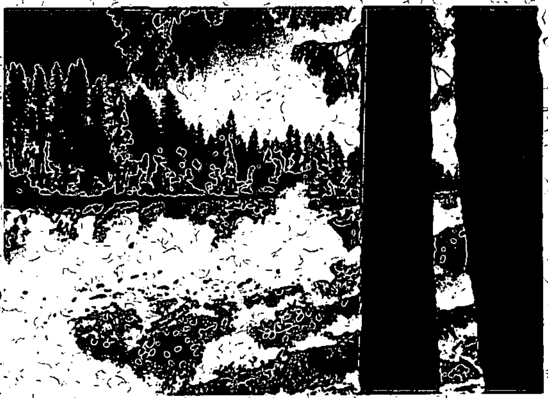
Canadian Water Quality Guidelines for Polychlorinated Biphenyls in Coastal and Estuarine Waters

D.R.J. Moore and S.L. Walker



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Canadian Water Quality Guidelines for PCBs in Coastal and Estuarine Waters
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P.23 References are:
1) Hansen *et al.* 1971
2) Hansen *et al.* 1974
3) Schimmel *et al.* 1974
4) Roesijadi *et al.* 1976a

Caption should read data set not date set

P.26 Mackay not MacKay

P.58 Duration units are days (d) not hours (h)

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Contents

Page

Contents (cont'd)

Contents (cont'd)

Tables

Illustrations

Abstract

A literature review was conducted on the physical and chemical properties of polychlorinated biphenyls (PCBs), their concentrations in the marine environment, their fate and persistence, their bioaccumulation, and their toxicity to marine biota. The information is summarized in this publication. From it, a Canadian water quality guideline is developed for PCBs to ensure the protection and maintenance of marine aquatic biota.

Résumé

On a fait une étude de la documentation sur les propriétés physiques et chimiques des polychlorobiphényles (PCB), leurs concentrations dans l'environnement marin, leur devenir et persistance, leur bioaccumulation, et leur toxicité pour le biote marin. On trouvera dans cette publication un résumé de l'information sur le sujet. Cette information a servi à l'élaboration de recommandations relatives aux PCB afin de protéger et préserver le biote aquatique marin.

Preface

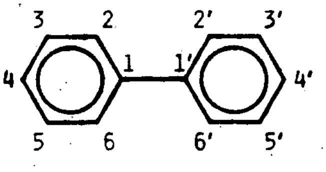
Polychlorinated biphenyls (PCBs) are a complex group of chlorinated hydrocarbon compounds that, because of their long-term production, multiplicity of uses, and environmental persistence, are widespread in the environment. Serious concerns about PCB contamination of the environment have been raised because of their persistence in the environment, their tendency to bioaccumulate in biota, and their chronic toxicity effects on biota. These concerns have resulted in Canadian government actions to halt the import and production of PCBs (Strachan, 1988) and ensure their safe disposal (e.g., Canadian Environmental Control Newsletter, 1988; Environment Canada, 1988). In Canada (CCREM, 1987) and the United States (U.S. EPA, 1980), water quality guidelines have been established at levels that ensure the protection of freshwater aquatic biota. However, whereas a water quality guideline of 0.03 $\mu g \cdot L^{-1}$ for PCBs has been established by the United States for marine aquatic biota (U.S. EPA, 1980), none has yet been established for Canada. The purpose of this report is to develop a Canadian water quality guideline for PCBs that ensures the protection and maintenance of marine aquatic biota by considering the physical and chemical properties of PCBs, their concentrations in the marine environment, their fate and persistence, their bioaccumulation, and their toxicity to marine biota.

Canadian Water Quality Guidelines for Polychlorinated Biphenyls in Coastal and Estuarine Waters

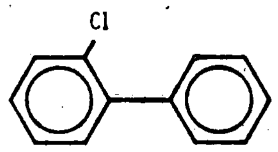
D.R.J. Moore and S.L. Walker

INTRODUCTION AND BACKGROUND

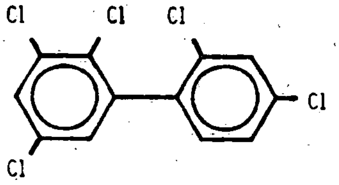
Polychlorinated biphenyls (PCBs) is the generic term for a group of 209 congeners that contain a varying number of substituted chlorine atoms in a biphenyl ring (Appendix B) (Tanabe, 1988). The potential positions for chlorine substitution are numbered according to the American Chemical Society standard notation (Fig. 1), and PCBs having the same number of chlorine atoms per biphenyl ring are grouped into a specific class and accorded a numeric prefix to define the number of substitutions (Table 1) (Rappe and Buser, 1980; Ghirelli *et al.*, 1983). Hence, tetrachlorobiphenyl has four chlorine atoms substituted on a biphenyl ring.



THE BIPHENYL MOLECULE



A MONOCHLOROBIPHENYL
(2-Chlorobiphenyl)



A PENTACHLOROBIPHENYL
(2,2',3,4',5-Pentachlorobiphenyl)

Figure 1. Numbering system and nomenclature for PCBs.

Table 1. Number of Possible Isomers and Percent Chlorine for the Ten Chlorobiphenyl Classes

Chlorobiphenyl	Empirical formula	No. of possible isomers	Percent chlorine by weight
Biphenyl	$C_{12}H_{10}$	1	0
Monochlorobiphenyl	$C_{12}H_9Cl$	3	18.8
Dichlorobiphenyl	$C_{12}H_8Cl_2$	12	31.8
Trichlorobiphenyl	$C_{12}H_7Cl_3$	24	41.3
Tetrachlorobiphenyl	$C_{12}H_6Cl_4$	42	48.6
Pentachlorobiphenyl	$C_{12}H_5Cl_5$	46	54.3
Hexachlorobiphenyl	$C_{12}H_4Cl_6$	42	58.9
Heptachlorobiphenyl	$C_{12}H_3Cl_7$	24	62.8
Octachlorobiphenyl	$C_{12}H_2Cl_8$	12	66.0
Nonachlorobiphenyl	$C_{12}HCl_9$	3	68.7
Decachlorobiphenyl	$C_{12}Cl_{10}$	1	71.2

Source : U.S. EPA, 1980.

Commercially, PCBs are produced as complex mixtures of chlorobiphenyls whose usage is primarily determined by the percentage of chlorine in the mixture. Mixtures containing 21%–54% chlorine by weight are used extensively in closed electrical systems as dielectric fluids (U.S. EPA, 1980). Both high and low chlorinated mixtures are used in a variety of other applications, including plasticizers, heat transfer fluids, hydraulic fluids, fluids in vacuum pumps and compressors, lubricants, wax extenders, and special adhesives, and as surface coatings for carbonless copy paper (Peakall and Lincer, 1970; CCREM, 1986). All these latter uses were curtailed in the United States in 1971 (U.S. EPA, 1980) and in Canada in 1977 (Strachan, 1988). In Canada, PCBs are currently used only in closed electrical equipment manufactured prior to 1980 (Strachan, 1988). The most common trade names under which PCB fluids have been sold include Aroclor, Askarel, Kanechlor, and Phenoclor (see Table 2 for other trade names) (McDonald and Tourangeau, 1986). All PCBs manufactured in North America were manufactured by Monsanto Co. under the trade name Aroclor. The

Table 2. Trade Names of Various PCB Formulations

Apirolio
Aroclor (U.S. & Can.)
Aroclor B
Asbestol
Askarel
Chlophen (Germany)
Chlorextol
Chlorinol
Diaclor
DK Decachlorobiphenyl
Dykanol
Elemex
Eucarel
Fenclor (Italy)
Inclor
Inerteen (U.S. & Can.)
Kanechlor (Japan)
Nepolin
Phenoclor (France)
Pydraul (U.S.)
Pyralene (France)
Pyranol (U.S. & Can.)
Pyroclor (Britain)
Saf-t-kuhl
Santotherm (Japan)
Santovec 1
Santovec 2
Sovol
Therminol Fr Series

Source: McDonald and Tourangeau, 1986.

commercially available Aroclor mixtures are designated by numbers such that the first two digits represent molecular type (12 = chlorinated biphenyls) and the last two digits represent percent chlorine by weight (Peakall and Lincer, 1970; Kalmaz and Kalmaz, 1979). Thus, Aroclor 1242 is a mixture of chlorinated biphenyls with 42% chlorine by weight.

Production

PCBs were first synthesized by Schmidt and Schulz in 1881 and have been in commercial production since 1929. From the mid-1950s, worldwide PCB production increased substantially to a peak at the end of the 1960s (Tanabe, 1988). Altogether, approximately 1.2 million tonnes of PCBs have been produced worldwide, of which only 4% has been degraded or incinerated, 65% is land stocked and potentially available to the environment, and another 31% has already been released to the environment (Hansen, 1987; Tanabe, 1988). With the discovery of widespread PCB contamination in the environment (Jensen *et al.*, 1969), production sharply declined through the 1970s and 1980s, although several countries, most notably the Soviet Union and Japan, still manufacture PCBs in significant quantities (Tanabe, 1988).

In North America, approximately 635 000 t of PCBs were produced before their import and manufacture were banned (CCREM, 1986). Of this total, 40 000 t were imported into Canada (no PCBs were manufactured in Canada) (Environment Canada, 1985). At present, 24 300 t are accounted for in storage for disposal or are in active use, mostly in electrical transformers (Table 3) (Environment Canada, 1985). Thus, 40% of all PCBs imported to Canada are unaccounted for, and most have probably been dispersed to the environment.

Table 3. PCB Inventory by Application for Canada

Application	Quantity (t)	% total
Electrical		
Capacitor	2 793.6	11.5
Transformer	14 852.8	61.1
Electromagnet	55.9	0.2
Other	79.4	0.3
Mechanical		
Hydraulic	17.5	0.1
Heat transfer	0.1	0.0004
Vacuum equipment	1.0	0.004
Storage for disposal	6 510.0	26.8

Source: Environment Canada, 1985.

Review of Guidelines and Legislation

Guidelines

The U.S. EPA (1980) PCB criterion to protect freshwater aquatic life is 0.014 $\mu g \cdot L^{-1}$ as a 24-h average. The corresponding marine water criterion is 0.03 $\mu g \cdot L^{-1}$. However, the authors of the U.S. EPA (1980) document stated that these criteria were probably an order of magnitude too high, because each criterion was based on bioconcentration factors measured in the laboratory, which underestimate field bioconcentration factors by at least 10 times. Several U.S. states (e.g., Indiana, Ohio, Pennsylvania) have more conservative water quality criteria for the protection of freshwater aquatic life (i.e., 0.001 $\mu g \cdot L^{-1}$) (Richardson, 1987).

The Canadian freshwater PCB guideline to protect aquatic life is 0.001 $\mu g \cdot L^{-1}$ (CCREM, 1987). This guideline is equivalent to the water quality objectives set by the International Joint Commission (IJC) for the Great Lakes and those set by the provinces of Quebec and Ontario for fresh water (Richardson, 1987). Canadian PCB guidelines for the protection of marine aquatic life do not yet exist.

Legislation

In Canada, PCBs first received serious attention after a 1976 task force report. Based on this report, the Canadian government passed legislation in 1977, under the new Environmental Contaminants Act (Strachan, 1988), banning all import, manufacture, and use of those PCBs containing three or more chlorine atoms (except for use as dielectric or heat transfer fluids). In 1980, this legislation was revised to prohibit the use of PCBs in new electrical equipment or for servicing existing equipment. Additional regulations implemented in 1985 restricted PCBs to dielectric fluids in existing electrical equipment. This equipment is being phased out of use, and all equipment using PCBs should be out of service within several decades (Strachan, 1988). PCBs appear on the Canadian Environmental Protection Act (CEPA) Toxic Substances List (Canada Gazette, 1989).

PHYSICAL AND CHEMICAL PROPERTIES

There are 209 possible PCB isomers and congeners for the biphenyl molecule, containing between 1 and 10 chlorine atoms (Appendix B) (Hansen, 1987; Safe *et al.*, 1987). Commercial production involves random chlorination, such that any given formulation can have from 40 to 80 individual PCB congeners (Safe *et al.*, 1987). Because of varying manufacturing conditions, congener profiles for each of the PCB formulations (e.g., Aroclor 1260) vary with time (Stout, 1986). Therefore, PCB formulations are not exact substances, which has three consequences: (1) the physical and chemical properties of PCB congeners and formulations vary; (2) analytical procedures are often inaccurate, and international collaborative studies have noted wide variation in identifying and quantifying PCB congeners from standard samples (Stout, 1986); and (3) similar formulations vary markedly in their biological activity, particularly in their partitioning behaviour, persistence/degradability, and toxicity (Hansen, 1987). In addition, few studies have considered the effects of salinity on the physical and chemical properties of PCBs. Thus, the detailed behaviour of PCBs in the marine environment is largely unknown.

With these points in mind, this section of the report will describe the physical and chemical properties of PCBs in the marine environment, the analytical methods for the detection of PCBs, and the methodological limits of detection.

Appearance

Chlorobiphenyls are colourless crystals when isolated in pure form by recrystallization (Rappe and Buser, 1980). However, when the biphenyl is catalytically chlorinated with anhydrous chlorine during commercial manufacture, the melting point of the resulting formulation is lowered and a liquid is produced (Rappe and Buser, 1980; Ghirelli *et al.*, 1983). Lower chlorinated PCB formulations (Aroclor 1016, 1221, 1232, 1242, 1248) are colourless mobile oils (U.S. EPA, 1980). Higher chlorinated formulations are yellow viscous liquids (Aroclor 1254), black resins (Aroclor 1260, 1262), or white powders (Aroclor 1268, 1270) (U.S. EPA, 1980).

Molecular Composition of Aroclor Formulations

The lower chlorinated Aroclor formulations (1221, 1232, 1016) are composed primarily of mono-, di-, and trichlorobiphenyls (Table 4) (Rappe and Buser, 1980). Aroclor 1016 was introduced by Monsanto Co. in 1971 to replace the more biologically persistent Aroclor 1242; it differs from the latter in having the penta- and higher chlorobiphenyls removed (Ghirelli *et al.*, 1983). With increasing chlorine content, mono-, di-, and trichlorobiphenyls gradually disappear from the formulations; these compounds are absent in Aroclors 1254 and 1260, which are rich in penta-, hexa-, and heptachlorobiphenyls (Table 4) (U.S. EPA, 1980).

Table 4. Approximate Molecular Composition (%) of Aroclor Mixtures

Chlorobiphenyl	1221	1232	1016	1242	1248	1254	1260
Biphenyl	11.0	6.0	Tr	–	–	–	–
Monochlorobiphenyl	51.0	26.0	1.0	1.0	–	–	–
Dichlorobiphenyl	32.0	29.0	20.0	17.0	1.0	–	–
Trichlorobiphenyl	4.0	24.0	57.0	40.0	23.0	–	–
Tetrachlorobiphenyl	2.0	15.0	21.0	32.0	50.0	16.0	–
Pentachlorobiphenyl	0.5	0.5	1.0	10.0	20.0	60.0	12.0
Hexachlorobiphenyl	–	–	Tr	0.5	1.0	23.0	46.0
Heptachlorobiphenyl	–	–	–	–	–	1.0	36.0
Octachlorobiphenyl	–	–	–	–	–	–	6.0
Nonachlorobiphenyl	–	–	–	–	–	–	–
Decachlorobiphenyl	–	–	–	–	–	–	–

Source: U.S. EPA, 1980.

Tr = trace (<0.01%)

Molecular Weight

The molecular weights of the Aroclor formulations range from 192 g·mol^{-1} for Aroclor 1221 to 372 g·mol^{-1} for Aroclor 1260 (Table 5) (Mackay *et al.*, 1983). The molecular weights for chlorobiphenyl congeners range from 189 to 499 g·mol^{-1} for mono- and decachlorobiphenyls, respectively.

Boiling Point

The boiling points for the Aroclor formulations and specific PCB congeners are listed in Table 5. In general, the boiling point increases by 29°C for the first chlorine added and then by increments that are 2°C lower for each subsequent chlorine addition (Mackay *et al.*, 1983). In general, Aroclors do not crystallize upon heating or cooling but do become resinous at a specific temperature known as the "pour point" (Table 5) (U.S. EPA, 1980).

Table 5. Physical and Chemical Properties of PCB Congeners and Aroclor Formulations

Compound	Molecular weight (g•mol^{-1})$^{(1)}$	Pour point (°C)$^{(2)}$	Boiling point (°C)$^{(1)}$	Vapour pressure (mm Hg*)$^{(2)}$	Henry's law constant (Pa•m^3•mol^{-1})$^{(1)}$	Solubility (μg•L^{-1})$^{(1)(2)}$	log $K_{ow}$$^{(1)}$
Mono CBP†	189	–	285	–	60	7200	4.66
Di CBP	223	–	312	–	60	2200	5.19
Tri CBP	257	–	337	–	77	670	5.76
Tetra CBP	292	–	360	–	76	230	–
Penta CBP	326	–	381	–	68	72	–
Hexa CBP	361	–	400	–	86	21	–
Hepta CBP	395	–	417	–	–	6	–
Octa CBP	430	–	432	–	–	2	–
Nona CBP	464	–	445	–	–	0.7	–
Deca CBP	499	–	456	–	–	0.2	–
Aroclor 1221	192	–	275–320	–	–	200	4.1–4.7
Aroclor 1232	221	-35.5	290–320	–	–	–	4.5–5.2
Aroclor 1016	257	–	325–356	–	–	225–250	4.4–5.8
Aroclor 1242	261	-19.0	325–366	4.06×10^{-4}	–	240	4.5–5.8
Aroclor 1248	288	-7.0	340–375	4.94×10^{-4}	–	54	5.8–6.3
Aroclor 1254	327	10.0	365–390	7.71×10^{-5}	–	12	6.1–6.8
Aroclor 1260	372	35–38.0	385–420	4.05×10^{-5}	–	2.7	6.3–7.5

(1) Mackay *et al.* (1983).
(2) U.S. EPA (1980).

* 1 mm Hg = 133.32 Pa
† CBP = Chlorobiphenyl

Volatility

Volatility is a difficult property to determine because there is no currently accepted means of measurement and because PCB formulations are mixtures of congeners that may volatilize at different rates. Laboratory studies often determine vapour pressure as an estimate of a compound's potential volatility in the environment. However, other factors in the environment, such as depth of the water body, air and water movement, concentration of PCBs in the water, sediment organic content, and presence of particulates in the water, all affect the volatility of PCB formulations.

Generally, vapour pressures of PCBs from solids fall by a factor of 2–3 per chlorine added (Mackay *et al.*, 1983) and range from 8.4×10^{-3} mm Hg for 2-monochlorobiphenyl to 3.7×10^{-5} mm Hg for 2,4,2',4'-tetrachlorobiphenyl at 25°C (1 mm Hg=133.32 Pa) (Neely, 1983). It has been suggested that sterically hindered isomers (those with chlorine atoms in the ortho position) such as 2,2'-dichlorobiphenyl have higher vapour pressures than do PCBs possessing no internal strain (e.g., 3,3'-dichlorobiphenyl) (Neely, 1983; Hansen, 1987). As with PCB congeners, the vapour pressure of PCB formulations decreases as chlorine content increases (U.S. EPA, 1980; Hansen, 1987).

Together with solubility, vapour pressures can be used to calculate an air/water partition coefficient (Henry's law constant). Using this constant, one can calculate rates of loss of PCB congeners from a water body. In general, Henry's law constant remains unchanged with chlorine number and is in the range of 60–86 Pa·m^3·mol^{-1} (Table 5) (Mackay *et al.*, 1983). These values would imply that PCB volatilization from water to air is relatively "fast" and that PCB congener distributions in water and air should be similar. However, environmental observations do not support these predictions, implying that other factors have important effects on the volatilization rates of PCBs. In natural aquatic environments, PCBs are strongly adsorbed to sediments and dissolved particulate matter (Biggs *et al.*, 1980; Harding and Addison, 1986), and this factor likely reduces rates of PCB volatilization (U.S. EPA, 1980). A better understanding of mass fluxes of PCBs in the environment will require further testing of the assumptions and data used to calculate Henry's law constant (Strachan, 1988).

Solubility

The solubility of PCB congeners in fresh water is extremely low and ranges from 7200 μg·L^{-1} for monochlorobiphenyl to 0.2 μg·L^{-1} for decachlorobiphenyl (Table 5) (Mackay *et al.*, 1983). In general, solubilities decline by a factor of 2–3 per chlorine added (Mackay *et al.*, 1983). This decline is not linear, however, because the steric effects of ortho-substituted chlorines increase the water solubility of these isomers (Hansen, 1987). The solubility of commercial formulations in fresh water ranges from 225–250 μg·L^{-1} for Aroclor 1016 to 2.7 μg·L^{-1} for Aroclor 1260 (Table 5) (U.S. EPA, 1980).

Because of the extremely non-polar nature of PCBs, their solubility in water is inversely related to salinity, such that a 25% reduction in solubility is predicted for a 33 ppt increase in salinity (Wiese and Griffin, 1978). However, there is conflicting evidence as to the "true" solubility of PCBs in salt water (Schoor, 1975). Reported solubilities of Aroclor 1254 in fresh water range from less than 0.1 μg·L^{-1} (Schoor, 1975) to 3000 μg·L^{-1} (Zitko, 1970) and in salt water from less than 0.04 μg·L^{-1} (Schoor, 1975) to 1500 μg·L^{-1} (Zitko, 1970). The differences arise from the fact that at low concentrations, long ultracentrifugation times are required to eliminate stable PCB aggregates from the emulsion (Schoor, 1975; Ghirelli *et al.*, 1983). Thus, the "true" solubility of PCBs in salt water is extremely difficult to determine but probably tends towards the lower reported values.

PCBs are readily soluble in non-polar organic solvents or in lipid-rich biological systems (U.S. EPA, 1980). To predict the partitioning behaviour of PCBs in the aquatic environment, one can use the 1-octanol/water partition coefficient (K_{ow}), which is a measure of the capacity of a molecule to pass from aqueous to non-polar phases (Shaw and Connell, 1984). Reported log K_{ow} values range from 4.1–4.7 for Aroclor 1221 to 6.3–7.5 for Aroclor 1260 (Mackay *et al.*, 1983). Because PCBs have very high K_{ow} values, they preferentially partition to the bottom sediments and the fat stores of aquatic biota (Kalmaz and Kalmaz, 1979). However, caution should be exercised before using K_{ow} to predict the detailed partitioning behaviour of PCBs, because congener structure and the rates at which congeners are metabolized have important consequences on their relative concentrations in water, sediment, and biota (Shaw and Connell, 1984; Farrington *et al.*, 1986).

Analytical Methodologies

Sample Purification

The methods for extracting PCBs from a sample and the clean-up procedure involved depend on the nature of the sample: air, water, sediment, or biota. Treated resins such as Amberlite, XAD-2, and Tenax have proved useful for extraction of PCBs from water. To separate PCBs from other organochlorine compounds, column chromatography on Florisil, silica gel, alumina, and charcoal has been successfully utilized (Rappe and Buser, 1980).

Separation of PCBs

When determining PCB residue levels, one should note that PCBs are complex mixtures composed of individual PCB congeners behaving in different ways. For example, low-molecular-weight compounds evaporate and degrade much more readily than high-molecular-weight compounds (Stout, 1986). With this point in mind, there are three major approaches to the characterization and quantification of PCBs (Table 6) (Safe *et al.*, 1983). Most analytical schemes utilize packed column gas chromatography (GC) for separation of PCB components and electron capture (EC) detection for quantification (Rappe and Buser, 1980; Safe *et al.*, 1987; Hansen, 1987). However, the low resolving power of this technique does not

Table 6. Characteristics of Low-, Medium-, and High-Resolution PCB Analyses

Separation	Detection	Quantification	Resolution
Packed column	EC	Peaks(s) or pattern matching	Low
Packed column	MS	Peaks(s) or pattern matching	Medium
Capillary column	EC or MS	Isomer–congener response	High

Source: Safe *et al.*, 1983.

separate the peaks of many PCB congeners (Safe *et al.*, 1987). Therefore, the technique has several disadvantages, including difficulty in quantifying severely degraded sample residues, difficulty in quantifying individual isomers, and a lack of data on potentially toxic PCB isomers (Safe *et al.*, 1983). Differences between laboratories are significant because of variability in peak selection, signal quantification (peak height vs. area), and inconsistent use of reference standards (Stout, 1986; Farrington *et al.*, 1986; Schwartz *et al.*, 1987). Studies have revealed up to 100-fold differences in the quantification of PCBs from standard samples (Stout 1986).

GC–mass spectrometry (MS) is a medium-resolution technique often used to confirm the molecular weights of individual PCB congeners and to identify co-contaminants such as dichlorodiphenyltrichloroethane (DDT), dichlorodiphenyldichloroethylene (DDE), hexachlorocyclohexane (lindane), and other organochlorine compounds (Safe *et al.*, 1987). Although this technique represents an improvement over the previous technique, it also suffers from many of the same disadvantages (Safe *et al.*, 1983).

Within the last decade, high-resolution isomer-specific PCB analysis has been developed as a viable analytical technique (Safe *et al.*, 1983). The technique uses high-resolution glass capillary column separation coupled with EC or MS detection and requires all 209 PCB congeners as reference standards (Safe *et al.*, 1983, 1987; Stout, 1986; Hansen, 1987). This analysis provides more accurate information regarding congener composition, although incorrect identification of congeners remains a problem (Stout, 1986). Further, the degree of sophistication required for this technique precludes its use in large-scale monitoring programs (Strachan, 1988).

At the National Water Quality Laboratory (NWQL) and National Water Research Institute (NWRI) in Canada, polychlorinated biphenyls are generally determined and reported as total PCBs by packed column GC (NWQL, 1985, 1987). A capillary column may be used first to qualitatively determine if the sample contains any PCBs. Quantitative analyses are then performed with a packed OV-101 column and compared with a standard 1:1:1 mixture of Aroclors 1242, 1254, and 1260 at a concentration of 200 $\mu g \cdot L^{-1}$.

The techniques described above and their associated difficulties point to an intrinsic analytical imprecision in characterizing and quantifying PCB residue levels in the environment. When these difficulties are combined with the knowledge that PCB levels vary with species, age, and size of specimens, sex, tissue type, timing of collection, and collection site (Stout, 1986), the enormous difficulty in assessing PCB pollution trends can be understood. Only intensive, regular sampling of individual PCB congeners will allow for fine-scale comparisons between sites and over time.

Limits of Detection

It has been determined that other organochlorine compounds may interfere in PCB analyses at levels below 0.01 $\mu g \cdot L^{-1}$ for water (Rappe and Buser, 1980) and 100 $\mu g \cdot kg^{-1}$ for sediment and biological tissue (Strachan, 1988). Levels down to 10%–20% of these values are generally reported as "trace" values (Strachan, 1988). Lower limits of detection are possible (e.g., by use of mass spectrometry), but the effort required precludes large-scale sampling programs (Rappe and Buser, 1980; Strachan, 1988).

At NWQL and NWRI, the detection limits for total PCBs are 0.02 $\mu g \cdot L^{-1}$ for water and 90 $\mu g \cdot kg^{-1}$ for sediment and biological tissue (NWQL, 1985, 1987). The sediment detection limit can be reduced to 10 $\mu g \cdot kg^{-1}$ by use of an ultrasonic extraction procedure.

ENVIRONMENTAL CONCENTRATIONS

Sources

PCB contamination in the marine environment has arisen exclusively from human activities. Consequently, areas of high contamination tend to occur near heavily industrialized areas (O'Connor and Huggett, 1988), such as the Hudson River estuary (Califano *et al.*, 1982) and Puget Sound (PTI Environmental Services, 1988). In Canada, PCB contamination of the marine environment is also highest near the more industrialized areas, such as the St. Lawrence estuary (Couillard, 1982), the Bay of Fundy (Gaskin *et al.*, 1983), and the Fraser River estuary (Rogers and Hall, 1987).

Because PCB congeners preferentially bind to sediments, transport through water from sources to distant sites is relatively slow. However, when the gradual transport of large quantities of PCBs through water is combined with the rapid transport of small quantities of PCBs through air, dispersion of PCBs to remote areas is inevitable (Tanabe, 1988). The widespread marine contamination of PCBs is evidenced by their detection in water, sediments, and biota from areas as remote as the Antarctic Ocean (Tanabe *et al.*, 1983; Subramanian *et al.*, 1986; Tanabe, 1988).

The principal route of PCB transport through the aquatic environment is from waste streams to receiving streams, downstream movement by means of solution and readsorption onto particulates and by the transport of sediment itself, until eventually reaching the coastal estuaries and open ocean (Kalmaz and Kalmaz, 1979). Evidence for this route of PCB movement through the environment can be gained by examining the global distribution of PCBs in the environment. At present, the total amount of PCBs released to the global environment is estimated to be 370 000 t, of which approximately 360 000 t (96% of total load) are retained in the coastal sediments (35% of total load) and open ocean water (61% of total load) (Tanabe, 1988). Further, despite a significant downward trend in PCB levels in many freshwater lakes and streams since the early 1970s (Noble and Elliott, 1986; Stout, 1986), the declines that have occurred in the marine environment have been modest (Young and Heesen, 1978; Olsson, 1987; Phillips and Spies, 1988; Andersson *et al.*, 1988; Tanabe, 1988). Therefore, the marine environment is the final sink for PCBs, and the problem of contamination of the marine environment is likely to continue long after major reductions of the dominant inputs have occurred.

PCBs enter the marine environment from a variety of sources. The purpose of this section is to briefly document several of the major sources of PCB contamination in the marine environment.

Air

Atmospheric transport and redeposition of PCBs are the primary mechanisms leading to PCB contamination of remote marine environments (Phillips, 1986; Tanabe, 1988). These mechanisms may account for up to 90% of the total PCBs dispersed to these areas (Ghirelli *et al.*, 1983). Although the atmosphere is an important transport mechanism for PCBs, concentrations in this medium are generally low. Major inputs of PCBs to the atmosphere are from incomplete incineration of PCB-contaminated landfill sites and as a result of volatilization of PCBs from contaminated aqueous sources (Ghirelli *et al.*, 1983; Strachan, 1988).

PCB dispersion by air is affected by the volatility of individual PCB isomers, prevailing winds and precipitation patterns, and possible photodegradation of higher chlorinated isomers (Ghirelli *et al.*, 1983).

Landfills

The land disposal of waste materials containing PCBs may result in release to the aquatic environment through the contamination of surface runoff and groundwater (Garrett, 1980). As up to 50% of the North American production of PCBs has been placed in landfills and dumps, this source is potentially significant (Strachan, 1988).

Insufficient evidence exists to adequately quantify the threat that PCBs in landfill sites pose to the marine environment. Limited monitoring of leachate samples from a drainage ditch at the Richmond landfill site in British Columbia indicated PCB concentrations up to 20 $\mu g \cdot L^{-1}$. This value was high considering that PCB levels in Ontario landfills range from not detected (ND) to 1.2 $\mu g \cdot L^{-1}$ (Garrett, 1980).

A recent case study indicates that unsecured landfills and illegal dumps pose the greatest threat to the marine environment (Ghirelli *et al.*, 1983; Weaver, 1984). In this case, over 300 000 kg of PCBs in liquid wastes, junked capacitors, and other contaminated materials from two General Electric capacitor plants were placed in unsecured landfills adjacent to the Hudson River in New York. This source was responsible for the addition of 3000 kg of PCBs to the Hudson River estuary each year, until remedial action was taken in 1977. PCB contamination of this estuary has led to restrictions on commercial and recreational fishing.

Industrial Effluents

The direct discharge of PCB-contaminated wastewater to coastal areas or to rivers that drain to coastal areas has been significantly reduced since the manufacture of PCBs was restricted (Weaver, 1984). However, industrial-effluent-associated discharges of PCBs to the marine environment were previously significant in such areas as the Southern California Bight (Young and Heesen, 1978; Gossett *et al.*, 1983) and the Acushnet River estuary–New Bedford Harbor area (Farrington, 1983; Weaver, 1984).

In Canada, several industrial processes, such as the recycling of PCB-contaminated waste oil and waste paper, have the potential for inadvertent releases of PCBs to the marine environment (Garrett, 1980; Strachan, 1988). Elevated PCB levels (up to 1500 $\mu g \cdot kg^{-1}$) have been detected in Fraser River sediments adjacent to a waste paper recycling plant (Garrett, 1980). The potential threat from these sources is unknown at present but will likely become insignificant as PCB content in recycled products becomes reduced with time.

Municipal Wastewater

Data from the Southern California Bight indicate that, in the early 1970s, the submarine discharge of municipal wastewater was the dominant source of PCB entry to this coastal marine ecosystem (Young and Heesen, 1978). In the New Bedford Harbor area, the discharge of PCBs from the city wastewater treatment plant remains significant, with 90–300 kg of PCBs being discharged annually (Weaver, 1984).

In British Columbia, Garrett (1980) found that, on average, 53% of influent PCBs were removed during wastewater treatment. It was further demonstrated that treatment plants having the largest industrial inputs were the plants with the highest PCB concentrations in the effluent. The use of "superchlorination" on sewage effluents for disinfection, odour control, and improved sedimentation may also lead to the formation of significant quantities of PCBs in the effluent. Generally, PCB concentrations in wastewater effluents do not exceed 1 $\mu g \cdot L^{-1}$ (Garrett, 1980).

Industrial and Municipal Wastewater Treatment Sludge

Disposal of industrial and municipal wastewater treatment sludges continues to be a significant source of PCBs the marine environment (Garrett, 1980; O'Connor *et al.*, 1983; Strachan, 1988). Of particular concern is the dumping of contaminated sludges to deep-water ocean sites. Projections by O'Connor *et al.* (1983) indicate that sediment concentrations near ocean discharge points could reach 200–300 $\mu g \cdot kg^{-1}$ after 100 years if present ocean discharge rates are continued off the New York coastline. Resuspension of sediments and transport in ocean currents are likely to cause further contamination of remote marine environments.

Worldwide Residue Levels

PCB residues have been widely reported in all media throughout the world. The purpose of this section is to briefly summarize PCB concentrations in coastal and estuarine waters, sediments, and biota.

Water Column

Total PCB concentrations in marine waters range from 0.04 $ng \cdot L^{-1}$ in the relatively unpolluted water of the Western North Pacific (Tanabe *et al.*, 1984) to 2.8 x 10^6 $ng \cdot L^{-1}$ in the Hudson River estuary (Table C-1) (Phillips, 1986). Table C-1 further indicates that estuaries and coastal waters, especially those near urban centres, are more contaminated by PCBs than are open ocean waters.

Many analyses of PCB residues in water often fail to filter out microparticulates, which tend to accumulate PCBs (Phillips, 1986). Therefore, the "true" PCB residue levels in the water column of many areas are unknown and probably overestimated. This margin of error may not be biologically significant. However, comparison of total PCBs between sites with varying microparticulate levels is not recommended.

Sediments

PCBs preferentially partition to the sediments, and thus environmental concentrations are much higher in the sediments than in the water column. Measured total PCB concentrations in marine sediments range from 0.3 $ng \cdot g^{-1}$ dry weight in the North Atlantic (Boehm, 1983) to <1.9 x 10^8 $ng \cdot g^{-1}$ dry weight in the polluted New Bedford Harbor (Table C-2) (Segar and Davis, 1984). As shown in Table C-2, coastal and estuarine sediments are orders of magnitude more contaminated by PCBs than are open ocean sediments.

Biota

Table C-3 summarizes whole-body PCB residue levels (wet weight) for organisms at different trophic levels and from various locations. Studies that measured PCB residue levels in specific tissues (e.g., brain, liver) or quantified residue levels on a dry-weight or lipid-weight basis were not included in Table C-3, because they were not comparable with whole-body, wet-weight measurements. As was shown with marine waters and sediments, the biota that were most contaminated were those from coastal areas and estuaries near urban centres (e.g., Baltic coast, New Bedford Harbor) (Table C-3).

Figures C-1 to C-4 indicate temporal changes in PCB residue levels in biota since the early 1970s. Several trends can be observed. First, in areas where PCB usage was curtailed in the early 1970s, PCB residue levels have declined (Figs. C-1 and C-2). Second, observed declines are most notable in areas that were previously highly contaminated (e.g., near sewer outfalls); declines in remote areas have been relatively minor (Figs. C-1 to C-4). Third, it appears that, since the late 1970s, PCB residue levels in biota have been relatively stable, and thus major declines in the near future are unlikely. In the above long-term studies, the collection and analytical procedures remained unchanged during the study, and thus the observed trends are likely "real" and not an artifact of changing methodologies.

Canadian Residue Levels

Water Column

Although the data are relatively sparse, PCB levels in Canadian marine waters parallel worldwide patterns; the most contaminated areas are those closest to urban centres (e.g., St. Lawrence estuary) (Germain and Langois, 1988) (Table C-4). However, even the most highly contaminated areas of the St. Lawrence estuary have relatively low PCB levels compared with other estuaries located near urban centres (e.g., Hudson River, U.S. Atlantic coast, Japanese coast).

Sediments

Sediment PCB levels in Canada also parallel worldwide patterns, although even the most contaminated site (an ocean dump site near Vancouver) had PCB concentrations several orders of magnitude below those found in the Hudson River estuary and New Bedford Harbor (Table C-5).

Biota

Table C-6 summarizes whole-body, wet-weight PCB residue levels in various Canadian biota from different trophic levels and from different locations. As before, studies that determined PCB residue levels in specific tissues or quantified levels on a dry- or lipid-weight basis were not included in Table C-6. In general, PCB levels in Canadian biota do not approach the levels observed in such polluted areas as New Bedford Harbor. However, the data do suggest that biota from the Bay of Fundy, St. Lawrence estuary, and portions of the British Columbia coast are contaminated by PCBs (Table C-6).

Several epidemiological studies have found that the decline in numbers of beluga whales (*Delphinapterus leucas*) in the St. Lawrence estuary may be attributed to their body burdens of toxic chemicals, particularly PCBs (as high as 575 $\mu g \cdot g^{-1}$ in blubber and 1725 $\mu g \cdot g^{-1}$ in mother's milk) (Masse *et al.*, 1986; Allan, 1988). PCB levels as high as 310 $\mu g \cdot g^{-1}$ in the blubber of harbour porpoises (*Phocoena phocoena*) from the Bay of Fundy have also been reported (Gaskin *et al.*, 1983). These studies show that cetaceans from the Bay of Fundy and St. Lawrence estuary may be experiencing deleterious effects from contamination by PCBs.

In an extensive analysis of PCB residue levels in the eggs of Canadian seabirds, Noble and Elliott (1986) showed that PCB levels are declining in the Bay of Fundy and on the British Columbia coast. However, little change or small increases in PCB residue levels have been observed on the outer continental shelf (Atlantic Ocean), in the St. Lawrence estuary and gulf, and in high-Arctic areas. Gaskin *et al.* (1983) found that PCB residue levels have not changed significantly in harbour porpoises from the Bay of Fundy between 1971 and 1977.

In summary, although PCB residue levels in Canadian biota are not as high as the levels in biota from the more polluted estuaries in the world, the problem of PCB contamination is an ongoing one, despite restrictions on usage since 1977.

ENVIRONMENTAL FATE AND PERSISTENCE

Behaviour in the Marine Environment

PCBs have a low solubility in water and high octanol/water partition coefficients, bioaccumulation potential, and resistance to degradation (Hamdy and Gooch, 1986). The physical and chemical properties of PCBs cause their removal from water by sorption onto sedimenting particles and bottom sediments (Fig. 2) (Halter and Johnson, 1977; Nau-Ritter and Wurster, 1983). Therefore, in the marine environment, PCBs are found at much higher concentrations in sediments than in the overlying water (see previous section). PCBs are associated particularly with suspended microparticulates of diameter less than 0.15 µm (U.S. EPA, 1980). PCB concentrations also increase with organic content of the particulates (Nau-Ritter *et al.*, 1982). PCBs sorbed to sediments can be released to the overlying water by slow desorption, especially when particulate PCB concentrations are high (Biggs *et al.*, 1980), resuspension and redistribution during periods of sudden hydrographic activity (e.g., flooding, dredging), and translocation through biological activity (Halter and Johnson, 1977). It has been shown that desorption from particulates favours the lower chlorinated, more water-soluble PCB congeners (Halter and Johnson, 1977; Wood *et al.*, 1987). However, the desorption behaviour of a PCB congener is affected not only by the number of chlorine atoms, but also by their positions. PCB congeners that are planar (e.g., those without chlorines in the ortho positions) are most efficiently adsorbed to sediments, and strength of adsorption decreases as congeners become less planar (Shaw and Connell, 1984).

For most marine organisms, PCB accumulation appears to be a two-step process: from sediments to water, then from water to organism (Fig. 2) (Halter and Johnson, 1977; Nau-Ritter and Wurster, 1983). However, benthic organisms that directly ingest PCB-contaminated sediments may accumulate significant levels of PCBs via this route (McElroy and Means, 1988). Because PCBs are highly lipophilic, they bioconcentrate in biota (Biggs *et al.*, 1980; Nau-Ritter and Wurster, 1983). The uptake and bioconcentration of PCB congeners in biota have been shown to be correlated with the 1-octanol/water partition coefficient and with the adsorption characteristics of PCB congeners on surfaces (Halter and Johnson, 1977; Shaw and Connell, 1984). Therefore, bioconcentration selectively favours higher chlorinated, planar PCB congeners.

PCBs rarely undergo typical chemical reactions, such as oxidation, reduction, addition, elimination, or electrophilic substitution, except under extreme conditions (U.S. EPA, 1980). Perhaps the only important chemical degradation pathway is that of photochemical transformation (Fig. 2). PCBs are also resistant to normal biodegradative processes and hence are extremely persistent in the marine environment. The purpose of the following discussion is to describe the persistence of PCBs and to determine major degradation pathways in the marine environment.

Persistence in the Marine Environment

Water and Sediments

Modelling studies have shown that photolysis may reduce the levels of higher chlorinated PCBs in large lakes or oceans, with resulting half-lives of 1–2 years (Bunce *et al.*, 1978). Given that PCBs volatilize fairly rapidly from water in laboratory aquaria (Mackay *et al.*, 1983), it would seem that PCB half-lives in the water column should be even shorter when volatilization is combined with photolytic degradation. However, the major accumulations of PCBs lie buried in aquatic sediments and are inaccessible to sunlight or volatilization unless disturbed (Brown *et al.*, 1987). Desorption of PCBs from sediments is a relatively slow process. For instance, in a 120-d test in flowing water, Halter and Johnson (1977) found that only 0.35% of PCBs in spiked sediments were lost during the entire study. Even under ideal, continuous-flow laboratory conditions, most PCB congeners require 20–30 d for partial depuration from the surface sediments. After this initial period of rapid depuration, depuration becomes even slower as rates become dependent on the redistribution rate of PCBs from deeper sediments (Wood *et al.*, 1987). Microbial transformation and degradation of PCBs in the sediments are limited to lower chlorinated (less than five) congeners under aerobic conditions (Ghirelli *et al.*, 1983).

In summary, PCB half-lives in marine waters and sediments are likely to be very long, in the order of years or decades.

Biota

Table 7 indicates that the lower chlorinated congeners and mixtures are depurated much more rapidly than the higher chlorinated congeners and mixtures. Further, this trend appears to be consistent

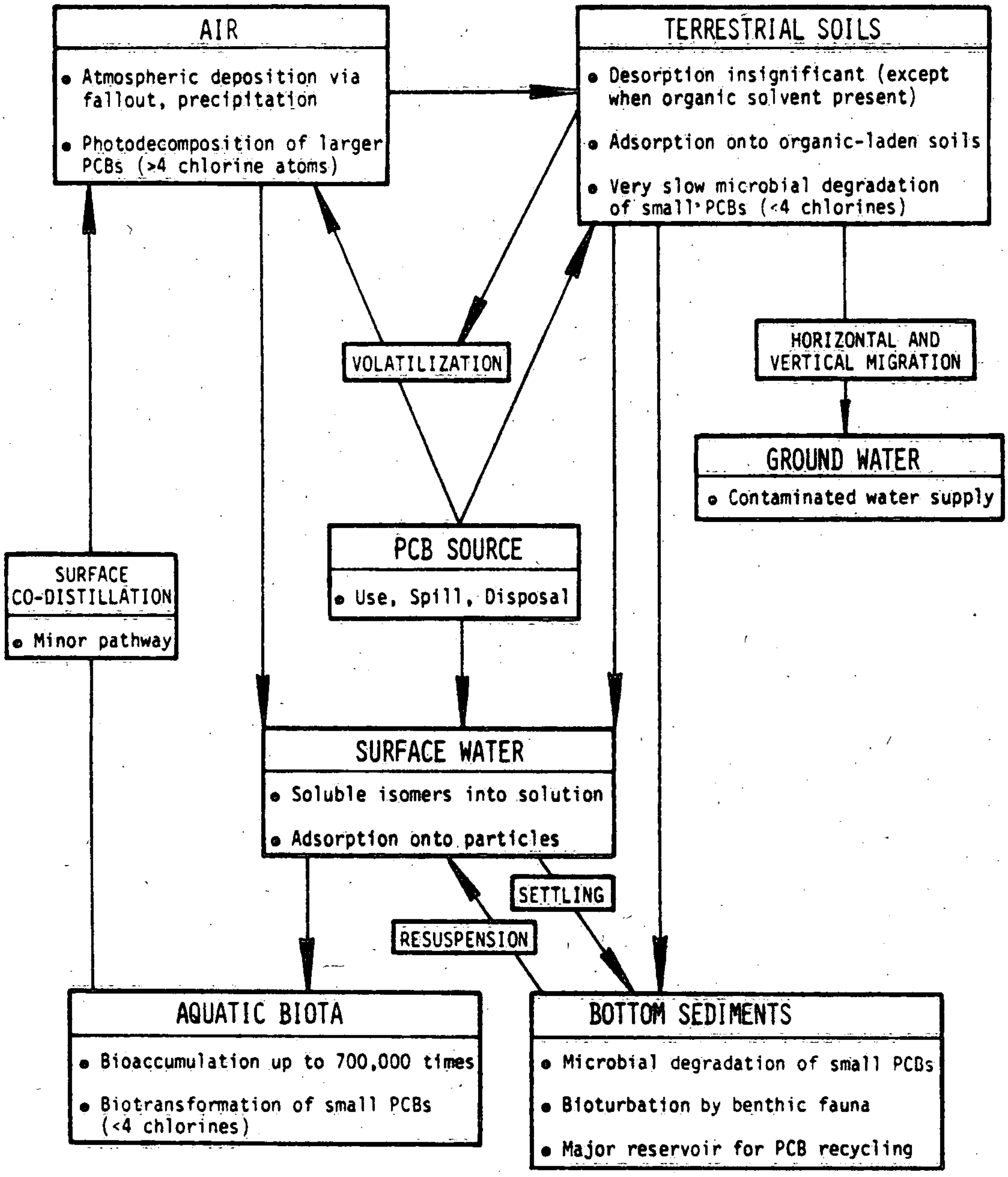


Figure 2. PCB environmental pathways and compartments (Ghirelli *et al.*, 1983).

Table 7. Persistence of PCB Congeners and Mixtures in Marine Biota

Organism	Aroclor/Congener	Half-life (d)	Reference
Invertebrates			
Polychaete worm	Aroclor 1242	12	Shaw and Connell,
(*Capitella capitata*)	Aroclor 1254	31	1987
	Aroclor 1260	34	
Pink shrimp	Aroclor 1254	17	Nimmo *et al.*,
(*Penaeus duorarum*)			1971
Cockle	Aroclor 1242	7–14	Langston, 1978
(*Cerastoderma edule*)	Aroclor 1254	>>21	
	Aroclor 1260	>>21	
	4,4'-CB	7–14	
	3,4,5,3',4',5'-CB	>21	
Cockle	4,4'-CB	>21	Langston, 1978
(*Macoma balthica*)	3,4,5,3',4',5'-CB	>21	
Mussel	2,4,4'-CB	16	Pruell *et al.*,
(*Mytilus edulis*)	2,4,5,2',5'-CB	28	1986
	2,4,5,2',4',5'-CB	46	
	2,3,4,2',3',4'-CB	37	
Fish			
Pinfish	Aroclor 1016	40	Hansen *et al.*,
(*Lagodon rhomboides*)			1974
Spot	Aroclor 1254	42	Hansen *et al.*,
(*Leiostomus xanthurus*)			1971
Mullet	Aroclor 1242	13	Shaw and Connell,
(*Chelon labrosus*)	Aroclor 1254	43	1987
	Aroclor 1260	38	
Striped bass	Aroclor 1254	5	O'Connor and
(*Morone saxatilis*)			Pizza, 1987
Cod (eggs)	2,4,5,2',4',5'-CB	>>13	Solbakken *et al.*,
(larvae)	2,4,5,2',4',5'-CB	>>15	1984
(*Gadus morhua*)			

CB = chlorobiphenyl

for both marine invertebrates and marine fish. Differences do arise between species in their ability to depurate PCBs. For instance, spot (*Leiostomus xanthurus*) exposed to Aroclor 1254 had a depuration half-life of 42 d (Hansen *et al.*, 1971), whereas striped bass (*Morone saxatilis*) had a half-life of 5 d (O'Connor and Pizza, 1987). O'Connor and Pizza (1987) speculated that these differences could be attributed to the lipid pools in different fish species, such that the species with low levels of deposited fat had more rapid rates of PCB loss when held under "clean" conditions. This effect was the result of proportionately greater amounts of PCBs remaining in the blood and thus available for rapid elimination via the hepatic pathway.

In summary, it appears that most marine biota require 1 to several weeks to eliminate 50% of the lower chlorinated PCBs and at least 1–2 months to eliminate 50% of the higher chlorinated PCBs, although this generalization may vary with species and congener structure.

Degradation Pathways

Photolysis

Photochemically catalyzed nucleophilic substitutions and photochemical free-radical substitutions are likely the only chemical degradation reactions of PCBs that are of environmental importance (U.S. EPA, 1980). Higher chlorinated PCB congeners are more photo-chemically reactive, especially under anaerobic conditions (U.S. EPA, 1980). The resulting 2-hydroxychlorobiphenyl is positioned to allow oxygen bonding at the ortho position of the other ring, resulting in the creation of highly toxic chlorodibenzofurans (CDFs) (Tanabe, 1988). Chemical analyses of PCB mixtures have demonstrated that CDFs are common contaminants at levels as high as 33 $\mu g \cdot g^{-1}$ (Nagayama *et al.*, 1976). When used as heat exchanger fluids in electrical capacitors and transformers, PCBs tend to decompose under the action of heat and electrical arcing to form CDFs at concentrations as high as 11 765 $\mu g \cdot g^{-1}$ (Nagayama *et al.*, 1976).

Microbial Degradation

The large quantities of PCBs released to polluted estuaries have likely exerted a strong selective effect on microorganisms that detoxify or utilize these compounds (Hamdy and Gooch, 1986). Seven genera of bacteria capable of degrading PCBs have been isolated from estuaries (*Acinetobacter, Aeromonas, Bacillus, Micrococcus, Pseudomonas, Streptomyces, Vibrio/Aeromonas*), whereas only two genera (*Pseudomonas, Vibrio*) have been isolated from seawater and ocean sediments (Sayler *et al.*, 1978). In general, microbial degradation of PCBs is effective only on the lower chlorinated congeners, although recent studies have indicated that several penta- and hexachlorobiphenyls can be dechlorinated by anaerobic bacteria (Bedard *et al.*, 1987; Brown *et al.*, 1987). Metabolites produced as a result of PCB degradation include benzoic acid, chlorobenzoic acid, chlorobiphenyldiol, and other aliphatic and aromatic hydrocarbons (Hamdy and Gooch, 1986). Brown *et al.* (1987) showed that microbial degradation in the Hudson River sediments may have significantly altered sediment congener profiles, with the end result that several PCB congeners of toxicological concern (i.e., those with chlorines in the meta and para positions of the biphenyl ring) (Tanabe, 1988) have been preferentially dechlorinated and subsequently biodegraded by aerobic bacteria. However, the evidence for microbial degradation of sediment PCBs was based solely on sediment core congener profiles and has been disputed because of the circumstantial nature of the data (Brown, 1989).

Metabolism by Fish, Birds, and Mammals

Few studies have provided concrete evidence that fish can metabolize PCBs. Bonn and Duinker (1985) found that sole (*Solea solea*) were able to metabolize PCB congeners with non-chlorinated meta and para carbons. Hydroxy metabolites have been isolated from fish exposed to lower chlorinated congeners (Sanborn *et al.*, 1975).

Laboratory studies have indicated that birds and mammals can metabolize lower chlorinated PCB congeners to more polar compounds, which are then excreted in the bile (Hamdy and Gooch, 1986). However, there is wide variation in the ability of different species to metabolize PCBs (Tanabe *et al.*, 1988; Cummins, 1988). For example, Tanabe *et al.* (1988) showed that the capacity to metabolize PCBs is lower in small cetaceans than in birds and terrestrial mammals. As well, certain chlorine substitution patterns are favoured for metabolism. The presence of two adjacent carbon atoms without chlorine substitution in either ring facilitates metabolism because it permits the formation of arene oxide intermediates (Tanabe, 1988). However, Tanabe *et al.* (1988) found that small cetaceans had no capacity to metabolize PCB congeners with adjacent non-chlorinated meta and para carbons.

In vitro experiments with rats have found that PCBs are metabolized by hepatic microsomal drug-metabolizing enzyme systems, and this metabolism is enhanced by microenzymes induced by phenobarbital (PB) and 3-methylcholanthrene (MC) (Tanabe *et al.*, 1988). It has been observed that higher activities of PB-type enzymes were in the order of terrestrial mammals > birds > marine mammals and MC-type enzymes in the order of terrestrial mammals > marine mammals > birds (Tanabe *et al.*, 1988). However, mink, which are extremely sensitive to PCBs, have very low activities for both types of enzymes (Aulerich and Ringer, 1977). It is unknown whether or not PCB metabolism by higher marine animals represents a major degradation pathway for PCBs in the marine environment.

BIOACCUMULATION

The accumulation of PCBs from the environment by an organism involves two opposing processes, those of uptake and loss (Harding and Addison, 1986). In aquatic biota, uptake can occur via three mechanisms (Shaw and Connell, 1986): absorption of PCBs from the water column through the gills, uptake through the epidermis, and consumption of contaminated food and sediments. Losses may occur through defecation of undigested material, excretion of metabolic waste, and production of young (Harding and Addison, 1986). Studies have indicated rapid uptake of PCBs by aquatic biota but rather low rates of metabolism and depuration (see previous section) (Biddinger and Gloss, 1984). Therefore, marine biota bioaccumulate PCBs. The purpose of this section is to compare bioaccumulation among different trophic levels of marine biota and to determine the importance of chlorine content to the bioaccumulation of specific PCB congeners.

Appendix D indicates that the bioaccumulation factors (BAFs) for PCBs in marine biota range from 8.0×10^2 to 4.4×10^7 in laboratory studies. Further, there is no apparent increase in bioaccumulation among organisms at higher trophic levels in these studies. However, this result may be an artifact of varying laboratory methodologies, as the two field studies from the Antarctic (Subramanian *et al.*, 1986) and Escambia Bay, Florida (Nimmo *et al.*, 1975), indicate increasing bioaccumulation in organisms from higher trophic levels. Appendix D also indicates that the bioaccumulation factors for individual species derived from laboratory experiments closely approximate corresponding values from the natural marine environment (e.g., *Penaeus* spp., *Leiostomus xanthurus*).

Several authors have argued that PCB bioaccumulation increases with trophic level (e.g., Tanabe, 1988). The apparent biomagnification of PCBs up the food web could be the result of organisms at higher trophic levels consuming contaminated food (Thomann, 1981; Sodergren, 1984; Rubinstein *et al.*, 1984) or the relationship of lipid content and age of organism with trophic level (Kalmaz and Kalmaz, 1979; Phillips, 1986; Tanabe, 1988). At present, the importance of food web biomagnification for organisms at higher trophic levels is unknown.

Appendix D indicates that bioaccumulation of PCB congeners may increase with chlorine content (Vreeland, 1974; Boon and Duinker, 1985; Shaw and Connell, 1987). This relationship has been attributed to the inverse relationship between water solubility of PCB congeners, which decreases with chlorine content, and bioaccumulation (Veith and Kosian, 1983; Boon and Duinker, 1985; Shaw and Connell, 1986). Also, PCB congeners with high chlorine content are more persistent in the environment, less volatile, and not metabolized in fish tissue. However, some PCBs exhibit significant deviations from this trend because of the effects of molecular stereochemistry. In particular, PCB congeners with meta–para unsubstituted adjacent carbon atoms are very persistent and have a high potential for bioaccumulation (Borlakoglu *et al.*, 1988; Boon and Eijgenraam, 1988).

In summary, PCB bioaccumulation is high in marine biota because of the highly lipophilic nature of PCBs and because these compounds are extremely persistent in the marine environment and in marine biota.

TOXICITY TO MARINE BIOTA

There are numerous studies that have considered the toxic (especially sublethal) effects of PCBs on aquatic biota. These studies range in scale from those measuring effects of PCBs at the biochemical level in individual species to those measuring effects at the community level. The purpose of this section of the report is to discuss the acute, chronic, and sublethal effects of PCBs on marine biota.

Aquatic toxicity tests can suffer from a variety of deficiencies, and tests with PCBs are no exception. Toxicity tests in any of the following categories were not considered for the purposes of water quality guideline development (Lloyd, 1986; Taylor, 1986): (1) epidemiological and biological surveillance studies; (2) studies that used impure test substances or that combined several substances together; (3) studies that measured responses not shown to be biologically important (e.g., adaptive responses); (4) studies that used solvent vehicles to assist in test solution preparation and that did not have appropriate solvent controls (reference to the literature was not considered to be a satisfactory control); and (5) studies in which the control population responded poorly to test conditions. Studies that were deemed to be adequate for water quality guideline development for PCBs in the marine environment are summarized in Appendix E.

Because standard protocols for toxicity testing may become outdated or are not always available or followed, a great deal of variability exists in the quality of published toxicity data. To ensure a consistent scientific evaluation, the data used in deriving a guideline must meet certain criteria, as outlined in CCREM (1987, Appendix IX). These criteria include information on test conditions/design (e.g., flow-through, static), test concentrations, temperature, water hardness, pH, experimental design (controls, number of replicates), and a description of the statistics used in evaluating the data. Each study is evaluated based on the above information and ranked as primary, secondary, or unacceptable (see CCREM 1987, Appendix IX, for a detailed description of the ranking criteria). All data included in the minimum data set must be primary in order for full guideline derivation to proceed. For interim guideline derivation, primary or secondary data may be used. Toxicity data that do not meet the criteria of primary or secondary data are unacceptable and cannot be used in either derivation procedure.

Acute Lethality Studies

The purpose of this section is to summarize concentrations of PCBs causing lethal effects to marine biota during acute exposures (up to 144 h). The results are tabulated in Table E-1.

Marine Plants

EC_{50}

Moore and Harriss (1972), in a study ranked as secondary because concentrations were not measured, found that natural phytoplankton assemblages experienced a 50% reduction in photosynthetic rate as measured by radiocarbon uptake (24-h EC_{50}) when exposed to nominal concentrations of 6.5, 15, and 30 $\mu g \cdot L^{-1}$ of Aroclor 1242, Aroclor 1254, and 2,4'-dichlorobiphenyl, respectively. This study was conducted *in situ* in bottles suspended from a floating platform near Escambia Bay, Florida, and thus approximated natural conditions.

In contrast, the green alga *Dunaliella tertiolecta* did not experience a 50% decline in photosynthetic rate until Aroclor 1254 levels were well above maximum solubility in salt water (24-h EC_{50} = 9000 $\mu g \cdot L^{-1}$) in a study ranked as secondary because concentrations were not measured (Luard, 1973). In both of the above studies, it was not determined whether the reduction in photosynthetic rate was due to reduced cell division, reduced photosynthetic rate per cell, or both. Other studies have confirmed that *D. tertiolecta* is unusually tolerant of high PCB levels (Fisher and Wurster, 1973; Craigie and Hutzinger, 1975; Biggs *et al.*, 1979).

Other Studies

Fisher and Wurster (1973) found that the marine diatom *Thalassiosira pseudonana* had a 17% reduction in growth rate after treatment with 10 $\mu g \cdot L^{-1}$ Aroclor 1254 for 72 h at the optimum temperature for this species (25°C). Complete photosynthetic inhibition for *T. pseudonana* was observed after a 48-h treatment with 50 $\mu g \cdot L^{-1}$ of Aroclor 1254 (Michaels *et al.*, 1982), and a significantly reduced growth rate (rates not stated by authors) was noted after a 96-h treatment at 25 $\mu g \cdot L^{-1}$ (Mosser *et al.*, 1972). Cosper *et al.* (1987) found that sensitive strains of the marine diatom *Ditylum brightwellii* exhibited reduced growth at 10 $\mu g \cdot L^{-1}$ and 100% mortality at 25 $\mu g \cdot L^{-1}$ when treated with Aroclor 1254 for 96 h. Another marine diatom, *Skeletonema costatum*, exhibited a significantly reduced growth rate (rates not stated by authors) after treatment with 10 $\mu g \cdot L^{-1}$ Aroclor 1254 for 96 h (Mosser *et al.*, 1972). As might be predicted from the above studies, Biggs *et al.* (1979) demonstrated a strong shift in species dominance from *T. pseudonana* in control mixed cultures to *D. tertiolecta* in mixed cultures treated for 72 h with 50 $\mu g \cdot L^{-1}$ Aroclor 1254.

The marine diatom *Rhizosolenia setigera* had a 13% reduction in growth rate after treatment with 1 $\mu g \cdot L^{-1}$ Aroclor 1254 for 144 h (Fisher and Wurster, 1973). In a study of open water and estuarine clones of three algal species (*Bellerochea polymorpha, Fragilaria pinnata,* and *T. pseudonana*), Fisher (1977) found that both the estuarine and especially the open water clones experienced reductions in final population densities (up to 99%) after treatment with 10 $\mu g \cdot L^{-1}$ Aroclor 1254 for 120 h.

Craigie and Hutzinger (1975) conducted a study of the effects of various Aroclor formulations (1221–1262) and PCB congeners on the growth of six classes of marine algae endemic to Canadian waters. They found that the effects varied greatly depending on Aroclor formulation and PCB congener. However, no consistent trends emerged as to which formulation or congener produced the strongest effects. At the lowest concentration tested (1000 $\mu g \cdot L^{-1}$), only *Olisthodiscus* sp. (Xanthophyceae) experienced a strong decline in growth.

In all of the acute toxicity tests on marine phytoplankton, the test conditions were static and PCB concentrations unmeasured; therefore, all studies were ranked as secondary (CCREM, 1987, Appendix IX).

Marine Invertebrates

LC_{50}

Adult grass shrimp (*Palaemonetes pugio*) had a 96-h LC_{50} of 41 $\mu g \cdot L^{-1}$ after treatment with Aroclor 1254 in a static test ranked as primary (Roesijadi *et al.*, 1976a). In the same study, juvenile grass shrimp were found to be more sensitive, with a 96-h LC_{50} of 7.8 $\mu g \cdot L^{-1}$. A similar response was demonstrated for juvenile grass shrimp (96-h LC_{50} = 12.5 $\mu g \cdot L^{-1}$) and brown shrimp (*Penaeus aztecus*) (96-h LC_{50} = 10.5 $\mu g \cdot L^{-1}$) after treatment with Aroclor 1016 in a flow-through test ranked as primary (Hansen *et al.*, 1974). McLeese and Metcalfe (1980) found almost no difference in the acute toxicity effects of Aroclor 1242 (96-h LC_{50} = 13.0 $\mu g \cdot L^{-1}$) and Aroclor 1254 (96-h LC_{50} = 12.0 $\mu g \cdot L^{-1}$) on the shrimp *Crangon septemspinosa* in static tests deemed unacceptable because only three shrimps were used and control treatments were not reported.

In the only other acute toxicity test found on marine invertebrates, Vernberg *et al.* (1977) determined that juvenile fiddler crabs (*Uca pugilator*) had a 96-h LC_{50} of 10 $\mu g \cdot L^{-1}$ after treatment with Aroclor 1254 in a static test ranked as secondary because concentrations were not measured. Conversely, adult fiddler crabs experienced no mortality at concentrations as high as 100 $\mu g \cdot L^{-1}$ in the same study.

Other Studies

Nimmo *et al.* (1971), in a study ranked as primary, and Duke *et al.* (1970), in a study ranked as secondary, found that juvenile pink shrimp (*Penaeus duorarum*) experienced 100% mortality after treatment with 100 $\mu g \cdot L^{-1}$ Aroclor 1254 in a flow-through test conducted for 96 and 48 h, respectively. Oysters (*Crassostrea virginica*) experienced no mortality after 96 h of the same treatment (Duke *et al.*, 1970).

The acute toxicity tests on juvenile grass shrimp and brown shrimp (Hansen *et al.*, 1974) (see above) and on juvenile pink shrimp (Nimmo *et al.*, 1971) were conducted in flow-through test conditions in which PCB concentrations in the water were measured throughout. In both studies, measured PCB concentrations were approximately 90% of the nominal concentrations reported above.

Marine Fish

LC_{50}

Kinter *et al.* (1972) found that adult killifish (*Fundulus heteroclitus*) had a 96-h LC_{50} of 16 000 $\mu g \cdot L^{-1}$ (calculated by extrapolation from the raw data presented in the paper) after treatment with Aroclor 1221 in a static test deemed unacceptable because experimental conditions were not reported in sufficient detail. No other acute toxicity studies on marine fish species were found from which an LC_{50} could be calculated. However, several freshwater fish species have also been shown to have high 96-h LC_{50} values (i.e., 278–60 900 $\mu g \cdot L^{-1}$) after treatment with various Aroclor formulations (Stalling and Mayer, 1972).

Other Studies

Juvenile pinfish (*Lagodon rhomboides*) experienced no mortality after a 48-h exposure to 100 $\mu g \cdot L^{-1}$ Aroclor 1254 in a flow-through test ranked as secondary because concentrations were not measured (Duke *et al.*, 1970). However, a 96-h exposure to a measured concentration of 56 $\mu g \cdot L^{-1}$ Aroclor 1016 in a flow-through test, ranked as primary, resulted in 18% mortality of juvenile pinfish (Hansen *et al.*, 1974).

Other Marine Biota

An estuarine clone of the bacteria *Escherichia coli* experienced a 99% reduction in cell density compared with controls after treatment with 10 000 $\mu g \cdot L^{-1}$ Aroclor 1254 for 24 h in a flow-through test ranked as secondary because concentrations were not measured (Colwell and Sayler, 1977). A ciliate protozoan (*Tetrahymena pyriformis*) experienced 96-h population density reductions of 10% at 1000 $\mu g \cdot L^{-1}$ Aroclor 1248 and 13%–22% at 1000 $\mu g \cdot L^{-1}$ Aroclor 1260 in static tests ranked as secondary because concentrations were not measured (Cooley *et al.*, 1973). Conversely, population density was reduced 8%–10% at 1.0 $\mu g \cdot L^{-1}$ Aroclor 1254 in a study deemed unacceptable because experimental conditions were not reported (Nimmo *et al.*, 1975).

Summary

Figure 3 indicates that most marine plankton and invertebrate species studied thus far have had an LC_{50}/EC_{50} in the range of 6.5–30 $\mu g \cdot L^{-1}$ total PCBs. The minimum observed EC_{50} was 6.5 $\mu g \cdot L^{-1}$ for a natural phytoplankton assemblage exposed to Aroclor 1242 for 24 h (Moore and Harriss, 1972). *Rhizosolenia setigera* experienced a 13% reduction in growth rate after exposure for 144 h to 1 $\mu g \cdot L^{-1}$ Aroclor 1254 (Fisher and Wurster, 1973). The most sensitive invertebrate, *P. pugio*, had a 96-h LC_{50} of 7.8 $\mu g \cdot L^{-1}$ (Roesijadi *et al.*, 1976a).

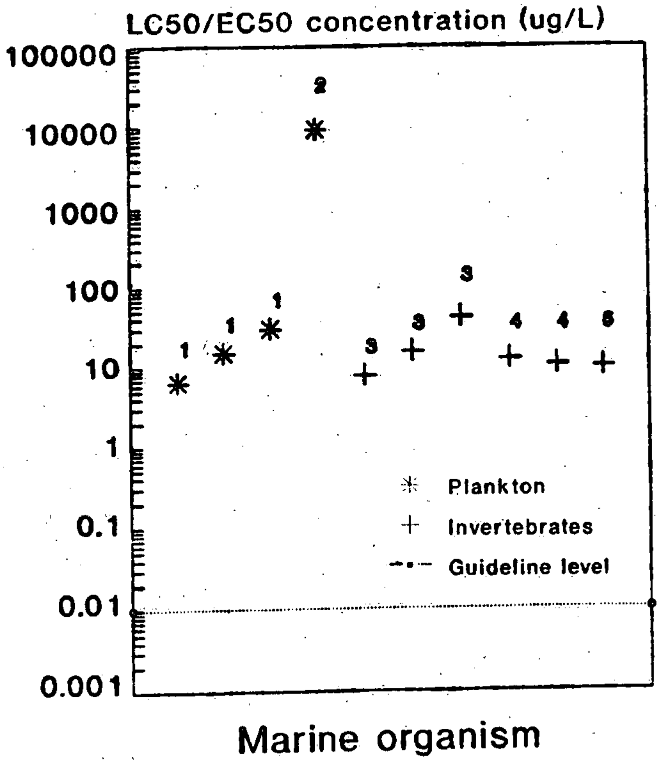


Figure 3. Observed LC_{50}/EC_{50} responses in marine biota after short term exposure to PCBs.

Chronic Lethality Studies

Previous studies have demonstrated that, like chlorinated hydrocarbon pesticides, effects of PCBs on biota occur at much lower concentrations when tests are conducted over a longer period of time (e.g., Mayer *et al.*, 1977). The purpose of this section is to report concentrations of PCBs causing lethal effects to marine biota during chronic exposures (>6 d). The results are summarized in Table E-2.

Marine Plants

The marine diatom *Cylindrotheca closterium* experienced an 83% reduction in population density relative to controls after exposure to 100 µg·L^{-1} Aroclor 1242 for 2 weeks in a static test ranked as secondary because concentrations were not measured (Keil *et al.*, 1971).

Marine Invertebrates

LC$_{50}$

Nimmo *et al.* (1971) found that pink shrimp (*P. duorarum*) were sensitive to chronic exposures of Aroclor 1254, such that juveniles had a 15-d LC$_{50}$ of 0.94 µg·L^{-1} and adults had a 35-d LC$_{50}$ of 3.5 µg·L^{-1} (measured concentrations) in a flow-through test ranked as primary. Similarly, Nimmo *et al.* (1974), in a study ranked as secondary, found that adult grass shrimp (*P. pugio*) experienced 50% mortality after 7 d of exposure to Aroclor 1254 at a nominal (unmeasured) concentration of 7.6 µg·L^{-1} (calculated by extrapolation from the raw data presented in the paper). In contrast, the marine amphipod *Gammarus oceanicus* was found to have a relatively high 35-d LC$_{50}$ of 200 µg·L^{-1} when treated with Aroclor 1254 in a static test ranked as secondary because concentrations were not measured (Wildish, 1972).

Other Studies

Roesijadi *et al.* (1976a) found that larval grass shrimp (*P. pugio*) exhibited 75% mortality after 5–6 d of exposure to 15.6 µg·L^{-1} Aroclor 1254 and 100% mortality after 11 d in a static test. Nimmo *et al.* (1971) found that juvenile pink shrimp (*P. duorarum*) exhibited significant mortality (30%) at the relatively low measured concentration of 0.57 µg·L^{-1} Aroclor 1254 after a 15-d exposure in a flow-through test. Both studies were ranked as primary.

Marine Fish

LC$_{50}$

Hansen *et al.* (1974) demonstrated that pinfish (*L. rhomboides*) experienced 50% mortality at a measured concentration of 59 µg·L^{-1} after an 18-d exposure to Aroclor 1016 in a flow-through test. However, the same end point was reached at a lower concentration of 21 µg·L^{-1} after a 33-d exposure. Similarly, spot (*L. xanthurus*) were sensitive to chronic exposures Aroclor 1254, with a 38-d LC$_{50}$ of 5 µg·L^{-1} in a flow-through test (Hansen *et al.*, 1971). In a study conducted under flow-through conditions, Schimmel *et al.* (1974) demonstrated that sheepshead minnow (*Cyprinodon variegatus*) embryos and fry were highly sensitive to exposure to Aroclor 1254. In both life cycle stages, 50% mortality was observed at a measured concentration of 0.32 µg·L^{-1} after 3 weeks, whereas adults were unaffected by concentrations an order of magnitude higher. These three studies were ranked as primary.

Other Studies

Hansen *et al.* (1971) found that juvenile pinfish (*L. rhomboides*) experienced 25% mortality after treatment with 5 µg·L^{-1} Aroclor 1254 for 17 d. Schimmel *et al.* (1974) found that sheepshead minnow (*C. variegatus*) fry had significantly higher mortality than did controls (38% and 11%, respectively) at a measured concentration of 0.16 µg·L^{-1} Aroclor 1254 but not at 0.06 µg·L^{-1}. Both studies were ranked as primary.

Summary

The lowest-observed-effect level (LOEL) found was with the sheepshead minnow fry, which experienced 38% greater mortality than controls after exposure to 0.16 µg·L^{-1} Aroclor 1254 for 3 weeks (Schimmel *et al.*, 1974). Figure 4 indicates that marine invertebrate and fish species experienced 50% mortality over a broad range of total PCB concentrations after long-term exposures (0.32–200 µg·L^{-1}). The lowest LC$_{50}$ was observed with sheepshead minnow embryos and fry after exposure to 0.32 µg·L^{-1} Aroclor 1254 for 3 weeks (Schimmel *et al.*, 1974).

Sublethality Studies

Most studies of the sublethal effects of PCBs on marine biota report results from relatively short

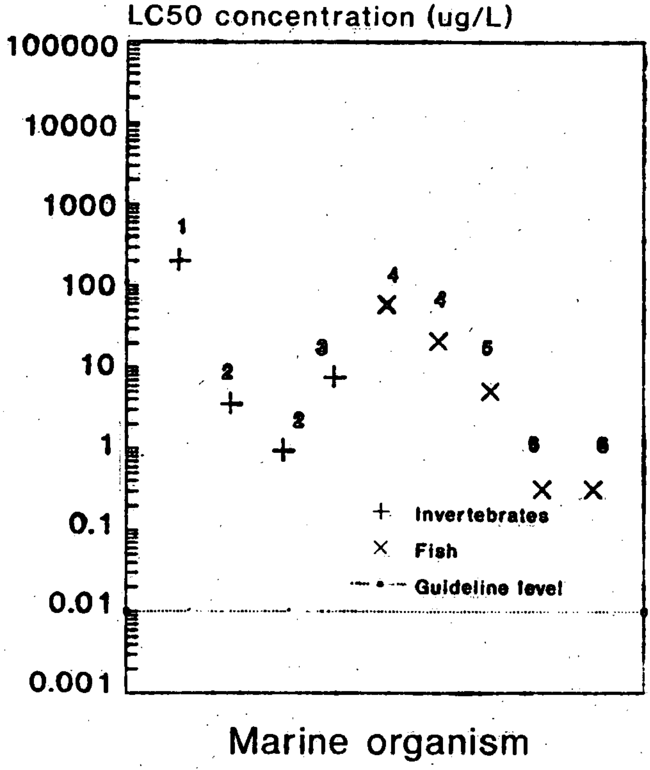


Figure 4. Observed LC_{50} responses in marine biota after long-term exposure to PCBs.

exposure periods (<24 h). In several of these studies, it was observed that with longer exposures to a given PCB concentration, sublethal effects were replaced by lethal effects. Therefore, the difference between lethal and sublethal effects was often a function of length of exposure period. All short- and long-term studies of the sublethal effects of PCBs on marine biota considered in this section are summarized in Table E-3.

Marine Plants

Studies that demonstrated reduced photosynthetic rates after exposure to PCBs but were unable to attribute the reduction to either reduced cell division or reduced photosynthetic rate per cell were previously discussed in the lethality sections of this report.

Larsson and Tillberg (1975) found that the green alga *Scenedesmus obtusiusculus* experienced a 21% reduction in phosphate uptake rates at 300 µg·L^{-1} and a 25% reduction in respiration (oxygen consumption) rates at 1000 µg·L^{-1} after a 24-h exposure to Aroclor 1242 in a static test ranked as secondary because concentrations were not measured. Keil *et al.* (1971) observed that the diatom *C. closterium* exhibited an 80% reduction in chlorophyll index (optical density at 670 µm) at 100 µg·L^{-1} and significantly reduced RNA levels at 10 µg·L^{-1} after a 14-d exposure to Aroclor 1242. This study was ranked as secondary.

Marine Invertebrates

Purple sea urchins (*Arbacia punctulata*) were found to have a significant 11% reduction in fertilization efficiency (cleaving eggs/total eggs counted x 100%) after a 25-min exposure to 5000 µg·L^{-1} Aroclor 1254 and 500 µg·L^{-1} Aroclor 1016 (Adams and Slaughter-Williams, 1988). Further, sea urchin embryos exposed to Aroclors 1254 and 1016 for 3 d experienced mutagenic effects as demonstrated by a significantly increased probability of developing into a deformed gastula at concentrations of 5000 µg·L^{-1} (41% deformed) and 1000 µg·L^{-1} (53% deformed), respectively. O'Connor *et al.* (1985) found that the amphipod *Gammarus tigrinus* experienced an 18% reduction in respiration rate after a 44-h exposure to 25 µg·L^{-1} Aroclor 1254 in a static test. Both tests were ranked as secondary because concentrations were not measured.

Grass shrimp (*P. pugio*) were shown to be sensitive to the effects of Aroclor 1254 during the larval stage, such that larval development was delayed 3.7 d when larvae were chronically exposed to a measured concentration of 3.2 µg·L^{-1} in a primary study (Roesijadi *et al.*, 1976a). In another study, grass shrimp exhibited an altered muscle amino acid profile (e.g., phenylalanine decreased 89%, glutamic acid increased 79%) after a 96-h exposure to a measured concentration of 29.4 µg·L^{-1} Aroclor 1254 in a static test (Roesijadi *et al.*, 1976b). This test was ranked as secondary because, although this response indicates an altered metabolic state, overall intracellular osmoregulation was not affected, and thus the biological significance of the response is unknown.

In two separate flow-through studies, oysters (*Crassostrea virginica*) were shown to be sensitive to the sublethal effects of PCBs. In the first study, ranked as primary, Hansen *et al.* (1974) found that oysters exposed for 96 h to a measured concentration of 0.6 µg·L^{-1} Aroclor 1016 experienced a 10% reduction in shell growth. In the second study, deemed unacceptable because of insufficient reporting of experimental conditions, Parrish *et al.* (1972) found that oysters had a significantly reduced growth rate (height and in-water weight) after a 24-week exposure to 5.0 µg·L^{-1} Aroclor 1254. No effect on growth rate was observed at 1.0 µg·L^{-1}.

Marine Fish

Weis and Weis (1982), in a study ranked as secondary because concentrations were not measured, found killifish (*Fundulus heteroclitus*) had a consistent retardation of hatching at 10 000 µg·L^{-1} Aroclor 1242 after a 7-d exposure. However, the retardation effect was not quantified by the authors. No effect was observed at the same concentration of Aroclor 1254. In response to a 24-h exposure to 75 000 µg·L^{-1} Aroclor 1221, the same species experienced significantly elevated serum osmolarity and sodium concentration in a study deemed unacceptable because experimental conditions were not reported in sufficient detail (Kinter *et al.*, 1972). This result indicated that the killifish had experienced a decline in the ability to osmoregulate.

Summary

Figure 5 indicates that sublethal effects of PCBs on marine biota were observed over a broad range of concentrations. The most sensitive response observed was for oysters, which experienced a 10% reduction in shell growth after a 96-h exposure to 0.6 µg·L^{-1} Aroclor 1016 (Hansen *et al.*, 1974).

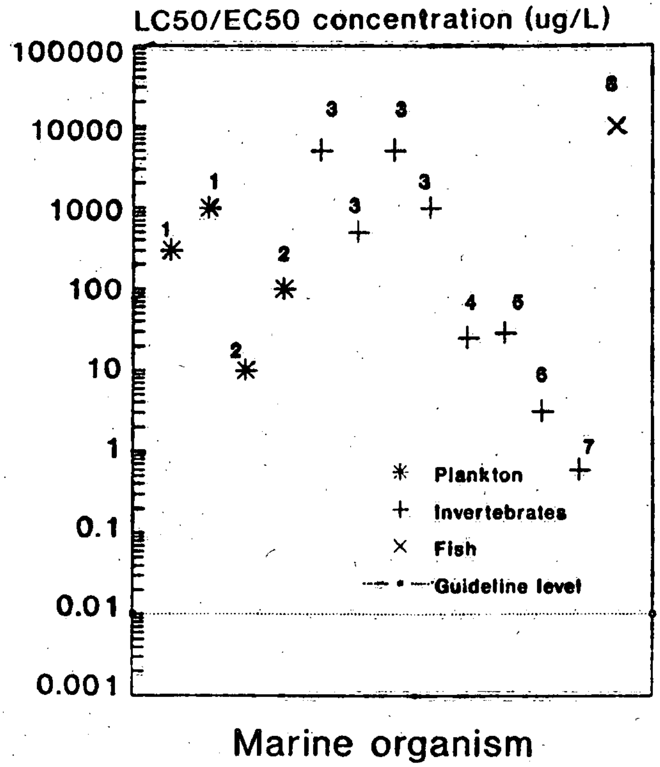


Figure 5. Observed sublethal responses in marine biota after exposure to PCBs.

Effects on Biota under Stress

Organisms rarely experience optimum conditions for growth and reproduction in nature. Therefore, the possible synergistic interaction between natural stresses in the environment and PCB contamination was also considered when developing PCB water quality guidelines. The purpose of this section is to report the effects of PCBs on marine biota already under stress from non-contaminant sources. The results are summarized in Table E-4.

Marine Plants

Fisher and Wurster (1973) determined the effects of Aroclor 1254 on the green alga *D. tertiolecta* and the diatoms *T. pseudonana* and R. setigera after an chronic exposure (72–192 h) in static conditions. At the highest concentration tested (10 µg·L^{-1}), no effect was observed on the growth rate of *D. tertiolecta*, even at suboptimal temperatures (i.e., 12°C). However, *T. pseudonana* experienced a 76% reduction in growth rate at 10 µg·L^{-1} at the suboptimal temperature of 12°C. At the lowest concentration tested (0.1 µg·L^{-1}), *R. setigera* exhibited an 18% decrease in growth rate at the suboptimal temperature of 10°C.

In nitrate-limited flow-through tests, Fisher *et al.* (1974) found that 0.1 µg·L^{-1} Aroclor 1254 caused a 68% reduction in the Shannon species diversity index and a change in the dominance hierarchy of a natural phytoplankton assemblage after a 6-d exposure. Similarly, Fisher *et al.* (1974) found that the measured concentration of 0.042 µg·L^{-1} Aroclor 1254 caused a dominance switch from *T. pseudonana* to *D. tertiolecta* after a 16-d exposure in nitrate-limited flow-through conditions.

Marine Invertebrates

Roesijadi *et al.* (1976a) found that salinity stress had little effect on the sensitivity of juvenile grass shrimp (*P. pugio*) to Aroclor 1254 in a static test. In all constant salinity treatments (1‰–35‰ salinity), the 96-h LC_{50} showed only minor variation, from 6.1 to 7.8 $\mu g \cdot L^{-1}$. However, when salinity conditions were varied during the exposure to Aroclor 1254, juvenile shrimp became more sensitive, with 1.8 $\mu g \cdot L^{-1}$ Aroclor 1254 causing a significantly greater mortality compared with controls (mortality effect not quantified) in a static test (Roesijadi *et al.*, 1976c).

Vernberg *et al.* (1977) determined the influence of stressful but sublethal combinations of temperature and salinity on the response of larval fiddler crabs (*U. pugilator*) subjected to sublethal concentrations of Aroclors 1016 and 1254. They found that at a concentration of 0.1 $\mu g \cdot L^{-1}$, extreme temperatures and salinity caused significantly greater mortality than was observed in non-stressful conditions or with controls. For instance, at 35‰ salinity and 15°C, larval mortality was 29% after exposure to 0.1 $\mu g \cdot L^{-1}$ Aroclor 1016, 78% after exposure to 0.1 $\mu g \cdot L^{-1}$ Aroclor 1254, and 15% in control treatments. In general, Aroclor 1254 was more toxic than Aroclor 1016.

Marine Fish

No studies of the effects of PCBs on marine fish under stress were found.

Marine Birds

Mallards *(Anas platyrhynchos)* fed Aroclor 1254 at concentrations ranging from 25 to 100 $mg \cdot kg^{-1}$ for 10 d exhibited reduced disease resistance when exposed to duck hepatitis virus (Friend and Trainer, 1970.

Summary

Figure 6 indicates that, in general, PCB effects on marine biota occurred at lower concentrations when the biota were under additional stress. The most sensitive response was a change in the dominance hierarchy of a *D. tertiolecta–T. pseudonana* mixed culture after a 16-d exposure to 0.042 $\mu g \cdot L^{-1}$ Aroclor 1254 when the culture was under nutrient limitation (Fisher *et al.*, 1974).

Dietary and Injection Studies

Benthic and higher-trophic-level marine biota may accumulate PCBs through their diet. The purpose of this section is to report the effects of PCBs in the diet or by injection to marine biota. The results are summarized in Table E-5.

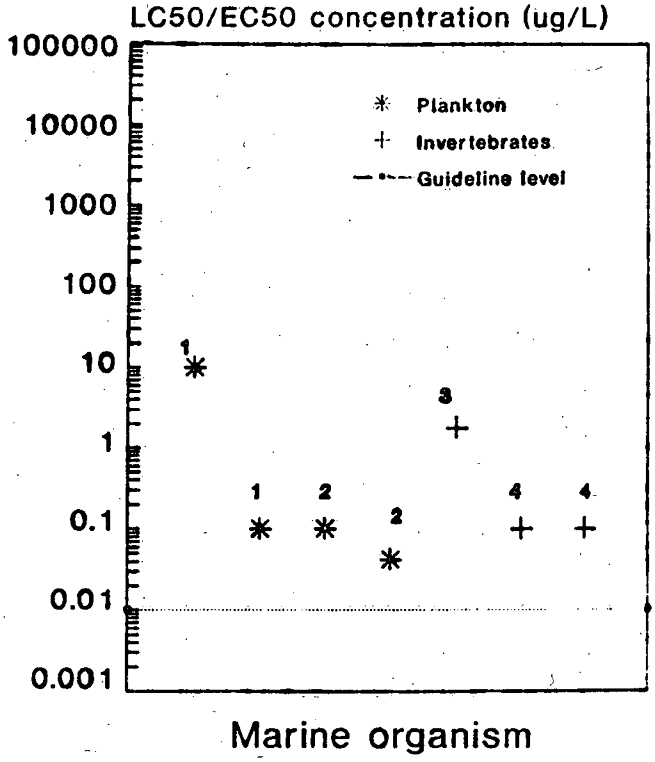


Figure 6. Observed lethal responses in marine biota under additional stress after exposure to PCBs.

Marine Invertebrates

After 4 and 8 weeks of exposure to Aroclor 1254 in the diet (500 $mg \cdot kg^{-1}$), the benthic worm *Nereis virens* was found to have elevated mixed-function oxidase and cytochrome P-450 activities (Fries and Lee, 1984). The biological significance of these biochemical changes is unknown (Lloyd, 1986).

Marine Fish

Anadromous yearling coho salmon (*Oncorhynchus kisutch*) injected with 0.15 $mg \cdot kg^{-1}$ Aroclor 1254 experienced significant mortality (6%) upon entry into seawater (Folmar *et al.*, 1982). Elevated thyroid activity was also observed in treated salmon.

Altered biosynthetic patterns, abnormalities in the testes, gills, liver, and kidneys, and delayed sexual maturity were observed in Atlantic cod (*Gadus morhua*) after a 5.5-month exposure to Aroclor 1254 (1, 5, 10, 25, 50 mg·kg^{-1}) in the diet (Freeman *et al.*, 1982). The PCB levels accumulated in the fish tissues in this study corresponded to fish tissue levels from several highly polluted areas in the North Sea.

Mullets (*Chelon labrosus*) exposed for 30 d to Phenochlor DP6 (a commercial PCB mixture containing six chlorine atoms per molecule) at a concentration of 48 mg·kg^{-1} in the diet experienced elevated glycohemia and glycogen content in the liver, heart, and muscle (Narbonne *et al.*, 1988). A decrease in plasma protein levels and an increase in hepatic proteins was also observed after PCB treatment. Only the elevated glycogen content in the liver remained after 30 d of depuration.

Marine Birds

Wild puffins (*Fratercula arctica*) injected with 30–35 mg of Aroclor 1254 under the skin experienced no effects on reproductive behaviour or hatching success under natural conditions over three breeding seasons (Harris and Osborn, 1981).

Lesser black-backed gulls (*Larus fuscus*) fed diets containing 50, 100, 200, and 400 mg·kg^{-1} of Aroclor 1254 for 8 weeks experienced increases in the weight, follicle size, and colloid content of the thyroid gland compared with controls (Jefferies and Parslow, 1972). Mean heart size also declined with increasing PCB dose.

At low dietary exposure levels (12 and 25 mg·kg^{-1}·d^{-1}) of Aroclor 1254, common murres (*Uria aalge*) experienced increases in gland weight, follicle size, and colloid content of the thyroid gland after 45 d of treatment (Jefferies and Parslow, 1976). However, these effects were reversed at high dietary exposure levels (400 mg·kg^{-1}·d^{-1}). These authors reported that the PCB levels accumulated in the lipids of the birds fell within the range of levels found in wrecks of dead murres in the Irish Sea (i.e., up to 79 µg·g^{-1} wet weight in the liver).

SUMMARY AND RECOMMENDED GUIDELINES

The acute toxicity data for PCBs indicate that this group of compounds is not highly toxic to marine biota. After short-term exposures to PCBs, the most sensitive organisms (the diatom *R. setigera* and the protozoan *T. pyriformis*) were found to have significant reductions in growth (13% and 8%–10%, respectively) at 1 µg·L^{-1} Aroclor 1254 (Fisher and Wurster, 1973; Cooley *et al.*, 1973; Nimmo *et al.*, 1975). Most of the marine plankton and invertebrate species studied had EC_{50}/LC_{50} values within 1–2 orders of magnitude of this level (i.e., <100 µg·L^{-1}). Fish were apparently not sensitive to short-term exposures to PCBs.

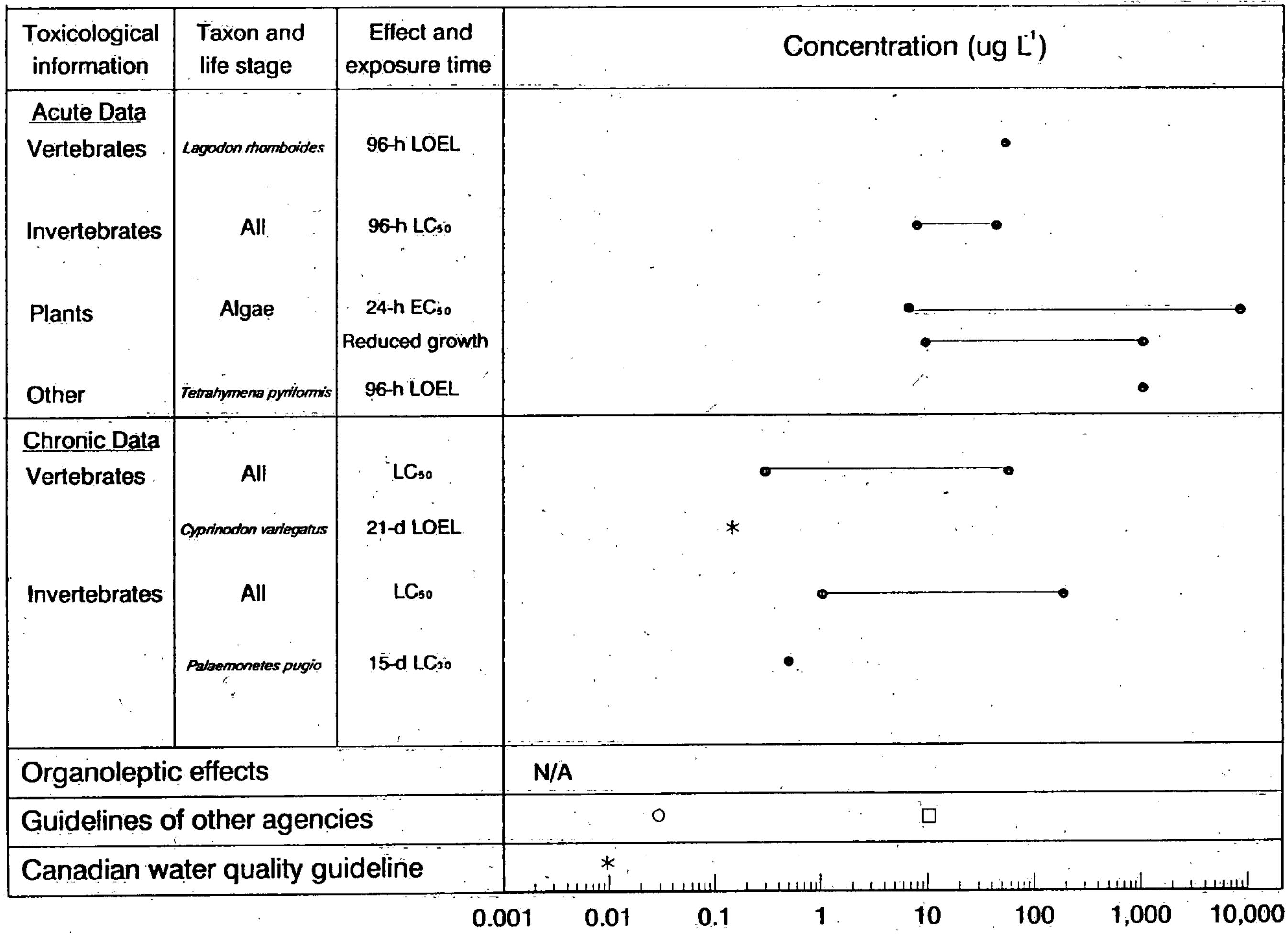
However, long-term exposures to PCBs lowered the lethality threshold for most marine biota (Fig. 7). The embryos and fry of the most sensitive species studied, sheepshead minnow (*C. variegatus*), had an LC_{50} of 0.32 µg·L^{-1} after a 3-week exposure to Aroclor 1254. Significant mortality (38%) was observed at 0.16 µg·L^{-1} Aroclor 1254, and only at levels as low as 0.06 µg·L^{-1} were there no observed effects (Schimmel *et al.*, 1974).

In general, the recorded levels for sublethal effects of PCBs on marine biota were not different from the levels at which lethal effects were observed. The most sensitive organism studied for sublethal effects, oysters (*C. virginica*), had a 10% reduction in shell growth rate after a 96-h exposure to 0.6 µg·L^{-1} Aroclor 1016 (Hansen *et al.*, 1974).

Marine biota under stress were much more sensitive to the effects of PCBs. The most sensitive stressed organisms studied, the diatom *R. setigera* and the fiddler crab *U. pugilator*, both experienced significant mortality after acute exposures to 0.1 µg·L^{-1} Aroclor 1254 (Fisher *et al.*, 1974; Vernberg *et al.*, 1977). Further, a 16-d exposure to 0.042 µg·L^{-1} Aroclor 1254 caused community structure changes in nitrate-limited phytoplankton (Fisher *et al.*, 1974). These levels of PCB exposure are within the ranges encountered in coastal and estuarine areas in Canada.

Variable amounts of toxicity data exist for different PCB congeners. The most frequently tested were Aroclors 1254 and 1016, although data were found for Aroclors 1242, 1248, 1221, 1260, 1262, 2,4'-CB, and Phenochlor DP6. Contaminants such as dioxins and furans in PCB mixtures used in earlier studies have resulted in discrepancies in toxicity data.

An interim water quality guideline of 0.01 µg·L^{-1} total PCBs is recommended for the protection and maintenance of coastal and estuarine aquatic organisms. This level was derived by applying a safety factor of 0.1 (CCREM, 1987, Appendix IX) to the LOEL (0.16 µg·L^{-1}) observed with the species most sensitive to long-term exposure to PCBs



∗ Toxicity test used to derive guideline value

○ U.S. EPA marine chronic criterion

□ U.S. EPA marine acute criterion

Figure 7. Guideline derivation graph for PCBs in the marine environment.

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Water Use: Protection of Marine Aquatic Life

Compound: Polychlorinated biphenyls (PCBs)

Canadian Water Quality <u>Guideline</u> Requirements : Minimum <u>Toxicity</u> Data Set

Aquatic Biota	Number of Studies Required	Primary	Temperate Species	Chronic Study	Two Classes Represented	Reference
Fish	1.	x	x	x		1
	2.	x	x	x		2
	3.	x	x			3
Invertebrates	1.	x	x	x	—	4
	2.	—	—	—		—
Plants	1.	—	—			—

Scientifically Justified Exemptions to Above Requirements: Yes ___ No x

Minimum Toxicity Data Set Requirements Met: Yes ___ No x If no, go to interim guideline section.

Canadian Water Quality <u>Guideline</u> Requirements: Minimum <u>Environmental Fate</u> Data Set

(1) Are the mobility of the compound and the compartments of the aquatic environment in which it is likely to be found known? Yes x No ___

(2) Are the kinds of chemical and biological reactions that take place during transport and after deposition known? Yes x No ___

(3) Are the eventual chemical metabolites known? Yes x No ___

(4) Is the persistence of the compound in water, sediments, and biota known? Yes x No ___

If the answer is no to any of the above, go to interim guideline section

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Canadian Water Quality <u>Interim Guideline</u> Requirements:

(1) Are there at least two acute and/or chronic studies for marine fish and for marine invertebrates? Yes x No ___

(2) Is one fish species a temperate species? Yes x No ___

(3) Are the two invertebrate species from different classes, and is one of the species temperate? Yes x No ___

If the answer is no to any of the above, then an interim guideline cannot be calculated. Primary or secondary studies may be used for the interim guideline data requirements

Figure 8. The minimum date set worksheet for the derivation of a Canadian water quality guideline for PCBs to protect marine aquatic life.

(*C. variegatus* fry) and rounding the value down to one significant figure. At this concentration, 38% mortality of *C. variegatus* fry was observed after a 3-week exposure to Aroclor 1254 (Schimmel *et al.*, 1974) (Fig. 7). Chronic exposures were deemed more appropriate for setting the water quality guideline because PCBs are not readily degraded in the marine environment. All studies considered in this report indicate that at 0.01 µg·L^{-1}, PCBs would exert no observable effect on marine biota, even when these biota are under stress by other factors (e.g., nutrient limitation) (Table E-4). One primary study on a temperate marine vascular plant or marine algal species and one primary chronic (partial or full life cycle) study on a temperate marine invertebrate species from a class other than Crustacea are required to upgrade the interim guideline to full guideline status (Fig. 8). Future water quality guideline efforts should focus on specific PCB congeners, particularly those congeners that are planar and therefore highly toxic (Tanabe, 1988). However, at present, the data are inadequate to attempt to recommend guidelines for specific PCB congeners in the marine environment.

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Appendix A

Literature Search

Appendix A

Literature Search

A literature search of the following data bases was conducted to retrieve any references that considered the environmental fate and behaviour of PCBs and the effects of PCBs on marine biota:

1. Water Resources Abstracts
2. Enviroline
3. Pollution Abstracts
4. Geological Reference File (Georef)
5. Micrololog
6. Aquatic Sciences and Fisheries Abstracts (ASFA)
7. Biosis (BA79)
8. Biosis (BA86)
9. Canadian Water Resources References (Aquaref)
10. National Technical Information Service (NTIS)

Where possible, the literature search was restricted to studies published after 1983. Studies published prior to 1983 were found by consulting review papers.

Appendix B

IUPAC Numbering System of PCB Compounds

Table B-1. IUPAC Numbering System of PCB Compounds

No.	Structure
1	2
2	3
3	4
4	2,2'
5	2,3
6	2,3'
7	2,4
8	2,4'
9	2,5
10	2,5'
11	3,3'
12	3,4
13	3,4'
14	3,5
15	4,4'
16	2,2',3
17	2,2',4
18	2,2',5
19	2,2',6
20	2,3,3'
21	2,3,4
22	2,3,4'
23	2,3,5
24	2,3,6
25	2,3',4
26	2,3',5
27	2,3',6
28	2,4,4'
29	2,4,5
30	2,4,6
31	2,4',5
32	2,4',6
33	2',3,4
34	2',3,5
35	3,3',4
36	3,3',5
37	3,4,4'
38	3,4,5
39	3,4',5
40	2,2',3,3'
41	2,2',3,4
42	2,2',3,4'
43	2,2',3,5
44	2,2',3,5'
45	2,2',3,6
46	2,2',3,6'
47	2,2',4,4'
48	2,2',4,5
49	2,2',4,5'
50	2,2',4,6
51	2,2',4,6'
52	2,2',5,5'
53	2,2',5,6'
54	2,2',6,6'
55	2,3,3',4
56	2,3,3',4'
57	2,3,3',5
58	2,3,3',5'
59	2,3,3',6
60	2,3,4,4'
61	2,3,4,5
62	2,3,4,6
63	2,3,4',5
64	2,3,4',6
65	2,3,5,6
66	2,3',4,4'
67	2,3',4,5
68	2,3',4,5'
69	2,3',4,6
70	2,3',4',5
71	2,3',4',6
72	2,3',5,5'
73	2,3',5',6
74	2,4,4',5
75	2,4,4',6
76	2',3,4,5
77	3,3',4,4'
78	3,3',4,5
79	3,3',4,5'
80	3,3',5,5'
81	3,4,4',5
82	2,2',3,3',4
83	2,2',3,3',5
84	2,2',3,3',6
85	2,2',3,4,4'
86	2,2',3,4,5
87	2,2',3,4,5'
88	2,2',3,4,6
89	2,2',3,4,6'
90	2,2',3,4',5
91	2,2',3,4',6
92	2,2',3,5,5'
93	2,2',3,5,6
94	2,2',3,5,6'
95	2,2',3,5',6
96	2,2',3,6,6'
97	2,2',3',4,5
98	2,2',3',4,6
99	2,2',4,4',5
100	2,2',4,4',6
101	2,2',4,5,5'
102	2,2',4,5,6'
103	2,2',4,5',6
104	2,2',4,6,6'
105	2,3,3',4,4'
106	2,3,3',4,5
107	2,3,3',4',5
108	2,3,3',4,5'
109	2,3,3',4,6
110	2,3,3',4',6
111	2,3,3',5,5'
112	2,3,3',5,6
113	2,3,3',5',6
114	2,3,4,4',5
115	2,3,4,4',6
116	2,3,4,5,6
117	2,3,4',5,6
118	2,3',4,4',5
119	2,3',4,4',6
120	2,3',4,5,5'
121	2,3',4,5',6
122	2',3,3',4,5
123	2',3,4,4',5
124	2',3,4,5,5'
125	2',3,4,5,6'
126	3,3',4,4',5
127	3,3',4,5,5'
128	2,2',3,3',4,4'
129	2,2',3,3',4,5
130	2,2',3,3',4,5'
131	2,2',3,3',4,6
132	2,2',3,3',4,6'
133	2,2',3,3',5,5'
134	2,2',3,3',5,6
135	2,2',3,3',5,6'
136	2,2',3,3',6,6'
137	2,2',3,4,4',5
138	2,2',3,4,4',5'
139	2,2',3,4,4',6
140	2,2',3,4,4',6'
141	2,2',3,4,5,5'
142	2,2',3,4,5,6
143	2,2',3,4,5,6'
144	2,2',3,4,5',6
145	2,2',3,4,6,6'
146	2,2',3,4',5,5'
147	2,2',3,4',5,6
148	2,2',3,4',5,6'
149	2,2',3,4',5',6
150	2,2',3,4',6,6'
151	2,2',3,5,5',6
152	2,2',3,5,6,6'
153	2,2',4,4',5,5'
154	2,2',4,4',5,6'
155	2,2',4,4',6,6'
156	2,3,3',4,4',5
157	2,3,3',4,4',5'
158	2,3,3',4,4',6
159	2,3,3',4,5,5'
160	2,3,3',4,5,6
161	2,3,3',4,5',6
162	2,3,3',4',5,5'
163	2,3,3',4',5,6
164	2,3,3',4',5',6
165	2,3,3',5,5',6
166	2,3,4,4',5,6
167	2,3',4,4',5,5'
168	2,3',4,4',5',6
169	3,3',4,4',5,5'
170	2,2',3,3',4,4',5
171	2,2',3,3',4,4',6
172	2,2',3,3',4,4',5'
173	2,2',3,3',4,5,6
174	2,2',3,3',4,5,6'
175	2,2',3,3',4,5',6
176	2,2',3,3',4,6,6'
177	2,2',3,3',4',5,6
178	2,2',3,3',5,5',6
179	2,2',3,3',5,6,6'
180	2,2',3,4,4',5,5'
181	2,2',3,4,4',5,6
182	2,2',3,4,4',5,6'
183	2,2',3,4,4',5',6
184	2,2',3,4,4',6,6'
185	2,2',3,4,5,5',6
186	2,2',3,4,5,6,6'
187	2,2',3,4',5,5',6
188	2,2',3,4',5,6,6'
189	2,3,3',4,4',5,5'
190	2,3,3',4,4',5,6
191	2,3,3',4,4',5',6
192	2,3,3',4,5,5',6
193	2,3,3',4',5,5',6
194	2,2',3,3',4,4',5,5'
195	2,2',3,3',4,4',5,6
196	2,2',3,3',4,4',5',6
197	2,2',3,3',4,4',6,6'
198	2,2',3,3',4,5,5',6
199	2,2',3,3',4,5,6,6'
200	2,2',3,3',4,5',6,6'
201	2,2',3,3',4',5,5',6
202	2,2',3,3',5,5',6,6'
203	2,2',3,4,4',5,5',6
204	2,2',3,4,4',5,6,6'
205	2,3,3',4,4',5,5',6
206	2,2',3,3',4,4',5,5',6
207	2,2',3,3',4,4',5,6,6'
208	2,2',3,3',4,5,5',6,6'
209	2,2',3,3',4,4',5,5',6,6'

Source: Ballschmiter and Zell, 1980.

Appendix C

PCB Levels

Table C-1. PCB Levels in Marine Waters Throughout the World

Area	Location	Range ($ng \cdot L^{-1}$)	Reference
Open ocean	North Atlantic	<1–150	Harvey *et al.*, 1973
	North Atlantic	0.4–41	Harvey *et al.*, 1973
	Sargasso Sea	0.9–3.6	Bidleman and Olney, 1974
	Western North Pacific	0.04–0.6	Tanabe *et al.*, 1984
	Mediterranean Sea	0.2–8.6	Elder and Villeneuve, 1977
	Atlantic north of 30°N	0.4–7.1	Harvey and Steinhauer, 1976
	Atlantic 30°N–37°N	4.0–8.0	Harvey and Steinhauer, 1976
	Baltic Sea	0.3–140	Ehrhardt, 1981
	Gulf of Mexico	0.8–4.1	Nisbet, 1976
Coast	Southern California	2.3–36	Scura and McClure, 1975
	Northwest Mediterranean	1.5–38	Elder *et al.*, 1976
	Atlantic coast, U.S.A.	10–700	Sayler *et al.*, 1978
	Baltic coast	0.1–28	Ehrhardt, 1981
	Dutch coast	0.7–8	Duinker and Hillebrand, 1979
	Japanese coast	600–900	FAO, 1976
	Puget Sound, U.S.A.	3.0–22	Pavlou and Dexter, 1979
Estuary	Rhine-Meuse, Holland	10–200	Duinker and Hillebrand, 1979
	Brisbane, Australia	ND–9.0	Shaw and Connell, 1980
	Hudson River, U.S.A.	$<100–2.8 \times 10^6$	Nisbet, 1976
	Escambia Bay, U.S.A.	120	Duke *et al.*, 1970

ND = not detected

Table C-2. PCB Levels in Marine Sediments Throughout the World

Area	Location	Range ($\mu g \cdot g^{-1}$ dry weight)	Reference
Open ocean	Mediterranean Sea	0.0008–0.009	Elder *et al.*, 1976
	Gulf of Mexico	<0.0002–0.035	Nisbet, 1976
	North Atlantic	0.0003–0.044	Boehm, 1983
Coast	New York Bight, U.S.A.	0.0005–2.2	West and Hatcher, 1980
	Massachusetts Bay, U.S.A.	0.001–0.03	Boehm, 1983
	Japanese coast	0.01–390	FAO, 1976
	Puget Sound, U.S.A.	0.008–0.64	Pavlou and Dexter, 1979
	Southern California, U.S.A.	10.9	Young and Heesen, 1978
	San Pablo Bay, U.S.A.	0.006–0.017	Phillips and Spies, 1988
	San Francisco Bay, U.S.A.	0.03–0.05	Bopp *et al.*, 1981
Estuary	Rhine-Meuse, Holland	0.05–1	Duinker and Hillebrand, 1979
	Brisbane, Australia	ND–0.058	Shaw and Connell, 1980
	Chesapeake Bay, U.S.A.	0.004–0.4	Colwell and Sayler, 1977
	Hudson River, U.S.A.	6.6–6700	Nadeau and Davis, 1976
	Escambia Bay, U.S.A.	ND–486	Nisbet, 1976
	New Bedford Harbor, U.S.A.	<190 000	Segar and Davis, 1984

ND = not detected

Table C-3. PCB Residue Levels in Marine Biota (Whole Body, Wet Weight) Throughout the World

Area	Location	Biota	Range ($\mu g \cdot g^{-1}$)	Reference
Open ocean	Western North Pacific	Plankton	1.8	Tanabe *et al.*, 1984
	Atlantic, Scotland	Plankton	0.01–0.12	Williams and Holden, 1973
	Atlantic/Gulf of Mexico	Gag	ND–0.13	Stout, 1980
	Atlantic/Gulf of Mexico	Grouper	ND–0.06	Stout, 1980
	Atlantic/Gulf of Mexico	Snapper	ND–0.46	Stout, 1980
	Atlantic/Gulf of Mexico	Mackerel	0.034–1.78	Stout, 1980
	Antarctic	Penguin	0.02–0.1	Subramanian *et al.*, 1986
	Western North Pacific	Dolphin	2.8–4.1	Tanabe *et al.*, 1984
Coast	Scottish coast	Plankton	0.01–0.92	Williams and Holden, 1973
	Puget Sound, U.S.A.	Plankton	0.2	Stout and Beezhold, 1981
	Washington, U.S.A.	Mussel	0.09–0.12	Stout and Beezhold, 1981
	Washington, U.S.A.	Sea cucumber	0.05	Stout and Beezhold, 1981
	Puget Sound, U.S.A.	Shrimp	0.17–0.43	Stout and Beezhold, 1981
	Southern California	Mussel	0.01–0.52	Young and Heesen, 1978
	Western U.S. coast	Bass	0.19–0.99	Stout and Beezhold, 1981
	Western U.S. coast	Bocaccio	0.14–0.16	Stout and Beezhold, 1981
	Western U.S. coast	Mackerel	0.08–0.92	Stout and Beezhold, 1981
	Western U.S. coast	Salmon	0.04–0.24	Stout and Beezhold, 1981
	Puget Sound, U.S.A.	Sole	1.71	Stout and Beezhold, 1981
	Southern California	Sole	0.74–2.1	Young and Heesen, 1978
	Baltic	Seal	12–4500	Olsson, 1987
Estuary	Clyde estuary, Scotland	Plankton	0.08–2.2	Williams and Holden, 1973
	New Bedford Harbor, U.S.A.	Lobster	2.9–21.7	O'Connor and Huggett, 1988
	Chesapeake Bay, U.S.A.	Shellfish	ND–0.07	Eisenberg *et al.*, 1980
	Weser estuary, Germany	Worm	0.051–0.091	Goerke *et al.*, 1979
	Weser estuary, Germany	Shrimp	0.056–0.064	Goerke *et al.*, 1979
	Weser estuary, Germany	Clam	0.023–0.059	Goerke *et al.*, 1979
	Weser estuary, Germany	Mussel	0.007–0.017	Goerke *et al.*, 1979
	Weser estuary, Germany	Sole	0.155–0.281	Goerke *et al.*, 1979
	New Bedford Harbor, U.S.A.	Finfish	ND–700	Weaver, 1984
	Chesapeake Bay, U.S.A.	Finfish	ND–0.98	Eisenberg *et al.*, 1980

ND = not detected

Table C-4. PCB Levels in Canadian Marine Waters

Area	Location	Range (ng·L^{-1})	Reference
Coast	Atlantic	0.015	Strachan, 1988
	Pacific	ND	Strachan, 1988
Estuary	St. Lawrence	10.3–22.0	Germain and Langois, 1988

ND = not detected

Table C-5. PCB Levels in Canadian Marine Sediments

Area	Location	Range (μg·g^{-1})	Reference
Coast	Atlantic, Nfld.	0.25	Strachan, 1988
	Chaleur Bay, N.B.	0.001–0.022	Matheson and Bradshaw, 1985
	Vancouver, B.C. (open-ocean dump site)	1.05	Garrett, 1980
Estuary	Fraser River, B.C.	ND–0.21	Garrett, 1980

ND = not detected

Table C-6. PCB Levels (Whole Body, Wet Weight) in Canadian Marine Biota

Area	Location	Biota	Range ($\mu g \cdot g^{-1}$)	Reference
Coast	St. Lawrence	Plankton	1.39	Ware and Addison, 1973
	Chaleur Bay, N.B.	Bivalves	0.008–0.78	Matheson and Bradshaw, 1985
	St. Lawrence	Eel	ND–3.85	Desjardins *et al.*, 1983
	Chaleur Bay, N.B.	Finfish	0.02	Strachan, 1988
	Arctic	Char	ND–0.03	Muir, 1985
	Bay of Fundy, N.S.	Shearwater	17–73	Gaskin *et al.*, 1978
	Arctic to Atlantic	Geese	0.04–0.31	Longcore *et al.*, 1983
	Labrador coast	Fulmar	3.05–19.2	Noble and Elliott, 1986
	Nfld. coast	Puffin	0.35–0.46	Noble and Elliott, 1986
	Bay of Fundy, N.S.	Puffin	3.97–7.05	Noble and Elliott, 1986
	Queen Charlotte Is.	Petrel	3.51–4.83	Noble and Elliott, 1986
	Queen Charlotte Is.	Murre	1.04	Noble and Elliott, 1986
	Queen Charlotte Is.	Murrelet	0.52–1.40	Noble and Elliott, 1986
	Queen Charlotte Is.	Auklet	0.77–3.61	Noble and Elliott, 1986
	B.C. coast	Auklet	0.28–0.52	Noble and Elliott, 1986
	Nfld. coast	Dolphin	9.62–87.0	Muir *et al.*, 1988
	Nfld. coast	Whale	0.52–14.7	Muir *et al.*, 1988
	Bay of Fundy, N.S.	Porpoise	67.5	Muir *et al.*, 1988
Estuary	St. Lawrence	Shellfish	ND–1.6	Couillard, 1982
	Fraser River, B.C.	Crab	ND–2.1	Garrett, 1980
	Fraser River, B.C.	Clam	ND–0.08	Garrett, 1980
	Fraser River, B.C.	Oyster	0.32	Garrett, 1980
	Fraser River, B.C.	Shrimp	0.30	Garrett, 1980
	St. Lawrence	Finfish	ND–13.0	Couillard, 1982
	St. Lawrence	Eel	ND–16.6	Desjardins *et al.*, 1983
	Fraser River, B.C.	Flounder	ND–0.65	Garrett 1980
	Fraser River, B.C.	Salmon	ND–0.09	Garrett 1980
	Fraser River, B.C.	Trout	ND–0.75	Garrett 1980
	Fraser River, B.C.	Sanddab	0.21	Garrett 1980
	St. Lawrence	Beluga	136.8	Muir *et al.*, 1988

ND = not detected

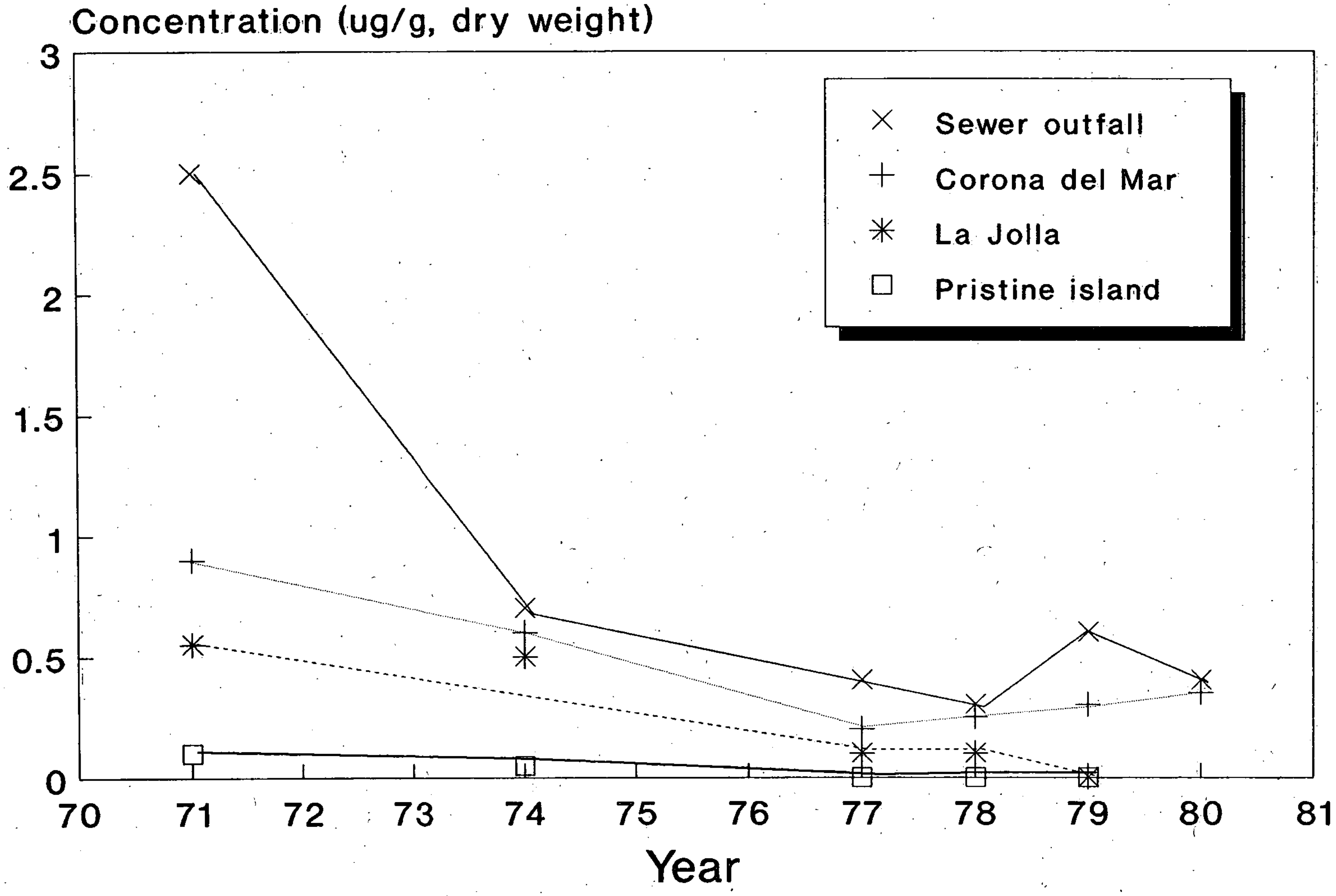

Figure C-1. PCB levels in mussels (*Mytilus californianus*) at the Los Angeles County sewer outfall, two coastal cities, and a pristine island site off the coast of southern California (data from Stout, 1986).

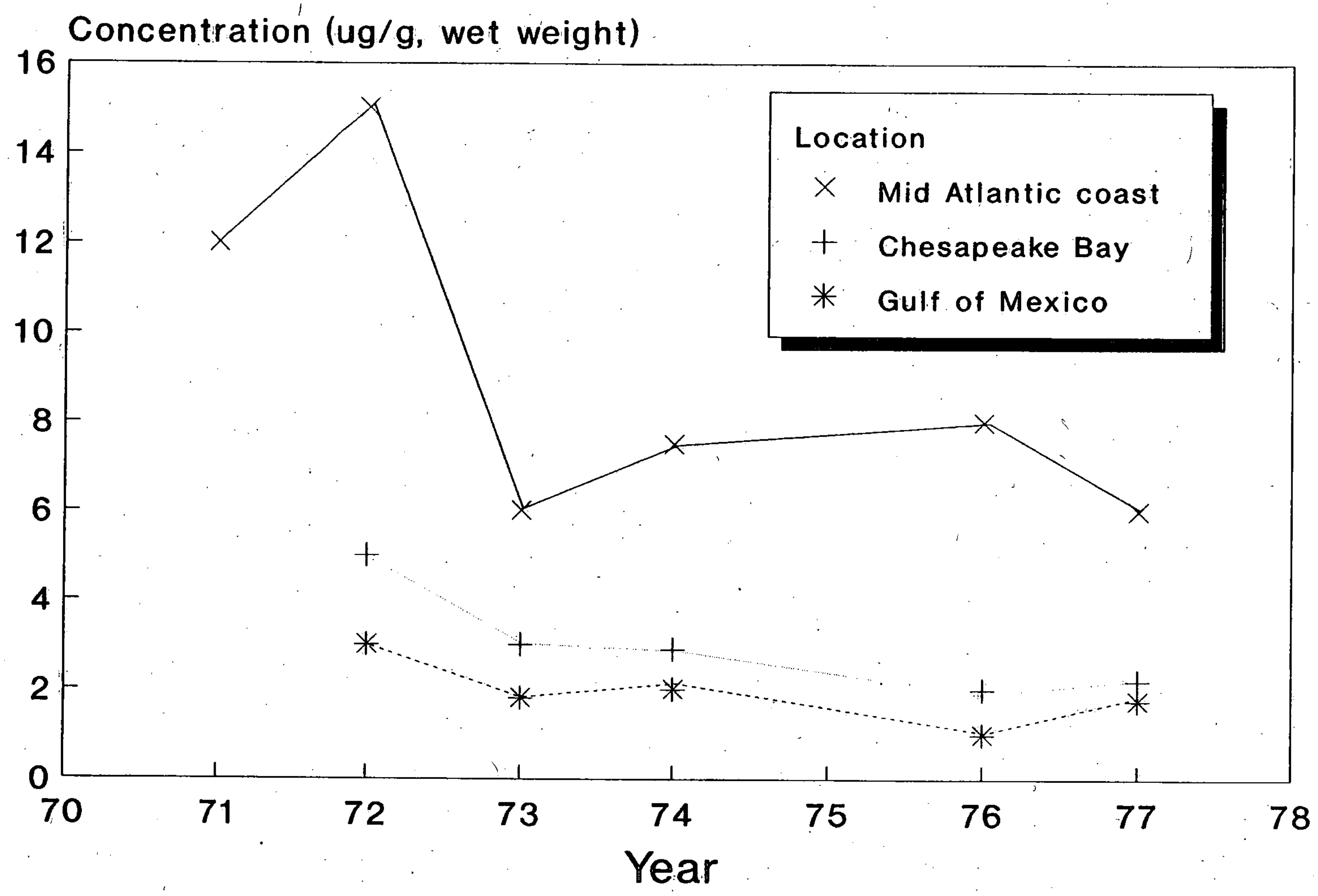

Figure C-2. **PCB levels in menhaden (*Brevoortia tyrannus* and *B. patronus*) oil from three areas of the fishery (data from Stout, 1986).**

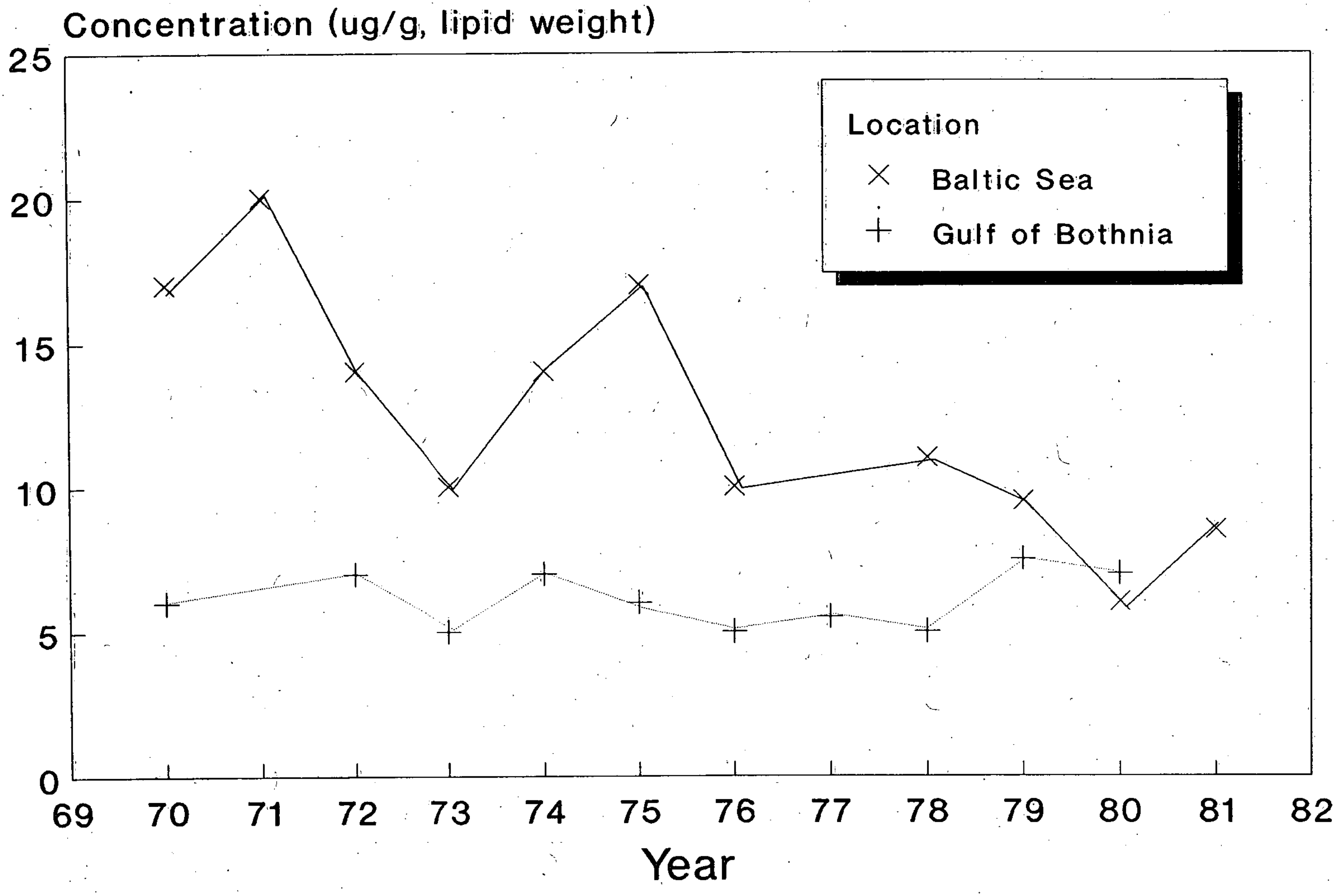

Figure C-3. PCB levels in herring muscle collected in the southern part of the Baltic and the Gulf of Bothnia (data from Olsson, 1987).

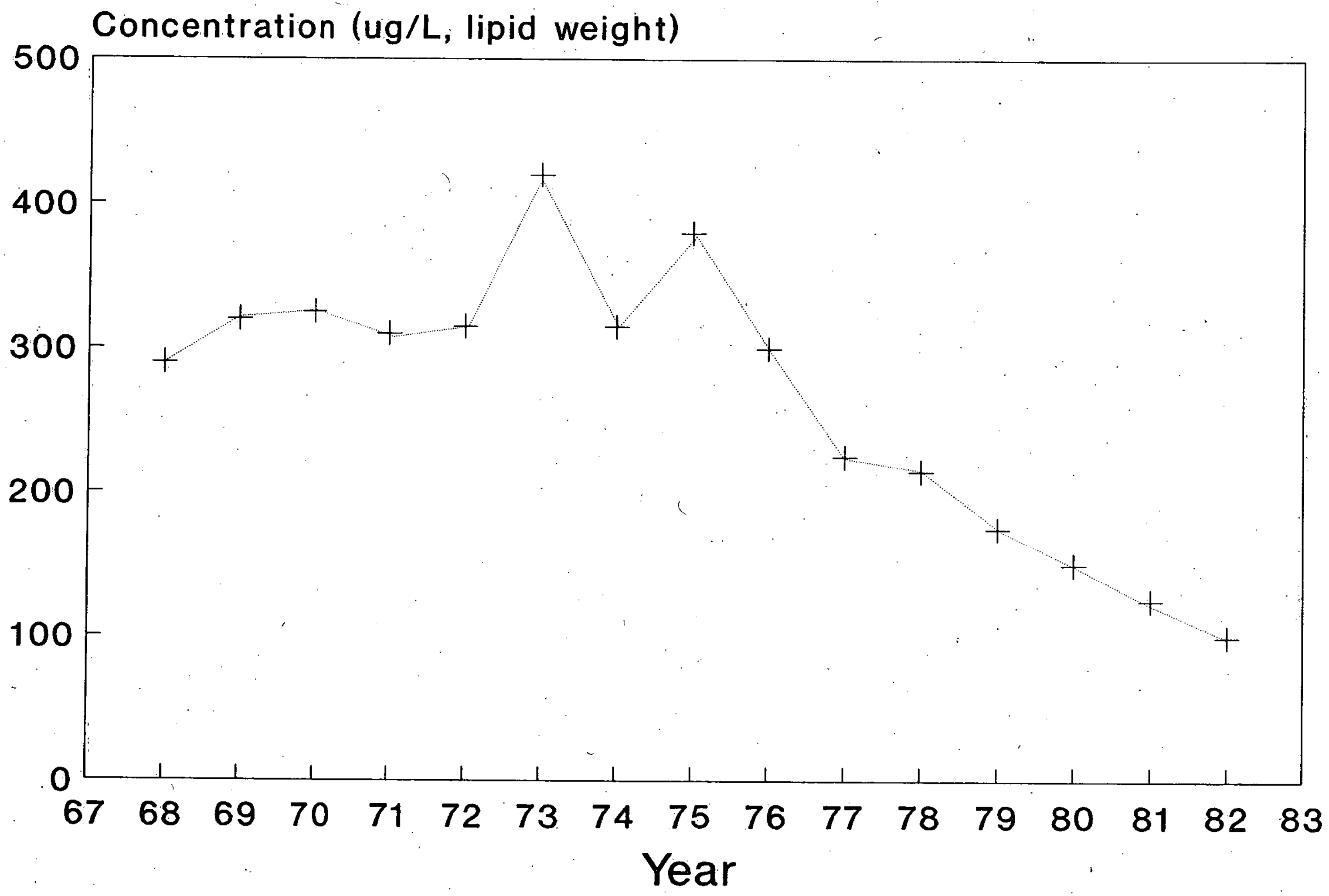

Figure C-4. PCB levels in guillemot eggs collected in the central part of the Baltic (data from Olsson, 1987).

Appendix D

PCB Bioaccumulation Factors (BAF) for Marine Biota from the Water Column

Table D-1. PCB Bioaccumulation Factors (BAF) for Marine Biota from the Water Column

Organism	Aroclor/Congener	BAF	Initial concentration	Duration	Reference
Plankton					
Cylindrotheca closterium	1254	1100	100 μg·L^{-1}	14 d	Keil *et al.*, 1971
Guinardia flaccida	2,4,5,2',5'-CB	460 000	NS	64 h	Harding and Phillips, 1978
Lithodesmium undulatum	2,4,5,2',5'-CB	360 000	NS	64 h	Harding and Phillips, 1978
Ditylum brightwellii	2,4,5,2',5'-CB	870 000	NS	64 h	Harding and Phillips, 1978
Lauderia borealis	2,4,5,2',5'-CB	1 400 000	NS	64 h	Harding and Phillips, 1978
Invertebrates					
Penaeus duorarum	1254	200 000	2.5 μg·L^{-1}	22 d	Nimmo *et al.*, 1971
Palaemonetes pugio	1254	5250–26 500	0.04–0.6 μg·L^{-1}	35 d	Nimmo *et al.*, 1974
Crassostrea virginica	1254	85 000–101 000	1.0–5.0 μg·L^{-1}	24–32 weeks	Parrish *et al.*, 1972
Crassostrea virginica	Di-CB	1200	NS	65 d	Vreeland, 1974
	Tri-CB	6200	NS	65 d	
	Tetra-CB	7400-11 000	NS	65 d	
	Penta-CB	27 000	NS	65 d	
	Hexa-CB	48 000	NS	65 d	
Nereis diversicolor	Phenoclor DP-5	800	0.6 μg·L^{-1}	14 d	Fowler *et al.*, 1978
Dipteran larvae	Mono-CB	1765–3319	9.0 ng·L^{-1}*	117 h	Wood *et al.*, 1987
	Di-CB	2458–8233	0.2–1.9 ng·L^{-1}	117 h	
	Tri-CB	4385–9732	0.04–0.7 ng·L^{-1}	117 h	
	Tetra-CB	3969–7226	0.03–0.3 ng·L^{-1}	117 h	
	Penta-CB	1514–4555	0.03–0.4 ng·L^{-1}	117 h	
	Hexa-CB	2232–4124	0.03–0.4 ng·L^{-1}	117 h	
	Hepta-CB	2965–4729	0.02–0.2 ng·L^{-1}	117 h	
	Octa-CB	3214–3852	0.03–0.05 ng·L^{-1}	117 h	
Fish					
Lagodon rhomboides	1016	17 000	1 μg·L^{-1}	56 d	Hansen *et al.*, 1974
Leiostomus xanthurus	1254	37 000	1 μg·L^{-1}	56 d	Hansen *et al.*, 1971
Mugil cephalus	1242	8700	2.2 μg·L^{-1}	70 d	Shaw and Connell, 1987
	1254	19 000	1.5 μg·L^{-1}	70 d	
	1260	16 000	1.3 μg·L^{-1}	70 d	
Morone saxatilis (Y-O-Y)	1254	34 700–96 000	0.4–0.9 μg·L^{-1}	48 h	Califano *et al.*, 1982
Solea solea (juveniles)	Tri-CB	290 000–1 100 000	0.9 pg·L^{-1}	42 d	Bonn and Duinker, 1985
	Tetra-CB	1 100 000–8 400 000	0.04–0.5 pg·L^{-1}	42 d	
	Penta-CB	800 000–30 000 000	0.01–0.2 pg·L^{-1}	42 d	
	Hexa-CB	3 400 000–30 000 000	0.1 pg·L^{-1}	42 d	
	Hepta-CB	3 000 000–44 000 000	0.01–0.1 pg·L^{-1}	42 d	

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*In this study, initial concentration and BAF varied between congeners of the same class.

NS = not stated
Y-O-Y = young-of-the-year
CB = chlorobiphenyl

Table D-1. Continued

Organism	Aroclor/Congener	BAF	Initial concentration	Duration	Reference
Cyprinodon variegatus (fry)	1254	16 000-32 000	$10.0\ \mu g\cdot L^{-1}$	21 d	Schimmel *et al.*, 1974
(adults)	1254	11 000-32 000	$10.0\ \mu g\cdot L^{-1}$	21 d	
C. variegatus (fry)	1016	14 400	$1\ \mu g\cdot L^{-1}$	28 d	Hansen *et al.*, 1975
(juveniles)	1016	43 100	$1\ \mu g\cdot L^{-1}$	28 d	
(adults)	1016	30 000	$1\ \mu g\cdot L^{-1}$	28 d	
Field Studies					
Krill	Total	400		Field	Subramanian *et al.*, 1986
Fish	Total	2500		Field	Subramanian *et al.*, 1986
Pygoscelis adeline (penguin)	Total	150 000		Field	Subramanian *et al.*, 1986
Leptonychotes weddelli (seal)	Total	250 000		Field	Subramanian *et al.*, 1986
Neritina reclivata (worm)	1254	> 7000		Field	Nimmo *et al.*, 1975
Penaeus spp. (shrimp)	1254	> 14 000		Field	Nimmo *et al.*, 1975
Callinectes spidus (crab)	1254	> 98 000		Field	Nimmo *et al.*, 1975
Anchoa mitchilli (anchovy)	1254	> 43 000		Field	Nimmo *et al.*, 1975
Arius spp. and *Bagre* spp. (catfish)	1254	> 54 000		Field	Nimmo *et al.*, 1975
Menidia beryllina (silverside)	1254	> 140 000		Field	Nimmo *et al.*, 1975
Bairdiella chrysura (perch)	1254	> 64 000		Field	Nimmo *et al.*, 1975
Cynoscion arenarius (seatrout)	1254	> 21 000		Field	Nimmo *et al.*, 1975
Leiostomus xanthurus (spot)	1254	> 26 000		Field	Nimmo *et al.*, 1975
Micropogon undulatus (croaker)	1254	> 23 000		Field	Nimmo *et al.*, 1975
Trinectes maculatus (hogchoaker)	1254	> 19 000		Field	Nimmo *et al.*, 1975
Trichiurus lepturus (cutlassfish)	1254	> 41 000		Field	Nimmo *et al.*, 1975

Appendix E

Studies

Table E-1. Acute Lethality Studies

Organism		Test rank	Test conditions	Aroclor/ Congener	Concentration ($\mu g \cdot L^{-1}$)	Duration (h)	Effect	Reference
<u>Plankton</u>								
Natural assemblage		SE	S,U	1242	6.5	24	EC_{50}	Moore and Harriss, 1972
				1254	15	24	EC_{50}	
				2,4'-CB	30	24	EC_{50}	
Dunaliella tertiolecta		SE	S,U	1254	9000	24	EC_{50}	Luard, 1973
Ditylum brightwellii		SE	S,U	1254	10	96	Reduced growth	Cosper *et al.*, 1987
					25	96	100% mortality	
Thalassiosira pseudonana		SE	S,U	1254	25	96	Reduced growth	Mosser *et al.*, 1972
T. pseudonana		SE	S,U	1254	50	48	Reduced photosynthesis	Michaels *et al.*, 1982
Skeletonema costatum		SE	S,U	1254	10	96	Reduced growth	Mosser *et al.*, 1972
D. tertiolecta		SE	S,U	1254	10	72	No effect	Fisher and Wurster, 1973
T. pseudonana		SE	S,U	1254	10	72	Reduced growth	Fisher and Wurster, 1973
Rhizosolenia setigera		SE	S,U	1254	1	144	Reduced growth	Fisher and Wurster, 1973
D. tertiolecta and *T. pseudonana*		UN	S,U	1254	50	72	Different dominant species	Biggs *et al.*, 1979
T. pseudonana	(open)	SE	S,U	1254	10	120	No effect	Fisher *et al.*, 1973; Fisher, 1977
	(estuary)						Reduced growth	
Fragilaria pinnata	(open)	SE	S,U	1254	10	120	Reduced growth	Fisher *et al.*, 1973; Fisher, 1977
	(estuary)						Reduced growth	

S = static conditions
F = flow-through conditions
U = unmeasured PCB concentrations
M = measured PCB concentrations

PR = primary study, which may be included in minimum data set for Canadian water quality guidelines or interim guidelines
SE = secondary study, which may be included in minimum data set for Canadian interim water quality guidelines
UN = unacceptable study, which cannot be included in minimum data set for Canadian water quality guidelines or interim guidelines

Table E-1. Continued

Organism	Test rank	Test conditions	Aroclor/ Congener	Concentration ($\mu g \cdot L^{-1}$)	Duration (h)	Effect	Reference
Bellerochea polymorpha (open)	SE	S,U	1254	10	120	Very reduced growth	Fisher *et al.*, 1973; Fisher, 1977
(estuary)						Very reduced growth	
S. costatum	SE	S,U	1221–1262	1000	144	No effect	Craigie and Hutzinger, 1975
Thalassiosira fluviatilis	SE	S,U	1221–1262	1000	144	No effect	Craigie and Hutzinger, 1975
D. tertiolecta	SE	S,U	1221–1262	1000	144	No effect	Craigie and Hutzinger, 1975
Monochrysis lutheri	SE	S,U	1248	1000	144	Reduced growth	Craigie and Hutzinger, 1975
Platymonas sp.	SE	S,U	1221–1262	1000	144	No effect	Craigie and Hutzinger, 1975
Porphyridium sp.	SE	S,U	1242–1254	1000	144	Reduced growth	Craigie and Hutzinger, 1975
Olisthodiscus sp.	SE	S,U	1242–1262	1000	144	Reduced growth (esp. 1248–1260)	Craigie and Hutzinger, 1975
<u>Invertebrates</u>							
Palaemonetes pugio (adult)	PR	S,M	1254	41	96	LC_{50}	Roesijadi *et al.*, 1976a
(juvenile)	PR	S,M	1254	7.8	96	LC_{50}	
(larvae)	PR	S,M	1254	15.6	96	LC_{50}	
P. pugio (juvenile)	PR	F,M	1016	12.5	96	LC_{50}	Hansen *et al.*, 1974
Penaeus aztecus	PR	F,M	1016	10.5	96	LC_{50}	Hansen *et al.*, 1974
Crangon septemspinosa	UN	S,M	1242	13.0	96	LC_{50}	McLeese and Metcalfe, 1980
			1254	12.0	96	LC_{50}	
Uca pugilator (adult)	SE	S,U	1254	100	96	No effect	Vernberg *et al.*, 1977
(juvenile)	SE	S,U	1254	10	96	LC_{50}	

Table E-1. Continued

Organism	Test rank	Test conditions	Aroclor/ Congener	Concentration ($\mu g \cdot L^{-1}$)	Duration (h)	Effect	Reference
Penaeus duorarum (juvenile)	PR	F,M	1254	100.0	96	100% mortality	Nimmo *et al.*, 1971
P. duorarum (juvenile)	SE	F,U	1254	100	48	100% mortality	Duke *et al.*, 1970
Crassostrea virginica	SE	F,U	1254	100	96	No mortality	Duke *et al.*, 1970
<u>Fish</u>							
Fundulus heteroclitus	UN	S,U	1221	16 000	96	LC_{50}	Kinter *et al.*, 1972
Lagodon rhomboides (juvenile)	SE	F,U	1254	100	48	No mortality	Duke *et al.*, 1970
L. rhomboides (juvenile)	PR	F,M	1016	56	96	18% mortality	Hansen *et al.*, 1974
<u>Other Biota</u>							
Tetrahymena pyriformis	UN	S,U	1254	1.0	96	8%-10% reduction in population density	Nimmo *et al.*, 1975
T. pyriformis	SE	S,U	1248	1000	96	10% reduction in population density	Cooley *et al.*, 1973
			1260	1000	96	13%-22% reduction in population density	
Escherichia coli	SE	F,U	1254	10 000	24	99% reduction in density	Colwell and Sayler, 1977

Table E-2. Chronic Lethality Studies

Organism		Test rank	Test conditions	Aroclor/ Congener	Concentration ($\mu g \cdot L^{-1}$)	Duration (h)	Effect	Reference
Plankton								
Cylindrotheca closterium		SE	S,U	1242	100	14	83% reduction in density	Keil *et al.*, 1971
Invertebrates								
Gammarus oceanicus		SE	S,U	1254	200	35	LC$_{50}$	Wildish, 1972
Penaeus duorarum	(adult)	PR	F,M	1254	3.5	35	LC$_{50}$	Nimmo *et al.*, 1971
	(juvenile)	PR	F,M	1254	0.94	15	LC$_{50}$	
Palaemonetes pugio		SE	F,U	1254	7.6	7	LC$_{50}$	Nimmo *et al.*, 1974
P. pugio (larvae)		PR	S,M	1254	15.6	5-6	75% mortality	Roesijadi *et al.*, 1976a
						11	100% mortality	
P. duorarum (juvenile)		PR	F,M	1254	0.57	15	30% mortality	Nimmo *et al.*, 1971
Fish								
Lagodon rhomboides		PR	F,M	1016	59	18	LC$_{50}$	Hansen *et al.*, 1974
					21	33	LC$_{50}$	Hansen *et al.*, 1974
Leiostomus xanthurus		PR	F,M	1254	5	38	LC$_{50}$	Hansen *et al.*, 1971
Cyprinodon variegatus	(adult)	PR	F,M	1254	3.5	21	No mortality	Schimmel *et al.*, 1974
	(juvenile)	PR	F,M	1254	3.5	21	24% mortality	
	(fry)	PR	F,M	1254	0.32	21	LC$_{50}$	
	(fry)	PR	F,M	1254	0.16	21	38% mortality	
	(fry)	PR	F,M	1254	0.06	21	No mortality	
	(embryo)	PR	F,M	1254	0.32	21	LC$_{50}$	
L. rhomboides (juvenile)		PR	F,M	1254	5	17	25% mortality	Hansen *et al.*, 1971

S = static conditions
F = flow-through conditions
U = unmeasured PCB concentrations,
M = measured PCB concentrations

PR = primary study, which may be included in minimum data set for Canadian water quality guidelines or interim guidelines
SE = secondary study, which may be included in minimum data set for Canadian interim water quality guidelines
UN = unacceptable study, which cannot be included in minimum data set for Canadian water quality guidelines or interim guidelines

Table E-3. Sublethality Studies

Organism	Test rank	Test conditions	Aroclor/ Congener	Concentration ($\mu g \cdot L^{-1}$)	Duration (h)	Effect	Reference
Plankton							
Scenedesmus obtusiusculus	SE	S,U	1242	300	24 h	Reduced phosphate uptake (21%)	Larsson and Tillberg, 1975
				1000	24 h	Reduced respiration rate (25%)	
Cylindrotheca closterium	SE	S,U	1242	10	14 d	Reduced RNA levels	Keil *et al.*, 1971
				100	14 d	Reduced chlorophyll index (80%)	
Invertebrates							
Arbacia punctulata	SE	F,U	1254	5000	25 min	Reduced fertilization (11%)	Adams and Slaughter-Williams, 1988
			1016	500	25 min	Reduced fertilization	
			1254	5000	3 d	Deformed gastulas (41%)	
			1016	1000	3 d	Deformed gastulas (53%)	
Gammarus tigrinus	SE	S,U	1254	25	44 h	Reduced respiration rate (18%)	O'Connor *et al.*, 1985
Palaemonetes pugio (adult)	SE	S,M	1254	29.4	96 h	Altered amino acid profile	Roesijadi *et al.*, 1976b
P. pugio (larvae)	PR	S,M	1254	3.2	11 d	Delayed larval development	Roesijadi *et al.*, 1976a
Crassostrea virginica	UN	F,U	1254	1.0	30 weeks	No effect	Parrish *et al.*, 1972
				5.0	24 weeks	Reduced growth rate	
C. virginica	PR	F,M	1016	0.6	96 h	Reduced growth rate (10%)	Hansen *et al.*, 1974
Fish							
Fundulus heteroclitus	UN	S,U	1221	75 000	24 h	Elevated serum osmolarity	Kinter *et al.*, 1972
F. heteroclitus (embryo)	SE	S,U	1242	10 000	7 d	Retardation of hatching	Weis and Weis, 1982
			1254	10 000	7 d	No effect	

S = static conditions
F = flow-through conditions

U = unmeasured PCB concentrations
M = measured PCB concentrations

PR = primary study, which may be included in minimum data set for Canadian water quality guidelines or interim guidelines
SE = secondary study, which may be included in minimum data set for Canadian interim water quality guidelines
UN = unacceptable study, which cannot be included in minimum data set for Canadian water quality guidelines or interim guidelines

Table E-4. PCB Effects on Biota under Stress

Organism	Test conditions	Aroclor/ Congener	Concentration ($\mu g \cdot L^{-1}$)*	Duration	Stress	Effect	Reference
<u>Plankton</u>							
Dunaliella tertiolecta	S,U	1254	10	5 d	Temperature	No effect	Fisher and Wurster, 1973
Thalassiosira pseudonana	S,U	1254	10	5 d	Temperature	Reduced growth rate (76%)	Fisher and Wurster, 1973
Rhizosolenia setigera	S,U	1254	0.1	8 d	Temperature	Reduced growth rate (18%)	Fisher and Wurster, 1973
Natural assemblage	F,U	1254	0.1	6 d	Nutrient	Reduced diversity (68%)	Fisher *et al.*, 1974
D. tertiolecta and *T. pseudonana*	F,M	1254	0.042	16 d	Nutrient	Change in dominant species	Fisher *et al.*, 1974
<u>Invertebrates</u>							
Palaemonetes pugio (juvenile)	S,M	1254	6.1–7.8	96 h	Salinity	LC_{50} unaffected by varying levels of constant salinity	Roesijadi *et al.*, 1976a
P. pugio (juvenile)	S,U	1254	1.8	96 h	Salinity	Increased mortality when salinity is not constant	Roesijadi *et al.*, 1976c
Uca pugilator (larvae)	S,U	1254	0.1	96 h	Salinity & temperature	Increased mortality	Vernberg *et al.*, 1977
		1016	0.1	96 h	Salinity & temperature	Increased mortality	
<u>Birds</u>							
Anas platyrhynchos	Feeding	1254	25–100 $mg \cdot kg^{-1}$	10 d	Duck hepatitis virus	Reduced disease resistance	Friend and Trainer, 1970

S = static conditions
F = flow-through conditions

U = unmeasured PCB concentrations
M = measured PCB concentrations

* Unless otherwise indicated.

Table E-5. Dietary and Exposure Studies

Organism	Test conditions	Aroclor/Congener	Concentration (mg·kg^{-1})	Duration	Effect	Reference
Invertebrates						
Nereis virens	Diet	1254	500	4 & 8 weeks	Elevated MFO & P-450	Fries and Lee, 1984
Fish						
Oncorhynhus kisutch	Injected	1254	0.15	-	6% mortality in seawater	Folmar *et al.*, 1982
Gadus morhua	Diet	1254	1–50	5.5 months	Altered biosynthetic patterns, deformed organs	Freeman *et al.*, 1982
Chelon labrosus	Diet	Phenochlor DP6	48	30 d	Altered biosynthetic patterns	Narbonne *et al.*, 1988
Birds						
Fratercula arctica	Injected	1254	30–35 mg/bird	3 years	No effect	Harris and Osborn, 1981
Larus fuscus	Diet	1254	50–400	8 weeks	Increase thyroid size, decreased heart size	Jefferies and Parslow, 1972
Uria aalge	Diet	1254	12–25 400	45 d 45 d	Increased thyroid size Decreased thyroid size	Jefferies and Parslow, 1976