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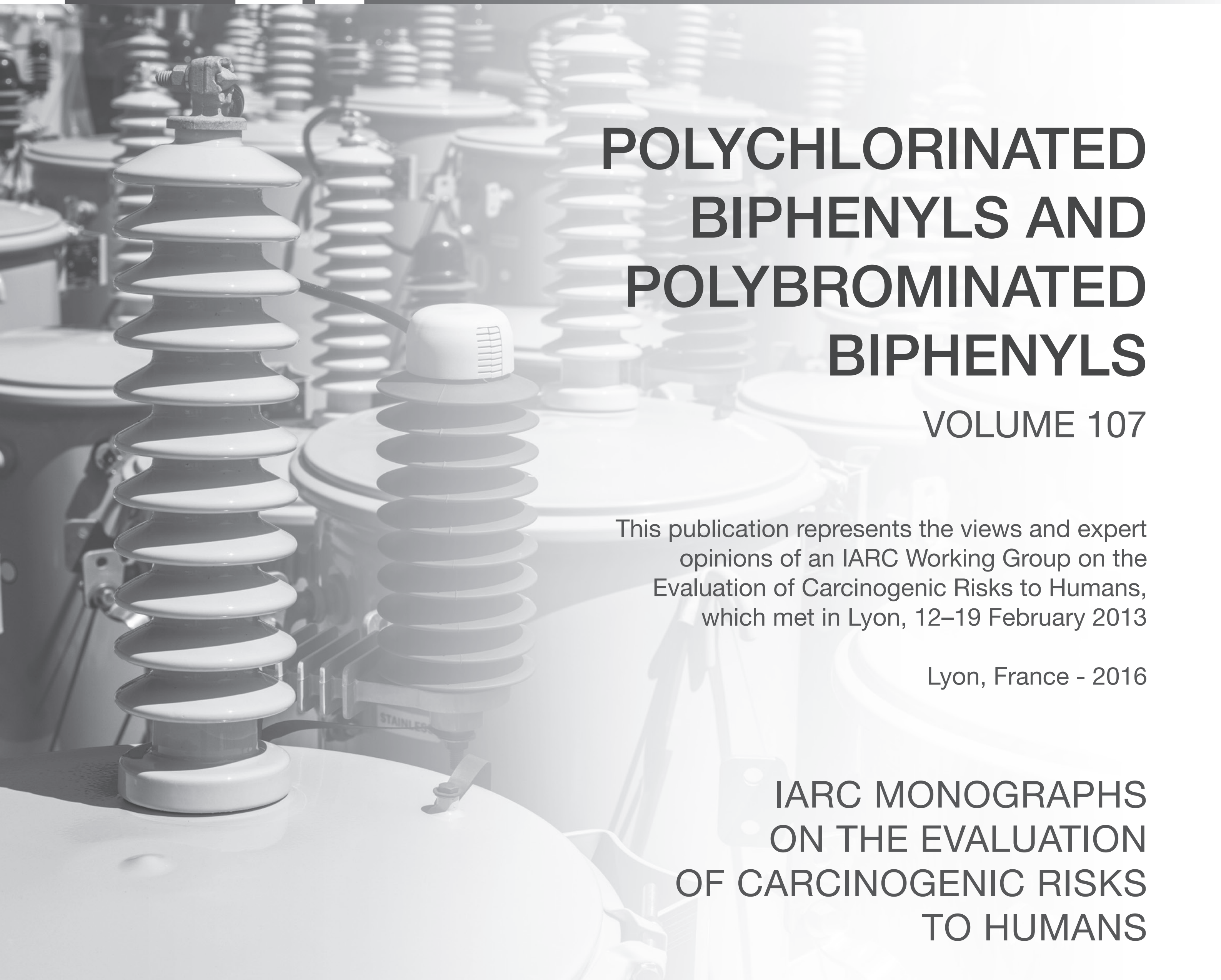

POLYCHLORINATED BIPHENYLS AND POLYBROMINATED BIPHENYLS

VOLUME 107

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ON THE EVALUATION
OF CARCINOGENIC RISKS
TO HUMANS

International Agency for Research on Cancer
World Health Organization

1. EXPOSURE DATA

1.1 Identification of the agent

1.1.1 Nomenclature

Polychlorinated biphenyls (PCBs) are a class of aromatic chemical compounds in which some or all hydrogen atoms attached to the biphenyl ring are substituted by chlorine atoms ($m + n = 1–10$) (Fig. 1.1). Synonyms for PCBs include chlorinated biphenyls, chlorinated diphenyls, chlorobiphenyls, or polychlorobiphenyls.

The general chemical formula is $C_{12}H_{(10\text{-m-n})}Cl_{(m+n)}$, where (m + n) is the number of chlorine atoms on the two rings. Depending on the position and number of the chlorine atoms, there are theoretically 209 individual PCB compounds (congeners). The carbon positions are numbered 1 to 6 on one ring, and 1′ to 6′ on the other. While positions 2,2′,6, and 6′ are called “*ortho*,” positions 3,3′,5 and 5′ are named “*meta*” and positions 4 and 4′ are called “*para*.”

Two different but correlated nomenclature systems are currently used. According to the International Union of Pure and Applied Chemistry (IUPAC) and in particular rule A-52.3 related to hydrocarbon systems, an unprimed number is considered lower (higher priority) than the same number when primed. Assemblies of unprimed and primed numbers are arranged in ascending numerical order. For a given PCB congener, the name lists the numbers sequentially [e.g. the PCB congener with chlorines on carbons 2,4,5, and 3′,4′ is identified as 2,3′,4,4′,5 (and not 2′,3,4,4′,5′)]. A deviation in that system lists the unprimed and primed chlorinated ring positions separately, sometimes eliminating the prime symbols and the commas for clarity and ease of typing (e.g. 245–3′4′5′ or 245–345).

In an additional strategy proposed by Ballschmiter & Zell (1980), a number (called “BZ number”) is attributed to each individual congener. This number correlates the structural arrangement of the PCB congener and ascending order of number of chlorine substitutions within each sequential homologue (Ballschmiter & Zell, 1980). This results in the congeners being numbered from PCB-1 to PCB-209. This shorthand nomenclature has become quite popular and is convenient for many uses, although it is important to note that it obscures the chemical identity of the congener and does not strictly follow the IUPAC rules.

Slight changes in the original BZ congener-numbering system were later recommended to correct some errors (Schulte & Malisch, 1983; Ballschmiter *et al.*, 1992), and this resulted in the renumbering of BZ numbers 199–201. Guitart *et al.* (1993) used a computer program to systematically renumber the PCBs according to the strict IUPAC rules. As a result, they recommended that the congeners previously numbered 107, 108, 109, 199, 200, and 201 be renumbered 109, 107, 108, 200, 201, and 199, respectively (reviewed in Mills *et al.*, 2007). The nomenclature for PCB congeners based on this report is shown in Table 1.1 and will be preferred in this *Monograph*. However, in the scientific literature,

Table 1.1 Correspondance between BZ number[a] and position of chlorine atoms on each phenyl ring of the PCBs[b]

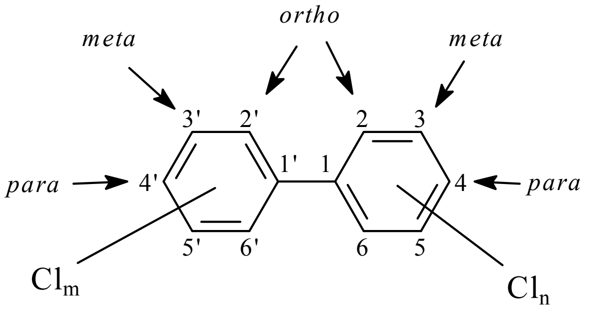
Position of chlorine atom on each ring	2	3	4	2,3	2,4	2,5	2,6	3,4	3,5	2,3,4	2,3,5	2,3,6	2,4,5	2,4,6	3,4,5	2,3,4,5	2,3,4,6	2,3,5,6	2,3,4,5,6
None	1	2	3	5	7	9	10	12	14	21	23	24	29	30	38	61	62	65	116
2′	4	6	8	16	17	18	19	33	34	41	43	45	48	50	76	86	88	93	142
3′		11	13	20	25	26	27	35	36	55	57	59	67	69	78	106	108	112	160
4′			15	22	28	31	32	37	39	60	63	64	74	75	**81**	**114**	115	117	166
2′,3′				40	42	44	46	56	58	82	83	84	97	98	122	129	131	134	173
2′,4′					47	49	51	66	68	85	90	91	99	100	**123**	137	139	147	181
2′,5′						52	53	70	72	87	92	95	101	103	124	141	144	151	185
2′,6′							54	71	73	89	94	96	102	104	125	143	145	152	186
3′,4′								**77**	79	**105**	109	110	**118**	119	**126**	**156**	158	163	190
3′,5′									80	107	111	113	120	121	127	159	161	165	192
2′,3′,4′										128	130	132	138	140	**157**	170	171	177	195
2′,3′,5′											133	135	146	148	162	172	175	178	198
2′,3′,6′												136	149	150	164	174	176	179	200
2′,4′,5′													153	154	**167**	180	183	187	203
2′,4′,6′														155	168	182	184	188	204
3′,4′,5′															**169**	**189**	191	193	205
2′,3′,4′,5′																194	196	199	206
2′,3′,4′,6′																	197	201	207
2′,3′,5′,6′																		202	208
2′,3′,4′,5′,6′																			209

[a] Revised PCB numbering system, including the revised numbering of congeners 107–109 and 199–201. For several PCB congeners, the indicated (truncated) structural names do not strictly adhere to the IUPAC rules (primed and unprimed numbers are interchanged). A comprehensive review of PCB nomenclature, including IUPAC names, is given in Mills *et al.* (2007).

[b] Dioxin-like PCBs are indicated in bold type

BZ, Ballschmiter and Zell; IUPAC, International Union of Pure and Applied Chemistry; PCB, polychlorinated biphenyl

Fig. 1.1 Chemical structure of PCBs and the IUPAC numbering system



Hydrogen atoms in positions 2,2′,6,6′ (*ortho*), 3,3′,5,5′ (*meta*) and/or 4,4′ (*para*) may be substituted by chlorine atoms; (m + n) is the number of chlorine atoms on the two rings
IUPAC, International Union of Pure and Applied Chemistry; PCB, polychlorinated biphenyl

the revised numbering of congeners 107–109 has not been adopted systematically; the numbering system commonly used has been that proposed by Ballschmiter *et al.* (1992) where only the original BZ numbers 199–201 are changed.

PCBs can be categorized by degree of chlorination (number of chlorine atoms) in 10 homologue groups (Table 1.2) from monochlorobiphenyls to decachlorobiphenyls. More than 60% of the PCBs are tetra- to hexachlorophenyls.

In the biphenyl molecule, the two aromatic rings can rotate about the connecting single 1,1′-bond (Fig. 1.1). As with all molecules, there is a low-energy preferred conformation. With PCBs, this conformation is dependent on the degree of chlorine substitution, since chlorine is larger than hydrogen and creates more steric hindrance to the rotation (Erickson, 2001). The two extreme theoretical configurations are “planar” or “coplanar,” in which the two benzene rings are in the same plane, and “non-planar” in which the benzene rings are at a 90° angle to each other (Faroon *et al.*, 2000). The probability of attaining a planar configuration is essentially determined by the number of substitutions in the *ortho* positions (2,2′,6,6′): the benzene rings of non-*ortho* substituted PCBs as well as mono-*ortho* substituted PCBs can assume a planar configuration and are referred to as “planar” or “coplanar” congeners (Erickson, 1997). The replacement of hydrogen atoms in the *ortho* positions with larger chlorine atoms forces the aromatic rings to rotate out of the planar configuration (Fig. 1.2); such structures are referred to as “non-planar” or “non-coplanar” congeners. [The Working Group does not recommend the use of this terminology, which is not technically appropriate since these PCBs do not easily assume a planar conformation.]

The relationship between PCB congener number and the Chemical Abstracts Service (CAS) registry number is given in Table 1.3. The congener numbering presented in this table follows that in Table 1.1, with the revised numbering of congeners 107–109. The congener lipophilicity is given in the same table, and was expressed against capacity to partition in octanol and water (K_{ow}) (see Section 1.1.2). Congeners can also be characterized by descriptors (CP0, CP1, 4Cl, PP, 2M) that give rapid access to geometry and substituent positions. The first descriptor, CP0, characterizes 20 congeners that are referred to as non-*ortho* congeners, consisting of those with chlorine substitution at none of the *ortho* positions on the biphenyl backbone. The second descriptor, CP1, comprises 48 congeners that are referred to as mono-*ortho* congeners and include those with chlorine substitution at only one of the *ortho* positions; CP0 and CP1 congeners can adopt a planar configuration. The 4Cl descriptor designates 169 congeners that have a total of four or more chlorine substituents, regardless of position. There are 54 PP congeners that have both *para* positions chlorinated. The 2M group contains 140 congeners that have two or more of the *meta* positions chlorinated. A total of 11 congeners have no descriptor.

The twelve congeners that display all descriptors are referred to as “dioxin-like” (Table 1.4). These twelve PCBs, namely PCB-77, PCB-81, PCB-105, PCB-114, PCB-118, PCB-123, PCB-126, PCB-156, PCB-157, PCB-167, PCB-169, and

Table 1.2 Physical and chemical properties of PCBs according to homologue group

Homologue group	CAS No.	Formula	No. of isomers	BZ No.	Relative molecular mass	Chlorine (% w/w)	Vapour pressure (Pa at 25 °C)[a]	Melting point (°C)[b]	Boiling point (°C)[c]
Monochlorobiphenyl	27323-18-8	$C_{12}H_9Cl$	3	1–3	188.66	18.79	1.1	25–77.9	285
Dichlorobiphenyl	25512-42-9	$C_{12}H_8Cl_2$	12	4–15	223.10	31.77	0.24	24.4–149	312
Trichlorobiphenyl	25323-68-6	$C_{12}H_7Cl_3$	24	16–39	257.55	41.30	0.054	28–87	337
Tetrachlorobiphenyl	26914-33-0	$C_{12}H_6Cl_4$	42	40–81	291.99	48.65	0.012	47–180	360
Pentachlorobiphenyl	25429-29-2	$C_{12}H_5Cl_5$	46	82–127	326.44	54.30	$2.6.10^{-3}$	76.5–124	381
Hexachlorobiphenyl	26601-64-9	$C_{12}H_4Cl_6$	42	128–169	360.88	58.93	$5.8.10^{-4}$	77–200	400
Heptachlorobiphenyl	28655-71-2	$C_{12}H_3Cl_7$	24	170–193	395.33	62.77	$1.3.10^{-4}$	83–149	417
Octachlorobiphenyl	55722-26-4	$C_{12}H_2Cl_8$	12	194–205	429.77	65.98	$2.8.10^{-5}$	159–162	432
Nonachlorobiphenyl	53742-07-7	$C_{12}HCl_9$	3	206–208	464.22	68.73	$6.3.10^{-6}$	182.8–206	445
Decachlorobiphenyl	2051-24-3	$C_{12}Cl_{10}$	1	209	498.66	71.10	$1.4.10^{-6}$	305.9	456

[a] Mean value for liquid.

[b] Values are approximations of the range across the isomers.

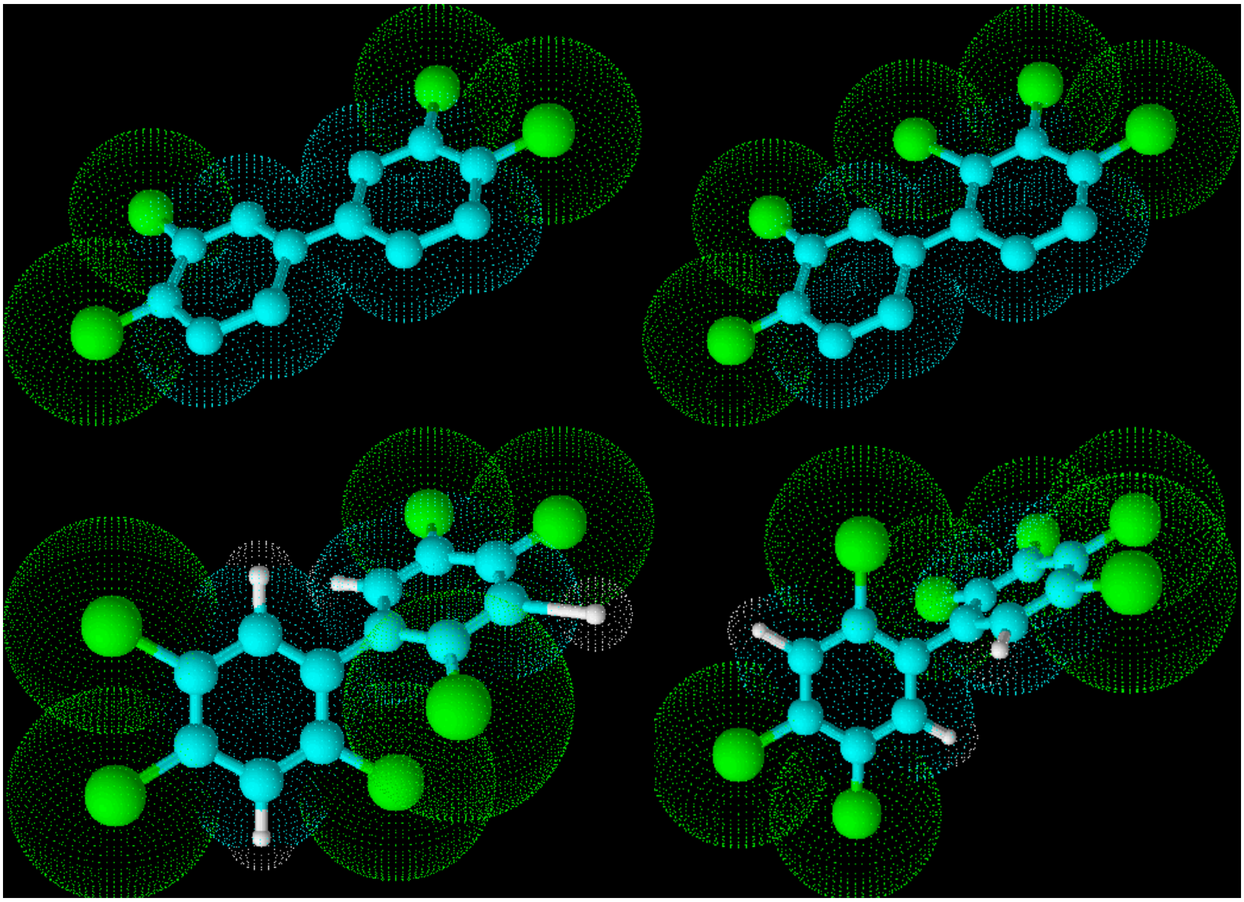
[c] Average value of all isomers in the group.

[The Working Group noted that the CAS No. for octachlorobiphenyl homologue group differs between ATSDR (2000) and Lindell (2012).]

BZ, Ballschmiter and Zell; CAS, Chemical Abstracts Service

From Shiu & Mackay (1986), ATSDR (2000), Erickson (2001), and Lindell (2012)

Fig. 1.2 Tridimensional chemical structures of selected PCBs



Upper panel: Spatial configuration of two dioxin-like PCBs: PCB-77 (3,3′,4,4′-tetrachlorobiphenyl), a non-*ortho* congener (left), and PCB-105 (2,3,3′,4,4′-pentachlorobiphenyl), a mono-*ortho* congener (right)
Lower panel: Spatial configuration of two di-*ortho* PCBs: PCB-153 (2,2′,4,4′,5,5′-hexachlorobiphenyl; left) and PCB-180 (2,2′,3,4,4′,5,5′-heptachlorobiphenyl; right)
Courtesy of Professor B. LeBizec

PCB-189, have been assigned toxicity equivalency factors (TEFs, assigned by WHO in 1998 and revised in 2005) (Van den Berg *et al.*, 2006). [The Working Group stressed that the activities of these PCB congeners are not solely dioxin-like.]

Depending on the context of the study or investigation, specific congeners may be monitored. For instance, the Stockholm Convention on Persistent Organic Pollutants (POPS) recommends measurement of six indicator PCBs (PCB-28, PCB-52, PCB-101, PCB-138, PCB-153, and PCB-180) to characterize contamination by PCBs. These congeners were chosen because they are found at higher concentrations in the environment, in food, or in human fluids/tissues. Depending on country and context, different lists of varying numbers of congeners may be used, e.g. 36 congeners for the Centers for Disease Control and Prevention, USA, or only PCB-138, PCB-153, and PCB-180 most frequently in epidemiological studies with human blood (see Section 2).

Of the 209 PCB congeners, 78 display axial chirality. Only 19 of these congeners, those with three or more chlorine atoms in the *ortho* position, exist as two mirror-image atropisomers, i.e. two chiral atropisomers (Lehmler & Robertson,

Table 1.3 Relationship between BZ number, CAS number, IUPAC name,[a] congener descriptor, and log K_{ow} for individual PCBs

BZ No.	IUPAC name	CAS No.	Descriptor[b]	Log K_{ow}	Vapour pressure (atm at 25 °C)[c]
1	2-CB	2051-60-7	CP1	4.46	
2	3-CB	2051-61-8	CP0	4.69	
3	4-CB	2051-62-9	CP0	4.69	
4	2,2′-DiCB	13029-08-8		4.65	1.5 to 4.2 × 10^{-6}
5	2,3-DiCB	16605-91-7	CP1	4.97	
6	2,3′-DiCB	25569-80-6	CP1	5.06	
7	2,4-DiCB	33284-50-3	CP1	5.07	9.9 × 10^{-7} to 2.1 × 10^{-6}
8	2,4′-DiCB	34883-43-7	CP1	5.07	
9	2,5-DiCB	34883-39-1	CP1	5.06	2.0 to 2.3 × 10^{-6}
10	2,6-DiCB	33146-45-1		4.84	
11	3,3′-DiCB	2050-67-1	CP0, 2M	5.28	4.1 to 9.1 × 10^{-7}
12	3,4-DiCB	2974-92-7	CP0	5.22	1.3 × 10^{-8} to 7.8 × 10^{-7}
13	3,4′-DiCB	2974-90-5	CP0	5.29	
14	3,5-DiCB	34883-41-5	CP0, 2M	5.28	
15	4,4′-DiCB	2050-68-2	CP0, PP	5.30	5.0 to 7.4 × 10^{-7}
16	2,2′,3-TriCB	38444-78-9		5.16	
17	2,2′,4-TriCB	37680-66-3		5.25	
18	2,2′,5-TriCB	37680-65-2		5.24	3.5 × 10^{-7} to 1.2 × 10^{-6}
19	2,2′,6-TriCB	38444-73-4		5.02	
20	2,3,3′-TriCB	38444-84-7	CP1, 2M	5.57	
21	2,3,4-TriCB	55702-46-0	CP1	5.51	
22	2,3,4′-TriCB	38444-85-8	CP1	5.58	
23	2,3,5-TriCB	55720-44-0	CP1, 2M	5.57	
24	2,3,6-TriCB	55702-45-9		5.35	
25	2,3′,4-TriCB	55712-37-3	CP1	5.67	
26	2,3′,5-TriCB	38444-81-4	CP1, 2M	5.66	1.8 to 4.5 × 10^{-7}
27	2,3′,6-TriCB	38444-76-7		5.44	
28	2,4,4′-TriCB	7012-37-5	CP1, PP	5.67	1.5 to 3.3 × 10^{-7}
29	2,4,5-TriCB	15862-07-4	CP1	5.60	
30	2,4,6-TriCB	35693-92-6		5.44	9.3 × 10^{-7} to 1.5 × 10^{-6}
31	2,4′,5-TriCB	16606-02-3	CP1	5.67	
32	2,4′,6-TriCB	38444-77-8		5.44	
33	2,3′,4′-TriCB	38444-86-9	CP1	5.60	
34	2,3′,5′-TriCB	37680-68-5	CP1, 2M	5.66	
35	3,3′,4-TriCB	37680-69-6	CP0, 2M	5.82	
36	3,3′,5-TriCB	38444-87-0	CP0, 2M	5.88	
37	3,4,4′-TriCB	38444-90-5	CP0, PP	5.83	
38	3,4,5-TriCB	53555-66-1	CP0, 2M	5.76	
39	3,4′,5-TriCB	38444-88-1	CP0, 2M	5.89	
40	2,2′,3,3′-TetraCB	38444-93-8	4CL, 2M	5.66	4.5 × 10^{-8} to 1.1 × 10^{-7}
41	2,2′,3,4-TetraCB	52663-59-9	4CL	5.69	
42	2,2′,3,4′-TetraCB	36559-22-5	4CL	5.76	
43	2,2′,3,5-TetraCB	70362-46-8	4CL, 2M	5.75	
44	2,2′,3,5′-TetraCB	41464-39-5	4CL, 2M	5.75	

Table 1.3 (continued)

BZ No.	IUPAC name	CAS No.	Descriptor[b]	Log K_{ow}	Vapour pressure (atm at 25 °C)[c]
45	2,2′,3,6-TetraCB	70362-45-7	4CL	5.53	
46	2,2′,3,6′-TetraCB	41464-47-5	4CL	5.53	
47	2,2′,4,4′-TetraCB	2437-79-8	4CL, PP	5.85	
48	2,2′,4,5-TetraCB	70362-47-9	4CL	5.78	
49	2,2′,4,5′-TetraCB	41464-40-8	4CL	5.85	
50	2,2′,4,6-TetraCB	62796-65-0	4CL	5.63	
51	2,2′,4,6′-TetraCB	68194-04-7	4CL	5.63	
52	2,2′,5,5′-TetraCB	35693-99-3	4CL, 2M	5.84	1.8 to 8.9 × 10^{-7}
53	2,2′,5,6′-TetraCB	41464-41-9	4CL	5.62	1.1 to 4.0 × 10^{-7}
54	2,2′,6,6′-TetraCB	15968-05-5	4CL	5.21	1.2 × 10^{-6} to 6.5 × 10^{-7}
55	2,3,3′,4-TetraCB	74338-24-2	CP1, 4CL, 2M	6.11	
56	2,3,3′,4′-TetraCB	41464-43-1	CP1, 4CL, 2M	6.11	
57	2,3,3′,5-TetraCB	70424-67-8	CP1, 4CL, 2M	6.17	
58	2,3,3′,5′-TetraCB	41464-49-7	CP1, 4CL, 2M	6.17	
59	2,3,3′,6-TetraCB	74472-33-6	4CL, 2M	5.95	
60	2,3,4,4′-TetraCB	33025-41-1	CP1, 4CL, PP	6.11	
61	2,3,4,5-TetraCB	33284-53-6	CP1, 4CL, 2M	6.04	
62	2,3,4,6-TetraCB	54230-22-7	4CL	5.89	
63	2,3,4′,5-TetraCB	74472-34-7	CP1, 4CL, 2M	6.17	
64	2,3,4′,6-TetraCB	52663-58-8	4CL	5.95	
65	2,3,5,6-TetraCB	33284-54-7	4CL, 2M	5.86	
66	2,3′,4,4′-TetraCB	32598-10-0	CP1, 4CL, PP	6.20	
67	2,3′,4,5-TetraCB	73575-53-8	CP1, 4CL, 2M	6.20	
68	2,3′,4,5′-TetraCB	73575-52-7	CP1, 4CL, 2M	6.26	
69	2,3′,4,6-TetraCB	60233-24-1	4CL	6.04	
70	2,3′,4′,5-TetraCB	32598-11-1	CP1, 4CL, 2M	6.20	
71	2,3′,4′,6-TetraCB	41464-46-4	4CL	5.98	
72	2,3′,5,5′-TetraCB	41464-42-0	CP1, 4CL, 2M	6.26	
73	2,3′,5′,6-TetraCB	74338-23-1	4CL, 2M	6.04	
74	2,4,4′,5-TetraCB	32690-93-0	CP1, 4CL, PP	6.20	
75	2,4,4′,6-TetraCB	32598-12-2	4CL, PP	6.05	
76	2,3′,4′,5′-TetraCB	70362-48-0	CP1, 4CL, 2M	6.13	
77	3,3′,4,4′-TetraCB	32598-13-3	**CP0, 4CL, PP, 2M**	6.36	5.2 × 10^{-9} to 2.1 × 10^{-8}
78	3,3′,4,5-TetraCB	70362-49-1	CP0, 4CL, 2M	6.35	
79	3,3′,4,5′-TetraCB	41464-48-6	CP0, 4CL, 2M	6.42	
80	3,3′,5,5′-TetraCB	33284-52-5	CP0, 4CL, 2M	6.48	
81	3,4,4′,5-TetraCB	70362-50-4	**CP0, 4CL, PP, 2M**	6.36	
82	2,2′,3,3′,4-PentaCB	52663-62-4	4CL, 2M	6.20	
83	2,2′,3,3′,5-PentaCB	60145-20-2	4CL, 2M	6.26	
84	2,2′,3,3′,6-PentaCB	52663-60-2	4CL, 2M	6.04	
85	2,2′,3,4,4′-PentaCB	65510-45-4	4CL, PP	6.30	
86	2,2′,3,4,5-PentaCB	55312-69-1	4CL, 2M	6.23	
87	2,2′,3,4,5′-PentaCB	38380-02-8	4CL, 2M	6.29	
88	2,2′,3,4,6-PentaCB	55215-17-3	4CL	6.07	
89	2,2′,3,4,6′-PentaCB	73575-57-2	4CL	6.07	

Table 1.3 (continued)

BZ No.	IUPAC name	CAS No.	Descriptor[b]	Log K_{ow}	Vapour pressure (atm at 25 °C)[c]
90	2,2′,3,4′,5-PentaCB	68194-07-0	4CL, 2M	6.36	
91	2,2′,3,4′,6-PentaCB	68194-05-8	4CL	6.13	
92	2,2′,3,5,5′-PentaCB	52663-61-3	4CL, 2M	6.35	
93	2,2′,3,5,6-PentaCB	73575-56-1	4CL, 2M	6.04	
94	2,2′,3,5,6′-PentaCB	73575-55-0	4CL, 2M	6.13	
95	2,2′,3,5′,6-PentaCB	38379-99-6	4CL, 2M	6.13	
96	2,2′,3,6,6′-PentaCB	73575-54-9	4CL	5.71	
97	2,2′,3,4′,5′-PentaCB	41464-51-1	4CL, 2M	6.29	
98	2,2′,3,4′,6′-PentaCB	60233-25-2	4CL	6.13	
99	2,2′,4,4′,5-PentaCB	38380-01-7	4CL, PP	6.39	
100	2,2′,4,4′,6-PentaCB	39485-83-1	4CL, PP	6.23	
101	2,2′,4,5,5′-PentaCB	37680-73-2	4CL, 2M	6.38	1.4 to 3.5 × 10^{-8}
102	2,2′,4,5,6′-PentaCB	68194-06-9	4CL	6.16	
103	2,2′,4,5′,6-PentaCB	60145-21-3	4CL	6.22	
104	2,2′,4,6,6′-PentaCB	56558-16-8	4CL	5.81	4.3 × 10^{-8} to 1.7 × 10^{-7}
105	2,3,3′,4,4′-PentaCB	32598-14-4	**CP1, 4CL, PP, 2M**	6.65	8.6 × 10^{-9}
106	2,3,3′,4,5-PentaCB	70424-69-0	CP1, 4CL, 2M	6.64	
107	2,3,3′,4,5′-PentaCB	70424-68-9	CP1, 4CL, 2M	6.71	
108	2,3,3′,4,6-PentaCB	70362-41-3	4CL, 2M	6.72	
109	2,3,3′,4′,5-PentaCB	74472-35-8	CP1, 4CL, 2M	6.48	
110	2,3,3′,4′,6-PentaCB	38380-03-9	4CL, 2M	6.48	
111	2,3,3′,5,5′-PentaCB	39635-32-0	CP1, 4CL, 2M	6.76	
112	2,3,3′,5,6-PentaCB	74472-36-9	4CL, 2M	6.45	
113	2,3,3′,5′,6-PentaCB	68194-10-5	4CL, 2M	6.54	
114	2,3,4,4′,5-PentaCB	74472-37-0	**CP1, 4CL, PP, 2M**	6.65	
115	2,3,4,4′,6-PentaCB	74472-38-1	4CL, PP	6.49	
116	2,3,4,5,6-PentaCB	18259-05-7	4CL, 2M	6.33	
117	2,3,4′,5,6-PentaCB	68194-11-6	4CL, 2M	6.46	
118	2,3′,4,4′,5-PentaCB	31508-00-6	**CP1, 4CL, PP, 2M**	6.74	1.2 × 10^{-8}
119	2,3′,4,4′,6-PentaCB	56558-17-9	4CL, PP	6.58	
120	2,3′,4,5,5′-PentaCB	68194-12-7	CP1, 4CL, 2M	6.79	
121	2,3′,4,5′,6-PentaCB	56558-18-0	4CL, 2M	6.64	
122	2,3,3′,4′,5′-PentaCB	76842-07-4	CP1, 4CL, 2M	6.64	
123	2,3′,4,4′,5′-PentaCB	65510-44-3	**CP1, 4CL, PP, 2M**	6.74	
124	2,3′,4′,5,5′-PentaCB	70424-70-3	CP1, 4CL, 2M	6.73	
125	2,3′,4′,5′,6-PentaCB	74472-39-2	4CL, 2M	6.51	
126	3,3′,4,4′,5-PentaCB	57465-28-8	**CP0, 4CL, PP, 2M**	6.89	
127	3,3′,4,5,5′-PentaCB	39635-33-1	CP0, 4CL, 2M	6.95	
128	2,2′,3,3′,4,4′-HexaCB	38380-07-3	4CL, PP, 2M	6.74	1.0 to 3.6 × 10^{-9}
129	2,2′,3,3′,4,5-HexaCB	55215-18-4	4CL, 2M	6.73	
130	2,2′,3,3′,4,5′-HexaCB	52663-66-8	4CL, 2M	6.80	
131	2,2′,3,3′,4,6-HexaCB	61798-70-7	4CL, 2M	6.58	
132	2,2′,3,3′,4,6′-HexaCB	38380-05-1	4CL, 2M	6.58	
133	2,2′,3,3′,5,5′-HexaCB	35694-04-3	4CL, 2M	6.86	
134	2,2′,3,3′,5,6-HexaCB	52704-70-8	4CL, 2M	6.55	

Table 1.3 (continued)

BZ No.	IUPAC name	CAS No.	Descriptor[b]	Log K_{ow}	Vapour pressure (atm at 25 °C)[c]
135	2,2′,3,3′,5,6′-HexaCB	52744-13-5	4CL, 2M	6.64	
136	2,2′,3,3′,6,6′-HexaCB	38411-22-2	4CL, 2M	6.22	
137	2,2′3,4,4′,5-HexaCB	35694-06-5	4CL, PP, 2M	6.83	
138	2,2′,3,4,4′,5′-HexaCB	35065-28-2	4CL, PP, 2M	6.83	5.2×10^{-9}
139	2,2′,3,4,4′,6-HexaCB	56030-56-9	4CL, PP	6.67	
140	2,2′,3,4,4′,6′-HexaCB	59291-64-4	4CL, PP	6.67	
141	2,2′,3,4,5,5′-HexaCB	52712-04-6	4CL, 2M	6.82	
142	2,2′,3,4,5,6-HexaCB	41411-61-4	4CL, 2M	6.51	
143	2,2′,3,4,5,6′-HexaCB	68194-15-0	4CL, 2M	6.60	
144	2,2′,3,4,5′,6-HexaCB	68194-14-9	4CL, 2M	6.67	
145	2,2′,3,4,6,6′-HexaCB	74472-40-5	4CL	6.25	
146	2,2′,3,4′,5,5′-HexaCB	51908-16-8	4CL, 2M	6.89	
147	2,2′,3,4′,5,6-HexaCB	68194-13-8	4CL, 2M	6.64	
148	2,2′,3,4′,5,6′-HexaCB	74472-41-6	4CL, 2M	6.73	
149	2,2′,3,4′,5′,6-HexaCB	38380-04-0	4CL, 2M	6.67	
150	2,2′,3,4′,6,6′-HexaCB	68194-08-1	4CL	6.32	
151	2,2′,3,5,5′,6-HexaCB	52663-63-5	4CL, 2M	6.64	
152	2,2′,3,5,6,6′-HexaCB	68194-09-2	4CL, 2M	6.22	
153	2,2′,4,4′,5,5′-HexaCB	35065-27-1	4CL, PP, 2M	6.92	1.9×10^{-9} to 6.9×10^{-8}
154	2,2′,4,4′,5,6′-HexaCB	60145-22-4	4CL, PP	6.76	
155	2,2′,4,4′,6,6′-HexaCB	33979-03-2	4CL, PP	6.41	3.5×10^{-9} to 4.4×10^{-8}
156	2,3,3′,4,4′,5-HexaCB	38380-08-4	**CP1, 4CL, PP, 2M**	7.18	2.1×10^{-9}
157	2,3,3′,4,4′,5′-HexaCB	69782-90-7	**CP1, 4CL, PP, 2M**	7.18	
158	2,3,3′,4,4′,6-HexaCB	74472-42-7	4CL, PP, 2M	7.02	
159	2,3,3′,4,5,5′-HexaCB	39635-35-3	CP1, 4CL, 2M	7.24	
160	2,3,3′,4,5,6-HexaCB	41411-62-5	4CL, 2M	6.93	
161	2,3,3′,4,5′,6-HexaCB	74472-43-8	4CL, 2M	7.08	
162	2,3,3′,4′,5,5′-HexaCB	39635-34-2	CP1, 4CL, 2M	7.24	
163	2,3,3′,4′,5,6-HexaCB	74472-44-9	4CL, 2M	6.99	7.9×10^{-10}
164	2,3,3′,4′,5′,6-HexaCB	74472-45-0	4CL, 2M	7.02	
165	2,3,3′,5,5′,6-HexaCB	74472-46-1	4CL, 2M	7.05	
166	2,3,4,4′,5,6-HexaCB	41411-63-6	4CL, PP, 2M	6.93	
167	2,3′,4,4′,5,5′-HexaCB	52663-72-6	**CP1, 4CL, PP, 2M**	7.27	
168	2,3′,4,4′,5′,6-HexaCB	59291-65-5	4CL, PP, 2M	7.11	
169	3,3′,4,4′,5,5′-HexaCB	32774-16-6	**CP0, 4CL, PP, 2M**	7.42	7.9×10^{-10}
170	2,2′,3,3′,4,4′,5-HeptaCB	35065-30-6	4CL, PP, 2M	7.27	
171	2,2′,3,3′,4,4′,6-HeptaCB	52663-71-5	4CL, PP, 2M	7.11	
172	2,2′,3,3′,4,5,5′-HeptaCB	52663-74-8	4CL, 2M	7.33	
173	2,2′,3,3′,4,5,6-HeptaCB	68194-16-1	4CL, 2M	7.02	
174	2,2′,3,3′,4,5,6′-HeptaCB	38411-25-5	4CL, 2M	7.11	
175	2,2′,3,3′,4,5′,6-HeptaCB	40186-70-7	4CL, 2M	7.17	
176	2,2′,3,3′,4,6,6′-HeptaCB	52663-65-7	4CL, 2M	6.76	
177	2,2′,3,3′,4,5′,6′-HeptaCB	52663-70-4	4CL, 2M	7.08	
178	2,2′,3,3′,5,5′,6-HeptaCB	52663-67-9	4CL, 2M	7.14	
179	2,2′,3,3′,5,6,6′-HeptaCB	52663-64-6	4CL, 2M	6.73	

Table 1.3 (continued)

BZ No.	IUPAC name	CAS No.	Descriptor[b]	Log K_{ow}	Vapour pressure (atm at 25 °C)[c]
180	2,2′,3,4,4′,5,5′-HeptaCB	35065-29-3	4CL, PP, 2M	7.36	1.3×10^{-9}
181	2,2′,3,4,4′,5,6-HeptaCB	74472-47-2	4CL, PP, 2M	7.11	
182	2,2′,3,4,4′,5,6′-HeptaCB	60145-23-5	4CL, PP, 2M	7.20	
183	2,2′,3,4,4′,5′,6-HeptaCB	52663-69-1	4CL, PP, 2M	7.20	
184	2,2′,3,4,4′,6,6′-HeptaCB	74472-48-3	4CL, PP	6.85	
185	2,2′,3,4,5,5′,6-HeptaCB	52712-05-7	4CL, 2M	7.11	
186	2,2′,3,4,5,6,6′-HeptaCB	74472-49-4	4CL, 2M	6.69	
187	2,2′,3,4′,5,5′,6-HeptaCB	52663-68-0	4CL, 2M	7.17	
188	2,2′,3,4′,5,6,6′-HeptaCB	74487-85-7	4CL, 2M	6.82	
189	2,3,3′,4,4′,5,5′-HeptaCB	39635-31-9	**CP1, 4CL, PP, 2M**	7.71	
190	2,3,3′,4,4′,5,6-HeptaCB	41411-64-7	4CL, PP, 2M	7.46	
191	2,3,3′,4,4′,5′,6-HeptaCB	74472-50-7	4CL, PP, 2M	7.55	
192	2,3,3′,4,5,5′,6-HeptaCB	74472-51-8	4CL, 2M	7.52	
193	2,3,3′,4′,5,5′,6-HeptaCB	69782-91-8	4CL, 2M	7.52	
194	2,2′,3,3′,4,4′,5,5′-OctaCB	35694-08-7	4CL, PP, 2M	7.80	
195	2,2′,3,3′,4,4′,5,6-OctaCB	52663-78-2	4CL, PP, 2M	7.56	
196	2,2′,3,3′,4,4′,5,6′-OctaCB	42740-50-1	4CL, PP, 2M	7.65	
197	2,2′,3,3′,4,4′,6,6′-OctaCB	33091-17-7	4CL, PP, 2M	7.30	
198	2,2′,3,3′,4,5,5′,6-OctaCB	68194-17-2	4CL, 2M	7.62	
199	2,2′,3,3′,4,5,5′,6′-OctaCB	52663-75-9	4CL, 2M	7.62	
200	2,2′,3,3′,4,5,6,6′-OctaCB	52663-73-7	4CL, 2M	7.20	
201	2,2′,3,3′,4,5′,6,6′-OctaCB	40186-71-8	4CL, 2M	7.27	
202	2,2′,3,3′,5,5′,6,6′-OctaCB	2136-99-4	4CL, 2M	7.24	
203	2,2′,3,4,4′,5,5′,6-OctaCB	52663-76-0	4CL, PP, 2M	7.65	
204	2,2′,3,4,4′,5,6,6′-OctaCB	74472-52-9	4CL, PP, 2M	7.30	
205	2,3,3′,4,4′,5,5′,6-OctaCB	74472-53-0	4CL, PP, 2M	8.00	
206	2,2′,3,3′,4,4′,5,5′,6-NonaCB	40186-72-9	4CL, PP, 2M	8.09	
207	2,2′,3,3′,4,4′,5,6,6′-NonaCB	52663-79-3	4CL, PP, 2M	7.74	
208	2,2′,3,3′,4,5,5′,6,6′-NonaCB	52663-77-1	4CL, 2M	7.71	
209	2,2′,3,3′,4,4′,5,5′,6,6′-DecaCB	2051-24-3	4CL, PP, 2M	8.18	

[a] The nomenclature in this table adheres to the IUPAC rules and thus primed and unprimed numbers may be interchanged compared with Table 1.1. Please see text for more details.

[b] Congener descriptors (CP0, CP1, 4Cl, PP, 2M) have been given where relevant; they give rapid access to geometry and substituent positions. 68 coplanar congeners fall into one of two groups CP0 or CP1.

The first group of 20 congeners consists of those without chlorine substitution at any of the "*ortho*" positions on the biphenyl backbone and are referred to as CP0 or non-"*ortho*" congeners. The second group of 48 congeners includes those with chlorine substitution at only one of the "*ortho*" positions and are referred to as CP1 or mono-"*ortho*" congeners. 175 congeners have a total of four or more chlorine substituents, regardless of position (4Cl). 54 congeners have both "*para*" positions chlorinated (PP). 146 congeners have two or more of the "*meta*" positions chlorinated (2M). The twelve congeners that have all four of the congener descriptors are referred to as being "dioxin-like," and are indicated in bold type.

In ATSDR (2000), PCB-63 was mistakenly attributed the CAS number of a pentachlorobiphenyl; for Henry's law constants, vapour pressure and solubility of most individual congeners, the reader is referred to Dunnivant & Elzerman (1988) and references within.

[c] Vapour pressures have been indicated for a selection of individual congeners.

BZ, Ballschmiter and Zell; CAS, Chemical Abstracts Service; CB, chlorinated biphenyl; IUPAC, International Union of Pure and Applied Chemistry

From Dunnivant & Elzerman (1988), ATSDR (2000), Mills *et al.* (2007), and Lindell (2012)

Table 1.4 The 12 dioxin-like PCBs, with corresponding CAS number, IUPAC name, and individual WHO_{1998}-TEF and WHO_{2005}-TEF values

PCB	IUPAC name	CAS No.	WHO_{1998}-TEF	WHO_{2005}-TEF
PCB-77	3,3′,4,4′-TetraCB	32598-13-3	0.0001	0.0001
PCB-81	3,4,4′,5-TetraCB	70362-50-4	0.0001	0.0003
PCB-105	2,3,3′,4,4′-PentaCB	32598-14-4	0.0001	0.00003
PCB-114	2,3,4,4′,5-PentaCB	74472-37-0	0.0005	0.00003
PCB-118	2,3′,4,4′,5-PentaCB	31508-00-6	0.0001	0.00003
PCB-123	2,3′,4,4′,5-PentaCB	65510-44-3	0.0001	0.00003
PCB-126	3,3′,4,4′,5-PentaCB	57465-28-8	0.1	0.1
PCB-156	2,3,3′,4,4′,5-HexaCB	38380-08-4	0.0005	0.00003
PCB-157	2,3,3′,4,4′,5′-HexaCB	68782-90-7	0.0005	0.00003
PCB-167	2,3′,4,4′,5,5′-HexaCB	52663-72-6	0.00001	0.00003
PCB-169	3,3′,4,4′,5,5′-HexaCB	32774-16-6	0.01	0.03
PCB-189	2,3,3′,4,4′,5,5′-HeptaCB	39635-31-9	0.0001	0.00003

CAS, Chemical Abstracts Service; CB, chlorinated biphenyl; IUPAC, International Union of Pure and Applied Chemistry; PCB, polychlorinated biphenyl; TEF, toxicity equivalency factor
From Van den Berg *et al.* (1998, 2006)

2001; Kania-Korwel & Lehmler, 2013). The IUPAC nomenclature and BZ number for the 19 atropisomeric PCBs are listed in Table 1.5. They are stereoisomers resulting from hindered rotation around single bonds where the steric-strain barrier to rotation is high enough to allow for the isolation of the enantiomers (Haglund & Wiberg, 1996; Harju & Haglund, 1999). Both atropisomers have the same chemical and physical behaviour, except for optical rotation (Lehmler *et al.*, 2010). They are stable at 25 °C, but at elevated temperatures it is necessary to separate the enantiomers via high-resolution chiral gas chromatography (GC) (Schurig & Reich, 1998; Harju & Haglund, 1999).

1.1.2 Chemical and physical properties of PCBs

Pure single PCB congeners are mostly colourless or slightly yellowish, often odourless, crystalline compounds. Commercial products, however, are viscous liquid mixtures of these compounds, with viscosity increasing with degree of chlorination, and colour ranging from light yellow to a dark colour. For example, Aroclor 1242 is a "mobile liquid" and Aroclor 1260 is a "sticky resin" (Erickson, 2001). These products do not crystallize at low temperatures, but turn into solid resins. An important property of PCBs is their general inertness; they resist acids, alkalis and oxidants and are fire-resistant because of their high flash-points (IPCS, 2003). However, under certain conditions, they may be destroyed by chemical, thermal and biochemical processes. PCBs show excellent dielectric (insulating) properties. This has made them useful in a wide variety of applications, including as dielectric fluids in transformers and capacitors, heat-transfer fluids, and lubricants.

The physical properties of PCBs are important in understanding their analytical, physiological, and environmental properties. However, the interactions of the various physical properties can be extremely complex (Erickson, 2001). Chemical and physical properties such as solubility, vapour pressure, and Henry's law constant have been reported for individual congeners (Shiu & Mackay, 1986; Murphy *et al.*, 1987; Sabljić & Güsten, 1989; Dunnivant *et al.*, 1992; Falconer & Bidleman, 1994). Data for homologue groups and for a selection of PCBs are presented in Table 1.2,

Table 1.5 PCB congeners that exist as chiral atropisomers

PCB	IUPAC name
PCB-45	2,2′,3,6-TetraCB
PCB-84	2,2′,3,3′,6-PentaCB
PCB-88	2,2′,3,4,6-PentaCB
PCB-91	2,2′,3,4′,6-PentaCB
PCB-95	2,2′,3,5′,6-PentaCB
PCB-131	2,2′,3,3′,4,6-HexaCB
PCB-132	2,2′,3,3′,4,6′-HexaCB
PCB-135	2,2′,3,3′,5,6′-HexaCB
PCB-136	2,2′,3,3′,6,6′-HexaCB
PCB-139	2,2′,3,4,4′,6-HexaCB
PCB-144	2,2′,3,4,5′,6-HexaCB
PCB-149	2,2′,3,4′,5′,6-HexaCB
PCB-171	2,2′,3,3′,4,4′,6-HeptaCB
PCB-174	2,2′,3,3′,4,5,6′-HeptaCB
PCB-175	2,2′,3,3′,4,5′,6-HeptaCB
PCB-176	2,2′,3,3′,4,6,6′-HeptaCB
PCB-183	2,2′,3,4,4′,5′,6-HeptaCB
PCB-196	2,2′,3,3′,4,4′,5,6′-OctaCB
PCB-197	2,2′,3,3′,4,4′,6,6′-OctaCB

CB, chlorinated biphenyl; IUPAC, International Union of Pure and Applied Chemistry; PCB, polychlorinated biphenyl

Table 1.3, and Table 1.6. Melting points range from 25 °C (PCB-2, PCB-7 and PCB-9) to 306 °C (PCB-209). Boiling points increase from low (monochlorobiphenyl, 285 °C) to highly (decachlorobiphenyl, 456 °C) chlorinated congeners (Hutzinger *et al.*, 1974; Shiu & Mackay, 1986).

The solubility of PCBs in water is extremely low, ranging from an average of 0.0012 to 4830 µg/L for the chlorobiphenyl congeners that occur commonly. The high solubility of the *ortho*-chlorinated congeners (4.8 mg/L for PCB-1) may be due to hydrogen bonding associated with the more polar character of these molecules. Solubility decreases rapidly in *ortho*-vacant congeners, especially as the *para* positions are filled, which may result in greater and more uniform perimeter electronegativity and interference with hydrogen bonding. PCBs are freely soluble in non-polar organic solvents, oils and biological lipids, and the shift from water to lipid solubility is linked to the degree of chlorination (Hutzinger *et al.*, 1974; Shiu & Mackay, 1986; ATSDR, 2000; IPCS, 2003).

The octanol/water partition coefficient (K_{ow}) is defined as the ratio of a chemical's concentration in the octanol phase to its concentration in the aqueous phase of a two-phase octanol/water system; values of K_{ow} are thus unitless (Table 1.3 and Table 1.6). The reported log K_{ow} values have been reviewed by Shiu & Mackay (1986). Fig. 1.3 shows the remarkable correlation between log K_{ow} (lipophilicity) and number of chlorine atoms (BZ numbers); log K_{ow} values ranged from 4.5 to 8.3. This partitioning plays a key role in environmental fate and transport. PCBs tend to favour the non-polar phase and will partition away from water to most solids, the organic portion being the preferred site (Erickson, 2001).

PCBs are characterized by Henry's law constants [a measure of the equilibrium distribution coefficient between air and water] that tend to decrease with a higher degree of chlorination. Less chlorinated PCB congeners have a considerably higher vapour pressure (1–2 Pa at 25 °C for monochlorobiphenyls) than the more highly chlorinated congeners (1.4×10^{-6} Pa for decachlorobiphenyl) (Shiu & Mackay, 1986). Therefore, the composition in air is dominated by the less chlorinated congeners and atropoisomers.

At high temperatures, PCBs are combustible, and the products of combustion include polychlorinated dibenzofurans (PCDFs) and hydrogen chloride, and polychlorinated dibenzodioxins (PCDDs) (IPCS, 1993; ATSDR, 2000).

Photochemical degradation may be one route for the breakdown of PCBs in the environment: photochemical experiments conducted under simulated natural conditions on several pure chlorobiphenyls and on commercial PCB products have indicated several degradative reactions, such as dechlorination, polymerization and solvolysis.

Table 1.6 Physical and chemical data for a selection of PCB congeners

PCB	No. of chlorine atoms	Melting point (°C)	Boiling point (°C)	Vapour pressure (10^{-6} kPa at 25 °C)	Log K_{ow}	Water solubility (µg/L)
PCB-1[a]	1	34	274	184	4.5	4830 (25 °C)
PCB-105	5	–	–	0.87	7.0	3.4 (25 °C)
PCB-118	5	–	–	1.20	7.1	13.4 (20 °C)
PCB-138	6	78.5–80	400[b]	0.53	6.5–7.4[b]	15.9[b]
PCB-153	6	103–104	–	0.05	6.7	0.9 (25 °C)
PCB-156	6	–	–	0.21	7.6	5.3 (20 °C)
PCB-163	6	–	–	0.08	7.2	1.2 (25 °C)
PCB-169	6	201–202	–	0.05	7.4	0.04–12.3[b]
PCB-180	7	109–110	240–280 (at 2.66 kPa)	0.13	6.7–7.2[b]	0.2 (25 °C)
PCB-183	7	83	–	–	8.3	4.9 (20 °C)

[a] Included based on its significantly different solubility and vapour pressure

[b] Calculated

K_{ow}, octanol/water partition coefficient; PCB, polychlorinated biphenyl

From Lindell (2012)

1.1.3 Trade names and composition of commercial products

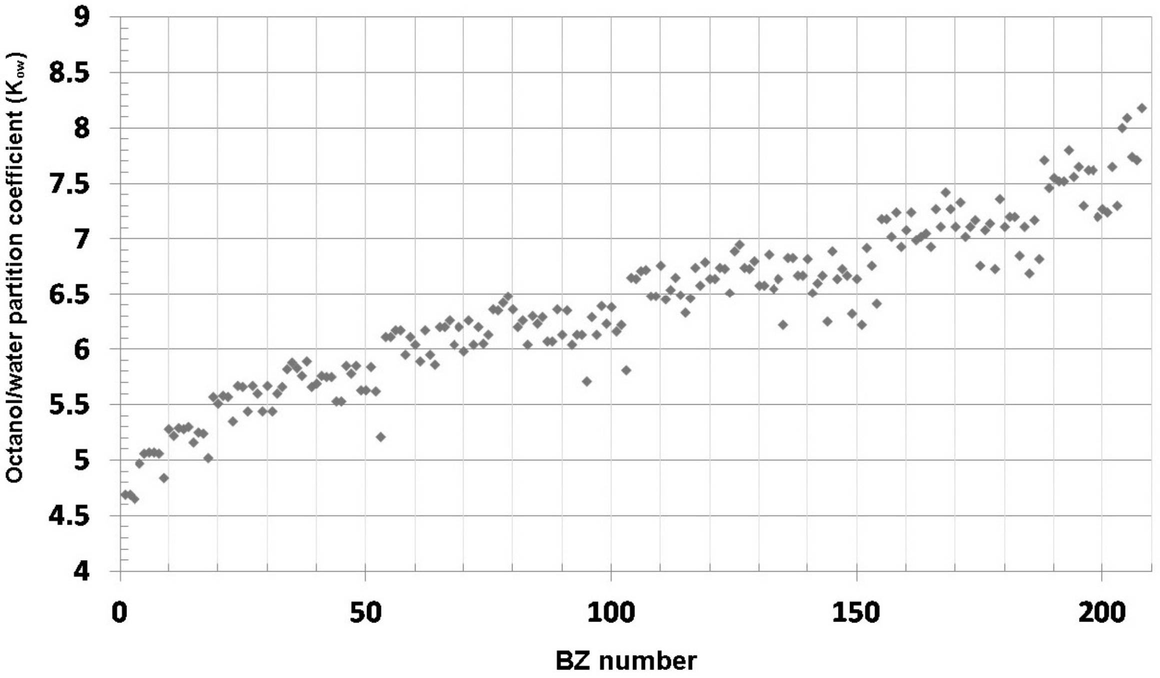
PCBs have never been used as single compounds, but rather as complex mixtures. The commercial products were manufactured to yield a certain degree of chlorination to fulfil technical requirements, generally between 21% and 68% chlorine.

Trade names for commercial products are given in Table 1.7. The most well known are Aroclor, Clophen, Phenochlor, Kanechlor, Pyralene, Fenclor, and Delor. The Aroclors, which were manufactured in the USA, are identified by a four-digit numbering code in which the first two digits indicate the type of mixture and the last two digits indicate the approximate chlorine content by percentage weight. Thus Aroclor 1242 is a chlorinated biphenyl mixture with an average chlorine content of 42%. The exception to this code is Aroclor 1016, which has an average chlorine content of 41% (Hutzinger *et al.*, 1974). Similarly, the Kanechlors are identified by a three-digit value indicating the average chlorine content (300 for 30%). Other products of similar chlorination content have been produced by different companies in Europe, Japan, and China.

Table 1.8 indicates equivalencies between main commercial formulations of PCBs. [The Working Group noted that these should be considered as approximate.] Since different production yield slight differences in the congener mixture, mixtures with comparable chlorine content but from different manufacturers (e.g. Aroclor 1260 and Clophen A60) show varying compositions, although with strong similarities (Johnson *et al.*, 2000).

The homologue composition of the commercial PCB products varies greatly according to chlorination degree achieved (Table 1.9). For example, Aroclor 1242 is a mixture of mono- to heptachlorobiphenyls, while Aroclor 1260 contains penta- to octachlorinated homologues. The concentrations of single congeners within each homologue group also differ between different products and batches (Fig. 1.4). About 130 of the 209 congeners have been identified in commercial formulations at concentrations above 0.05%. Generally, commercial PCB products consist of about 100–140 PCB congeners, with mono- and non-*ortho* substituted PCBs as minor or trace constituents (Frame *et al.*, 1996a, b; Johnson *et al.*, 2000).

Fig. 1.3 Octanol/water partition coefficient (K_{ow}) of PCB congeners according to the degree of chlorination (BZ number)



BZ, Ballschmiter and Zell; PCB, polychlorinated biphenyl
Compiled by the Working Group

An archetypal distribution of PCB congeners was detected in Aroclor 1254, lot 124–191 (corresponding to the historical G4 production process), while lot 6024 showed a profile characteristic of the A4 production process used between 1974 and 1976 (Kodavanti *et al.*, 2001). Indeed, Aroclor 1254 was produced by two different chlorination procedures (two-step versus single-step chlorination) (Frame *et al.*, 1996a, b). The differences in composition of the two lots are given in Table 1.10. Although Aroclor 1254 A4 probably represented less than 1% of the total production of Aroclor 1254, this PCB product was extensively used by standard suppliers and thus by researchers (Frame, 1999).

Chiral PCB congeners are important constituents of both technical and environmental mixtures of PCBs. For example, the total concentration of chiral PCB congeners in the commercial mixtures Aroclor 1242 and Aroclor 1260 is 6% and 30% by weight, respectively (Kania-Korwel *et al.*, 2007). Chiral enantiomers may have different biological and toxicological properties (Püttmann *et al.*, 1989; Rodman *et al.*, 1991). There is evidence that PCB atropisomers differ in their biological activities (Kania-Korwel *et al.*, 2006, 2008). They have been found in non-racemic proportions in many species (Lehmler *et al.*, 2010; Wong & Warner, 2009). While physical and chemical processes in the environment generally affect the two enantiomers of a known compound at the same rate, biological processes may result in the enrichment of one of the enantiomers, because of enantio-selective interactions with biological macromolecules (Buser & Mueller, 1993).

Table 1.7 Trade names for commercial PCB products[a, b]

Asbestol (trans, cap)	Hydol (trans, cap)
Askarel	Montar
Bakola 131 (trans, cap)	Nepolin
Biclor (cap)	No-Flamol (trans, cap)
Chlorextol (trans)	Phenoclor (trans, cap)
Chlorinol	Pydraul
Clophen (trans, cap)	Pyralene (trans, cap)
Clorphen (trans)	Pyranol (trans, cap)
Delor	Pyroclor (trans)
Duconol (cap)	Saf-T-Kuhl (trans, cap)
Dykanol (trans, cap)	Santotherm FR
EEC-18	Santovac 1 and 2
Elemex (trans, cap)	Siclonyl (cap)
Eucarel	Solvol (trans, cap)
Fenchlor (trans, cap)	Sovol
Elemex (trans, cap)	Therminol FR
Hivar (cap)	

[a] Each trade name may correspond to one or several products with varying chlorine content (see Table 1.8).
[b] Products may be used in transformers (trans) or capacitors (cap).
PCB, polychlorinated biphenyl
From IPCS (1993)

1.1.4 Contaminants and impurities of commercial products

Commercial PCB products have been reported to be contaminated with other chlorinated aromatic compounds, such as polychlorinated naphthalenes and PCDFs (IARC, 1978). Vos & Koeman (1970) were able to identify tetrachlorodibenzofurans, pentachlorodibenzofurans, and chlorinated naphthalenes in samples of Phenoclor DP-6 and Clophen A60. Bowes *et al.* (1975) examined samples of Aroclor 1248, 1254 and 1260 produced in 1969, samples of Aroclor 1254 from 1970 and Aroclor 1016 from 1972, and samples of Aroclor 1260, Phenoclor DP-6 and Clophen A60. They found PCDFs in all Aroclor preparations except Aroclor 1016, and in Clophen A60 and Phenoclor DP-6 (Table 1.11). The levels of PCDFs were in the low microgram per gram range (Erickson, 2001), but additional PCDFs may be formed from PCBs on heating. Impurities such as 2,3,7,8-tetrachlorodibenzofuran and 2,3,4,7,8-pentachlorodibenzofuran have been reported in different amounts under various manufacturing conditions in Aroclor 1248, Aroclor 1254, Clophen A-60, Phenoclor DP-6, and Kanechlor 400 (de Voogt & Brinkman, 1989). Rappe & Gara (1977) confirmed by capillary gas chromatography–mass spectrometry (GC–MS) that 2,3,7,8-tetrachlorodibenzofuran was one of the main PCDFs in “Yusho oil,” as reported by Nagayama *et al.* (1976).

The proportion of impurities may vary between batches. For example, Aroclor 1254 with lot numbers 6024 and 124–191, which were produced by the same company by two different production processes, showed a 3.4-fold difference in the total concentration of PCDFs (Table 1.10).

It is important to note that PCDDs are not found in commercial PCB preparations (Erickson, 2001).

Overall, differences in composition as well as the presence of toxicologically relevant impurities may have had a significant impact on the results of toxicological studies with commercial

Table 1.8 Comparison of commercial PCB products based on percentage chlorination

Average number of chlorine atoms/ molecule	Range of chlorination (%)	Aroclor (USA)	Clophen (Germany)	Phenoclor (France)	Pyralene (France)	Kanechlor (Japan)	Fenoclor (Italy)	Delor (former Czechoslovakia)	PCB (China)
1.15	21	1221							
2	32–33	1232			2000	200			
2.5	38				1500				
3	40–42	1242, 1016	A30	CP3	3000	300	42	2; 103	PCB_3
4	48	1248	A40	DP4		400		3; 104	
5	52–54	1254	A50	DP5		500	54	4 and 5; 105	PCB_5
6–6.8	60–62	1260, 1262	A60	DP6		600	64	106	
8.7	68	1268					70		
10	71	1270					DK		

PCB, polychlorinated biphenyl

Adapted from de Voogt & Brinkman (1989), Erickson (1997), and Johnson *et al.* (2000)

Table 1.9 Homologue composition and physical properties of selected commercial PCB products

	Aroclor							Kanechlor		
	1221	**1232**	**1016**	**1242**	**1248**	**1254**	**1260**	**300**	**400**	**500**
Composition (%)										
Biphenyl	11	6	< 0.01	-	-	-	-	-	-	-
Monochlorobiphenyl	51	26	1	1	-	-	-	-	-	-
Dichlorobiphenyl	32	29	20	17	1	-	-	17	3	-
Trichlorobiphenyl	4	24	57	40	23	-	-	60	33	5
Tetrachlorobiphenyl	2	15	21	32	50	16	-	23	44	27
Pentachlorobiphenyl	0.5	0.5	1	10	20	60	12	0.6	16	55
Hexachlorobiphenyl	-	-	< 0.01	0.5	1	23	46	-	-	13
Heptachlorobiphenyl	-	-	-	-	-	1	36	-	-	-
Octachlorobiphenyl	-	-	-	-	-	-	6	-	-	-
Nonachlorobiphenyl	-	-	-	-	-	-	-	-	-	-
Properties										
Relative molecular mass	200.7	232.2	257.9	266.5		328.0	357.7			
Colour	Clear	Clear	Clear	Clear		Light yellow	Light yellow			
Density (g/cm^3 at 25 °C)	1.18	1.26 1.27	1.37	1.38	1.41 1.44	1.50 1.54	1.56 1.62			
Viscosity (cP at 38 °C)	5	8	20	24	70	700	Resin			
Physical state	Oil	Oil	Oil	Oil		Viscous liquid	Viscous liquid			
Boiling point (°C)	275–320	290–325	325–356	325–366		365–390	385–420			
Water solubility (µg/L at 25 °C)	200 15 000[a]	1450[a]	240 420	240	52 54	12	3			
Vapour pressure (10^{-6} kPa at 25 °C)	893	613	53	53	53	11	5.3			
Henry's law K (atm.m^3/mol, 25 °C)	3.5×10^{-3}		2.9×10^{-4}	5.2×10^{-4}		2.0×10^{-3}	4.6×10^{-3}			
Log K_{ow}[b]	2.8	3.2	4.4	4.1	6.1	6.5	6.9			
Flashpoint (°C)	141–150	152–154	170	176–180	193–196	None to boiling	None to boiling			

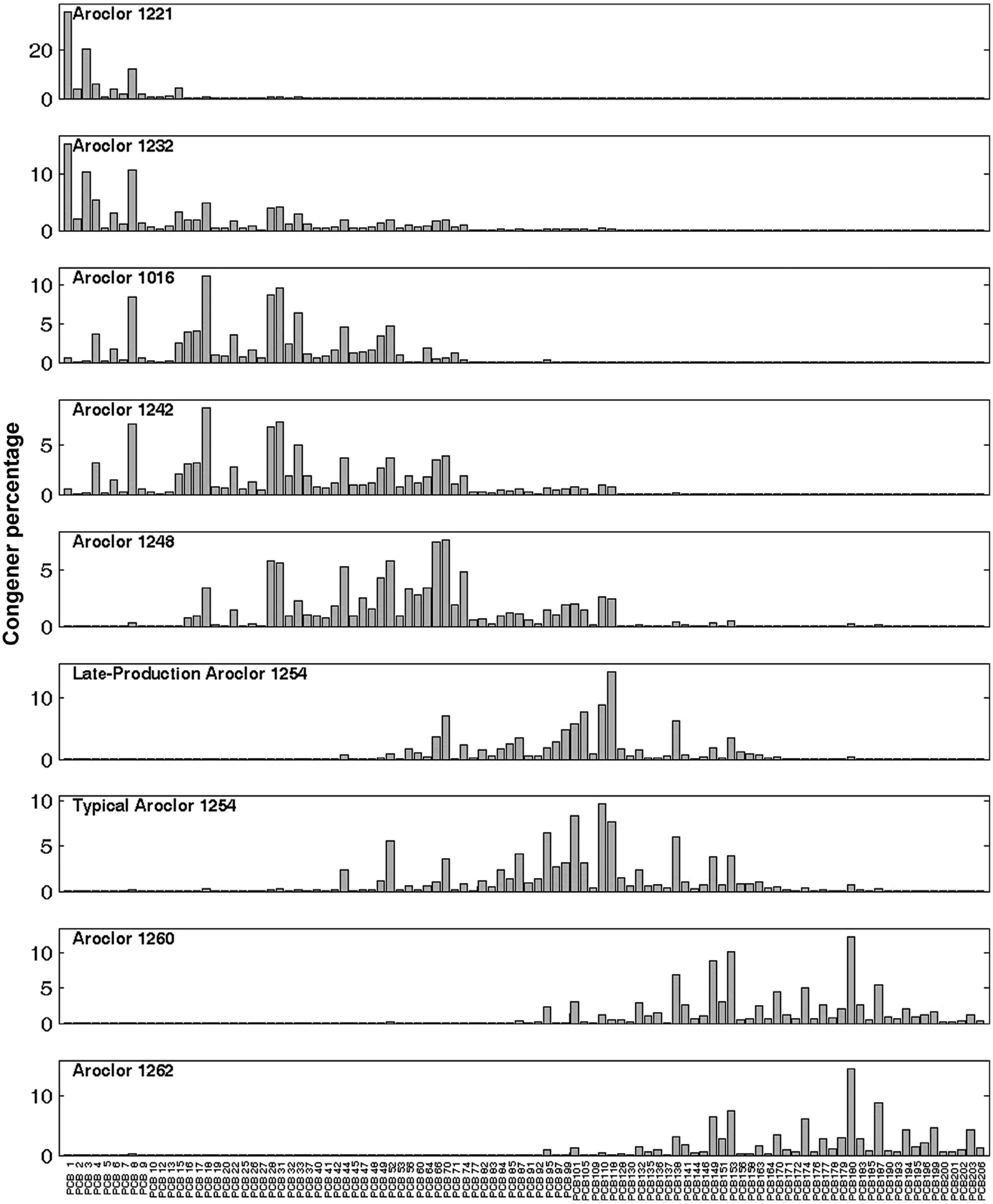
[a] Estimated value

[b] Log K_{ow} represents an average value for the major components of the Arochlor mixture. The Henry's law constants were estimated by dividing the vapour pressure by the water solubility (Cohen & Mercer, 1993; Erickson, 1997).

PCB, polychlorinated biphenyl

From Hutzinger *et al.* (1974), Pellet *et al.* (1993), and Lindell (2012).

Fig. 1.4 Congener-specific composition of Aroclor formulations



Only the 100 most abundant congeners are shown in this figure.
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Table 1.10 Chemical profile and impurities (polychlorodibenzodioxins, polychlorodibenzofurans and polychlorinated naphthalenes) in lots 124–191 and 6024 of Aroclor 1254

PCBs and impurities	Aroclor 1254	
	Lots 124–191 (G4 process)	Lot 6024 (A4 process)
*Non-*ortho *congeners*		
PCB-77	0.01 mg/g	27.2 mg/g
PCB-81	0.01 mg/g	0.28 mg/g
PCB-126	0.17 mg/g	3.24 mg/g
PCB-169	0.01 mg/g	0.02 mg/g
*Mono-*ortho *congeners*		
PCB-105	51.00 mg/g	130.00 mg/g
PCB-114	0.05 mg/g	0.78 mg/g
PCB-118	127.00 mg/g	124.00 mg/g
PCB-123	0.57 mg/g	2.14 mg/g
PCB-156	4.80 mg/g	51.00 mg/g
PCB-157	0.36 mg/g	26.30 mg/g
PCB-167	ND	ND
PCB-189	ND	ND
PCDFs		
2,3,7,8-TetraCDF	129.9 ng/g	350.1 ng/g
1,2,3,7,8-PentaCDF	295 ng/g	1920.2 ng/g
2,3,4,7,8-PentaCDF	821 ng/g	4049.2 ng/g
1,2,3,4,7,8-HexaCDF	1638.1 ng/g	4571.4 ng/g
1,2,3,6,7,8-HexaCDF	733.7 ng/g	3190.5 ng/g
1,2,3,7,8,9-HexaCDF	ND	ND
2,3,4,6,7,8-HexaCDF	213.3 ng/g	1333.3 ng/g
1,2,3,4,6,7,8-HeptaCDF	581.8 ng/g	1506.5 ng/g
1,2,3,4,7,8,9-HeptaCDF	533.3 ng/g	1459.4 ng/g
1,2,3,4,6,7,8,9-OctaCDF	356 ng/g	945.6 ng/g
Σ polychlorinated dibenzofurans (PCDF)	11.3 µg/g	38.7 µg/g
Σ polychlorinated dibenzo-p-dioxins (PCDD)	< 2 ng/g	< 2 ng/g
Σ polychlorinated naphtalenes	155 µg/g	171 µg/g
Σ non-*ortho* congeners-TEQ	17.3 µg WHO-TEQ/g	353 µg WHO-TEQ/g
Σ mono-*ortho* congeners-TEQ	5.51 µg WHO-TEQ/g	10 µg WHO-TEQ/g
Σ PCDF-TEQ	0.54 µg WHO-TEQ/g	2.25 µg WHO-TEQ/g
Total PCDD+PCDF+PCB-TEQ	23.4 µg WHO-TEQ/g	365.3 µg WHO-TEQ/g

CDF, chlorodibenzofuran; ND, not detected; PCB, polychlorinated biphenyl; PCDFs, polychlorodibenzofurans; TEQ, toxic equivalent
Adapted from Kodavanti *et al.* (2001) and EFSA (2005)

Table 1.11 Concentrations of chlorodibenzofurans in Aroclor, Clophen, and Phenoclor

Commercial PCB mixture (date of production)	Polychlorodibenzofurans (concentrations in mg/g)			
	Tetra-CDF	Penta-CDF	Hexa-CDF	Total
Aroclor 1248 (1969)	0.5	1.2	0.3	2.0
Aroclor 1254 (1969)	0.1	0.2	1.4	1.7
Aroclor 1254 (1970)	0.2	0.4	0.9	1.5
Aroclor 1260 (1969)	0.1	0.4	0.5	1.0
Aroclor 1260 (lot AK3)	0.2	0.3	0.3	0.8
Aroclor 1016 (1972)	< 0.001	< 0.001	< 0.001	-
Clophen A60	1.4	5.0	2.2	8.6
Phenoclor DP6	0.7	10.0	2.9	13.6

CDF, chlorodibenzofuran
Adapted from Bowes *et al.* (1975)

PCB products and mixtures (EFSA, 2005). Consistent interpretation of the results of such studies, especially differentiation of the effects caused by respective PCBs, may only be achieved if the congener composition of these mixtures is known. The determination of the content in specific congeners was not feasible in most cases due to the lower sensitivity of analytical techniques available in the past.

1.2 Analysis

1.2.1 General considerations

Past and current methods for the chemical analysis of PCBs have been reviewed recently (Le Bizec *et al.*, 2015). Since the 1960s, PCBs have been determined using GC techniques with electron capture detection (ECD), initially using packed columns. Today the separation has been improved by the use of capillary columns and the selectivity by the use of MS detectors. Increase in sensitivity, expressed as decreasing detection limits, has been achieved as analytical techniques have improved.

Originally, PCB concentrations were determined on the basis of commercial products, e.g. various Aroclor products with different chlorination levels. Later, PCB concentrations were determined based on homologue groups, while today congener-specific analysis is a common practice. These methodological changes, including differences in the basis of quantification, are an obstacle when comparing older with more recent studies.

Even when comparing studies from the same period, it can be difficult to compare PCB concentrations reported by different laboratories, if information on data quality is not available and if the results for different numbers of congeners are summarized. Often “total” PCB concentrations are reported, summing up all the congeners included in the laboratory’s method and assumed to approach the true total PCB concentration. Operational sum parameters have been defined to harmonize congener lists and improve comparability, for example, the six indicator PCB congeners (PCB-28, PCB-52, PCB-101, PCB-138, PCB-153, and PCB-180), expressed as PCB_6. The six congeners were not selected from a toxicological point of view, but were considered as indicators for the different PCB patterns in various sample types and are most suitable for evaluating non-dioxin-like PCBs (NDL-PCBs) (EFSA, 2005). This parameter is used, for example, in the European food and feed regulation (EC, 2011a).

Some agencies, such as the International Council for the Exploration of the Sea (ICES), recommend reporting PCB_7, which includes the mono-*ortho* congener PCB-118 in addition to the PCB_6 (ICES, 2012; Webster *et al.*, 2013). In the Arctic Monitoring and Assessment Programme (AMAP), the sum of 10 PCB congeners is often reported (PCB-28, PCB-31, PCB-52, PCB-101, PCB-105, PCB-118, PCB-138, PCB-153, PCB-156, PCB-180), which includes the six indicator PCBs. However, reports of individual concentrations in scientific studies have the advantage of allowing sum calculations as required. In food analyses, PCB concentrations are preferred to compliance/non-compliance reports (EFSA, 2005).

Depending on the sample type and the purpose of the study, PCB concentrations may be reported in different units. Concentrations in solid samples are generally reported in mass per mass. Normalizations to dry weight or lipid weight are common for abiotic matrices (e.g. soil and sediment) and those with a high lipid content (e.g. fatty food products), respectively. For liquid and air samples, the concentrations are often given in mass per volume. However, as liquid volumes are susceptible to small changes during sample storage and cannot be determined as precisely as masses, concentrations in small liquid volume samples (e.g. blood) are increasingly related to mass instead of volume.

Apart from the adjustment of mass for fresh weight (also referred to as raw weight, wet weight), lipid normalization of PCB concentrations in blood samples is also common. (Schisterman *et al.*, 2005; Phillips *et al.*, 1989; Grimvall *et al.*, 1997). [The Working Group has acknowledged that a variety of lipid determination methods for blood are used and that there is no consensus on how to determine lipid concentrations.]

Given the low concentrations of PCBs in some matrices, reliable quality assurance and quality control are particularly important, including for example monitoring of recovery rates and procedural blanks, duplicate analyses, analyses of in-house reference material and external quality control in proficiency testing schemes.

The transport and storage of samples can be a source of error through PCB loss or contamination. Studying the effects of storage conditions on PCBs in biological material, De Boer & Smedes (1997) generally did not find temperature effects as long as the temperature was < 5 °C, or downward trends in PCB contents. Practical guidance on the storage and transport of marine samples intended for PCB analysis is given by OSPAR (1999, 2002) and Webster *et al.* (2013).

[The Working Group stressed the importance of how the “non-detects” were reported and treated in the data analysis. There are a variety of methods used and there is currently no global consensus.]

1.2.2 Analytical tools

Instrumental analysis is essentially identical for all matrices. Dioxin-like PCBs (DL-PCBs) are often analysed together with dioxins and furans by gas chromatography-high resolution mass spectrometry (GC-HRMS). For this purpose, DL-PCBs are separated from other PCB congeners as part of the clean-up and fractionation process, for example using activated carbon, porous graphite columns, or 2-(1-pyrenyl) ethyldimethylsilyated (PYE) silica (Hess *et al.*, 1995).

Gas chromatography-electron capture detection (GC-ECD) provides low detection limits and high precision, but is less specific than MS, as it separates PCB congeners only by retention time. MS adds a second dimension in terms of different mass spectra. Therefore, ^{13}C-labelled PCB congeners can be separated from the native molecule on a mass basis. In contrast, as retention times are identical to the native analogues, ^{13}C-labelled PCB congeners cannot be used in GC-ECD analyses.

Due to lower selectivity and the risk of interference, GC-ECD is often based on two GC capillary columns of different polarity (dual column GC) (Covaci & Schepens, 2001). Webster *et al.* (2013) recommend that retention times be checked for shifts between analytical runs, usually with the help of characteristic peaks, for example those added as injection standards. Coelution of PCB-138 and PCB-163 occurs on many common capillary columns.

Among the MS techniques, electron capture negative ionization (ECNI) is very sensitive for detection of penta- to decachlorinated PCBs (Webster *et al.*, 2013). However, electron impact (EI) has better selectivity than ECNI and comparable sensitivity when combined with large-volume injection, which requires rigorous sample clean-up (Covaci *et al.*, 2002a). Suitable target and qualifier ions for PCBs are listed by Webster *et al.* (2013).

Some studies have applied gas chromatography-ion trap mass spectrometry (GC-ITMS), for example for the analysis of PCBs in human milk (Gómara *et al.*, 2011). GC-ITMS with its MS/MS option offers increased selectivity while being less expensive than HRMS (Webster *et al.*, 2013). Triple quadrupole mass spectrometry (LRMS/MS) operated in the selected reaction monitoring mode has also been shown to provide selectivity and sensitivity comparable to that of HRMS in food analyses (Ingelido *et al.*, 2012).

Bioassays are an alternative method of determining PCB concentrations and have been suggested as screening tools for monitoring PCDD/Fs and DL-PCBs in foodstuffs by the European Commission Directive 2002/69 (EC, 2002). The dioxin-responsive chemically activated luciferase (CALUX or lux) assay is mechanism-specific and uses the interaction with the aryl hydrocarbon (Ah) receptor. Differences between results of the bioassay and of the conventional targeted high resolution gas chromatography-high-resolution mass spectrometry (HRGC-HRMS) analysis of PCDD/Fs and DL-PCBs have been shown (van Leeuwen *et al.*, 2007), possibly caused by other compounds capable of interactions with the AhR (Vorkamp *et al.*, 2012).

Enzyme-linked immunosorbent assays (ELISA) have been successfully applied to PCB analyses in environmental samples, showing reasonable agreement with conventional GC analyses, but with a high dependence on sample pretreatment (Johnson & Van Emon, 1996; Deng *et al.*, 2002). Recent developments include, for example, immunosensors for applications in situ (Lin *et al.*, 2008) and immunoaffinity chromatography for sample purification (Van Emon & Chuang, 2013).

1.2.3 Analysis of environmental samples

Selected methods for analysis of PCBs in environmental matrices are presented in Table 1.12.

Supplementary material on analysis of PCBs in soil and sediment is available online at: http://monographs.iarc.fr/ENG/Monographs/vol107/suppl_S1.pdf.

(a) Air and dust

Both active and passive sampling are used for PCB analysis in air. Passive sampling has been applied to the analysis of outdoor air using semi-permeable membrane devices (Ockenden *et al.*, 2001) and polyurethane foam (Mari *et al.*, 2008). Vegetation is used as a natural passive sampler, for example tree bark integrating atmospheric PCB concentrations over the life time of the tree (Hermanson and Hites, 1990) or pine needles reflecting up to several years of PCB exposure (Kylin *et al.*, 1994).

Polyurethane foam has also been used for indoor air collection (Hazrati & Harrad, 2006), but active sampling is often the preferred method (EPA, 1999; Kohler *et al.*, 2005). To account for concentration differences and the limited air volume in an indoor setting, outdoor air is usually sampled by high-volume sampling, while low-volume sampling is used for indoor air.

Once retained on a solid matrix (filter, sorbent), PCBs are solvent-extracted using the same techniques as commonly applied for soil,

Table 1.12 Selected methods of analysis of PCBs in environmental matrices

Sample matrix	Sample preparation	Assay method	Detection limit[a]	Reference
Air	Collection on sorbent/filter, solvent extraction, evaporation, acid treatment and/or other clean-up (if necessary), separation of dioxin-like and non-dioxin-like PCBs if required.	GC-ECD; GC-HRMS	0.03–10 pg/m^3; 10 pg/m^3	McConnell *et al.* (1998), Mari *et al.* (2008)
Dust	Sieving of samples (during sampling or afterwards), solvent extraction, evaporation, acid treatment, back extraction, clean-up, evaporation.	GC-MS	NA	Harrad *et al.* (2009)
Water	Liquid-liquid extraction or SPE of unfiltered or filtered water, evaporation, clean-up if necessary. Alternative technique: Passive sampling	GC-ECD; GC-HRMS	0.22–3 ng/L; 0.004–0.5 ng/L	Hope *et al.* (1997), EPA (2008a)
Soil	(Water removal), extraction, evaporation, clean-up, including sulfur removal, separation of dioxin-like and non-dioxin-like PCBs if required.	GC-HRMS	1.5 ng/kg; 0.4–46 ng/kg	Wang *et al.* (2010), EPA (2008a)
Sediment	(Water removal), extraction, possibly in combination with sulfur removal, evaporation, clean-up, including sulfur removal, separation of dioxin-like and non-dioxin-like PCBs if required.	GC-ECD; GC-MS; GC-HRMS	NA; 0.4–46 ng/kg	Webster *et al.* (2013), EPA (2008a)

[a] Detection limits are given for individual PCB congeners

PCB, polychlorinated biphenyl; ECD, electron capture detection; EI, electron impact; GC, gas chromatography; HRMS, high-resolution mass spectrometry; MS, mass spectrometry; NA, not available; SPE, solid-phase extraction sediment or biota. Before extraction, recovery/internal standards are added, e.g. PCB congeners that are not present in the environment, or ^{13}C-labelled PCB congeners. Extraction is often performed by Soxhlet (EPA, 1999; Menichini *et al.*, 2007). Ultrasonic extraction and pressurized liquid extraction (PLE) have also been described (Aydin *et al.*, 2007; Mari *et al.*, 2008). Barro *et al.* (2005) applied headspace-solid phase micro extraction, which does not involve solvents.

Whether or not clean-up steps are required depends on potential interferences from the matrix (e.g. particles) and co-extracted compounds as well as on expected concentrations. Adsorption chromatography can be applied, for example using alumina (Zhang *et al.*, 2011a) or silica. As all PCB congeners are acid stable, acid treatment is possible.

Dust for PCB analysis has been collected in several ways, for example from the residents' vacuum cleaner bags (Franzblau *et al.*, 2009; Knobeloch *et al.*, 2012), by vacuuming (Wilson *et al.*, 2001; Harrad *et al.*, 2009) and from air conditioning units (Tan *et al.*, 2007). Dust samples originating from vacuum bags might be sieved, but cut-off sizes differ, e.g. 150 μm (Wilson *et al.*, 2001) and 1 mm (Knobeloch *et al.*, 2012) have been described.

Extraction techniques are basically the same as described for sorbents, including Soxhlet extraction (Dirtu & Covaci, 2010), PLE (Harrad *et al.*, 2009) and ultrasonic extraction (Wilson *et al.*, 2001). Before extraction, internal/recovery standards should be added, as described for air samples. Due to interferences from the matrix and co-extraction of other compounds, clean-up of dust samples will be required. PLE can be combined with simultaneous clean-up by adding adsorption materials to the cells; however, additional clean-up steps may be necessary (Harrad *et al.*, 2009). Various sorbents have been used for clean-up of dust samples, including Florisil (Wilson *et al.*, 2001; Harrad *et al.*, 2009), silica gel (Dirtu & Covaci, 2010), and combinations of both (Knobeloch *et al.*, 2012). As described for air samples, acid treatment has also been applied (Harrad *et al.*, 2009).

(b) Water

The PCB content in a water sample is strongly influenced by the amount of suspended particulate matter (SPM) that adsorbs PCBs. Depending on the objectives of the analysis, different approaches can be chosen, resulting in different fractions to be analysed:

- Unfiltered water includes dissolved components and those bound to colloids and SPM.
- Filtered water gives PCB concentration on SPM (residue on the filter) and dissolved or bound to colloids (filtrate).
- Passive sampling targets the dissolved fraction.

Passive sampling devices integrate PCB concentrations over time, which reduces temporal variability. Common formats for water sampling include semipermeable membrane devices, low density polyethylene, and silicone rubber (Lohmann *et al.*, 2012). Passive sampling techniques have been applied for analysis of PCBs in river water (Grabic *et al.*, 2010) and seawater (Granmo *et al.*, 2000; Fernandez *et al.*, 2012).

In water bodies with a low SPM content, e.g. seawater, PCB concentrations will likely be low, and large amounts of water will have to be sampled and processed, while avoiding contamination. Guidelines for seawater sampling and the subsequent analysis of organic contaminants have been established by OSPAR (OSPAR, 2013). Studies have shown that the critical part of such analysis occurs outside the laboratory, i.e. during sampling, transport, and storage (Wolska *et al.*, 2005). As described for air and dust, recovery/internal standards should be added before extraction.

PCBs from water samples are typically extracted by either liquid–liquid extraction (LLE), i.e. the direct extraction of PCBs with a non-polar solvent (Hope *et al.*, 1997), or solid-phase extraction (SPE), where PCBs are retained on a solid phase and subsequently eluted with a non-polar solvent (Russo *et al.*, 1999). The United States Environmental Protection Agency (EPA) method 1668B for determination of PCBs in several matrices describes SPE, continuous LLE, and separatory funnel extraction as suitable extraction methods for aqueous samples (EPA, 2008a).

The amount of SPM in the sample is a critical factor, as LLE might be insufficient and SPE cartridges might become blocked by samples with a high SPM content (Erger *et al.*, 2012). Alternatively, SPM might be removed by filtration and analysed separately, for example by Soxhlet or ultrasonic extraction. This could be the method of choice for water samples with a high SPM content, for example wastewater samples or landfall leachate (Zorita & Mathiasson, 2005).

To what extent purification is necessary depends on the nature of the sample, its SPM content, PCB concentration and that of interfering compounds. Although sampling only freely dissolved PCBs, some passive sampling approaches add a clean-up step after extraction, for example using acid silica or aluminium (Grabic *et al.*, 2010). Surface-water samples, however, have often been analysed without clean-up (Hope *et al.*, 1997; Erger *et al.*, 2012), while other studies have included adsorption chromatographic steps (Khim *et al.*, 2001). Gel permeation chromatography (GPC) may be used for water extracts that contain organic compounds of high relative molecular mass (EPA, 2008a).

PCB exposure from snow can be considered insignificant, with the exception of polar regions where snow may be a source of drinking-water. Analytical methods are similar to those for water (Carrera *et al.*, 1998).

1.2.4 Analysis of biological samples

Several matrices have been analysed to determine internal exposure to PCBs, or body burden, including adipose tissue, meconium, placenta, blood, umbilical cord blood, human milk, and hair (Table 1.13).

(a) Tissues (adipose tissue and placenta)

The analytical methods applied to the analysis of PCBs in tissues such as adipose and placenta are similar to those used for environmental samples. The characteristically high lipid content of adipose and other tissues, however, requires rigorous lipid removal before instrumental analysis.

Different ways of sample pretreatment have been applied after or as part of the homogenization procedure, for example sample drying with Na_2SO_4 or hydromatrix (Covaci *et al.*, 2002a; Saito *et al.*, 2004), melting of fat (De Saeger *et al.*, 2005), mixing with base (Kim & Fisher, 2008) and addition of ethanol for protein precipitation (Whitcomb *et al.*, 2005).

Extraction is generally carried out with a non-polar solvent such as toluene or hexane, in some cases in a mixture with acetone, dichloromethane or propanol (Guvenius *et al.*, 2002; Saito *et al.*, 2004; Fernandez *et al.*, 2008). The extraction could often proceed by shaking or rotating, for example in an Ultra Turrex or Vortex (Guvenius *et al.*, 2002). Other extraction techniques are the same as those applied in environmental analyses, including ultrasonic extraction (Suzuki *et al.*, 2005), Soxhlet (Fernandez *et al.*, 2008), PLE (Saito *et al.*, 2004), and MAE (Li *et al.*, 2006). Supercritical fluid extraction with carbon dioxide (sometimes modified with dichloromethane) has also been applied (Stellman *et al.*, 1998). For the extraction of placenta, Gómara *et al.* (2012) additionally described the preparation of a suspension that was liquid–liquid extracted. As for other matrices, recovery/internal standards are generally added before extraction.

Table 1.13 Selected methods for analysis of PCBs in biological matrices

Sample matrix	Sample preparation	Assay method	Detection limit[a]	Reference
Adipose tissue	Pre-treatment (drying and/or protein denaturation), extraction, evaporation, lipid removal, further clean-up, separation of dioxin-like and non-dioxin-like PCBs if required.	GC-ECD; GC-HRMS	0.009–1.1 ng/g lipid; 0.002–0.2 ng/g lipid	Whitcomb *et al.* (2005), Fernandez *et al.* (2008)
Placenta	Pretreatment (drying and/or protein denaturation), extraction, evaporation, lipid removal, further clean-up, separation of dioxin-like and non-dioxin-like PCBs if required.	GC-ECD; GC-MS (ECNI)	NA	Gómara *et al.* (2012), Ma *et al.* (2012)
Blood	Protein denaturation, extraction, evaporation, lipid removal, further clean-up, separation of dioxin-like and non-dioxin-like PCBs if required.	GC-ECD; GC-MS; GC-HRMS	10–100 pg/mL; 2–5 pg/mL	Covaci & Schepens (2001), Lu *et al.* (2012)
Urine (hydroxylated PCBs)	Acidification, extraction, evaporation, derivatization.	GC-MS (EI)	0.02–0.04 ng/mL	Hong *et al.* (2005a, b)
Human milk	Drying or protein denaturation + fat globules dispersion, extraction, evaporation, lipid removal, further clean-up, separation of dioxin-like and non-dioxin-like PCBs if required.	GC-ECD; GC-MS; GC-HRMS	NA; 0.01–0.03 ng/mL; NA	Duarte-Davidson *et al.* (1991), Covaci *et al.* (2001), Fürst (2006)
Hair	Washing, incubation with HCl, extraction, evaporation, lipid removal, further clean-up.	GC-ECD; GC-MS (EI)	0.3–2 ng/g	Covaci *et al.* (2002b)

[a] Detection limits are given for individual PCB congeners

ECD, electron capture detection; ECNI, electron capture negative ionization; EI, electron impact; GC, gas chromatography; HCl, hydrochloric acid; HRMS, high-resolution mass spectrometry; MS, mass spectrometry; NA, not available

A common method for lipid removal is treatment of the sample with acid, usually sulfuric acid (Whitcomb *et al.*, 2005). GPC is another suitable method (Ma *et al.*, 2012), but may not achieve complete removal of lipids. The use of partially deactivated neutral aluminium for lipid removal has also been described (Stellman *et al.*, 1998).

For further clean-up of the extracts, the same techniques are applied as in the environmental analyses, either individually or in combinations. These include silica gel (Suzuki *et al.*, 2005; Fernandez *et al.*, 2008), alumina (Covaci *et al.*, 2002a), Florisil (Whitcomb *et al.*, 2005) and GPC (Saito *et al.*, 2004). Impregnating the silica gel with acid is a common way of combining adsorption chromatography with lipid removal (Covaci *et al.*, 2002a; Fernandez *et al.*, 2008).

Some studies have analysed PCB metabolites in adipose tissue and placenta, e.g. hydroxylated PCBs and methylsulfonyl-PCBs. These methods usually included a fractionation by adsorption chromatography and elution with different solvents (Guvenius *et al.*, 2002; Saito *et al.*, 2004). In the method by Gómara *et al.* (2012), hydroxylated PCBs were separated from the parent compounds during liquid-liquid extraction (LLE). After derivatization, the fraction containing hydroxylated PCBs was cleaned up in the same way as described for the parent compounds.

(b) Blood (including umbilical cord blood)

Numerous studies have analysed PCBs in blood, mostly in serum, but also in plasma (Schettgen *et al.*, 2011). The analytical methods used generally do not differ for serum and plasma. Given the low lipid content of blood, PCB concentrations are generally low and the sample amount available for analysis may be a challenge. Most studies work with volumes of 0.5–2 mL. Methods have recently been developed to extract PCBs from only 50 µL of plasma and from dried blood spots (Lu *et al.*, 2012).

Umbilical cord blood has often been analysed in combination with maternal blood, using the same methods. Given the lower lipid content and usually lower PCB concentrations in cord blood, adjustments of the sample intake might be useful; however, sample availability is usually the limiting factor.

Apart from the addition of internal standards, the first step in PCB analysis of serum, plasma or cord blood is generally the denaturation of protein, e.g. by addition of formic acid (Kang *et al.*, 2008), methanol (Korrick *et al.*, 2000), or acetonitrile (Agudo *et al.*, 2009). Different extraction techniques have been described, among which the simple mixing of the sample with solvent (Apostoli *et al.*, 2005; Schettgen *et al.*, 2011). LLE has also been used (Kawashiro *et al.*, 2008; Bachelet *et al.*, 2011) as well as SPE on C_{18} or hydrophilic–lipophilic balanced reversed phase sorbent (Covaci & Schepens, 2001; Lee *et al.*, 2011). Guvenius *et al.* (2003) used Lipidex 5000, a lipophilic gel, for extraction of PCBs from cord blood.

Since they are present at low concentration, lipids are not always removed from the extract (Lu *et al.*, 2012). Lipids can be removed by direct addition of acid to the extracts (Atuma & Aune, 1999) or by clean-up methods on acidified silica (Covaci & Schepens, 2001). Further clean-up sorbents include Florisil (Whitcomb *et al.*, 2005), alumina (Stellman *et al.*, 1998), neutral silica gel (Atuma & Aune, 1999), or combinations of these (Guvenius *et al.*, 2003; Apostoli *et al.*, 2005).

To account for the low concentrations of PCBs in blood, extracts are often reduced to very small volumes, e.g. 50 µL (Covaci & Schepens, 2001). This is achieved by addition of non-volatile keepers (Covaci & Schepens, 2001), or by evaporation to dryness (Apostoli *et al.*, 2005) and reconstitution in the desired solvent or a solution of syringe standards in this solvent. Evaporation to dryness carries the risk of loss of volatile PCB congeners.

Hydroxylated PCB metabolites have been analysed in blood and umbilical cord blood. Guvenius *et al.* (2003) used the same extraction method as for parent PCBs, but obtained hydroxylated PCBs in an isolated fraction, based on different elution solvents. Park *et al.* (2009) treated the sample with hydrochloric acid and 2-propanol, and extracted hydroxylated PCBs by LLE. Hydroxylated PCBs require derivatization to non-polar molecules before separation by GC (Sandau *et al.*, 2000).

(c) Urine

A few studies have assessed PCB metabolites (hydroxylated PCBs) in urine samples. Hydroxylated PCBs are more polar than their parent compounds and act as weak acids, which has to be taken into account in extraction, clean-up, and separation by GC.

Hong *et al.* (2005a, b) presented two methods for the extraction of hydroxylated PCBs from urine. The first method combined SPE testing of four different phases, with five derivatization methods. Best recoveries and GC separations were found for hydroxylated PCBs extracted on a C_2 phase and derivatized with iodopropane under basic conditions (Hong *et al.*, 2005a). The second method used headspace solid-phase microextraction and on-fibre derivatization, achieved by placing the needle in the headspace of a solution of bis(trimethylsilyl)trifluoroacetamide (BSTFA) (Hong *et al.*, 2005b). The derivatized hydroxylated PCBs were transferred to the GC injector by thermic desorption. Several fibre materials were tested, of which polydimethylsiloxane-divinylbenzene (PDMS-DVB) gave the highest signal in the analysis.

(d) Human milk

Breast milk is the most extensively analysed matrix for the estimation of PCB body burden in humans. The first studies date back to the 1970s (Musial *et al.*, 1974), and programmes for the biomonitoring of human milk have been established in several countries or regions (Wilhelm *et al.*, 2007; Krauthacker *et al.*, 2009; Cerná *et al.*, 2012). Analytical methods are very diverse: the milk samples may be treated as liquid, or the lipid phase may be isolated, or the sample may be freeze-dried and treated as solid.

When the whole milk sample is treated as a liquid, the first steps usually include protein denaturation and dispersion of fat globules by addition of sodium oxalate, or acetic acid and methanol, sometimes in combination with ultrasound treatment (Dmitrovic & Chan, 2002; Fürst, 2006). Before or after this step, internal standards are usually added, and the sample is extracted by LLE (Chovancová *et al.*, 2011), or SPE (Covaci *et al.*, 2001; Dmitrovic & Chan, 2002). Hexane is a commonly used solvent, although a large variety of solvent combinations and solvent sequences have been described in the literature.

In some studies, the lipid phase of the milk sample is separated or extracted and a defined amount of fat used for further analysis (Fürst, 2006; Pérez *et al.*, 2012).

In the third approach, milk samples are freeze-dried and a defined amount is extracted with techniques commonly applied to solid samples, e.g. Soxhlet extraction (Duarte-Davidson *et al.*, 1991) and PLE (She *et al.*, 2007). Matrix solid-phase dispersion has also been described (Gómara *et al.*, 2011). However, freeze-drying always runs the risk of loss of volatile PCBs and cross-contamination.

As described for other human matrices, lipids in the extract are removed before instrumental analysis. Furthermore, the extracts usually contain co-extracted compounds that are likely to interfere with PCBs in the instrumental analysis. The clean-up techniques therefore generally include lipid destruction by acid treatment, either directly in the extract (Duarte-Davidson *et al.*, 1991), or by acidified silica gel (Covaci *et al.*, 2001). Alternatively, GPC has been used, but usually in combination with acid treatment (She *et al.*, 2007). Further clean-up techniques include adsorption chromatography on neutral or basic silica (She *et al.*, 2007), alumina (Chovancová *et al.*, 2011), and Florisil (Pérez *et al.*, 2012), also in combinations (Fürst, 2006). Ingelido *et al.* (2007) described clean-up by supercritical fluid extraction.

(e) Human hair

With a lipid content of about 2% (Altshul *et al.*, 2004), hair accumulates lipophilic compounds such as PCBs and has the advantage of being sampled non-invasively. However, to what extent hair PCB content reflects internal exposure to PCBs is difficult to determine, even if the hair is washed before analysis to avoid co-extraction of dust particles. Comparisons of serum and hair samples showed weak correlations for most PCB congeners and considerably higher PCB concentrations in hair, also on a lipid-normalized basis (Altshul *et al.*, 2004). Effects of hair colour (natural or dyed) cannot be ruled out (Covaci *et al.*, 2002b).

Sample amounts of less than 1 g are sufficient for detection of PCBs. The hair samples are washed, and cut or pulverized, and then spiked with internal or recovery standards and incubated with hydrochloric acid (Covaci *et al.*, 2002b). Extraction techniques applied in hair analyses include LLE (Covaci *et al.*, 2002b), Soxhlet (Zhang *et al.*, 2007), and ultrasonic extraction (Barbounis *et al.*, 2012). The same methods for lipid removal and extraction clean-up as for other biological matrices have been used, e.g. adsorption chromatography on acidified silica gel, alumina (Covaci *et al.*, 2002b), and Florisil (Zhang *et al.*, 2007). A comparison between three laboratories analysing the same hair sample but using different internal standards, extraction techniques and analytical instruments (GC-ECD, GC-LRMS and GC-HRMS) showed good agreement, with a relative standard deviation of 15% (Gill *et al.*, 2004).

1.2.5 Analysis of food samples

Food items are regularly analysed for PCBs in various national and international food-monitoring programmes (Fromberg *et al.*, 2011; EFSA, 2005), and market-basket or duplicate-diet studies have been performed to identify PCB intake from food (Voorspoels *et al.*, 2008; Fromme *et al.*, 2009).

These studies have often applied methods that are sufficiently versatile to allow analysis of different kinds of food item with varying lipid content and consistency. The first step is often a drying of the food material with sodium sulfate (Voorspoels *et al.*, 2008; Schecter *et al.*, 2010), followed by the addition of recovery or internal standards, and Soxhlet extraction using hexane:acetone (Voorspoels *et al.*, 2008), or toluene (Kiviranta *et al.*, 2004). The clean-up usually includes lipid removal by acid treatment, either as direct addition to the extracts (Fromme *et al.*, 2009), or via acid-impregnated silica gel (Voorspoels *et al.*, 2008). Further clean-up steps can include neutral and basic silica gel (Son *et al.*, 2012), alumina (Kiviranta *et al.*, 2004), and Florisil (Schecter *et al.*, 2010); however, the extent of purification and fractionation is highly dependent on the target analytes.

Food monitoring sometimes focuses on DL-PCBs, which are analysed together with dioxins and furans. These are separated from other PCB congeners by fractionation on a carbon column, which separates the molecules by planarity (Fernandes *et al.*, 2004). Given the low concentrations of DL-PCBs, the fractions are sometimes further purified before instrumental analysis (Fromme *et al.*, 2009).

Some studies have used more specific methods for different food items, for example, protein denaturation and dispersion of fat globules in dairy products, by the addition of sodium oxalate, or potassium oxalate and ethanol (Fromberg *et al.*, 2011; Sirot *et al.*, 2012), followed by LLE. In other studies using cows' milk, the samples are freeze-dried before extraction (Lake *et al.*, 2013), or the fat is separated using a detergent (Pérez *et al.*, 2012). The clean-up steps may be the same as for other lipid-containing matrices. For the analysis of butter and vegetable oil, Roszko *et al.* (2012) described a dialysis method based on low-density polyethylene semi-permeable membranes, followed by GPC and common column clean-up.

Numerous studies have dealt with analysis of PCBs in fish, as summarized by Domingo & Bocio (2007). Analyses of meat and fish basically follow the same methods (Su *et al.*, 2012). Samples are often dried as the first step, e.g. by freeze-drying (Abalos *et al.*, 2010; Liu *et al.*, 2011) or addition of anhydrous sodium sulfate (de Boer *et al.*, 2010). After addition of internal standards, the samples are extracted on a Soxhlet apparatus (Su *et al.*, 2012), by PLE (Pérez-Fuentetaja *et al.*, 2010), or ultrasonic extraction (Son *et al.*, 2012). The clean-up techniques are the same as described for other food matrices, including acid treatment (Su *et al.*, 2012), acid and neutral silica gel, and alumina (Liu *et al.*, 2011), and Florisil (Villa *et al.*, 2011), sometimes in an automated PowerPrep system (Abalos *et al.*, 2010). A rapid extraction and purification method was presented by Kalachova *et al.* (2011), combining PCB partitioning into ethyl acetate and lipid removal on a silica gel microcolumn.

Eggs are commonly analysed for PCBs, with a focus on the egg yolk (Kiviranta *et al.*, 2004; Voorspoels *et al.*, 2008). While the same methods could be applied as for other food samples, recent publications have only equilibrated the sample with solvents (Fromberg *et al.*, 2011; Rawn *et al.*, 2012). The clean-up steps include lipid removal by direct acid treatment and adsorption chromatography on acid silica and Florisil (Rawn *et al.*, 2012).

Fruit and vegetables are analysed less frequently than lipid-rich food items. In the methods described by Grassi *et al.* (2010) and Sirot *et al.* (2012), freeze-drying, extraction using Soxhlet or PLE, and acid treatment were applied, in a manner very similar to that used for analyses of other food items.

1.3 Production and uses

1.3.1 Production processes

PCBs have commonly been synthesized commercially by catalytic chlorination of biphenyl. The catalysts used include iron, iodine, and chlorides of aluminium, tin, and antimony. The synthesis is performed as a one-step chlorination process, or in two steps with further chlorination of residues from the first step. The crude products are purified by alkali wash to remove hydrogen chloride and ferric chloride, blown with air, and sometimes also by distillation (IARC, 1978). The degree of chlorination is controlled by the time (range, 12–36 hours) in the reactor.

The manufacturing process for Aroclors involved the chlorination of biphenyl with anhydrous chlorine in the presence of a catalyst, such as iron filings or ferric chloride. In 1974–1977, "late production" Aroclor 1254 was made by a two-stage chlorination procedure. In the first stage, biphenyl was chlorinated to 42% chlorine content by weight as for Aroclor 1242. This was then fractionated to give a distillate (Aroclor 1016). The residue (mostly mono-*ortho* tetrachlorobiphenyls and higher homologues) was further chlorinated to 54% chlorine by weight, resulting in a lot (Monsanto lot KI-02–6024) with markedly higher levels of the high non-*ortho* and mono-*ortho* PCB congeners than the Aroclor 1254 lots produced earlier. The differences between the early and late lots of Aroclor 1254 are discussed in more detail above (see Section 1.1.3 and Table 1.10).

1.3.2 Production volumes and trends

Although the commercial production of PCBs began in the 1920s, it was not until after 1945 that production reached substantial volumes. Production peaked in the 1960s and 1970s, and had ceased in most countries by the end of the 1970s or early 1980s.

Estimates of the total cumulative worldwide production of PCBs indicate that 1 to 1.5 million tonnes (or more) of commercial PCB products were manufactured. Production volumes from former Czechoslovakia, France, Germany, Italy, Japan, China, Poland, the Russian Federation and the former Soviet Union, Spain, the United Kingdom, and the USA, as reported by Tatsukawa (1976), de Voogt & Brinkman (1989), Jiang *et al.* (1997), AMAP (2000), Holoubek *et al.* (2001a), and Sułkowski *et al.* (2003), add up to around 1 325 000 tonnes for 1930–1993 (Table 1.14).

In the USA, annual production peaked in 1970 with a total volume of 39 000 tonnes. From 1957 to 1971, 12 different types of Aroclor with chlorine contents ranging from 21% to 68% were produced in the USA by Monsanto Chemicals Co. (see Section 1.1). In addition, Geneva Industries produced a smaller amount of PCBs from 1972 to 1974 (EPA, 2008b).

In China, the production of PCBs began in 1965 and was gradually stopped between 1974 and the 1980s. According to preliminary investigation and analysis, 7000–10 000 tonnes of PCBs were produced in China from 1965 to 1974, with 9000 tonnes as PCB_3 [similar to Aroclor 1242] and 1000 tonnes as PCB_5 [similar to Aroclor 1254] (Xing *et al.*, 2005; NIP China, 2007).

Information from the Democratic People's Republic of Korea (NIP Korea DPR, 2008) indicated that production of PCBs has been ongoing at two sites since the 1960s. The initial production capacity for PCBs was 1200 tonnes per year, with a tendency to increase until the 1980s; however, capacity has decreased since the early 1990s, and the average annual production volume in 2001–2006 was 411.6 tonnes. The total amount produced up to 2006 could be estimated at around 30 000 tonnes. According to this report, the Democratic People's Republic of Korea planned to reconsider its production of PCBs in 2012.

The commercial products were marketed under more than one hundred different trade names, depending on place of manufacture, production process, and chlorine content. Aroclors comprised at least 10 different commercial PCB products, under the names Aroclor 1016, 1221, 1232, 1242, 1248, 1254, 1260, 1262, 1268, and 1270. It should be noted that Aroclor 5460 was not a PCB product, but consisted of polychlorinated terphenyls. Other commercial PCB products include Clophen (four products), Delor (three products), Fenclor (five products), Kanechlor (five products), Phenochlor (four products), Pyralene (three products), Sovol, and Therminol (see Section 1.1.3).

1.3.3 Uses

Due to the physical and chemical properties of PCBs, such as non-flammability, chemical stability, high boiling point, and high dielectric constants, PCBs were widely used in several industrial and commercial open and closed applications (Table 1.15). PCBs have also been used in corresponding military applications, but detailed information on military use is typically very scarce.

As a result of the production process, PCBs were never used as individual congeners, but as technical products composed of multiple congeners. The commercial PCB products were generally used as such, but mixtures with other compounds were also produced to obtain specific properties. For example, the PCB product Sovol may have been mixed with α-nitronaphtalene to increase volatility, and sold as Nitrosovol (UNEP, 1988). Similarly, Galbestos was a mixture of PCBs and asbestos used on galvanized steel and

Table 1.14 Volume and duration of PCB production in countries with known production (by production volume)

Producer	Country	Duration		Volume (tonnes)	Reference
		Start	Stop		
Monsanto	USA	1930	1977	641 246	de Voogt & Brinkman (1989)
Bayer AG	Germany, western	1930	1983	159 062	de Voogt & Brinkman (1989)
Orgsteklo	Russian Federation	1939	1990	141 800	AMAP (2000)
Prodelec	France	1930	1984	134 654	de Voogt & Brinkman (1989)
Monsanto	United Kingdom	1954	1977	66 542	de Voogt & Brinkman (1989)
Kanegafuchi	Japan	1954	1972	56 326	Tatsukawa (1976)
Orgsintez	Russian Federation	1972	1993	32 000	AMAP (2000)
Caffaro	Italy	1958	1983	31 092	de Voogt & Brinkman (1989)
2.8 Vinalon and the Sunchon Vinalon Complex	Democratic Republic of Korea	1960[a]	2012[b]	30 000[c]	NIP Korea DPR (2008)
SA Cros	Spain	1955	1984	29 012	de Voogt & Brinkman (1989)
Chemko	Former Czechoslovakia	1959	1984	21 482	Schlosserová (1994)
Xi’an	China	1965	1980	10 000	Jiang *et al.* (1997), NIP China (2007)
Mitsubishi	Japan	1969	1972	2 461	Tatsukawa (1976)
Electrochemical Co.	Poland	1966	1970	1 000	Sułkowski *et al.* (2003)
Zaklady Azotowe Tarnow-Moscice	Poland	1974	1977	679	Sułkowski *et al.* (2003)
Geneva Industries	USA	1972	1974	454	EPA (2008b)
Total		*1930*	*2012*	*1 355 810*	

[a] During the 1960s

[b] “The Ministry of Chemical Industry will, by 2012, take measures to dismantle the PCBs production process and establish a new process of producing an alternative.”

[c] Estimated from Republic of Korea 2008, National Implementation Plan for the Stockholm Convention on Persistent Organic Pollutants.

PCB, polychlorinated biphenyl

Adapted from Breivik *et al.* (2007)

galvanized corrugated sliding panels in various industrial and military applications.

(a) Closed applications

The predominant applications for PCBs were in dielectric fluids in capacitors and transformers. These applications are considered to be closed applications, since PCBs are not expected to leak out of the system. However, transformers had occasionally to be topped up with PCBs so that these systems were not completely closed.

While applications in hydraulic and heat transfer, and cooling systems are also usually considered to be closed applications, there have been reports of accidental leaks from such systems, and thus these applications are often referred to as “normally closed.”

During the 1960s, dielectric fluid in capacitors and transformers represented 50–60% of the sales of PCBs in the USA (IARC, 1978). In 1972, Monsanto restricted its sale of PCBs to capacitor and transformer applications (Erickson, 2001); after this date, these applications represented some 99% of the total use of PCBs in the USA (Durfee *et al.*, 1976). In China, PCB_3 [similar to Aroclor 1242] was used primarily in power capacitors applied in electricity production, distribution and transmission, while PCB_5 [similar to Aroclor 1254] was used mainly as a paint additive (see Table 1.8).

Table 1.15 Industrial uses of PCBs

System/category	Aroclor									DecaCB
	1221	1232	1016	1242	1248	1254	1260	1262	1268	
Dielectric fluids										
Capacitors	✓		✓+	✓+		✓				
Transformers			✓	✓		✓+	✓+			
Hydraulic/lubricants/heat-transfer fluids										
Heat transfer				✓	✓	✓				
Hydraulic fluids		✓		✓	✓	✓	✓			
Vacuum pumps					✓	✓	✓			
Gas transmission turbines	✓			✓						
Immersion oil for microscopes							✓			✓
PCBs incorporated into products and materials										
Rubber	✓	✓		✓+	✓	✓			✓	
Synthetic resins					✓	✓	✓	✓	✓	
Carbonless copy paper				✓+						
Pipeline valve grease									✓	
Adhesives	✓	✓		✓+	✓	✓				
Wax extenders				✓+		✓			✓	
Caulk and joint sealants						✓[a]				
Insulation and other building materials						✓			✓	
De-dusting agents						✓	✓			
Inks						✓				
Cutting oils						✓				
Wire and cable coatings						✓	✓			
Die or investment casting										✓
Pesticide extenders						✓				

[a] Also others
✓ Denotes use of given Aroclor in a specific end-use
✓+ Denotes principal use
PCB, polychlorinated biphenyl
Adapted from Johnson *et al.* (2000) and Erickson & Kaley (2011)

As production and use of PCBs became banned, outdated PCB-containing equipment (equipment filled with PCBs as dielectric fluid) was generally removed from use and stored for disposal (Xing *et al.* 2005). In this equipment, about 6000 tonnes of PCBs came from capacitors (NIP China, 2007).

(b) Open applications

PCBs were also used in several open applications as a major constituent of permanent elastic sealants and as flame-retardant coatings (Heinzow *et al.*, 2007).

The use as plasticizer in sealants (caulking material) and flooring material was common in many countries, representing up to 15–20% of the total use of PCBs in Sweden (Jansson *et al.* 1997). The sealants were mainly used in outdoor applications, but indoor use was not uncommon. Use in flooring material was limited to indoor use.

Sealants that were mixed with PCBs were mainly of the polysulfide type. The mixing was often performed on site. Information on concentrations to be used were not available to the Working Group; however, from a technical point of view, PCB concentrations were likely to be above 5%. Sealants analysed some 40 years after application often contained concentrations of PCBs of 5–15%, with concentrations of up to 35% being reported. The concentration may vary not only between sites, but also within a building. These variations may be the result of use of sealants with different PCB content, or of secondary processes, such as migration out of the matrix. There are reports indicating that inner parts of sealants could contain higher concentrations than the superficial parts (Johansson *et al.*, 2003).

In addition to the use as sealants and flame-retardant coatings, PCBs have also been used in other open applications, such as in inks, adhesives, microencapsulation of dyes for carbonless duplicating paper, conveyor belts, rubber products, paints, pesticide fillers, plasticizers, polyolefin catalyst carriers, immersion oil for microscopes, cutting and lubricating oils, surface coatings, wire insulators, and metal coatings (ATSDR, 2000; Erickson, 2001; Erickson & Kaley, 2011). Also, use in small ballasts for fluorescent lights could be regarded as an open application, especially after long-lasting usage.

(c) Disposal of equipment containing PCBs

Improper handling of electronic waste (e-waste) has been identified as a source of environmental contamination with PCBs, especially for old equipment (Leung *et al.*, 2006). Dismantling of ships has also been identified as a potentially important source of occupational exposure to and environmental contamination with PCBs (Basel Convention, 2003).

With the complete ban on the use of PCBs, stockpiles awaiting elimination have successively appeared in many countries.

In 2000, 23 companies worldwide had facilities for the disposal of equipment containing PCBs, of which 11 were in Europe. The use of solvent for decontamination represents the most common procedure of disposal, followed by destruction by incineration, dechlorination with sodium, retrofilling and vitrification. The most common technology used for destruction of PCBs is by incineration, with an efficiency of between 99% and 99.99999% (IOMC, 1998). For exemple, France has an installed capacity for incineration of PCB residues amounting to around 20 000 tonnes per year (INERIS, 2013).

1.4 Environmental occurrence and exposure

PCBs are found worldwide at measurable levels in all environmental media (soils and sediments, water, air), in wildlife, and also probably in the body of every human. Human exposure to PCBs occurs mostly via ingestion of contaminated food (see Section 1.4.7), but also via inhalation and dermal absorption.

Soils are natural sinks for persistent and lipophilic compounds such as PCBs; PCBs are absorbed by the organic carbon of the soil, and once absorbed they are relatively persistent (Buckley-Golder, 1999) (see Section 1.4.5). PCBs enter the soil via different pathways: industrial releases from manufacture, use and disposal, accidental releases, atmospheric deposition, application of sewage sludge, and erosion and leachate from nearby contaminated areas. PCBs in organic liquids may be dissolved by soils and then migrate with the solvent.

The congener patterns of PCBs in soils and sediments change over time as a result of the activity of aerobic bacteria (that degrade less chlorinated congeners) and anaerobic bacteria (that can cause partial dechlorination of more highly chlorinated congeners) (Hardell *et al.*, 2010). The patterns found in environmental biota are often referred to as "weathered," since they result from alterations in the composition of a mixture (e.g. resulting from bio accumulative and metabolic processes in higher biological organisms and through bacterial action, exposure to ultraviolet radiation, etc.). "Weathering" processes result in PCB patterns with either a higher chlorinated fraction or congeners with higher bioaccumulative properties compared with the commercial products. "Weathering" must be considered when assessing PCB-associated risks based on studies with experimental animals exposed to commercial PCB products.

Water is a major pathway for migration of PCBs, both in solution and particulate-bound, although PCBs are lipophilic and generally not very soluble in water (see Section 1.4.6). Less chlorinated PCB congeners have greater solubility than more highly chlorinated congeners.

Air is another major pathway for PCB migration (see Sections 1.4.3 and 1.4.4). PCBs are semi-volatile compounds and, as with water solubility, less chlorinated congeners are more volatile than more highly chlorinated ones (Totten *et al.*, 2006). There is extensive evidence that PCBs in aquatic systems exchange with PCBs in air (Bamford *et al.*, 2002). Air transport of PCBs can occur in either the vapour phase or particulate-bound, thus contributing to global pollution and PCB contamination of remote regions of the earth. PCBs in air come from several direct or indirect sources, including industrial facilities, military sites, contaminated bodies of water, landfills and hazardous waste sites, electric arc furnaces, incineration and other forms of combustion, sewage sludge applied to agricultural lands, and construction materials, including in paints (Hu & Hornbuckle, 2010), caulking, light ballasts, floor sealants, and adhesives and plasticizers in older buildings (Wallace *et al.*, 1996).

PCBs from soil, sediment, air and water enter the food-chain by uptake and bioaccumulation in plants and animal fats. There is significant biological magnification of PCB concentration as PCBs move up the food-chain. PCB concentrations vary depending on the degree of bioaccumulation, and are usually highest in carnivorous fish coming from contaminated waters. PCBs are found in the fat of all meat animals, in all dairy products containing fat, and in eggs (ATSDR, 2000; IOM, 2003), sometimes at high concentrations due to local contamination of grasses, and feeding practices in some countries (see Section 1.4.2). Also, it is not uncommon to feed domestic animals with fish meal or oil, or waste animal fats, which results in recycling of PCBs (IOM, 2003). For example, farmed salmon fed with concentrated fish meal or fish oil containing significant amounts of PCBs showed elevated concentrations of PCBs (Hites *et al.*, 2004). PCBs found in food are typically of higher chlorination, since they are less volatile and more biologically persistent in plants and animals than the lower congeners.

Another important route of exposure to PCBs is inhalation; however, it is difficult to determine the relative contribution of inhalation compared with ingestion. Harrad *et al.* (2006) have suggested that inhalation may account for 4–63% (median, 15%) of overall exposure in humans. PCBs may be attached to indoor dust, which can be either ingested or inhaled. Individuals who spend significant periods of time in the presence of either outdoor or indoor vapour-phase PCBs will have continuous exposure that is not reflected in measurements of "total" PCBs, because the less chlorinated congeners are more rapidly metabolized and excreted by the human body (Fig. 1.5; Johansson *et al.*, 2003). Concentrations of different PCB congeners were measured in blood from individuals living in houses where PCB-containing sealant was used. Concentrations of most congeners were only slightly elevated (1.2 to 3.2 times), but the two congeners with a low level of chlorination (PCB-28 and PCB-66) were detected at much higher concentrations (30 and 9 times, respectively) in contaminated flats than in control flats.

Dermal absorption of PCBs may occur primarily in the occupational setting, but also through contact with contaminated sediments or other applications to the skin (Wester *et al.*, 1987, 1993). Less chlorinated congeners are more rapidly absorbed through the skin than more highly chlorinated congeners (Garner & Matthews, 1998).

Congener patterns in the general human population are always different from any pattern found in commercial PCB products (Patterson *et al.*, 2009). The factors that may explain this are:

- The general public is exposed to multiple sources of PCBs, only rarely to a single commercial product.
- There may be more than one route of exposure for almost all matrices/animals/humans.
- Dechlorination occurs to varying degrees in sediments, soils, water and air. Commercial PCB products will volatize to some degree, and in doing so, will lose less chlorinated congeners.
- PCBs ingested by fish and animals will be metabolized (to less chlorinated and hydroxylated congeners) to different degrees. Thus most food stuffs will demonstrate a shift in the congener profile compared to the commercial product.
- When inhalation is the major route of exposure, there is selective exposure to the more volatile, less chlorinated and less persistent congeners.
- Genetic differences among individuals may confer differences in metabolic activity and selective metabolism of different congeners.

1.4.1 Diffuse sources of PCBs worldwide

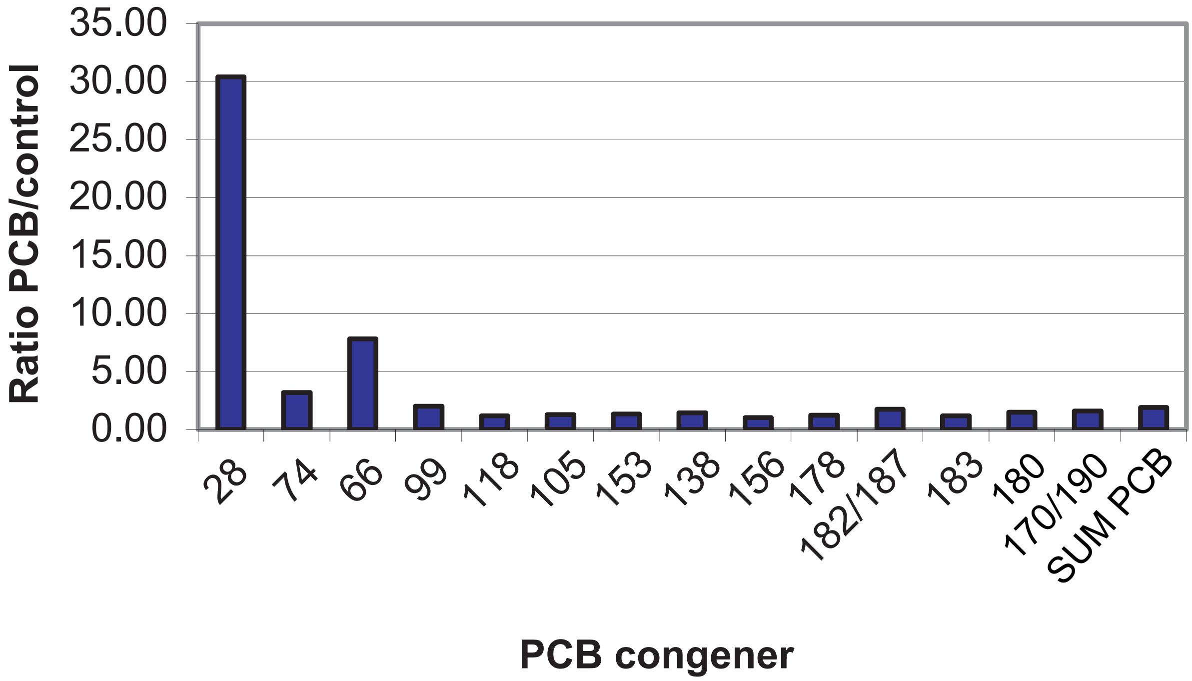
(a) North America

The two Monsanto facilities that manufactured PCBs in the USA were located in Anniston, Alabama, and Sauget, Illinois. In Anniston, more than 400 000 tonnes of PCBs were produced, at least 4550 tonnes were discarded in two landfills, and at least 20.5 tonnes were released into the atmosphere (Hermanson & Johnson, 2007). Many of the large industries using PCBs manufactured by Monsanto were located near major bodies of water, and PCBs were released into the environment as a result of unintentional leaks, volatilization during the production process, and migration from associated landfills and waste products. There was also production, at lower quantities, by Geneva Industries in Houston, Texas (de Voogt & Brinkman, 1989). As a result, contamination has occurred in many rivers and streams near these sites of production (see Section 1.4.6(a)).

(b) Europe

In western Europe, many chemical plants are located along major rivers (i.e. Rhine, Rhone, and Seine) and there have been several isolated incidents of organic chemical pollution. The Seine estuary remains one of the most polluted

Fig. 1.5 Blood PCB concentrations in individuals living in PCB-contaminated flats relative to individuals living in control flats



PCB, polychlorinated biphenyl
From Johansson et al. (2003)

in Europe (RNO, 2012). Also, the Venice lagoon in Italy is particularly polluted owing to the proximity of an important industrial district (the Marghera Harbour) (see Section 1.4.6(b)).

In the Slovak Republic, the Chemko Chemical Co. (based in the Michalovce district) produced 21 000 tonnes of commercial PCB mixtures between 1959 and 1984 (Delor 103, 104, 105, 106, Delotherm DK and DH, Hydelor 137). Improper disposal from the Chemko plant via release of effluent directly into the Laborec river resulted in long-term environmental contamination.

During the conflict of the former state union of Serbia and Montenegro throughout the 1990s, the burning or damaging of industrial and military targets resulted in the release of large amounts of PCBs into the environment: more than 1000 electro-transformer stations that contained PCB oil were damaged. After the bombardment of Kragujevac, Serbia, 2500 kg of PCB-based oil from the transformers of the Zastava automobile industry were spilled.

A French inventory reported that the number of installed transformers containing at least 100 kg of PCBs was 100 000 units in 1987, corresponding to 50 000 tonnes of fluids containing 60% PCBs (Pyralene), and to 50 000 tonnes of carcasses with 5% of PCB residues. The 250 000 medium-voltage capacitors represented about 3000–5000 tonnes of pure PCBs, while the low-voltage capacitors represented 1500–2000 tonnes of hardly extractable PCBs.

In Spain, an inventory in 1997 reported some 6000 tonnes of PCBs, although the amount of material containing or contaminated with PCBs could reach 200 000 tonnes.

(c) Asia

Contamination of soil and sediments has been reported in the Russian Federation, China, Viet Nam, and Japan. Such contamination may originate from PCB producing plants (e.g. China, Japan, Democratic People's Republic of Korea), or from e-waste recycling facilities (e.g. China). In addition, two major accidents of food contamination occurred in Taiwan, China and Japan (see Section 1.4.2(a)).

(d) South and Central America

There has been no manufacture of PCBs in South and Central America, but there has been widespread use of PCB-containing transformers and other PCB-containing devices.

(e) Africa

There has been no manufacture of PCBs in Africa, but there has been widespread use of PCB-containing transformers and other PCB-containing devices. In Africa, several studies showed an increase in the number of sources of PCBs, due to leakage and wrongly disposed transformers, shipwrecks, and biomass burning.

Another major source of exposure is the importing of e-waste and increase of e-waste recycling facilities, usually illegal, but common in Ghana, Senegal, Nigeria, Kenya, and the United Republic of Tanzania. A report by the United Nations Environment Programme (UNEP) documented issues concerning e-waste in South Africa, Kenya, Uganda, Morocco, and Senegal (UNEP, 2009).

In spite of the lack of homogenous data, an attempt has been made to compare the main PCB stocks that reside in the various countries of the region. [These data should only be seen on the relative scale since lacking the accuracy to make them valuable in the absolute sense.]

In Algeria, the national inventory of electrical equipment and PCB wastes identified 6770 appliances and around 4000 tonnes of oil to remove. The deposit of transformers, capacitors and various equipment containing PCBs was estimated at 1700 tonnes in Tunisia and 1150 tonnes in Morocco (Business Med, 2010).

1.4.2 Accidental releases into the food-chain

(a) Asia

Cooking oil contaminated by Kanechlor has been the source of two accidental mass poisonings in western Japan (later called "Yusho," oil disease in Japanese) and in Taiwan, China (later called "Yucheng," oil disease in Chinese). Commercial PCB mixtures were used as heat-transfer media in oil tanks; leakage of the pipes caused exposure to the PCB mixture and PCB pyrolytic products, mainly PCDFs and polychlorinated quaterphenyls (PCQs) (Masuda *et al.*, 1986). Patients from both countries have been exposed to comparable quantities of PCBs and PCDFs. The PCB/PCDF concentrations in the Yusho oil were higher (several hundred ppm to 3000 ppm) than those in the Yucheng oil (53 to 100 ppm) (Guo *et al.*, 2003); however, on average, Yucheng patients consumed the contaminated oil for a longer duration than the Yusho patients.

(i) Yusho incident, Japan

In 1968, the Yusho incident involved approximately 1800 people who ingested rice oil contaminated by Kanechlor 400 and its pyrolytic products, mainly in Fukuoka and Nagasaki prefectures (Masuda, 1994a, b; Kuratsune, 1996; Matsueda *et al.*, 1993; Todaka *et al.*, 2007a; Nagayama *et al.*, 1977; Tanabe *et al.*, 1989; Masuda *et al.*, 1998; Ohta *et al.*, 2008a). Affected people developed a "strange skin disease," including acne-form eruption, follicular accentuation, and pigmentation, as well as eye discharge and swelling of eyelids. The mean concentrations of seven PCB congeners (PCB-105, PCB-118, PCB-138, PCB-153, PCB-157, PCB-170, and PCB-180) detected in blood were 6.7 ppb and 3.84 ppb (95% confidence interval, 3.54–4.17), 5 and 20 years after being exposed, respectively (Masuda & Yoshimura, 1982). Mortality data among registered Yusho patients were identified by follow-up studies to 1990 and 2007. The first of these two reports (Ikeda & Yoshimura, 1996) reported serum PCB concentrations in the range of 0 to 35 ppb in 1972, and a decrease to about 5 ppb in 1984 (Iida *et al.*, 1999) (see Sections 1.4.9(b)(iii) and (c)(iv) for additional data on PCB concentrations in blood and adipose tissue, respectively).

(ii) Yucheng incident, Taiwan, China

In 1978–9, the Yucheng incident involved approximately 2000 people who ingested rice oil contaminated with Kanechlor 500 and its pyrolytic products (Hsu *et al.*, 1985). After a few months, these people developed chloracne, hyperpigmentation, severe fatigue, peripheral neuropathy, and other signs and symptoms similar to Yusho disease. On the basis of a dietary questionnaire, it was estimated that Yucheng patients had consumed on average about 1 g (range, 0.7–1.4) of PCBs and 3.8 mg (range, 1.8–5.6) of PCDFs (Lan *et al.*, 1981). Another study estimated the intakes of PCBs, PCDFs, and PCQs by Yucheng patients at 673, 3.8, and 490 mg, respectively (Masuda *et al.*, 1986). DL-PCBs contributed to approximately 30% and 20% of the total TEQ (toxic equivalent) in Yucheng men and women, respectively. Compared with the general population in Taiwan, China, the mean total serum PCB concentrations in the Yucheng victims were still nine times higher 15 years after exposure (see Sections 1.4.9(b)(iii) and (c)(iv) for additional data on PCB concentrations in blood and adipose tissue, respectively).

(b) Europe

In Europe, the “Belgian dioxin crisis” was caused by the accidental release of 50 kg of a commercial PCB mixture contaminated with 1 g of dioxins commonly found in transformers, to a stock of recycled fat used for the production of 500 tonnes of animal feed. In May 1999, it appeared that more than 2500 poultry and pig farms could have been contaminated. Chickens showed the classical signs of oedema disease.

In Ireland in 2008, a tank for storage of pork fat was contaminated with heat-transfer fluid containing PCBs (Hovander *et al.*, 2006). [The Working Group noted that the label of “dioxin crisis” attributed to these episodes of PCB feed contamination was inappropriate.]

1.4.3 Outdoor air

PCBs in outdoor air may be a significant source of exposure. Concentrations of PCBs in air depend on a variety of factors, including temperature and proximity to local sources. Temperature is particularly important in controlling the cycle of volatilization and precipitation. Proximity to local sources, such as industrial facilities, landfills, or contaminated bodies of water, results in elevated air concentrations of both vapour phase and particulate-bound PCBs that dissipate with distance at different rates, resulting in both local and distant contamination. Combustion and other high-temperature processes generate PCBs, in particular during combustion of highly chlorinated compounds; however, this route of unintentional formation is considered to contribute little to total airborne PCBs. Migration to the outdoor environment has also been shown to occur as a result of erosion of exposed sealants.

(a) North America

PCB concentrations in outdoor air vary greatly between urban and rural sites in North America, and may be very high near industrial facilities and other contaminated sites (Table 1.16). These differences reflect primarily the impact of local sources and dilution in air, but also the deposition of PCBs at lower temperatures.

The major sources in Chicago are from landfills, sewage sludge drying beds, and transformer storage yards (Hsu *et al.*, 2003). Shen *et al.*

Table 1.16 PCB concentrations in outdoor air in North America

Reference	Location, sources	PCBs measured	PCB concentration in pg/m³ as mean and/or range	Comments
Vorhees *et al.* (1997)	Near a PCB-contaminated site, New Bedford Harbor, Massachusetts	"PCB concentrations"	400–61 000	
	Comparison neighbourhood		100–8200	
Hung *et al.* (2001)	Canadian Arctic	Sum of 102 congeners	28 in 1993; 23 in 1997	PCB-28, PCB-52, and PCB-118 showed little or no decline over time
Hermanson *et al.* (2003)	Near the former Monsanto PCB-manufacturing facility in Anniston, Alabama	Sum of 120 congeners	8700–82 000 [annual average, 27 000]	
Totten *et al.* (2004)	Urban sites (Camden and Jersey City, New Jersey)	Sum of 116 congeners	Average, 3250 and 1260, respectively	
	Remote and suburban areas at various sites near the New York City metropolitan region		Averages of 150–220	
Sun *et al.* (2006)	Six sites near near USA–Canadian Great Lakes (Lake Michigan near Chicago)	Sum of 84 congeners	± 100–1400	
Hermanson & Johnson (2007)	Near the former Monsanto PCB-manufacturing facility in Anniston, Alabama	PCBs in tree bark	171 927 ng/g (ppb) lipid near the site, to 35 ng/g (ppb) lipid at a distance of 7 km	Tree bark serves as passive vapour-phase air sampler
Sun *et al.* (2007)	Six sites distant from urban areas near USA–Canadian Great Lakes (Lakes Superior and Huron)	Sum of 84 congeners	60–86	
	Six sites near near USA–Canadian Great Lakes (Lake Erie)		± 1.1–230	
Palmer *et al.* (2008)	Near the contaminated Hudson River, downstream communities	Sum of 84 congeners	Median, 711	Concentrations were higher closer to the river than further away, and higher in warmer than cooler months of the year. The congener pattern in air was primarily PCBs with three or four chlorines
	City upstream of the industrial sites that caused the contamination		Median, 431	
Palmer *et al.* (2008)	Contaminated portion of the Hudson River	Sum of 84 congeners	102–4011 (median, 711)	
	Community upstream of the contamination		80–2366	
Harrad *et al.* (2009)	Toronto, Canada	Sum of 8 congeners	100–1400 (mean, 350)	
Li *et al.* (2010)	North America	Sum of PCBs	79 (49–120)	
	Remote sites in Alaska and rural sites in the lower 48 states of the USA		1–50	
	Large urban areas like Chicago		1000 and 150 000	
Persoon *et al.* (2010)	Cleveland, Ohio	Sum of 151 congeners	1730–4240	
	Chicago, Illinois		1130–2690	

PCB, polychlorinated biphenyl

(2006) found large relative differences in air PCB concentrations between urban, rural and remote sites, with the highest concentrations in Toronto, Canada, and the Eastern third of the USA [absolute concentrations could not be quantified] using results from passive air samplers in 31 stations in Canada and the USA.

(b) Europe

In Europe, the reported PCB concentrations in outdoor air range from ~10 up to ~1000 pg/m³ in western European countries and from ~50 up to ~9000 pg/m³ in eastern European contries.

Measurement in the Baltic region showed PCB concentrations in southern Norway to be rather high and similar to those in urban areas (Backe *et al.*, 2000; Agrell *et al.*, 2001). Results from the Czech national monitoring system and European Monitoring and Evaluation Programme (EMEP) background monitoring stations also showed relatively high PCB concentrations in this country (EC, 2004). Typical values for background sites usually range up to ~100 pg/m³ and up to several 100s pg/m³ for contaminated areas (Kocan, 2000, 2001).

PCB concentrations in outdoor air may also be measured in precipitation as total deposition rates (ng/m² per day). In southern Sweden (Backe *et al.*, 2002), PCB concentrations ranged from 1.18 to 81.4 ng/L, with no seasonal trends. In Paris, France, average PCB concentrations (sum of seven congeners) in rain during 1986–2001 remained approximately constant at about 40 ng/L (Chevreuil *et al.*, 2001).

Declining concentrations of PCBs have been observed since the early 1960s and 1970s, decreasing by 67% in France (EC, 2004) and by 78% in the United Kingdom (CITEPA, 2013) over 20 years. The difference observed between the steady concentrations in rain and the decrease in general atmospheric emissions may be partly explained by water solubility limits and differences between point sources and global emissions.

Air concentrations of the seven indicator PCBs 28, 52, 101, 118, 153, 138 and 180 were measured at four locations in the Czech Republic, Finland, Sweden, and the Netherlands, from 1996 to 2001. Measured values did not vary noticeably during this period at any location (Fig. 1.6). This suggests that a steady-state has been reached between degradation and environmental cycling, with an ongoing low-level input from existing equipment and contaminated material (Holoubek *et al.*, 2003).

(c) Asia

Limited information on the concentrations of PCBs in air and dust has been reported in Asian countries (Table 1.17). One of the most extensive studies reported results for outdoor air samples from 55 sites in Japan, 20 in China, 30 in the Republic of Korea, and 1 in Taiwan, China. The range of concentrations was 100–1000 pg/m³.

(d) South and Central America

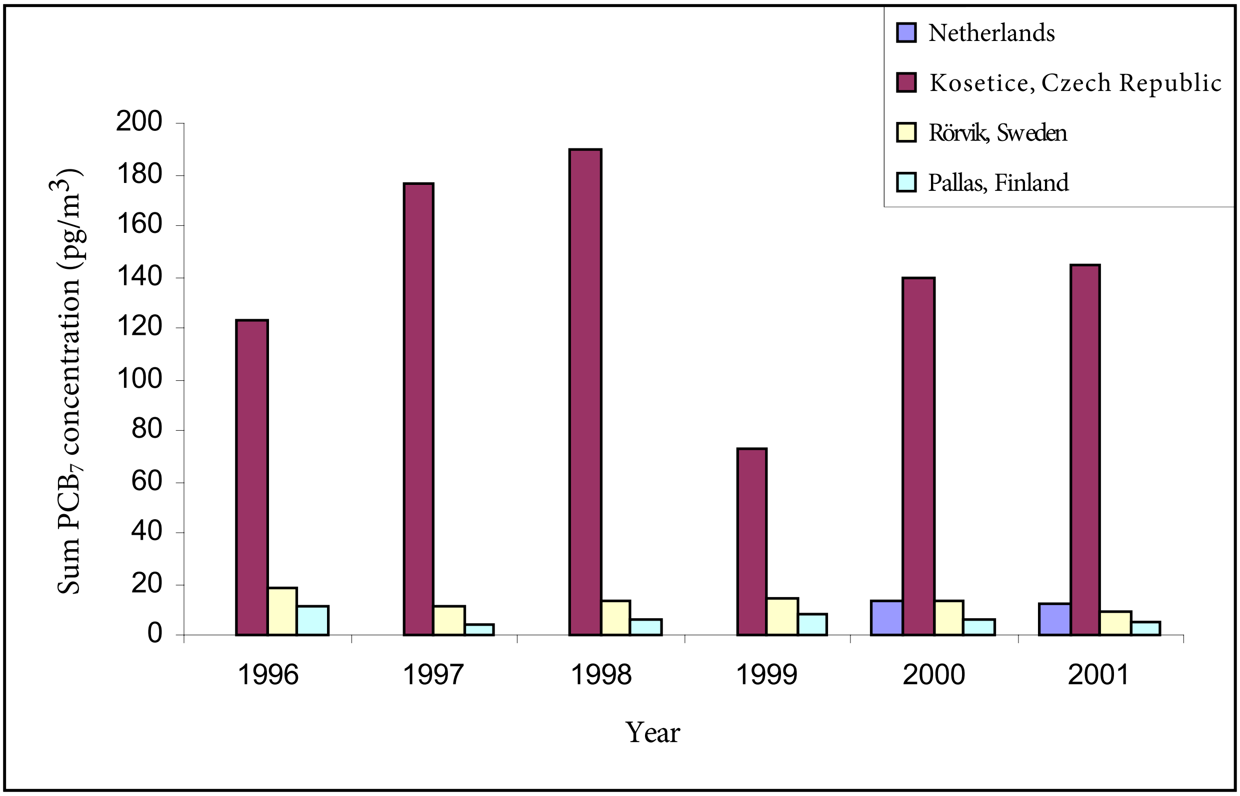
Shen *et al.* (2006) found large relative differences in air PCB concentrations between urban, rural and remote sites using passive air samplers in 4 stations in Mexico, Belize, and Costa Rica. One site in Mexico had higher concentrations than sites in Central America and in Canada.

Li *et al.* (2010) reviewed information from various research groups around the world and reported the average concentration of the sum of PCBs in air to be 66 pg/m³ (range, 9–670 pg/m³) for South America, and 59 pg/m³ (range, 17–150 pg/m³) for Central America.

(e) Africa

Only recently have data from passive air samplers deployed on the African continent become available. PCB concentrations have been reported as very high in Senegal (500 pg/m³) (Klánová *et al.*, 2009), Côte d’Ivoire, and the Gambia (up to 300 pg/m³) (Gioia *et al.*, 2011). Concentrations in some areas in South Africa, Kenya, Egypt, the Democratic Republic of the

Fig. 1.6 Annual average atmospheric concentrations of seven indicator PCBs (PCB_7) from four European Monitoring and Evaluation Programme stations in Europe, 1996–2001



PCB_7, sum concentration of PCB-28, PCB-52, PCB-101, PCB-118, PCB-153, PCB-138, and PCB-180
From Holoubek *et al.* (2003)

Congo, Ghana, Mali, and the Sudan were also high, and comparable to those in urban areas in more developed countries. These levels could not be explained by biomass burning or primary emissions, and were probably due to e-waste dumps. Lower concentrations have been measured in the Congo, Ethiopia, Mauritius, Nigeria, the Togolese Republic, Tunisia, and Zambia.

(f) Vegetation used for monitoring studies

Plant foliage is a reliable proxy for monitoring levels of vapour-phase compounds in outdoor air since it bioaccumulates organic pollutants. Several researchers have used vegetation, grass, conifer needles, mosses, pollen, and leafy vegetable species (cabbage and lettuce) as biomonitors to evaluate patterns of PCB contamination (Larsen *et al.*, 1985; Reischl *et al.*, 1989; Kylin, 1994; Simonich & Hites, 1995). This method has been employed in high-mountain ecosystems (Daly & Wania, 2005), and in several countries, including the Czech Republic (Holoubek *et al.*, 1994), Poland (Migaszewski, 1999), western Finland (Sinkkonen *et al.*, 1995), Germany (Reischl *et al.*, 1987), Italy (Gaggi *et al.*, 1985), and France (Granier & Chevreuil, 1992).

1.4.4 Indoor air

PCBs have been shown to migrate into surrounding materials, such as concrete or wood, and to indoor air. The major sources are PCB-containing caulk, paint (where PCB-11 is the main marker), floor sealants, and ballasts

Table 1.17 PCB concentrations in outdoor air and dust in Asian countries

Reference	Country, region Date of study	Sources	PCBs measured Comments	Concentrations
Iwata *et al.* (1995)	Russian Federation, Lake Baikal May 1992	Six outdoor air samples from research vessel	Kanechlors 300, 400, 500, 600 as standards	Range, 8.7–23 pg/L
McConnell *et al.* (1996)	Russian Federation, Lake Baikal June 1991	A total of 19 outdoor air samples	Aroclors 1242 and 1254 as standards	Mean, 196 ± 65 pg/m^3
Hogarh *et al.* (2012)	Taiwan, China; China; Japan; Republic of Korea March–May, 2008	Outdoor air samples from 55 sites in Japan (37 rural, 4 suburban and 14 urban), 20 in China (3 rural and 17 urban), 30 in the Republic of Korea (12 rural, 2 suburban and 16 urban), and 1 in Taiwan, China	Sum of 202 congeners	Japan, 40–760 pg/m^3 China, 300–2500 pg/m^3 Taiwan, China, about 317 pg/m^3 Republic of Korea, 36–600 pg/m^3
Thacker *et al.* (2013)	India, central and western regions 2009–2010	Outdoor air samples from various cities	Sum of dioxin-like PCBs	Range, 0.0001 × 10^{-1} to 0.0295 ng TEQ/Nm^3

PCB, polychlorinated biphenyl; TEQ, toxic equivalent in lighting devices. Outgassing from contaminated dust may also contribute. Joint sealants are increasingly recognized as important diffuse sources of indoor air contamination by PCBs.

(a) North America

PCBs have been measured in indoor air in several studies (Vorhees *et al.*, 1997; Vorhees *et al.*, 1999; Herrick *et al.*, 2004; Colt *et al.*, 2005; Franzblau *et al.*, 2009; Harrad *et al.*, 2009). In the USA it was reported that indoor air concentrations of PCBs were 5–300 times greater than those in outdoor air (Wallace *et al.*, 1996), and that concentrations were higher in older buildings. The concentrations of PCBs in indoor air in North America are summarized in Table 1.18.

(b) Europe

The highest indoor concentrations (up to 7500 ng/m^3) have been reported in buildings constructed between 1960 and 1975 from prefabricated concrete elements sealed with elastic materials containing PCBs (Balfanz *et al.*, 1993). Joint sealants containing PCB were discovered in various public buildings in Europe (Kohler *et al.*, 2005; Wilkins *et al.*, 2002). Estimated indoor PCB concentrations in contaminated sections were the lowest in microenvironments such as cars (8.92 ng/m^3), and were inversely related to the degree of chlorination of the PCB mixtures used (Hammar 1992; Harrad *et al.*, 2006; Kuusisto *et al.*, 2006, 2007; Frederiksen *et al.*, 2012). The concentrations of PCBs in indoor air in Europe are summarized in Table 1.19.

(c) Asia

Indoor floor dust samples ($n = 43$) collected from rural homes and mosques in Gujarat, Pakistan, showed median total PCB concentrations of 0.67 ng/g (range, 0.3–6.1 ng/g) (Ali *et al.*, 2012). The PCB profile was dominated by PCB-153 (> 60% of the sum of PCBs), with concentrations between < 0.2 and 2.4 ng/g. These PCB concentrations were 10 times lower than those reported in house dust in Singapore (Tan *et al.*, 2007).

1.4.5 Soil and sediments

PCBs can enter soil and sediments through various routes. Sediments constitute an important sink for PCBs entering the marine environment. Sewage sludges are monitored for PCBs in countries where they are largely used (60%) in agriculture. The dumping of incinerator-related materials and/or the inadequate management of commercial PCBs have resulted in significantly elevated PCB concentrations.

1.4.6 Water

Inputs of PCBs to the hydrological cycle are principally via discharges of sewage and industrial effluents, urban run-off, leachates from solid waste landfill sites, atmospheric deposition and, of increasing concern, via agricultural run-off (Scrimshaw *et al.*, 1996).

Water can contain PCBs either in solution or bound to particulates. While PCBs are not very water-soluble, water can be a significant source of exposure to less chlorinated congeners that have a greater solubility than more highly chlorinated congeners. PCB concentrations in sea and freshwater are summarized in Table 1.20.

(a) North America

(i) Drinking-water

In the USA, the EPA has set a goal for PCBs in drinking-water of zero, and a maximum contaminant concentration of 500 ng/L (500 ppt), with sources being primarily landfills, and discharge of waste chemicals (EPA, 2014). While conventional treatment of drinking-water will remove particulate-bound PCBs, those that are soluble are often not completely removed. Solubilities of individual PCB congeners vary from about 4 ppm for monochlorobiphenyl to as low as 0.0007

Table 1.18 PCB concentrations in indoor air in North America

Reference	Location	Source	PCBs measured	Concentration	Comments
Vorhees *et al.* (1997)	New Bedford Harbor, Massachusetts, USA	18 homes	Sum of 65 congeners	Geometric mean concentration, 18 ng/m^3 (range, 7.9–61 ng/m^3)	
		Comparison neighbourhood		Geometric mean concentration, 10 ng/m^3 (range, 5.2–51 ng/m^3)	
Vorhees *et al.* (1999)	New Bedford Harbor, Massachusetts, USA	House dust in homes surrounding the Superfund site	Sum of 65 congeners	1400 (range, 320–23 000) ng/g dry weight	
		Comparison neighbourhood		60 (15–290) ng/g	
Herrick *et al.* (2004)	Greater Boston, USA	24 university buildings		> 36 200 ppm 111–395 ng/m^3	One third of the 24 buildings investigated contained caulk at concentrations > 50 ppm (the EPA limit)
Colt *et al.* (2005)	Four geographical regions in the USA	PCBs in carpet dust, 443 homes of Caucasian Americans who served as controls in a case–control study on non-Hodgkin lymphoma		Specific concentrations not reported	PCB concentration in dust was significantly related to age of the house, being greatest in homes built before 1940, and significantly greater in homes built in 1960–1979 (when PCBs were being manufactured in the USA) than in homes constructed after 1980
Franzblau *et al.* (2009)	Five counties in Michigan, USA	House dust House dust	PCB-123 PCB-118	439 000 ppt 33 600 000 ppt	Dioxin-like PCBs contributed 66.2% of the total WHO TEQ found in dust
Harrad *et al.* (2009)	Texas, USA	20 homes	Sum of 9 tri- to heptachlorinated congeners	200 ng/g (ppb); (range, 0.71–620 ng/g)	
	Toronto, Ontario, Canada	10 homes	Sum of 9 tri- to heptachlorinated congeners	260 ng/g (ppb) (range, 51–820 ng/g)	Levels were more than four times higher than those measured in cities in the United Kingdom and New Zealand

EPA, United States Environmental Protection Agency; PCB, polychlorinated biphenyl

Table 1.19 PCB concentrations in indoor air in Europe

Reference	Country	Source	PCB concentration (mean or range)	Comments
Hammar (1992)	Sweden	Joint sealants	80 ng/m^3	Outside the building, mean concentrations were 0.5–4.6 ng/m^3
Balfanz *et al.* (1993)	Germany	Air from contaminated buildings	Range, > 300–7500 ng/m^3	Indoor PCB concentrations were inversely related to the degree of chlorination of the PCB mixtures used
Wilkins *et al.* (2002)	Denmark (Organization of Sealant Branch’s Manufacturers and Distributors)	Dust from public and residential buildings with excessive microbial growth	Estimated inventory of 75 tonnes in caulking materials	Concentration in polluted buildings was 10–20 times higher than the amount found in samples from other buildings
Kohler *et al.* (2005)	Switzerland	Joint sealants in public buildings	> 10 g/kg in 48% of samples	70% of samples contained PCB mixtures such as Clophen A50, Aroclor 1248, and Aroclor 1254
Harrad *et al.* (2006)	United Kingdom	Homes, offices, cars, public microenvironments	8.92 ng/m^3	The least contaminated microenvironment was the car (average, 1391 pg/m^3)
Kuusisto *et al.* (2007)	Finland	Walls/floor	110–540 µg/m^2	Detected PCBs were highly chlorinated
Frederiksen, *et al.* (2012)	Denmark	Air from uncontaminated apartments Elastic sealants from contaminated apartments	168–3843 ng/m^3 187–221 680 mg/kg	Significant correlations were observed between the lower chlorinated congeners in air and sealant

PCB, polychlorinated biphenyl

Table 1.20 PCB concentrations in various types of water around the world

Reference	Type of water	Location	PCB measured	Concentration	Comments
North America					
Jeremiason *et al.* (1994)	Lake	Lake Superior, USA		2.4 ng/L in 1980; 0.18 ng/L in 1992	
Connolly *et al.* (2000)	River	Hudson River		Sometimes > 1300 ng/L	Varied greatly with season and water flow
Rowe *et al.* (2007)	River	Delaware River	Sum of 116 congeners	420–1650 pg/L	
Wang *et al.* (2012)	River	Mississippi River	Sum of 27 congeners	86 and 254 ng/L	
	Lake	Lake Pontchartrain		134–728 ng/L	In some months the PCBs in river water were primarily in the liquid phase, whereas in other months primarily in the sediment
South and Central America					
Rissato *et al.* (2006)	River	Sao Paulo State, Brazil	Sum of seven congeners	0.02–0.5 ng/L	Predominantly lower chlorinated congeners
Africa					
Scarpato *et al.* (2010)	Sea	Tunisia	Sum of 10 congeners	10–12 ng/g	PCB contamination evaluated by mussel-caging technique (exposure, 12 weeks)
		Morocco–Algeria coastal sites		7–8 ng/g	
Jayed *et al.* (2010)	Ocean	Thirteen sites along the Atlantic Moroccan coast	Sum of PCB-28, PCB-153, PCB-138	Wet season: 11 ng/g Dry season: 8.2 ng/g	Concentrations in mussels during wet and dry seasons not significantly different, but values in the northern sites exceeded 2–3 times the medians registered for the other sampling sites
Vorkamp *et al.* (2010)	Ocean	Cape Town harbour	Sum of congeners	81 ng/g dw	Bivalve samples
		Cape Town sea shore		15 ng/g dw	
		Ghana coast		5 ng/g dw	
Europe					
Nondek & Frolikova (1991)	Lake	Sumava lakes, Czech Republic		1900 ng/g	Contamination due to atmospheric transport to non-industrialized areas
Winkels *et al.* (1998)	River	River Danube, Czech Republic		< 5 ng/g dw	Contamination due to flood disaster in the Moravian part of the Czech Republic in July 1997

Table 1.20 (continued)

Reference	Type of water	Location	PCB measured	Concentration	Comments
Fillmann *et al.* (2002)			Sum of seven congeners	≤ 700 ng/g dw (2–196 µg/kg ww in fish)	
UNEP (2002)	River	Krupa, Sana and Lepenica rivers, Balkan area, Slovenia		380 ng/L (in 1988) 100 ng/L (in 1997)	The factory in Semič was storing 5–6 tonnes of waste oil containing PCBs
Desmet *et al.* (2012)	River	Rhone river, France	Sum of PCB_7	1–40 ng/g dw	Concentrations consistently lower than those found during the previous decade (Burns & Villeneuve, 1987). Maximum PCB concentration was identified in 1960–75. The downward trends in concentration followed emission reductions, although soil concentrations decreased at much slower rates (Tolosa *et al.*, 1995)
ADEME (1998), Blanchard *et al.* (2001)	Wastewater	Wastewater treatment plants, France	Sum of seven congeners	Input water, 100–300 ng/L Output water, 15–54 ng/L	In 1999, average concentration was 15–26 ng/L. High levels of DL-PCBs in eel from Dutch freshwater were reported in a screening of Dutch fishery products (Van Leeuwen *et al.*, 2002)
Asia					
Kucklick *et al.* (1994)	Lake	Lake Baikal, Siberia, the Russian Federation, June 1991	61 PCB congeners using standards of Aroclor 1242, 1254, and 1260	Mean, 560 ± 180 pg/L for dissolved phase, and 420 ± 400 pg/L for particulate phase	
Iwata *et al.* (1995) May 1992	Lake	Lake Baikal, the Russian Federation, June 1991	Kanechlors 300, 400, 500, 600 as standards	Range, 8.7–23 pg/m^3	
McConnell *et al.* (1996) June 1991	Lake	Lake Baikal, the Russian Federation, June 1991	Aroclors 1242, 1254 as standards	Mean, 1 324 ± 96 pg/m^3	

DL-PCB, dioxin-like polychlorinated biphenyl; dw, dry weight; PCB, polychlorinated biphenyl; ww, wet weight ppm for the decachlorobiphenyl (Erickson, 1997). Thus under certain circumstances, drinking-water can still be a source of exposure to less chlorinated congeners.

(ii) Sea and freshwater

The USA–Canadian Great Lakes are contaminated by multiple sources of PCBs (Bhavsar *et al.*, 2007; Turyk *et al.*, 2012). It has been shown that industrial sites on rivers feeding Lake Erie received the largest quantities of PCBs, with 26% derived from atmospheric deposition (Kelly *et al.*, 1991). The Hudson River in New York is highly contaminated with PCBs because of releases from two large capacitor plants (Carpenter & Welfinger-Smith, 2011), the Fox River in Wisconsin is highly contaminated because of releases from a manufacturer of carbonless copy paper, and a paper mill (Imamoglu *et al.*, 2004), and the St Lawrence River and several of its tributaries have been contaminated by releases from aluminium foundries operated by companies that discarded hydraulic fluids containing PCBs in drains (Fitzgerald *et al.*, 1996). The Hudson and Fox Rivers are being dredged to remove these contaminants.

(b) Europe

(i) Sea

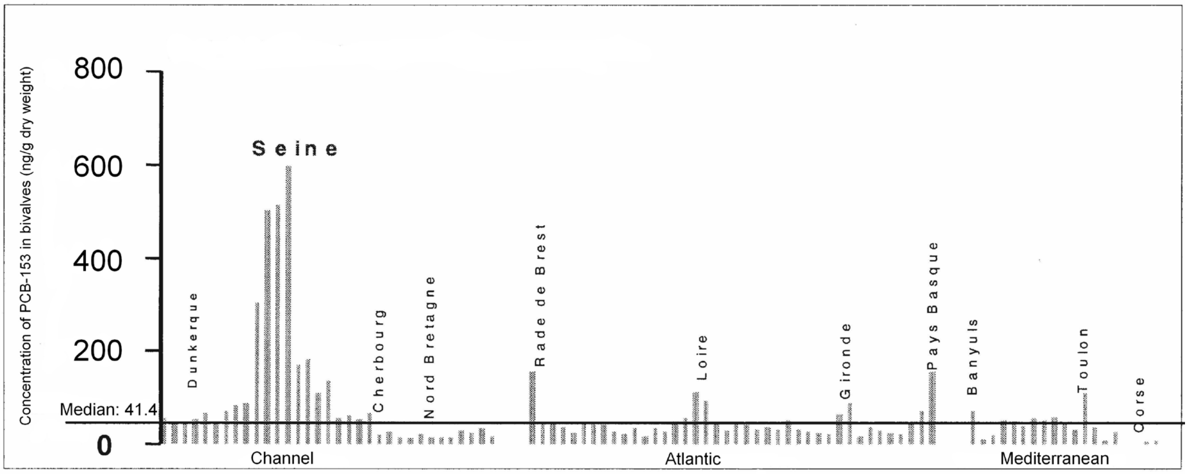
An extensive review of data obtained during the 1980s has been published (Tolosa *et al.*, 1995). In general, the concentrations of PCBs for all the investigated areas in the Mediterranean Sea were similar except in the Ligurian Sea where concentrations were higher. Predictably, the highest concentrations were reported in urban and industrial wastewaters (e.g. from Marseille and Barcelona) as well as in river discharges (e.g. from the Rhone), and decreasing concentration gradients have been found in transects offshore from these sources. PCB concentrations in the suspended particulate matter from coastal and open Western Mediterranean waters were of 5–35 pg/L in 1990, of the same order of magnitude as those reported in other regions, e.g. North Sea and North Atlantic. A more recent study covering the whole Western basin also shows a spatial gradient from the continental shelf (3.5–26.6 pg/L) towards the open sea (1.7–6.6 pg/L); a relatively important enrichment (8.4 pg/L) in open sea stations located in higher productivity frontal zones was observed (Dachs *et al.*, 1997). The dissolved PCBs (Σ12 congeners) amounted to 28–63 pg/L. Total concentrations of PCBs in estuarine and coastal sediment samples of the Mediterranean Sea ranged from 0.04 to 1684 ng/g dw (Koci, 1998; Vale *et al.*, 2002; Vojinovic-Miloradov *et al.*, 2002; Cardellicchio *et al.*, 2007).

During 1974–82, PCB concentrations decreased by a factor of 3 in offshore Monaco (Burns & Villeneuve, 1987), while the surface sediments of the Adriatic coast did not show a temporal trend (Picer & Picer, 1991).

Concentrations of PCBs in ocean water are usually in the low picogram per litre range. The general trend for concentrations in the Baltic Proper suggests an increase in PCB concentrations from the early 1970s onwards (ICES, 2000). This is an opposing trend to the decreasing concentration trends for PCBs in biota from the Baltic Proper (HELCOM, 1996; Roots, 1996).

The monitoring of PCBs in coastal areas may be based on measurements in mussels. Trends in PCB concentrations in the Seine estuary in France are reported in Fig. 1.7 (RNO, 2012). The rate of decrease was 3.5% per year. As reported by the Arctic Monitoring and Assessment Programme (AMAP), several time-series of PCB-153 concentrations in blue mussels from around Iceland showed significant decreasing trends; however, one time-series from a fjord system showed a significant increase (Rigét *et al.*, 2010). Active mussel watching (mussel transplantation) has also been applied in monitoring programmes in Africa (see Table 1.20).

Fig. 1.7 PCB contamination along the coast of France



Concentrations of PCB-153 in mussels or oysters (used as "sentinel species") sampled from coastal areas of France. The Seine estuary and bay are heavily exposed to manmade chemicals of terrestrial origin derived from the urbanized and industrialized river Seine.
Data from The French pollution monitoring programme (Réseau National d'Observation de la qualité du milieu marin)
Reproduced from Abarnou *et al.* (2002), with permission from the publisher

(ii) Freshwater

The major source of freshwater contamination in Europe comes from diffuse leaching of products from users, households, and industries into wastewater streams (UNEP, 2002). The areas most polluted by flood disasters are in Poland (the River Odra) (Wolska *et al.*, 1999; Protasowicki *et al.*, 1999) and in the Czech Republic. The River Danube is a major source of contamination to the Black Sea; however, many chlorinated hydrocarbons have been banned by several European and other countries in the past 10 years (Winkels *et al.*, 1998; Covaci *et al.*, 2002c; Fillmann *et al.*, 2002; see Table 1.20).

Industrial contamination is known to have occurred in Germany (the Rivers Elbe and Rhine and their tributaries) (Brauch, 1993), in former Czechoslovakia (the Sumava Lakes) (Nondek & Frolikova, 1991), in England and Ireland (where however approximately a 50% decline in concentrations between 1970 and 1990, was recorded) (Sanders *et al.*, 1992; Harrad *et al.*, 1994) and in Slovenia through the dumping of industrial waste in the Krupa river during the manufacture of transformers. PCB contamination also occurred in the Balkan area, in the cities of Pancevo, Novi Sad, Belgrade, Kragujevac, in Serbia, after military intervention by NATO in spring 1999.

1.4.7 Food products

Since the early 1990s, food has been identified as the major route of human exposure to lipophilic and persistent organochlorines such as PCBs, PCDDs, and PCDFs. In populations that are not exposed to other known sources, dietary intake contributes to about 90% of the total daily intake of dioxin-like compounds including dioxin-like PCBs, and of this, food of animal origin contributes about 90% in various regions of the world (Schecter *et al.*, 1997; Büchert *et al.*, 2001; Llobet *et al.*, 2003a, b; Päpke & Fürst, 2003; Schecter *et al.*, 2003a, b; Charnley & Doull, 2005; Huwe & Larsen, 2005).

Similarly, it is generally accepted that the major route of exposure to non-dioxin-like PCBs, namely to PCB_6, is dietary intake, by consumption of fatty foodstuffs (IARC, 1978; IPCS, 1993; EFSA, 2005; Lindell, 2012). However, inhalation can also be a significant source of exposure (see Section 1.4.4).

Human food can become contaminated by PCBs via three main routes:

- uptake from the environment, by fish, birds, livestock (via food-chains), and crops;
- contamination of animal feed, by regular practices or accidentally;
- direct contamination of food, accidentally.

Data on PCB concentrations in food are reported in many different ways, making comparisons difficult. The number of congeners analysed differs between studies and often congeners are summed according to groups, such as indicator PCBs, DL-PCBs, or some other number of congeners. When using TEQs, the scheme used should be noted; also some studies report TEQ on the basis of bioassays such as the CALUX system as biological equivalents (BEQ). Results have been reported with different reference units (wet weight, dry weight, or lipid weight). Further difficulties in interpretation arise since different parts of fish or seafood are analysed (muscle, liver, skin, etc.) and PCB concentrations are also sometimes reported on the basis of prepared food (to account for changes by cooking or frying). Finally, the objectives of a study may bias the sampling strategy, often resulting in reporting of higher concentrations.

(a) PCB concentrations in food

Concentrations of DL-PCBs in various meats and dairy products from selected countries and regions are presented in Table 1.21.

(i) Polar regions and North America

PCB concentrations in food for polar regions and North America are summarized in Table 1.22. Domingo & Bocio (2007) reviewed the concentrations of PCB and PCDD/PCDF in marine species and human intake through fish and seafood consumption by different region-specific sections.

The traditional food items for indigenous peoples in the Arctic include lipid-rich tissue of high trophic-level animals. After long-range transport and biomagnification of PCBs in the Arctic marine food-chain, PCBs accumulate in edible animals like fish, seals and whales (AMAP, 2004). This dietary exposure led to PCB concentrations in Arctic inhabitants that exceeded those of individuals living at temperate latitudes (Dewailly *et al.*, 1993), but levels have been shown to decrease (AMAP, 2009). Likewise, PCBs in traditional food items have generally decreased (Rigét *et al.*, 2010).

(ii) Africa

Loutfy *et al.* (2006) investigated levels of WHO-TEQs from diet in Egypt, and determined a range of 6.59–9.98 pg TEQ/kg per day, with about 40% of this value due to DL-PCBs. This value exceeds the maximum WHO tolerable daily intake (TDI) of 4 pg TEQ/kg per day. The primary source was found to be dairy products, in which PCB concentrations were several times higher than in such products in more developed countries. Loutfy *et al.* (2007) determined the concentrations of PCDD/PCDF and dioxin-like PCBs in samples of fish and seafood (mullet fish, bolti fish, bivalves and crab) randomly acquired in local markets in Egypt. The upper-bound concentrations of dioxin-like PCBs ranged from 0.14 (bivalves) to 0.76 (mullet) pg WHO-TEQ/g wet weight, respectively.

Adu-Kumi *et al.* (2010) reported an average TEQ for dioxin-like PCBs in fish from two lakes in Ghana to be 0.7 pg WHO-TEQ/g.

(iii) Australia and New Zealand

In 2000–2001, 168 samples of 22 foods collected for the Australian Total Diet Survey were analysed for DL-PCBs and compared with those from other areas of the world (Table 1.21; Food Standards Australia New Zealand, 2004).

A more recent study reported PCB concentrations from composite samples of Australian farmed yellowtail kingfish (mean, 21 µg/kg; range, 8.6–29 µg/kg), mulloway (mean, 5.4 µg/kg; range,

Table 1.21 Concentrations of dioxin-like PCBs in selected foods from various countries and regions

Food	PCB concentration (range of means), pg TEQ/g lipid					
	Australia	Europe[a]	New Zealand[a,b]	North America[a]	Netherlands[c]	United Kingdom
Beef	0.03–0.11	–	0.0036–0.092	0.5	1.24	0.25–0.31[f]
Pork	0.04–0.07[d]	0.8	0.15–0.43[e]	0.02–1.7	0.23	–
Lamb	0.02–0.06	–	0.01–0.045	–	–	–
Poultry	0.18–0.24	0.7	0.018–0.14	0.3	1.72	0.47–0.53
Fish	9.46–9.5	0.03–9[h]	0.77	0.11–0.28[h]	0.412[g,h]	3.57–3.57
Eggs	0.04–0.11	0.2–0.6	0.05–0.11	0.029[h]	0.87	0.11–0.20
Milk	0.04–0.11	0.2–1.8	0.027–0.15	0.5	0.69	0.34–0.43
Bread	0.0003–0.005	–	0.00099–0.004	–	–	0.06–0.15
Butter	0.021–0.086	–	0.15–0.15	–	0.96	–

[a] Results reported in international toxic equivalents (I-TEQ), which are 10–20% lower than WHO-TEQs
[b] Results reported in the range of lower to middle bound
[c] Results reported as lower bound only
[d] Assumes bacon is representative of all pork products
[e] Pork meat
[f] Carcass meat
[g] Lean fish
[h] Reported on a fresh-weight basis
From Food Standards Australia New Zealand (2004)
PCB, polychlorinated biphenyl; TEQ, toxic equivalent

4.7–6 μg/kg) and manufactured feed (Padula *et al.*, 2012). The mean concentration of DL-PCBs was 2.1 pg TEQ/g (range, 1.2–2.8 pg TEQ/g) in kingfish, and 0.51 pg TEQ/g (range, 0.41–0.61 pg TEQ/g) in mulloway.

(iv) Asia

Concentrations of specific PCB congeners in samples of food from Asia are summarized in Table 1.23. In Japan, a study sponsored by the Ministry of Health and Welfare showed a more than 50% decrease in concentrations of three non-*ortho* substituted PCBs in human milk samples between 1973 and 1996 (Environment Agency of Japan, 1999). A report from the Republic of Korea demonstrated regular dietary exposure (Son *et al.*, 2012; Table 1.23). In China, Liu *et al.* (2011) determined concentrations of seven indicator PCBs in marine fish. The sum of PCB$_7$ ranged from 0.3 to 3.1 μg/g wet weight, with median and mean values of 6.4 ng/g wet weight and 398 ng/g wet weight, respectively (Table 1.23). The average concentrations and contributions of the seven specific congeners at four different sites are presented in Table 1.24. [It was noted that the concentrations found in this study were higher than in other parts of the world.]

(v) Europe

The major contributors to total exposure in Europe appeared to be milk and dairy products for almost all groups of infants and toddlers (Barr *et al.*, 2006; Becker *et al.*, 2009), and fish and seafood products for most of the adolescents, adults, elderly and very elderly groups (Langer *et al.*, 2007; Fréry *et al.*, 2009; ANSES, 2011).

The most comprehensive assessment of PCB concentrations in food was undertaken by the European Food Safety Agency (EFSA) (EFSA, 2005, 2010, 2012). For the 27 European Union Member States, and Switzerland and Norway, in a report that took all food groups together, the

Table 1.22 PCB concentrations in marine foods and estimated dietary intake in polar regions and North America

Country	Food analysed	PCB concentration	Estimated dietary intake	Comments	References
Polar regions					
Inuit of Quebec, Canada	ΣPCB10 in: Polar bear fat Seal blubber Arctic char muscle	 7 μg/g lipid 1 μg/g lipid 150 ng/g lipid		Female consumers of these foods had higher PCB concentrations in milk than a group in Southern Quebec	Dewailly *et al.* (1993)
West Greenland	ΣPCB10 in: Minke whale, beluga and narwhal blubber Halibut liver, kittiwake liver and muscle, minke whale skin, and seal blubber	 > 500 ng/g 50–500 ng/g	23 μg/day per person (3 μg/day per person if blubber food items are excluded from the diet)	Compared with the marine animals, concentrations in food sources from the terrestrial environment were characterized as low	Johansen *et al.* (2004)
North-western Territory, Canada	Food including cooked sucker flesh, raw beluga mattak (skin/blubber) and boiled Canada goose meat	Foodstuffs in the 50–500 ng/g group (Berti *et al.*, 1998)	Mean, 23 ng/kg bw per day Median, 11 ng/kg bw per day	Provisional tolerable daily intake was 300 ng/kg bw per day, based on Health Canada	Johansen *et al.* (2004)
Canada	Fish products from retail market	Geometric mean WHO-TEQ (pg/g wet weight): 0.06 (shrimp), 0.08 (tilapia), 0.92 (salmon)		No information on human exposure	Rawn *et al.* (2006)
North America					
USA (California coast)	Samples of a variety of fish	Mean I-TEQ: 109 pg/g lipid (non-*ortho* PCBs 77, 126, 169)		No information on human exposure	Brown *et al.* (2006)
USA, Maryland, Washington, DC, and North Carolina	Commercially wild caught and farm-raised fish	Bluefish, 800 ng/g ww (highest) Coho salmon, 0.35 ng/g ww (lowest)			Hayward *et al.* (2007)
USA	Salmon and canned sardines	Salmon: PCB-153, 1.2 ng/g ww; PCB-138, 0.93 ng/g ww Canned sardines: PCB-153 and PCB-138, 1.8 ng/g ww		Six of seven NDL-PCBs congeners were detected, with PCB-153 and PCB-138 at highest levels	Schecter *et al.* (2010)

NDL-PCB, non-dioxin-like polychlorinated biphenyls; ww, wet weight

Table 1.23 PCB concentrations in food in Asia

Country, region	Date	Source	PCBs measured	Concentration	Reference
Russian Federation					
Lake Baikal, Siberia	June 1991	Pelagic sculpin, omul, Baikal seal	61 PCB congeners using standards of Aroclor 1242, 1254, and 1260	Ranges, 2.7–2.8 mg/kg of lipid for pelagic sculpin, and 0.73–1.6 mg/kg of lipid for omul	Kucklick *et al.* (1994)
Lake Baikal, Siberia	May–June 1992	Five species of 35 fresh fish samples collected from Lake Baikal in 1993	Total PCBs using an equivalent mixture of Kanechlors 300, 400, 500, and 600 as standards	Mean, 1.7 ± 0.96 μg/g lipid	Nakata *et al.* (1995)
Lake Baikal, Siberia	1993	Three species of fish collected from Lake Baikal in 1993	Total PCBs using an equivalent mixture of Kanechlors 300, 400, 500, and 600 as standards	350 ± 350 ng/g ww	Nakata *et al.* (1997)
China					
Shanghai and its vicinity	2000–1	Various fish and seafood	Kanechlor-300, 400, 500, 600 as standards	Range, 0.20 (shrimp and mussel) to 2.5 (mackerel) ng/g ww	Nakata *et al.* (2002b)
North-eastern, Bohai Sea coastline	Early 2000s	Bivalves and gastropods	PCB mixture (EPA 68A-LCS)	Range, 62.3–344.9 ng/g lipid, for bivalves Range, 81.6–583.6 ng/g lipid, for gastropods	Zhao *et al.* (2005)
Dalian, Tianjin, and Shanghai		Fish and shellfish collected from local supermarkets	PCB-138 and PCB-153 were dominant, followed by PCB-101 and PCB-180	3.60 (0.83–8.04) ng/g ww Estimated daily intake: 1.83 ng/kg bw	Yang *et al.* (2006)
Guangzhou and Zhoushan	2003–4	Seafood (mainly harvested locally) purchased from local markets in Guangzhou and Zhoushan	PCBs 81, 77, 123, 118, 114, 105, 126, 167, 156, 157, 169, 189	Range, 1510–10 200 pg/g lipid	Jiang *et al.* (2007)
South China Sea, Bohai Sea, East China Sea, and Yellow Sea	2006–9	Marine fish	7 PCB congeners (28, 52, 101, 118, 138, 153, and 180); details in Table 1.24	Mean, 398 ng/g ww Median, 6.4 ng/g ww Range, 0.3–3100 ng/g ww	Liu *et al.* (2011)
South, Daya Bay and Hailing Bay	July 2007, December 2007	Fish	PCBs 31/28, 52, 44, 99, 149/118, 153, 138, 180, 170, 194, 101, 110, 147, 146, 187	Range, 1.5–4.0 ng/g ww	Yu *et al.* (2011a, b)
Nanjing	July, 2006	Fish and meat from 10 markets	PCBs 8, 18, 28, 52, 44, 66, 101, 81, 77, 123, 118, 114, 105, 153, 126, 138, 128, 187, 167, 156, 157, 170, 180, 189, 169, 195, 206, 209	Range, 0.87–15 ng/g ww for different fishery product; 5.1–20 ng/g ww for meat product	Su *et al.* (2012)
Fengjiang town (Taizhou)	2005–9	Rice hulls from a waste electrical and electronic-equipment dismantling area	PCBs 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, 28, 52, 101, 138, 153, 180, 3, 15, 19, 202, 205, 208, 209 (dry weight basis)	44.1 ng/g (range, 12.8–124 ng/g) in 2005, 16.3 ng/g (range, 5.44–24.9 ng/g) in 2006, 9.01 ng/g (range, 2.57–22.8 ng/g) in 2007, 7.90 ng/g (range, 3.08–16.5 ng/g) in 2008, 7.39 ng/g (range, 3.80–10.7 ng/g) in 2009	Fu *et al.* (2012)

Table 1.23 (continued)

Country, region	Date	Source	PCBs measured	Concentration	Reference
Japan					
Ariake Sea		Shrimp, mussel, and mackerel		Range, 0.20–2.5 ng/g ww	Nakata *et al.* (2002a)
Japan		Fish and shellfish	PCB-126 and PCB-118 were the highest contributing congeners	In 1999: 0.98×10^{-3} $\text{WHO-TEQ}_{\text{PCDD/PCDF/PCB}}$ In 2004: 0.91×10^{-3} $\text{WHO-TEQ}_{\text{PCDD/PCDF/PCB}}$	Sasamoto *et al.* (2006)
Hirakata city, Osaka Prefecture	Unspecified	Domestic and imported seafood purchased from three food markets	PCBs 81, 77, 123, 118, 114. 105, 126, 167, 156, 157, 169, 180, 170, 189	Range, 13–40 182 pg/g ww	Ohta *et al.* (2008b)
Lao People's Democratic Republic					
Vientiane (Agent Orange-non-sprayed capital)	2001	Meat, fish, and dairy products from food markets	PCBs 37, 77, 126, 169, 81, 28, 33, 55, 60, 66, 74, 105, 114, 118, 122, 123, 124, 156, 157, 167, 189, 52, 101, 128, 138, 153, 170, 180, 187, 194, 206, 209	Range, 0.004–0.186 pg TEQ/g in fish samples; 0.011– 0.063 pg TEQ/g in meat and dairy products	Schecter *et al.* (2003a)
Republic of Korea					
		Muscle of sport and market fish	22 PCB congeners	23.0 (4.48–95.6) ng/g ww (sport fish) 8.91 (2.96–68.2) ng/g ww (market fish)	Yim *et al.* (2005)
		40 species of marine organism	DL-PCBs	0.4×10^{-3} $(0.008–0.6) \times 10^{-3}$ WHO-TEQ ww	Moon & Ok (2006)
	2005 to 2007	26 marine species ($n = 78$) collected annually during 2005–2007 from a large fish market in Busan	PCBs 8, 18, 28, 29, 44, 52, 87, 101, 105, 110, 118, 128, 138, 153, 170, 180, 187, 194, 195, 200, 205, 206	Range, 0.2–41 ng/g ww	Moon *et al.* (2009)
Singapore					
Singapore (cont.)	June 2002 to June 2003	Twenty types of seafood from local supermarkets	PCBs 17, 18, 28/31, 33, 44, 49, 52, 70, 74, 82, 87, 90, 101, 95, 99, 105, 110, 118, 128, 132, 138, 149, 151, 153, 156, 169, 170, 171, 177, 180, 183, 187, 194, 199, 201, 205, 206, 208, 209	Mean, 3.72 ng/g ww (range, 0.61–28.47 ng/g ww)	Bayen *et al.* (2005)

DL-PCBs, dioxin-like polychlorinated biphenyl; ww, wet weight

Table 1.24 PCB concentrations in marine fish from China

PCB	PCB concentration (n/g ww)							
	South China Sea		Boahi Sea		East China Sea		Yellow Sea	
	Average concentration (n/g ww)	Contribution (%)	Average concentration (n/g ww)	Contribution (%)	Average concentration (n/g ww)	Contribution (%)	Average concentration (n/g ww)	Contribution (%)
PCB-25	0.10	5.0	6.7	10.7	38.8	7.5	111.9	11.1
PCB-52	0.13	6.4	4.6	7.3	40.8	7.8	64.2	6.4
PCB-101	0.35	17.3	8.6	13.7	48.3	9.3	88.1	8.7
PCB-118	0.22	10.7	12.1	19.3	43.9	8.4	106.2	10.5
PCB-138	0.66	32.4	11.2	17.8	167.1	32.1	336.5	33.4
PCB-153	0.39	18.8	16.3	26.0	136	26.2	248.8	24.7
PCB-180	0.19	9.4	3.3	5.3	45.0	8.7	52.4	5.2
Σ 7 PCBs	2.0	–	62.8	–	520	–	1008	–

PCB, polychlorinated biphenyl; ww, wet weight
Data from Liu *et al.* (2011)

upper bound (lower bound) for the 50th, 90th and 95th percentiles were < 0.005 (< 0.005), 0.02 (0.01) and 0.03 (0.01) pg WHO_{2005}-TEQ/g wet weight for PCDD/PCDF, respectively. For the total TEQ, the upper bound (lower bound) concentrations were 0.01 (< 0.005), 0.04 (0.02) and 0.07 (0.04) pg WHO_{2005}-TEQ/g wet weight, respectively (EFSA CONTAM, 2012). Infant formulae showed upper bound concentrations below the current maximum levels (0.2 pg WHO_{2005}-TEQ/g wet weight), with highest concentrations found in ready-to-eat meals containing fish or meat. Overall, a decrease in concentrations of DL-PCBs was observed for the three food groups available: "raw milk and dairy products," "hen eggs and egg products" and "muscle meat from fishes other than eels." Feed and food of animal origin contained higher concentrations of PCDD/PCDF and DL-PCBs combined (the non-*ortho* PCBs were the main contributors to the total TEQs) than foods from plant origin. PCB-153, PCB-138, and PCB-180 represented altogether 36.9–97.8% of the sum of PCB_6. The maximum levels were exceeded in 9.7% of the food samples and 2.3% of the feed samples for PCDD/PCDF and DL-PCBs combined, and in 3.0% of the food samples and 2.4% of the feed samples for the PCB_6. With respect to food categories, lower PCB concentrations were found in meat from sheep, eggs from battery rearing, farmed salmon and trout, and farm milk (which however showed higher concentrations of PCDD/PCDF and DL-PCBs combined than milk from bulk) (EFSA, 2012).

The Baltic Sea area is heavily contaminated with persistent organochlorine compounds, including PCBs (Kiviranta *et al.*, 2003), as is clearly attested by samples of fatty fish from the eastern coast in Sweden (Svensson *et al.*, 1995). In the most contaminated feed group, the highest relative contribution to the WHO_{2005}-TEQ_{total} came from non-*ortho* PCBs, up to twice the average contribution (EFSA, 2012).

(b) Estimated daily dietary intake

In Europe, more than 90% of PCB exposure in the general population is via food consumption (EFSA, 2005; Table 1.25). Average daily dietary intakes of the sum of PCB_6 are in the range of 10–45 ng/kg bw for adults, and two and a half times higher in children. Limited exposure data for young children indicate that the average daily intake (breastfeeding excluded) of the sum of PCB_6 is about 27–50 ng/kg bw. Overall, the non-*ortho* PCBs represented 21.0–74.9% of the WHO_{2005}-TEQ_{total} of PCDD/PCDF and DL-PCBs combined in food (EFSA, 2012), and the mono-*ortho* PCBs represented no more than 12% of the WHO_{2005}-TEQ_{total}. In the most contaminated samples, such as products from aquatic animals and from ruminants, the relative contribution of the non-*ortho* PCBs ranged from 34.2% to 86.1%. Most likely due to an effect of the European risk management measures, a decrease in exposure to the sum of PCB_6 was observed between 2002–2004 and 2008–2010 in most but not all population groups, and it was estimated between 2.0% and 75.6%.

In the USA, the daily dietary intake of PCBs for adults decreased from 1978 (0.027 µg/kg bw) until 1986–1991 (< 1 ng/kg bw) (IPCS, 2003). Mean daily intakes for infants during the same period decreased from 11 to < 1 ng/kg bw. However, trends during 1991–1997 did not appear to decrease, and ranges of daily dietary intake were 3–5 ng/kg bw for adults, and 2–12 ng/kg bw for children of different ages (IPCS, 2003).

Daily dietary intake of PCBs from countries in Asia are presented in Table 1.26. In China, the estimated daily intake from four food groups of animal origin ranged from 0.09 to 0.59 pg TEQ/kg bw for DL-PCBs, which is lower than the daily intake in some developed countries (Liu *et al.*, 2013). A survey of food items on the market and typical consumption patterns in Japan reported a daily intake for the general population of 2.60 pg TEQ/kg bw per day (Koizumi *et al.*, 2005). Of

Table 1.25 Dietary exposure to PCBs for an average consumer on the European market

Food group	Mean ΣPCBs (ng/g)	Consumption (g/day)			Exposure (ng/day)		
		Italy	France	Sweden	Italy	France	Sweden
Cereals and cereal products	0.0213	270	218	292	6	5	6
Fruits and vegetables	0.0495	498	313	387	25	15	19
Eggs	0.73	18	17	15	13	12	11
Fats and oils	5.05	38	18	24	192	91	121
Meat and meat products	1.52	134	117	143	204	178	218
Offals	0.74	3	3	7	2	2	5
Fish and fish products	12.50	43	32	35	538	400	438
Milk	0.17	124	106	343	21	18	59
Cheese and dairy products	0.98	87	100	45	86	98	44
Total (ng/kg bw per day)		–	–	–	18.1	13.7	15.4
Total (ng/kg bw per day) for a high consumer of meat and meat products		–	–	–	22.0	17.6	18.9
Total (ng/kg bw per day) for a high consumer of fish and fish products		–	–	–	40.4	31.8	33.3

PCB, polychlorinated biphenyl
Adapted from EFSA (2005)

Table 1.26 Estimated daily dietary intake of PCBs in Asia

Country, region	Date	Source	PCBs measured	Mean daily intake	Reference
Japan, Fukuoka Prefecture	1969–70	Patients Individual consumption of oil was estimated by taking into account age, sex and the number of meals at home	PCBs, PCDFs, and PCQs	Estimated total intake: PCBs, 633 mg PCDFs, 3.4 mg PCQs, 596 mg	Hayabuchi *et al.* (1979)
Japan, eight sites from Hokkaido to Okinawa	1995 survey	Food duplicate study 40 women (mean age, 52 years)	11 PCB congeners (74, 99, 118, 138, 146, 153, 156, 163, 170, 180, and 182)	165.9 ng/day	Koizumi *et al.* (2005)
Japan, 75 different areas of 25 prefectures	Not reported	Food duplicate study 374 subjects, 86 men and 288 women (mean age, 48.0 years; range, 17–72 years)	12 PCBs	Mean PCB intake, 0.59 pg/kg bw per day Median PCB intake, 0.39 pg/kg bw per day	Arisawa *et al.* (2008)
Republic of Korea	2010	Estimated dietary intake 200 individual food samples from 40 different foodstuffs	62 PCB congeners, including 7 indicator PCBs and 12 DL-PCBs (PCB-1, 3, 4, 8, 10, 15, 18, 19, 22, 33, 37, 44, 49, 54, 70, 74, 87, 95, 99, 104, 110, 112, 128, 149, 151, 155, 158, 168, 170, 171, 177, 178, 183, 187, 188, 191, 194, 199, 201, 202, 205, 206, 208, and 209)	9.9 ng/kg bw per day	Son *et al.* (2012)

DL-PCBs, dioxin-like polychlorinated biphenyls; PCDFs, polychlorinated dibenzofurans; PCQs, polychlorinated quaterphenyls; ww, wet weight these, 2.41 pg TEQ/kg bw per day was from ingestion of food, while inhalation and soil ingestion contributed only to 0.19 pg TEQ/kg bw per day. A "typical" Japanese person receives 120.7 pg TEQ per day through food consumption (mainly fish/shellfish, followed by meat/eggs).

In specific subpopulations with high dietary PCB exposure, such as Baltic Sea fishermen, the daily intake from fish of the sum of PCB_6 was estimated at 40 ng/kg bw, corresponding to a total daily intake of the sum of non-dioxin-like PCBs of 80 ng/kg bw, before taking into account the rest of the diet (Lindell, 2012).

In breastfed infants, the most recent WHO study of PCB exposure reported a mean daily intake of about 1600 ng/kg bw (range, 230–7300 ng/kg bw per day) for total PCB_6. Thus, exposure of infants to PCB_6 (and DL-PCBs) through human milk is about two orders of magnitude higher than the average daily intake by adults.

1.4.8 Occurrence in manufactured products other than commercial PCB preparations

In addition to commercial PCB preparations, many manufactured products contain PCBs as a result of contact with PCB products, as contaminants during manufacture, or as degradation products of other chlorinated compounds. For example, PCBs have been found in various paint pigments (Hu & Hornbuckle, 2010; Kuusisto *et al.*, 2006). Electronic equipment contains PCBs, which are released during dismantling.

Since the sampling and determination of the presence of PCBs is a difficult process, the Basel Convention has established a so-called "grey list" of materials and equipment that are suspected to contain PCBs (Basel Convention, 2003):

- Cable insulation
- Rubber and felt gaskets
- Thermal insulation material including fibreglass, felt, foam and cork
- Transformers, capacitors (also contained in electronic equipment)
- Voltage regulators, switches, bushings and electromagnets
- Adhesives and tapes
- Oil, including that contained in electrical equipment and motors, anchor windlasses, hydraulic systems
- Surface contamination of machinery and other solid surfaces
- Oil-based paint
- Caulking
- Rubber isolation mounts
- Foundations mounts
- Pipe hangers
- Light ballasts
- Plasticizers.

1.4.9 Population biomonitoring

(a) Blood

The presence of PCBs in serum or blood may reflect exposure from any source (Dewailly *et al.*, 1988). Results from different studies in humans have indicated that measurements of PCBs in serum generally reflect cumulative past exposure. Many PCB congeners can remain in the body for years after exposure, although some of the less chlorinated congeners are more volatile and consequently show shorter residence times.

(i) North America

Hopf *et al.* (2009a) provided an extensive review of reports on background levels of PCBs in the USA population. They concluded that serum concentrations increased up to 1979 and decreased after that, but that the background levels are still of concern. The NHANES survey over the period 2002–2004 reported increasing concentrations of PCBs with age, and concentrations were higher in men than in women, and higher in African-Americans and Caucasians than in Mexican-Americans (Patterson *et al.* 2009). Sjödin *et al.* (2004) showed a decline in concentrations of PCB-153 between 1985 and 2002 in pooled samples from the NHANES study.

Several studies have looked at specific populations living near specific contaminated sites or eating contaminated fish (Table 1.27).

Serum concentrations for the sum of 17 congeners in Viet Nam veterans were 167.5 ng/L lipid adjusted, of which the major portion (116.6 ng/L) were di-*ortho* congeners (Schecter *et al.*, 1996).

Jarrell *et al.* (2005) determined the sum of 24 congeners in pregnant women in Canada, and reported a mean value of 0.78 ng/L wet weight.

Because the less chlorinated PCBs are more volatile, teachers working in a school where caulk containing PCBs was used showed serum congener profiles that were enriched in less chlorinated congeners (Herrick *et al.*, 2011).

DeCaprio *et al.* (2005) reported finding a pattern of PCB congeners in serum specific of young native Americans living near a PCB-contaminated waste site. This pattern was not clearly observed in older individuals because it was obscured by the greater concentrations of more persistent congeners, coming primarily from dietary exposure.

(ii) Europe

Several European studies on human biomonitoring have reported blood PCB concentrations in adults or children (summarized in Table 1.28). Past environmental contamination in industrial areas has polluted surrounding soils and forage, leading in turn to high blood PCB concentrations in the adult population. Age-related accumulation of PCBs has been observed in many studies (Patterson *et al.*, 1994; Apostoli *et al.*, 2005; Park *et al.*, 2007), and may be partially explained by historical high levels of exposure in the 1970s.

In Germany, Environmental Surveys (GerES) were carried out in 1998 (Becker *et al.*, 2002) and during 2003–2006 (Becker *et al.*, 2009). GerES data show mean blood concentrations for the sum of PCBs of 1.3–1.7 µg/L in 1998 and of 286 ng/L in the more recent survey, with strong difference (factor of 5.6) between age groups 18–25 and 66–69 years. In Belgium in 2007–2011 (Schoeters *et al.*, 2011), the Flemish Human Environmental Survey reported average blood PCB concentrations of 333 ng/g lipid. Average concentrations in the United Kingdom in 2003 were 170 ng/g lipid (Thomas *et al.*, 2006). In Spain in 2004–2008, concentrations of the most common PCBs were in the range of 21.8 to 38.9 ng/g lipid (Ibarluzea *et al.*, 2011). In France, blood analysis in the general adult population was first carried out in 1986 (Dewailly *et al.*, 1988) and then in 2006–2007 (French Nutrition and Health survey; Fréry *et al.*, 2013). The reported blood PCB concentrations in populations in industrial polluted areas such as Italy (Turci *et al.*, 2004; Apostoli *et al.*, 2005; Turrio-Baldassarri *et al.*, 2008) and Slovakia (Jursa *et al.*, 2006) were high compared with those in non-occupationally exposed populations such as in Sweden (Salihovic *et al.*, 2012). In the Faroe Islands (Denmark), high concentrations of PCBs and hydroxylated PCBs in serum samples from pregnant women were attributed to the traditional diet, made of pilot whale meat, blubber and other marine food (Fängström *et al.*, 2002).

The most frequently detected di-*ortho*-chlorine-substituted PCBs in population studies are PCB-138, PCB-153, and PCB-180 (Glynn *et al.*, 2000), accounting for 65–78% of the measured sum of total PCBs (Needham *et al.*, 2005). The seven PCB indicator congeners (118, 138, 153, 156, 170, 180, and 194) contributed to 99% of the total PCB levels, with a modest contribution from dioxin-like congeners (Apostoli *et al.*, 2005).

In several countries in the European Union, a clear decrease in blood concentrations of PCBs has been observed in the last two decades. Overall,

Table 1.27 Serum concentrations of PCBs after consumption of PCB-contaminated fish, North America

Country, region	Sample	PCBs measured	Mean ng/g (ppb)	Reference
North Canada, Nunavik	Inuit women, *n* = 159	Sum of 14 congeners	313.2 ± 2 Range, 71.3–1951.3	Muckle *et al.* (2001)
USA, St Lawrence River	Native American adults, *n* = 753	Sum of 101 congeners	4.39 ± 4.18	DeCaprio *et al.* (2005)
USA, St Lawrence River	Native American adolescents		0.71 ± 0.668 (if not breastfed) 0.95 ± 0.806 (if breastfed)	Schell *et al.* (2008)
USA, Great Lakes	Fish consumers, *n* = 293	Sum of 89 congeners in µg/L (ppb) wet weight	4.2 (2.7), in 1994–95 2.8 (2.0), in 2001–05	Knobeloch *et al.* (2009)
	Fishing-ship captains, men		6.3 (5.0), in 1994–95 1.2 (0.9) – 3.8 (3.0), in 2001–05	
USA, Anniston, Alabama	Adult residents, *n* = 394	Sum of 35 congeners	4.72 ± 11.05 Range, 0.09–170.42	Goncharov *et al.* (2011)

PCB, polychlorinated biphenyl

mean whole blood concentrations of PCB-138, PCB-153, and PCB-180 appear to have decreased by approximately 80% in 20 years (Link *et al.*, 2005; Hagmar *et al.*, 2006; Agudo *et al.*, 2009; AMAP, 2009). Nevertheless, compared with North America (CDC, 2005), serum concentrations of PCB-138, PCB-153, and PCB-180 were higher by two- to fivefold in Germany in 1998 (Heudorf *et al.*, 2002), or Italy in 2001–2003 (Turci *et al.*, 2004; Apostoli *et al.*, 2005; Needham *et al.*, 2005). Similarly, serum concentrations of hydroxylated PCBs and methylsulfonyl-substituted metabolites of PCBs were higher by two to threefold in a contaminated area in a study in Slovakia (Hovander *et al.*, 2006).

(iii) Asia

In Asia, PCB concentrations in several biological samples (including serum or whole blood, umbilical cord blood, hair, breast milk, adipose tissue, liver, kidney, and lung tissues) showed a wide range (Table 1.29; Schecter *et al.*, 2003a). Data specific to the Yusho and Yucheng patients are presented in Table 1.30 and Table 1.31, respectively.

(iv) South and Central America

Rodríguez-Dozal *et al.* (2012) analysed serum samples from pregnant women in Mexico for 19 congeners and Aroclor 1260. For Aroclor 1260 [calculated as the sum of PCB-138 and PCB-153 multiplied by 5.2], they reported regional differences (mean concentration, 31.1 ng/g lipid) and elevated concentrations from residents of Merida (maximum, 546.2 ng/g lipid). Trejo-Acevedo *et al.* (2012) measured serum PCB concentrations (sum of 14 congeners) from children living in a malaria-endemic area of Mexico, and reported a mean serum PCB concentration of 5892 ± 3895.7 ng/g lipid. In an analysis of PCB congeners in maternal blood of women in Sao Paulo State, Brazil, PCB-118, PCB-138, and PCB-153 were detectable in more than 70% of samples, and their concentrations were almost double in women from industrial areas compared with women from rural areas (Rudge *et al.*, 2012).

(v) Africa

Röllin *et al.* (2009) reported overall low blood concentrations of PCBs (99, 118, 138, 153, 170, 180 and 187) in delivering mothers from seven geographical regions in South Africa. Large regional differences were observed, with women

Table 1.28 Blood concentrations of PCBs in various European countries

Country	Reference/study	Period	Age (years)	Number	PCBs measured	Mean	95th percentile
France	Dewailly *et al.* (1988)	1986	Men: 38	569	Σ 7 PCBi	4020 ng/L	5000 ng/L
	Fréry *et al.* (2009)	2005	30–65	1030	138, 153, 180	347 ng/g lipid	714 ng/g lipid
	ANSES (2011)	2009–10	18–75	606 16	138, 153, 180	305 ng/g lipid 681 ng/g lipid	1368 ng/g lipid 3150 ng/g lipid
	Fréry *et al.* (2013)	2006–7	18–74	386	Σ 6 PCBi	287 ng/g lipid 1858 ng/L	721 ng/g lipid 4977 ng/L
Germany	GerES III GerES IV (2008)	1998 2003–6	18–69 7–14	2815 1079	138, 153, 180	1570 ng/L 286 ng/L	5000 ng/L 980 ng/L
United Kingdom	Thomas *et al.* (2006)	2003	22–80	151	Σ 31 congeners	170 ng/g lipid	670 ng/g lipid
Belgium	Schoeters *et al.* (2011)	2007–11	50–65	1530	138, 153, 180	333 ng/g lipid	
Italy	Turci *et al.* (2004)	2001–3		162	Total PCBs	2480 ng/L	5240 ng/L
	Apostoli *et al.* (2005)	2003	20–79	311	Σ 24 congeners	897 ng/g lipid	2643 ng/g lipid
	Turrio-Baldassarri *et al.* (2008)	2004	Men: 51	94	Σ 6 congeners	866 ng/g lipid	
Slovakia	Jursa *et al.* (2006)	2001–2	20–70	315	Σ 45 congeners	CA:5863 ng/g lipid RA:1245 ng/g lipid	Max: 55 334 ng/g lipid Max: 9015 ng/g lipid
	Park *et al.* (2007)	2002–4	Pregnant	CA: 762 RA: 341	118, 153, 105, 138, 180, 170	CA: 734 ng/g lipid RA: 351 ng/g lipid	CA:2105 ng/g lipid RA: 469 ng/g lipid
Spain	Agudo *et al.* (2009)	1992–6	35–64	953	138, 153, 180	459 ng/g lipid	
	Ibarluzea *et al.* (2011)	2004–8	Pregnant	1259	138, 153, 180	88 ng/g lipid	
Sweden	Salihovic *et al.* (2012)	2001–4	70	Men: 495 Women: 517	138, 153, 180	Men: 600 ng/g lipid Women: 517 ng/g lipid	753 ng/g lipid 664 ng/g lipid

CA, contaminated area; PCB, polychlorinated biphenyl; RA, reference area; PCBi, indicator PCBs

Table 1.29 PCB concentrations in biological samples from populations in Asia

Country, region, Year	Subjects, participants	Samples	PCBs measured	Mean concentrations (standard deviation or range)	Reference
Taiwan, China 1994	Pooled blood of 50 women	Blood serum	PCBs 28, 52, 74, 66, 101, 153, 138, 187, 183, 156, 157, 180, 170	386 ng/g lipid	Guo *et al.* (1997)
Central Taiwan, China 2001	30 primiparous women (mean age, 27.8 yr; range, 20–35 yr)	Breast milk	PCBs 77, 81, 126, 169, 105, 114, 118, 123, 156, 157, 167, 189	4.87 (SD 8.04) pg TEQ/g lipid	Chao *et al.* (2003)
Central Taiwan, China 2000–1	20 pregnant women; mean age, 28 yr (range, 25–35 yr)	Placenta, milk, venous blood, and cord blood	12 DL-PCBs and 6 indicator PCBs	DL-PCBs: 5292 pg/g lipid in placenta, 10 170 pg/g lipid in milk, 9496 pg/g lipid in venous blood, and 3577 pg/g lipid in cord blood Indicator PCBs: 32 457 pg/g lipid in placenta, 55 425 pg/g lipid in milk, 36 416 pg/g lipid in venous blood, and 37 758 pg/g lipid in cord blood	Wang *et al.* (2004)
Taiwan, China 2004	Pooled blood plasma of 10 blood donors	Blood plasma	33 PCB congeners included PCB-8, 37, 44, 49, 52, 60, 66, 70, 74, 77, 82, 87, 99, 101, 105, 110, 114, 118, 126, 128, 138, 153, 156, 157, 158, 166, 169, 170, 179, 180, 183, 187, and 189.	187 ng/g lipid	Hsu *et al.* (2005)
East China July 11–13, 2006	64 male workers, aged 18–60 yr	Hair	PCBs (1668A-LCS, 1668A-IS)	Mean 1 600 pg/g dw (55 400–7 200 000 pg/g dw)	Wen *et al.* (2008)
China, Zhejiang April 2007 to January 2008	Surgical patients newly diagnosed for cancer (mean age, 65 yr; range, 32 to 94 yr)	Kidney, liver and lung tissues	27 PCB congeners	Median (range) in ng/g lipid: 382.15 (86.92–1403.92) (kidney); 460.00 (89.19–1742.57) (liver); 304.64 (104.85–373.25) (lung)	Zhao *et al.* (2009)
China, Shenzhen July to November 2007	60 samples from primiparous women living in areas not polluted by POPs (mean age, 28 yr; range 20–34 yr)	Breast milk	PCBs 28, 52, 101, 138, 153, 180, 77, 81, 126, 169, 105, 114, 118, 123, 156, 157, 167, 189	DL-PCBs: median (range): 4580 (1964–13 967) pg/g fat Indicator PCBs: 13.2 (3.4–39.2) pg/g fat	Deng *et al.* (2012)
China, Zhejiang Province 2008	74 women in rural areas (mean age, 25.0 yr; range, 19–29 yr) and in urban areas (mean age, 26.5 yr; range, 22–29 yr)	Breast milk	PCBs 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, 189, 28, 52, 101, 138, 153, 180	42 774 ± 27 841 pg/g lipid (urban group) 26 546 ± 11 375 pg/g lipid (rural group)	Shen *et al.* (2012)
India, six different locations, 2009	55 mothers, reproductive age, ranged 21–38 yr	Breast milk	$^{13}C_{12}$-labelled PCBs	3.1 to 5 400 ng/g lipid weight	Devanathan *et al.* (2012)

Table 1.29 (continued)

Country, region, Year	Subjects, participants	Samples	PCBs measured	Mean concentrations (standard deviation or range)	Reference
India, Bangalore and Chidambaram 2007	25 e-waste recycling workers	Serum	62 PCB congeners	360 pg/g ww	Eguchi *et al.* (2012)
India, Bangalore and Chidambaram 2007	20 residents near a coastal area	Serum	62 PCB congeners	140 pg/g ww	Eguchi *et al.* (2012)
Islamic Republic of Iran, Ahvaz and Noushahr cities, and the countryside of Noushahr November 2007 to January 2008	16 pregnant women in Noushahr (mean age, 26 yr; range, 16–43 yr) 21 pregnant women in Ahvaz (mean age, 27 yr; range, 18–36 yr) 19 pregnant women in countryside of Noushahr (mean age, 25 yr; range, 15–36 yr)	Hair	PCBs 28, 52, 101, 118, 138, 143, 153, 180	Median (range): 9 (4–140) ng/g in Noushahr 8 (4–14) ng/g in Ahvaz 2 (undetected –15) ng/g in Noushahr countryside	Dahmardeh Behrooz *et al.* (2012)
Japan, Fukuoka Prefecture April to June, 1991	Nine normal women (mean age, 30 yr; range, 25–32 yr)	Breast milk	PCB-77, PCB-126, PCB-169	Mean coplanar PCBs, 21.3 pg TEQ/g fat Mean PCB-77: 12.4 pg/g fat; Mean PCB-126: 183.7 pg/g fat; Mean PCB-169: 65.7 pg/g fat TEFs as proposed by the NATO-CCMS (1988), and those of the coplanar PCBs were calculated using data reported by Safe (1990).	Matsueda *et al.* (1993)
Japan September 1994 to November 1996	31 normal volunteers (age, 20–61 yr)	Sebum, and blood	PCB-77, PCB-126, PCB-169	Mean PCBs, 447.3 pg/g lipid (sebum), and 204.6 pg/g lipid (blood)	Iida *et al.* (1999)
Japan 1998–9	28 patients with various illnesses (age, 19–87 yr)	Liver and adipose tissue	Non-*ortho*-PCBs	Mean (range): 20 (2.8–91) TEQ/g lipid (liver tissue) 17 (2.7–57) pg TEQ/g lipid (adipose tissue)	Takenaka *et al.* (2002)
Japan 1999–2000	80 women (mean age, 36.9 yr; range, 26–43 yr)	Serum	36 PCBs	Median, 0.46 (25th percentile, 0.35; 75th percentile, 0.66) nmol/g lipid	Tsukino *et al.* (2006)

Table 1.29 (continued)

Country, region, Year	Subjects, participants	Samples	PCBs measured	Mean concentrations (standard deviation or range)	Reference
Japan, Fukuoka Prefecture 2002–3	127 normal controls (age, 68.0 yr; SD, 5.4 yr)	Blood/serum	PCB-77, PCB-126, PCB-169	11.9 pg TEQ/g lipid	Todaka *et al.* (2007a)
Japan Born 1950–86	15 samples from 9 healthy subjects	Preserved umbilical cord	Dioxin-like PCBs (81, 77, 123, 118, 114, 105, 126, 167, 156, 157, 169, 189)	Mean (range): 2700 (250–12 000) pg/g	Aozasa *et al.* (2008)
Japan, Sapporo City July 2002 to July 2004	101 primiparous pregnant women (mean age, 28.8 yr; range, 18–40 yr) and 94 multiparous pregnant women (mean age, 32.3 yr; range, 28–47 yr)	Blood	PCBs 28, 44, 47/48, 49, 52/69, 56/60, 63, 66, 70, 71, 74, 85, 87, 92, 93/95/98, 99, 101, 107/108, 110, 117, 128, 130, 132, 134, 135, 137, 138, 139, 141, 146, 147, 151, 153, 163/164, 165, 170, 172, 177, 178, 179, 180, 181, 182/187, 183, 191, 194, 195, 196/203, 198/201, 200, 202, 205, 206, 207, 208, and 209	Mean (range): 114.5 ± 61.0 (42.2–329.3) ng/ g lipid (primiparous) 100.2 ± 48.2 ng/g lipid (31.5–258.0) (multiparous)	Todaka *et al.* (2008a)
Japan, Sapporo City, Hokkaido Prefecture July 2002 to July 2004	60 mothers (mean age, 31 yr; range, 21–47 yr)	Blood and breast milk	PCBs 77, 81, 126, 169, 105, 114, 118, 123, 156, 157, 167, 189	Mono-*ortho* PCBs, 13.4 ± 5.8 ng/g lipid (blood) and 14.4 ± 8.2 ng/g lipid (breast milk) Non-*ortho* PCBs 97 ± 10 pg/g lipid (blood); and 60 ± 28 pg/g lipid (breast milk)	Todaka *et al.* (2008b)
Japan, Fukuoka and Nagasaki prefectures Born 1970–3	Five babies born to healthy mothers	Preserved umbilical cord	DL-PCBs (77, 81, 126, 169, 105, 114, 118, 123, 156, 157, 167, 189)	0.1 pg TEQ/g dw	Nagayama *et al.* (2010)
Japan, Sapporo City July 2002 to October 2005	119 primiparous mothers (mean age, 30 yr; range, 21–40 yr)	Blood and breast milk	Non-*ortho* PCBs, mono-*ortho* PCBs, and 56 NDL-PCBs	120.2 ± 67.3 ng/g lipid (blood) 90.4 ± 51.6 ng/g lipid (breast milk)	Todaka *et al.* (2010)
Japan, Sapporo City July 2002 to October 2005	514 pregnant women (mean age, 32 yr; range, 22–41 yr)	Blood and breast milk	Non-*ortho* PCBs, mono-*ortho* PCBs, and 56 NDL-PCBs	Non-*ortho* PCBs, 77 ± 32 pg/g lipid (blood) and 51 ± 21 pg/g lipid (breast milk) Mono-*ortho* PCBs, 11.7 ± 5.7 pg/g lipid (blood) and 10.0 ± 5.2 ng/g lipid (breast milk) NDL-PCBs, 107 (16–326) ng/g lipid (blood) and 73 (12–252) ng/g lipid (breast milk)	Todaka *et al.* (2011)

Table 1.29 (continued)

Country, region, Year	Subjects, participants	Samples	PCBs measured	Mean concentrations (standard deviation or range)	Reference
Republic of Korea, Kyungpook May 2007 to May 2008	53 female myoma patients (mean age, 47 yr; range, 40–68 yr)	Adipose tissue	PCBs 8, 18, 28, 29, 44, 52, 87, 101, 105, 110, 118, 128, 138, 153, 170, 180, 187, 194, 195, 200, 205, 206	270 ± 140 ng/g lipid	Moon *et al.* (2012)
Russian Federation, Irkutsk Region 1992	Three groups of Siberians	Blood	PCBs 77, 126, 169	Mean TEQ, 2.0–25.2 ppt	Schecter *et al.* (2002)
Viet Nam, areas sprayed with Agent Orange 2006	Potentially exposed persons	Blood	Coplanar PCBs, mono-*ortho* PCBs	Coplanar PCBs TEQ, 1.1–5.6 pg/g lipid Mono-*ortho* PCBs TEQ, 1.8–7.3 pg/g lipid	Schecter *et al.* (2006)

DL-PCB, dioxin-like polychlorinated biphenyl; dw, dry weight; NDL-PCB, non-dioxin-like polychlorinated biphenyl; TEQ, toxic equivalent; ww, wet weight; yr, year

Table 1.30 PCB, PCDF, and PCDD concentrations in biological samples from the Yusho population, Japan

Region Period	Subjects/ participants	Sample	PCB measured	Concentration (mean, median, range)	Reference
Fukuoka, Saga, and Ishigaki cities 1970 in Saga; 1972 in Fukuoka; and 1972 in Ishigaki	*n* = 11 in Saga; *n* = 19 in Fukuoka; and *n* = 12 in Ishigaki	Adipose tissue, and breast milk	Mean PCBs	In Saga, PCBs in adipose tissue, mean, 2.6 (range, 0.5–5.3) ppm fat basis In Fukuoka, PCBs in breast milk, mean, 1.2 (range, 0.3–5.6) ppm fat basis In Ishigaki, PCBs in breast milk, mean, 0.4 (0.1–0.7) ppm fat basis	Masuda et al. (1974)
Japan 1973		Blood (*n* = 41), adipose tissue (*n* = 6), liver (*n* = 5)	PCB-118, 105, 153, 132, 156, 170, 180	Mean, 6.7 ppb in blood Mean, 2.5 ppm in adipose tissue Mean, 0.1 ppm in the liver	Masuda & Yoshimura (1982)
Fukuoka Prefecture 1981	59 Yusho patients aged > 40 years not receiving antihypertensive treatment	Blood/serum	Total PCBs	5.1 ± 2.3 ppb for men 6.4 ± 5.3 ppb for women	Akagi & Okumura (1985)
Japan 1988	259 patients (136 men and 123 women)	Blood/serum	Specific congeners not mentioned	Geometric means of PCBs and triglyceride: 3.84 (95% CI, 3.54–4.17) ppb and 114.3 (95% CI, 106.6–122.6) mg/dL, respectively Arithmetic mean of PCBs: 4.8 ppb (range, 0.6–320 ppb)	Hirota et al. (1993)
Japan September 1994 to November 1996	39 Yusho patients	Sebum, blood serum	PCB-77, PCB-126, PCB-169	428.1 pg/g lipid in sebum, and 390.7 pg/g lipid in blood	Iida et al. (1999)
Japan 2002	279 Yusho patients	Blood/serum	PCB-77, PCB-81, PCB-126, PCB-169	3.383 ± 2.765 (range 0.25–25.1) ppb	Uenotsuchi et al. (2005)
Japan 2002–3	279 Yusho patients in 2002 and 269 Yusho patients in 2003.	Blood/serum	PCB-77, PCB-81, PCB-126, PCB-169	125.0 pg-TEQ/g lipid	Todaka et al. (2005)
Fukuoka Prefecture 2002	279 Yusho patients and 92 Yusho-suspected persons	Blood/serum	PCBs 81, 77, 126, 169, 105, 114, 118, 123, 156, 157, 167, 189	Yusho patients: Non-*ortho* PCBs, 12.3 pg TEQ/g lipid; mono-*ortho* PCBs, 25.0 pg TEQ/g lipid Yusho-suspected persons: Non-*ortho* PCBs, 10.0 pg TEQ/g lipid; mono-*ortho* PCBs, 8.8 pg TEQ/g lipid	Todaka et al. (2007a)

Table 1.30 (continued)

Region Period	Subjects/ participants	Sample	PCB measured	Concentration (mean, median, range)	Reference
Fukuoka Prefecture 2002–5	242 Yusho patients, 74 Yusho-suspected persons in 2004, and 237 Yusho patients and 114 Yusho-suspected persons in 2005	Blood/serum	PCB-77, PCB-81, PCB-126, PCB-169	Yusho patients: 12.3, 11.7, 10.6, and 11.0 pg TEQ/g lipid in 2002, 2003, 2004, and 2005, respectively Yusho-suspected persons: 10.0, 8.3, 8.3, and 10.5 pg TEQ/g lipid in 2002, 2003, 2004, and 2005, respectively	Todaka et al. (2007b)
Japan 2001–3	359 Yusho patients	Blood/serum	PCBs 81, 77, 126, 169, 105, 114, 118, 123, 156, 157, 167, 189	3.14 ng/g blood	Imamura et al. (2007)
Fukuoka Prefecture 2004–7	242, 237, 300, and 96 Yusho patients from 2004 to 2007, respectively, and 74, 113, 125, and 148 Yusho-suspected persons, respectively	Blood/serum	Concentrations of 64 PCB congeners: TriCB-(28, 29), TetraCB-(44, 47/48, 49, 52/69, 56/60, 63, 66, 70, 71, 74), PentaCB-(85, 87, 92, 93/95/98, 99, 101, 105, 107/108, 110, 114, 117, 118, 123), HexaCB-(128, 130, 132, 134, 135, 137, 138, 139, 141, 146, 151, 153, 156, 157, 163/164, 167), HeptaCB-(170, 172, 177, 178, 179, 180, 181, 182/187, 183, 189, 191), OctaCB-(194, 195, 196/203, 198/201, 200, 202, 205), NonaCB-(206, 207, 208), DecaCB-209	Yusho patients: 2004, 645 (40–3032) ng/g lipid; 2005, 760 (40–4723) ng/g lipid; 2006, 667 (74–2432) ng/g lipid; and 2007, 510 (51–2252) ng/g lipid Yusho-suspected persons: 2004, 355 (20–1418) ng/g lipid; 2005, 490 (64–4055) ng/g lipid; 2006, 397 (18–1850) ng/g lipid; and 440 (19–2183) ng/g lipid	Todaka et al. (2009a, b)
Fukuoka Prefecture 2002–8	26 pairs of Yusho mothers and their children (19 mothers, 26 children)	Blood/serum	PCB-77, PCB-81, PCB-126, PCB-169	In the formula-fed group: 12.65 pg TEQ/g lipid for the mothers, and 3.85 pg TEQ/g lipid for the children In the breast-fed group: 10.64 pg TEQ/g lipid for the mothers; and 3.27 pg TEQ/g lipid for the children	Tsukimori et al. (2011)
Japan [Period not specified]	27 Yusho patients	Blood/serum	Hydroxylated PCBs (4-OH-CB109, 4-OH-CB146 + 3-OH-CB153, 4-OH-CB187, 4’-OH-CB172)	Total mean (range), 687 (95–1740) pg/g ww Range of the major hydroxylated PCB metabolites: 4-OH-CB187 (54–906 pg/g ww), 4-OH-CB146 +3-OH-CB153 (32–527 pg/g ww), 4-OH-CB109 (ND–229 pg/g ww) and 4’-OH-CB172 (ND–143 pg/g ww).	Tobiishi et al. (2011)
Japan 1968–2006 (the time of delivery of Yusho descendants)	64 Yusho mothers and 117 descendants (10 with FYD and 107 without FYD)	Maternal blood/serum	DL-PCBs (77, 81, 126, 169)	Black baby group, 57.6 pg TEQ/g lipid Non-black baby group, 31.8 pg TEQ/g lipid	Tsukimori et al. (2012)

Table 1.30 (continued)

Region Period	Subjects/ participants	Sample	PCB measured	Concentration (mean, median, range)	Reference
Umbilical cord					
Japan Yusho victims (1968–2000)	11 samples from 6 Yusho babies	Preserved umbilical cord	DL-PCBs (77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, 189)	6500 (130–11 000) pg/g in three designated patients 580 (130–1400) pg/g in eight suspected patients	Aozasa et al. (2008)
Fukuoka and Nagasaki prefectures Born 1970–3	7 babies born to Yusho mothers	Preserved umbilical cord	DL-PCBs (77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, 189)	0.3 pg TEQ/g dw	Nagayama et al. (2010)

dw, dry weight; FYD, fetal Yusho disease; PCB, polychlorinated biphenyl; PCDD, polychlorinated dibenzodioxins; PCDF, polychlorinated dibenzofurans; TEQ, toxic equivalent; ww, wet weight

Table 1.31 PCB concentrations in biological samples from Yucheng patients, Taichung County, Taiwan, China

Date of study	Patients	Sample	PCB measured	Concentration (mean, median, range)	Reference
1979–81	Children (*n* = 113)	Blood	PCBs	39 000 pg/g	Kashimoto *et al.* (1985)
1992	Mothers (*n* = 56)	Adipose tissue		2820 ± 300 (SE) ng/g	Guo *et al.* (1997)
1994–6	Adults (*n* = 42)	Sebum Blood	Dioxin-like PCBs	868.6 pg/g 714.4 pg/g	Iida *et al.* (1999)
1994	Adults (*n* = 414)	Serum	NR	1500 ng/g lipid (PCB-138 represented 29% of all measured PCBs)	Lung *et al.* (2005)
1994–5	Adults (*n* = 41)	Blood	NR	2468 ng/g lipid (13.3 ng/g sample)	Hsu *et al.* (2005)
				133 pg/g (PCB TEQ in men) 127 pg/g (PCB TEQ in women)	Lambert *et al.* (2006)

NR, not reported; PCB, polychlorinated biphenyl; SE, standard error; TEQ, toxic equivalent

from rural areas having the lowest levels of PCBs. PCB-138 and PCB-153 were found in the blood of mothers from all of the 61 sites studied at geometric mean concentrations of 3.56 and 3.2 ng/g lipid, respectively. Ahmed *et al.* (2002) reported the sum concentration of 29 congeners in blood from Egyptian women to be 61.9 ng/g. Weiss *et al.* (2006) reported concentrations of PCB-153 in infertile women in the United Republic of Tanzania to be 0.17 µg/kg. Sum PCB concentrations in serum samples from Bizerte, Tunisia, ranged from 37.5 to 284.6 ng/g lipid, with mean and median value of 136.1 ng/g lipid and 123.2 ng/g lipid, respectively. The PCB profile consisted mainly of persistent congeners such as PCB-138, PCB-153, and PCB-180 (82.7% of the sum of PCBs). PCB concentrations were significantly higher in men ($P < 0.05$) than in women (Ben Hassine *et al.*, 2014).

(b) Human milk

Due to its high fat content, human milk can accumulate large amounts of PCBs, thus making it an ideal matrix for the determination of concentrations of PCBs and other lipophilic compounds, and can be sampled using non-invasive techniques. In addition, human milk represents a good indicator of the body burden of lipophilic non-metabolized PCBs, since fat is mobilized for the production of milk during lactation. Animal studies and mass balance studies for humans have revealed that large amounts of PCBs can be eliminated through lactation (Lindell, 2012). Data are summarized in Table 1.32.

(i) Global assessment

The transfer of PCBs from mother to infants via breast milk is an important source of exposure, and several factors (including maternal residence, age, and parity) can potentially affect levels of contaminants in breast milk. Because of the importance of breastfeeding for infants, contamination of human milk is of specific public concern.

The Stockholm Convention on Persistent Organic Pollutants (POPs) is a guidance document, the objective of which is to document the effectiveness of the implementation of the obligations under the Convention. The World Health Organization (WHO) introduced worldwide measurement campaigns to determine the exposure of infants to dioxin-like PCBs (UNEP, 2012).

Table 1.32 PCB concentrations in human milk, by country

Country, population	PCBs (WHO-TEQ pg/g fat)			Sum indicator PCBs (ng/g fat)[a]			Reference
	Mean	Median	Range	Mean	Median	Range	
Europe, 1992–2003							
Czech Republic	–	15.24	14.32–28.5	–	502	496–1009	Van Leeuwen & Malisch (2002)
Germany	– DL-PCBs: 12.60 (in Duisburg) 6.31 (in Munich)	13.67 –	12.8–14.3 –	– –	220 –	188–238 –	Ulaszewska *et al.* (2011)
Greece	–	6.56 DL-PCBs	–	–	–	–	Costopoulou *et al.* (2006)
Italy (Milan, Rome, Venice)	– –	16.29 –	11.02–19.33 DL-PCBs, 6.02–19.21 pg WHO_{2005}-TEQ/g lipid	– –	253 –	195–323 –	Weiss *et al.* (2003), Ingelido *et al.* (2007), Abballe *et al.* (2008)
Norway	–	8.9	6.56–9.61	–	119	106–132	Polder *et al.* (2008)
Spain	–	–	–		(Sum of PCB-138, PCB-153, PCB-180) × 1.7: 1355 (in 1994) 653 (in 2000)	–	Cerná *et al.* (2008)
Spain	– DL-PCBs, 4.8 pg WHO_{2005} TEQ/g lipid	– –	– –	– –	241 –	162–467 –	Schuhmacher *et al.* (2009)
Sweden	–	9.71	–	–	146	–	Norén & Meironyté (2000)
North America							
Canada, *n* = 86 women eating fish from Lake Ontario	–	–	153 (50th percentile)	–	–	–	Stewart *et al.* (2003)
Western Canada, *n* = 47 women	–	38.20	–	–	–	–	Jarrell *et al.* (2005)
Canada, Northern Quebec, Inuit women from Nunavik	–	385.0 ± 1.9 SD	75.7–1915.8	–	–	–	Muckle *et al.* (2001)
USA, North Carolina, *n* = 331 women	–	77	9–708	–	–	–	Pan *et al.* (2010)

Table 1.32 (continued)

Country, population	PCBs (WHO-TEQ pg/g fat)			Sum indicator PCBs (ng/g fat)[a]			Reference
	Mean	Median	Range	Mean	Median	Range	
South and Central America							
Brazil, Rio de Janeiro, $n = 40$ mothers	9.7	–	150	–	–	–	Paumgartten *et al.* (2000)
Africa							
Ghana, $n = 67$ mothers	62	–	15–160	–	–	–	Asante *et al.* (2011)
South Africa, Limpopo Provence	10	–	–	–	–	–	Darnerud *et al.* (2011)
Tunisia	180	–	–	–	–	–	Ennaceur *et al.* (2008)
Zimbabwe	26	–	–	–	–	–	Chikuni *et al.* (1997)
Asia							
Japan	1.30×10^3 (in 1972) 1.51×10^3 (in 1974) 0.20×10^3 (in 1998)						Environment Agency of Japan (1999)
China, $n = 1237$				–	–	–	Li *et al.* (2009)
Total TEQ	5.42	5.11	Upper bound, 2.59–9.92	–	–	–	
Estimated dietary intake of PCDD/PCDF + DL-PCBs in infants	28.0 pg TEQ/kg bw per day	–	14.2–48.6 pg TEQ/kg bw per day	–	–	–	

[a] Indicator PCBs are PCBs 28, 52, 101, 138, 153 and 180

DL-PCB, dioxin-like polychlorinated biphenyl; PCB, polychlorinated biphenyl; TEQ, toxic equivalent

The evaluation of the Stockholm Convention has been applied (with slight changes) for five rounds of the UNEP/WHO survey. Often, human milk from primiparae mothers (for detail, see UNEP, 2012) is preferred to human blood, since sampling is non-invasive and PCBs are easier to detect (due to the higher lipid content of milk). It should be noted that for global assessment, the concentrations of dioxin-like PCBs (DL-PCBs) on a TEQ basis for the last three rounds of the UNEP/WHO survey on mothers' milk may be lower by 30% (range, 2–60%) if WHO toxic equivalency factors (TEFs) for 2005 (WHO_{2005}-TEF) are applied, rather than those for 1998 (WHO_{1998}-TEF). International chemical assessments report that the average concentration of PCBs in human milk fat ranges from 0.5 to 4 µg/g (IPCS, 2003). For the sum of PCB_6, the median is between 10.8–30.7 ng/g lipid, and maxima are between 37.1–65.8 ng/g lipid. Overall, the UNEP/WHO survey showed a correlation between maternal age and concentrations of DL-PCBs in breast milk, and lower concentrations of PCBs in breast milk of multiparous women when compared with primiparous women.

(ii) Americas

The mean concentration of PCBs in whole breast milk in Canadian women steadily increased from 6 µg/kg in 1970 to 12 µg/kg in 1975, and to 26 µg/kg in 1982, before declining to 6 µg/kg in 1986 (IPCS, 2003).

Recent data on concentrations of PCDD/PCDFs and DL-PCBs in human milk from South America were reported only for Brazil (Paumgartten *et al.*, 2000).

(iii) Europe

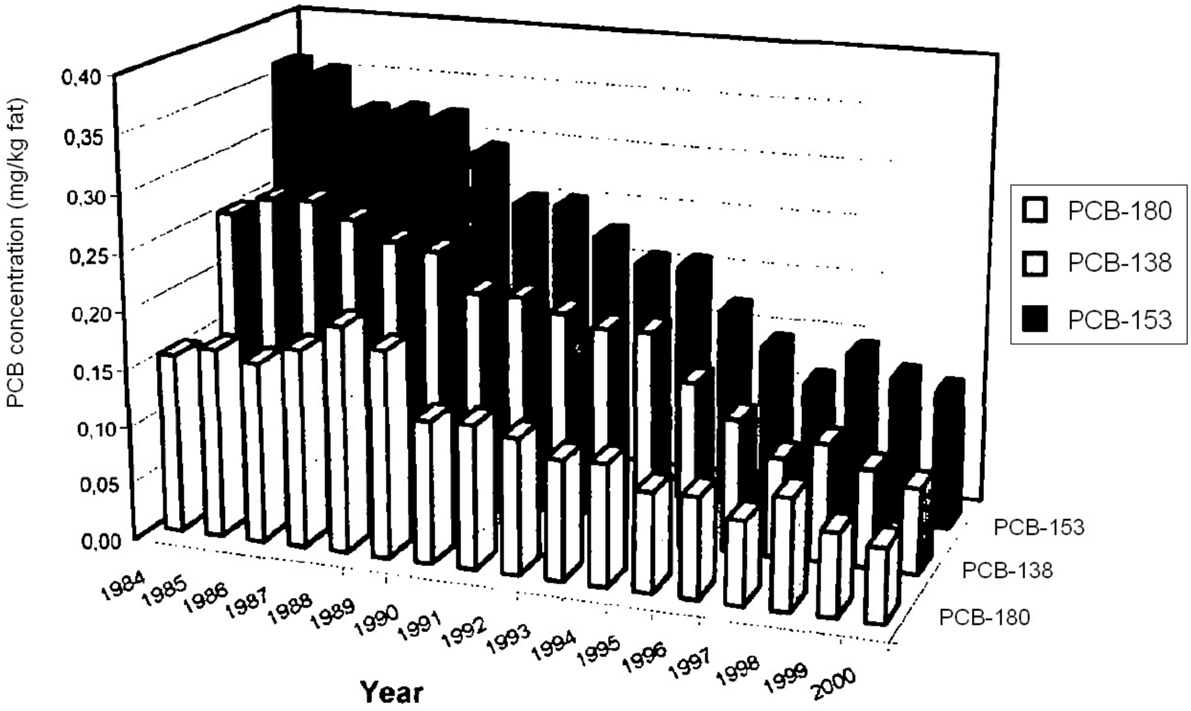
In Europe, concentrations of DL-PCBs (on a TEQ basis) and PCB indicators in human milk are considerably higher than in other regions of the world, a legacy from past exposures. For the sum of PCB_6, the median of 115.3 ng/g lipid is between 3.8 times and 10.7 times higher than in other regions of the world, and maximum concentrations are up to 14.9 times higher (IPCS, 2003). However, a WHO survey identified a decrease in WHO-TEQ PCDD/PCDF and PCB concentrations in human milk over the last decade (Van Leeuwen *et al.*, 2002). It was assumed that this decrease was the result of the ban of PCB use in open systems, and the strict regulations on the use of PCBs and on their disposal in closed systems.

Norén & Meironyté (referenced in IPCS, 2003) reported a steady decrease (from 910 to 324 ng/g lipid) in total PCB concentrations in the breast milk of Swedish women between 1967 and 1997. A declining trend could be observed for the sum of the PCB_6 in Germany, and mean values for the congeners PCB-138, PCB-153 and PCB-180 were approximately 60–70% lower in 2000 than in 1984 (Fürst, 2001; Fig. 1.8). An approximately 74% decrease in DL-PCB concentrations during the last decade was reported in Italy (Di Domenico & Turrio Baldassarri, 1990; Weiss *et al.*, 2003; Abballe *et al.*, 2008). Analyses of milk samples from the Czech Republic also revealed a decline in median concentrations between 1994 and 2000, the strongest decrease being observed between 1994 and 1997 (Cerná *et al.*, 2008). Nevertheless, it should be noted that concentrations in areas with heavy contamination did not show a significant decline in exposure over the past 10 years.

(iv) Asia

In Japan, a time-trend study showed that average PCB concentrations in human milk increased from 1.3 ng/g in 1972 to a peak of 1.5 ng/g in 1974, and then decreased by about 13% in 1998 (Environment Agency of Japan, 1999). In contrast, daily intake of PCBs from breast milk was estimated to decrease from 22.3 µg/g to 0.31 µg/g during this same period. [This trend reflects a change in PCB concentrations in food, due to both a decrease in contamination and more dependence on imported foods, which were less

Fig. 1.8 PCB concentrations in human milk in Germany, 1984–2000



From Fürst (2001)
PCB, polychlorinated biphenyl

contaminated than domestic foods (IPCS, 2003), and is consistent with the observed decline in PCB concentrations in the environment and in human tissues.]

In China, a national investigation of individuals in 12 provinces representing approximately 50% of the total Chinese population reported PCDD/PCDF-TEQ and total-TEQ in human milk from rural areas to be lower than those from urban areas (Li *et al.*, 2009). Positive correlations were found between total-TEQ in human milk and the consumption of aquatic food and meat.

PCB levels in breast milk samples from women in Asia are summarized in Table 1.29 and Table 1.30.

(c) Adipose tissue

(i) North America

Lordo *et al.* (1996) reported PCB concentrations (sum of tetra- to octochlorobiphenyls) in pooled adipose tissue to be 672 ng/g in 1986, compared with 407 ng/g in 1982, and 508 ng/g in 1984. Stellman *et al.* (1998) reported a total PCB concentration of 267 ng/g in breast adipose tissue of healthy women from Long Island, New York. An approximation of Aroclor 1260 [summed concentrations of PCB-138 and PCB-153 multiplied by 5.2] measured in breast adipose tissue, was reported to be 870 ng/g (Aronson *et al.*, 2000). Muscat *et al.* (2003) measured PCB concentrations (sum of 14 congeners) in breast adipose tissue in women without metastatic breast cancer to be 361 ± 235.9 ng/g and 395.4 ± 279.3 ng/g, in women who did not have recurrence and women who did have recurrence, respectively.

(ii) Europe

The results of a study conducted in 1993–94 suggested that concentrations of PCBs in adipose tissue are the best indicator of long-term exposure or of total body burden of PCBs, compared with human milk or blood (Kocan *et al.*, 1994).

PCB concentrations in adipose tissue of the general population in industrialized countries vary very widely, ranging from < 1000 to 5000 ng/g fat (Falandysz *et al.*, 1994; Holoubek *et al.*, 1995, 2001b). In a comparative study in Europe (Van Bavel *et al.*, 2003), PCB concentrations in the population in Sweden were one third (mean, 661.9 ng/g fat; range, 247.2–1651.2 ng/g fat; Σ37 PCBs) of those in the Hungarian samples.

(iii) South and Central America

Breast adipose tissue in 76 women from an agricultural region of north-eastern Argentina contained eight PCBs at very low levels (only 1.3% above detection limits), but high levels of *p*,*p*-dichlorodiphenyldichloroethane (DDE) and other pesticides (Muñoz-de-Toro *et al.*, 2006).

The sum of four PCB congeners in children from Nicaragua was 530 ng/g lipid weight (2.0 ng/g wet weight) in those living and working near a waste-disposal site and eating fish from contaminated Lake Managua, 230 ng/g lipid weight (0.9 ng/g wet weight) in those living nearby but not working at the waste site and not eating fish, and 160 ng/g lipid weight (0.6 ng/g wet weight) in those living at a distance from the waste site and not eating fish (Cuadra *et al.*, 2006).

(iv) Asia

PCB concentrations in adipose tissue were reported from Yusho and Yucheng patients (see Table 1.30 and Table 1.31).

(v) Adipose versus serum measurements

Arrebola *et al.* (2012a, b) measured concentrations of three PCB congeners in serum and adipose tissue in adults from Bolivia. PCB-138 had median concentrations of 0.2 ng/mL in serum [33.7 ng/g lipid] and 84 ng/g in adipose tissue [105 ng/g lipid]. The median values for PCB-153 was 0.3 ng/mL in serum [59.0 ng/g lipid], and 52.7 ng/g in adipose tissue [65.8 ng/g lipid]. PCB-180 had median values of 0.1 ng/mL in serum [26.7 ng/g lipid] and 32.8 ng/g in adipose tissue [41.0 ng/g lipid].

(d) Umbilical cord blood, placenta, and fetal tissue

(i) North America

Stewart *et al.* (2000) reported cord blood PCB concentrations from women living along Lake Ontario and eating contaminated fish. The average cord blood PCB concentration was 0.525 ng/g wet weight [25th percentile, 0.174 ng/g wet weight; 75th percentile, 1.11 ng/g wet weight]. In plasma from umbilical cord in Inuit women from northern Canada, the geometric mean for the sum of 14 PCB congeners was 279.9 ng/g lipid (range, 70.8–1420.1 ng/g lipid) (Muckle *et al.* 2001). Dallaire *et al.* (2003) reported changes in concentrations in umbilical cord blood in this population over time, and found a 7.9% annual decrease between 1994 and 2000. Choi *et al.* (2006) measured 51 congeners in cord blood from women living near a PCB-contaminated site in Massachusetts, and reported a geometric mean of 0.40 ng/g (range, 0.068–18.14), with no consistent relationship with residential distance from the waste site. Consumption of meat and local dairy products (but not fish) were associated with higher cord blood PCB concentrations.

In women from New York state, Schecter *et al.* (1998) reported the concentration of three dioxin-like PCBs to be 18.2 pg/g lipid in placenta, giving a TEQ of 1.05. The concentrations of 14 single PCB congeners in plasma from Inuit women from Nunavik and southern Quebec were highly correlated with those in placenta (Pearson's $r = 0.77–0.97$; $P < 0.001$), and concentrations in Inuit women were on average four times higher than in women from southern Quebec (Pereg *et al.*, 2002). Doucet *et al.* (2009) analysed placenta from Canadian women having elective abortions in 1998–2006 and reported annual average total PCB concentrations ranging from 7 to 70 ng/g lipid, with no clear time trend.

(ii) Europe

Koopman-Esseboom *et al.* (1994) used the concentrations of four congeners (PCB-118, PCB-138, PCB-153, and PCB-180), as measured in umbilical cord blood and in breast milk, as indicators of exposure of the developing fetus and breastfed infant. For these congeners, the correlation coefficients between maternal plasma, cord plasma and human milk were highly significant.

Soechitram *et al.* (2004) analysed PCBs (PCB-118, PCB-138, PCB-146, PCB-153, PCB-156, PCB-180) and hydroxylated metabolites of PCBs (PCB-107, PCB-136, PCB-146, PCB-153, PCB-172, PCB-187) in samples of maternal plasma and corresponding cord blood in the Netherlands. The calculated ratio for cord versus maternal blood was 1.28 ± 0.56 for PCBs and 2.11 ± 1.33 for hydroxylated PCBs, expressed per gram of lipid. A significant correlation between the respective maternal and cord concentrations for both PCBs and hydroxylated PCBs was found. The results indicated that approximately 50% and 30% of hydroxylated PCBs and PCBs, respectively, was transferred across the placenta to the fetus.

(e) Hair

(i) North America

Altshul *et al.* (2004) reported median PCB concentrations (sum of 57 congeners) in hair of 2640 ng/g fat (range, 1180–3620 ng/g fat) in a population of students in Boston, USA. Washing hair with shampoo decreased concentrations of PCBs by 25–33% on average, and up to 62% for less chlorinated congeners. [The Working Group considered that the analytical method was reliable and reproducible.] The concentrations of PCBs in hair were higher than in serum. Correlation between concentrations in hair and blood was moderate for the more persistent PCB congeners, with no or little correlation for the other congeners.

(ii) Europe

Covaci *et al.* (2002b) assessed PCB exposure in hair samples from Greece, Romania, and Belgium. Mean PCB concentrations in samples from Belgium were up to 14 ng/g hair, while concentrations in samples from Greece were about three times lower. Similar ratios of PCB-153 over total PCBs were found for all three countries.

(iii) Asia

One study measured PCB concentrations in the hair of pregnant women in various cities in the Islamic Republic of Iran (see Table 1.29).

1.5 Occupational exposure to PCBs

In 1978, an estimated 12 000 persons in the USA were exposed occupationally to PCBs (Lloyd *et al.*, 1976; NIOSH, 1977). Since the previous IARC evaluations of PCBs (IARC, 1978, 1987), occupational exposures to PCBs have changed, since most industrial countries have banned or partially banned their use. Nevertheless, the earlier references cited previously have been incorporated in the present monograph.

Earlier occupational exposures to PCBs occurred during PCB manufacture, capacitor and transformer manufacture and repair, production of carbonless copy paper, and accidental releases from these processes. More recent occupational exposures to PCB usually occur through PCB emissions via waste incineration, fires, and waste recycling.

1.5.1 PCB manufacture

The few available studies of occupational exposure among PCB-manufacturing workers have been performed in France, Italy, Japan, Poland and the USA. Workers' exposures during PCB manufacture have been mentioned since 1936 (Jones & Alden, 1936); air PCB concentrations ranged from 26 to 163 μg/m³. Evidence of extensive exposures was available from two larger clinical studies in Slovakia in which workers' blood concentrations were measured at 1160–9600 ng/g lipid (n = 242) (Langer *et al.*, 1997), and 4905–6540 ng/g lipid (n = 240) (Langer *et al.*, 2002). These workers had been employed at the plant for at least 5 years, but no information was given regarding the type of activities that they had performed.

1.5.2 Capacitor manufacture

Before PCBs were banned, capacitor manufacturers filled (impregnated) casings with wound paper and foil/plastic with the PCB-containing oil before the top was fastened (crimped, sealed, soldered). PCB exposure (probably via the dermal route) occurred during filling: the capacitors were either flood-filled or manually filled, resulting in spills and worker exposures. The brand of PCB oil used differed geographically (Aroclors were used in the USA, Pyralene/Phenochlor in Sweden and Italy) and temporally (the percentage chlorination was reduced, e.g there was a switch from Aroclor 1254, with 54% chlorination, to Aroclor 1242, with 42% chlorination.

Other chemical exposures in capacitor manufacturing were possible, such as from other impregnation oils (e.g. mineral oils), degreasing agents such as trichloroethylene (Brown & Jones, 1981; Bertazzi *et al.*, 1987), dibenzofurans (Gustavsson *et al.*, 1986), chlorinated naphthalenes, lead solder, epoxies, and methyl ethyl ketone (MEK) (Mallin *et al.*, 2004; Persky *et al.*, 2012). Ageing capacitors can from time to time explode [due to the physical stress of metal ageing], thus further exposing workers.

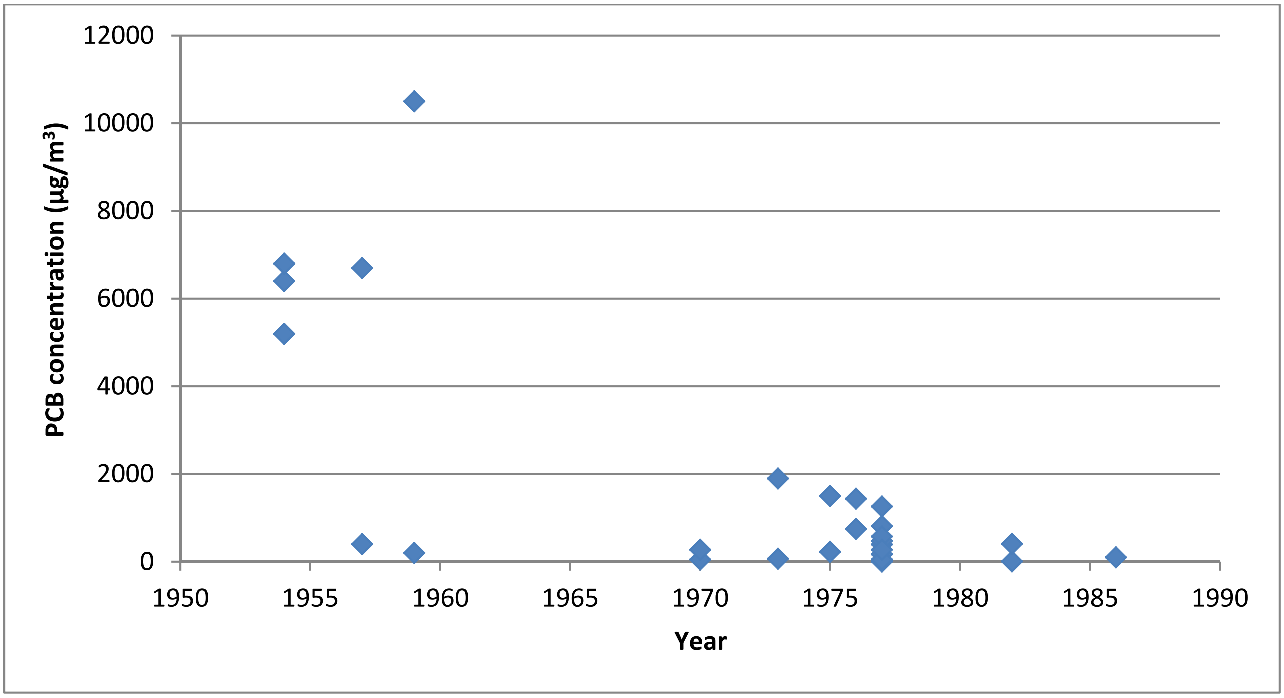
Fig. 1.9 shows ranges of air PCB concentrations in capacitor-manufacturing sites (Ouw *et al.*, 1976; Brown & Jones, 1981; Bertazzi *et al.*, 1982; Fischbein *et al.*, 1982; Lawton *et al.*, 1985; Gustavsson *et al.*, 1986). The earlier concentrations (measured in the 1950s–1960s) were highest (Bertazzi *et al.*, 1982), and decreased in later years.

In the 1950s, maximum PCB concentrations in workroom air in several plants in the USA (Massachusetts) were reported to be 200–10 500 μg/m³ (Elkins, 1959). No details on the number of plants surveyed, nor the number of samples collected, or the work performed in the plants were given, but four different jobs were surveyed: for impregnating with PCBs, average air concentrations ranged from 200 to 5800 μg/m³; for soldering, 800 μg/m³; mixing oil, 600 μg/m³, and regulator filling, 100 μg/m³. No toxic effects were noted at these concentrations; however, it was noted that air PCB concentrations of > 10 000 μg/m³ were "unbearably irritating." This is contrary to a report in a Japanese capacitor-manufacturing plant where a dermatitis outbreak occurred when air PCB concentrations reached 100 μg/m³ (Meigs *et al.*, 1954). Air PCB concentrations between 1953 and 1957 in a Japanese capacitor-manufacturing factory ranged from 400 to 6700 μg/m³ (NIEHS, 1976).

Ouw *et al.* (1976) reported that workers in the electrical industry in Australia were exposed to Aroclor 1242 at air concentrations of 320–2220 μg/m³, with a mean of 1270 μg/m³; and were found to have PCB blood concentrations of approximately 0.4 g/kg bw. Contact with PCBs was primarily via the skin.

Brown & Jones (1981) measured air concentrations of Aroclor 1016 in two plants in the USA (New York and Massachusetts) plants in 1977. The time-weighted averages (TWA) were different for the two plants, with air concentrations at the New York plant being lower than at the Massachusetts

Fig. 1.9 Air PCB concentrations in capacitor manufacturing plants (µg/m³) by year



Compiled by the Working Group using data from Ouw *et al.* (1976), Brown & Jones (1981), Bertazzi *et al.* (1982), Fischbein *et al.* (1982), Lawton *et al.*(1985) and Gustavsson *et al.*(1986)
PCB, polychlorinated biphenyl

plant. For the New York plant, PCB concentrations in personal air samples ranged from 24 to 393 µg/m³ (*n* = 28), and in area air samples from 3 to 476 µg/m³ (*n* = 19). For the Massachusetts plant, PCB concentrations in personal air samples ranged from 170 to 1260 µg/m³ (*n* = 29), and in area air samples from 50 to 810 µg/m³ (*n* = 25). Air PCB concentrations (TWA) were extremely high during capacitor impregnation (New York: 160 µg/m³, Massachusetts: 850 µg/m³), degreasing (Massachusetts; 1260 µg/m³), and sealing/soldering (New York: 393 µg/m³, Massachusetts: 720 and 1060 µg/m³). Capacitors that failed were sent for repair where they were re-opened and manually drained. Repair workers' PCB exposures were measured as 298 µg/m³ (recovery), and 50 µg/m³ (repair) in the New York plant. [These workers would also have had extensive dermal exposures, which were not assessed.]

Eight studies have reported PCB concentrations in workers' blood in Australia, Finland, Italy, Germany, and the USA (Karppanen & Kolho, 1972; Ouw *et al.*, 1976; Maroni *et al.*, 1981a; Bertazzi *et al.*, 1982; Acquavella *et al.*, 1986; Wolff *et al.*, 1992; Kannan *et al.*, 1994; Seegal *et al.*, 2011; Persky *et al.*, 2012). The reporting of PCB blood concentrations was not uniform, which hindered comparison across studies.

Karppanen & Kolho (1972) compared blood PCB concentrations in workers in a capacitor factory in Finland where Aroclor 1242 had been used as the impregnating fluid: the groups comprised laboratory workers handling PCBs (*n* = 6), impregnation workers (*n* = 11) employed for 4 years, and a control group (*n* = 9) that had never been professionally exposed to PCBs. Blood PCB concentrations were approximately 50 times greater in impregnation workers (0.07–1.9 µg/g) than in the control group (0.003–0.012 µg/g), and were also higher in laboratory workers (0.036–0.062 µg/g) than in the controls. The pattern of PCB congeners in the exposed workers differed markedly from that of the PCBs actually used. More highly chlorinated PCBs persisted in the blood, while the less chlorinated PCBs contained in Aroclor 1242 had been eliminated from the body. [Consequently, the total PCB intake must have been higher than that reflected by the levels detected in blood.]

Total serum PCB concentrations have been reported to be 18.2 ppb (Acquavella *et al.*, 1986) in a clinical survey among 205 workers at a capacitor-manufacturing plant in the USA. This mean value represented workers (*n* = 205) with (39%) and without (61%) potential for PCB exposure in their jobs. [The Working Group noted that PCB concentrations were not reported separately for workers with and without occupational PCB exposure.] Log serum PCB concentrations were found to be significantly correlated with duration of employment, age, cumulative occupational exposure, and fish and wine consumption, as confirmed by multiple linear regression.

In a study of mortality in Italy (Bertazzi *et al.*, 1982), workers in the autoclave room were exposed to air PCB concentrations of 5200–6800 µg/m^3 in 1954 (*n* = 3), and 48–275 µg/m^3 in 1977 (*n* = 9). Eighteen workplace surface-wipe samples showed extensive PCB contamination (0.2–159 µg/cm^2), as did nine hand-wipe samples (0.3–9.2 µg/cm^2). Workers' serum PCB concentrations were reported by type of PCBs: for highly chlorinated PCBs (54% chlorination) (*n* = 67), the mean was 230.5 ppb (SD, 174.5), while for less chlorinated PCBs (42% chlorination) (*n* = 67) the mean was 114.1 ppb (SD, 79.6). In a later study (Bertazzi *et al.*, 1987), the corresponding values were 202.8 ppb (SD, 111.7; *n* = 37) and 42.9 ppb (SD, 34.7; *n* = 37), respectively.

Wolff *et al.* (1992) studied PCB blood concentrations in capacitor workers in 1976 and 1979 in the USA. For the first sampling year, the mean concentration of the less chlorinated PCBs (di-, tri-, and tetrachlorobiphenyls) was 55 ng/mL (range, 6–2257 ng/mL), and for the latter year was 41 ng/mL (range, 6–350 ng/mL). Mean concentrations of highly chlorinated PCBs were 10 ng/mL (range, 1–308 ng/mL) in 1976, and 13 ng/mL (range, 2–350 ng/mL) in 1979. These capacitor workers (*n* = 60) were also surveyed by the National Institute for Occupational Health and Safety in 1977 (NIOSH, 1977), when the following blood PCB concentrations were reported as follows: less chlorinated PCBs (quantified as Aroclor 1242), 2–3300 ppb (ng/mL), and more highly chlorinated PCBs (quantified as Aroclor 1254), 5–250 ng/mL.

About 30 years later (in 2003–2006), Seegal *et al.* (2011) measured blood concentrations of individual PCB congeners in some of these former capacitor workers, and found that concentrations had dropped statistically significantly: mean concentration of less chlorinated PCBs (PCBs 28, 56, 66, 74, 99, 101) was 2.84 ng/g or 0.45 µg/g lipid in men, and 2.29 ng/g or 0.34 µg/g in women; mean concentration of highly chlorinated PCBs (PCBs 105, 118, 138, 146, 153, 156, 167, 170, 172, 174, 177, 178, 180, 183, 187, 199, 203) was 4.09 ng/g or 0.65 µg/g lipid in men, and 3.21 ng/g or 0.47 µg/g lipid in women; and total PCB concentration was 7.47 ng/g or 1.19 µg/g lipid in men, and 5.81 ng/g or 0.86 µg/g lipid in women.

Maroni *et al.* (1981a) carried out a study in two Italian electrical-capacitor manufacturing plants using PCBs as a dielectric fluid. Plant A produced electric capacitors filled with a mixture of mineral oils and PCBs. PCBs with 54% chlorination were used from 1949 to 1965, and subsequently replaced with Pyralene 3010 with 42% chlorination. The power-capacitor casings were filled with PCBs in autoclaves, and were manually removed when cooled from 70 °C to 40 °C before they were welded, tested, and finished externally. Electric "filters" (small capacitor systems used in electrical household appliances) were impregnated with PCBs. Plant B performed short-circuit testing of high-power capacitors filled with Apirolio, a PCB mixture with 42% chlorination. Stress-testing the capacitors often included explosions. Airborne PCBs were mainly trichlorobiphenyls and concentrations ranged from 48 µg/m^3 (filter operations) to 275 µg/m^3 (power-capacitor manufacturing). Surface-wipe samples showed both tri- and pentachlorobiphenyl mixtures, with the highest amounts being found on the capacitor basket rolling carrier: trichlorobiphenyls, 127 mg; and pentachlorobiphenyls, 15 mg. Plant A employed 67 workers (40 women and 27 men): 48 were currently employed in the capacitor-manufacturing departments, 16 had been employed there for at least 6 months before the beginning of the study, and 3 had always been employed in other non-manufacturing departments without direct exposure to PCBs. PCB recovery from the palms of the hands of power-capacitor workers (plant A) showed total PCB (tri- and pentachlorinated biphenyls) skin-surface concentrations to be 4–28 µg/cm^2. Mean (± SD) blood PCB concentrations differed between current (377 ± 190 µg/kg) and past exposed workers (292 ± 161 µg/kg); workers with occasional exposure had the lowest mean total PCB exposures (110 ± 31 µg/kg). Blood PCB concentrations by job performed were highest for welders (1259 µg/kg), followed by impregnation workers (556 ± 337 µg/kg), assembly of capacitors (406 ± 173 µg/kg), and finally assembly of filters (246 ± 130 µg/kg). The blood PCB concentrations were not correlated with duration of exposure, but with the percentage ratio of hours per year spent with direct exposure to PCBs. Plant B included 13 workers (all men) exposed to PCBs during handling of the capacitors contaminated with Apirolio, dispersed from explosions sometimes caused by stress-testing. Blood PCB concentrations in currently exposed workers in plant B (200 ± 146 µg/kg) were between occasionally exposed (110 ± 31 µg/kg) and past exposed workers (292 ± 161 µg/kg) in plant A. Although the PCB mixture used in both plants had a chlorine content of 42%, the workers differed in their ratio of penta- to trichlorobiphenyls; plant A workers had higher concentrations of pentachlorobiphenyls than of trichlorobiphenyls, while the reverse was true in plant B workers. This difference was attributed to the heavy past exposure to highly chlorinated PCBs used until 1965 in plant A. Workers with abnormal liver findings (*n* = 16) had twice the concentrations of tri- (215 ± 95 µg/kg) and pentachlorobiphenyls (308 ± 306 µg/kg) compared with workers (*n* = 64) without abnormal liver findings (tri- and pentachlorobiphenyl concentrations were 92 ± 64 and 176 ± 108 µg/kg, respectively) (Maroni *et al.*, 1981b). Duration of exposure did not explain this observed difference.

One German capacitor-manufacturing worker was reported to have a blood PCB-169 concentration of 11 ng/g (Kannan *et al.*, 1994).

After a capacitor explosion at a Finnish paper mill, workers' (*n* = 15) blood PCB concentrations were 3.5–48.3 µg/L (Luotamo *et al.*, 1984). [These levels were much lower than during capacitor manufacturing itself.]

In a recent cross-sectional study, Persky *et al.* (2012) reported blood PCB concentrations separately for diseased (having diabetes) and non-diseased (without diabetes) workers. In diseased workers, the concentrations were: DL-PCBs, 2.5 ng/g; NDL-PCBs, 17.0 ng/g; estrogenic PCBs [PCB-52, 99, 101, 110, 153], 3.6 ng/g; anti-estrogenic PCBs [PCB-105, PCB-156], 3.6 ng/g; and PCB-74, 4.9; PCB-99, 1.0 ng/g; PCB-118, 1.4 ng/g; PCB-138, 2.5 ng/g; PCB-146, 0.4 ng/g; PCB-153, 2.8 ng/g; PCB-156, 0.6 ng/g; PCB-170, 0.7 ng/g; PCB-180, 1.1 ng/g; PCB-187, 0.3 ng/g; PCB-194, 0.2 ng/g; PCB-201, 0.2 ng/g; PCB-203, 0.2 ng/g; and PCB-206, 0.1 ng/g. In non-diseased workers, the concentrations were: DL-PCBs, 0.4 ng/g; NDL-PCBs, 4.3 ng/g; estrogenic PCBs, 1.0 ng/g; anti-estrogenic PCBs, 0.1 ng/g; PCB-74, ng/g; 0.8, PCB-99; 0.3 ng/g; PCB-118, 0.2 ng/g; PCB-138, 0.6 ng/g; PCB-146, 0.1 ng/g; PCB-153, 0.8 ng/g; PCB-156, 0.1 ng/g; PCB-170, 0.2 ng/g; PCB-180, 0.4 ng/g; PCB-187, 0.1 ng/g; PCB-194, 0.1 ng/g; PCB-201, 0.1 ng/g; PCB-203, 0.1 ng/g; and PCB-206, 0.04 ng/g.

1.5.3 Transformer manufacture and repair

Transformer manufacture was very similar to capacitor manufacture. Transformers were filled with PCBs, but the impregnation fluid was usually diluted with other chlorinated solvents (e.g. trichlorobenzene; Greenland *et al.*, 1994), and sold under different names such as Askarel (Inerteen), Pyranol, Chlophen, Apirolio, and Derol (Kerns, 1975; Lees *et al.*, 1987; Emmett *et al.*, 1988; Kalina *et al.*, 1991; Greenland *et al.*, 1994; Yassi *et al.*, 1994; Altenkirch *et al.*, 1996; Loomis *et al.*, 1997; Caironi *et al.*, 2005).

Although air concentrations from transformer manufacture were not available, two studies reported air PCB concentrations during transformer repair in two different USA plants (Lees *et al.*, 1987; Emmett *et al.*, 1988). Work activities were sampling and testing transformer fluids for dielectric properties, topping up transformers when oil levels were low, clean-up of any spills or leaks, repair of transformers by drainage of transformer oil to replace parts, and periodic filtering of the transformer oil to upgrade its dielectric properties. Ranges of air PCB concentrations for several job tasks were reported: repair and clean-up ($n = 3$), 43.1–60.0 µg/m^3 and TWA, 16.7–24.0 µg/m^3; clean-up of PCB leakage ($n = 3$), 0.1–3.1 µg/m^3 and TWA, 0.01–0.4 µg/m^3; and secondary oil leak repair and clean-up ($n = 15$), 2.1–17.1 and TWA, 0.7–12.4 µg/m^3 (Emmett *et al.*, 1988). Other job tasks for which concentrations were reported were draining and pumping transformer oil ($n = 9$), 1.1 µg/m^3; transformer repair ($n = 15$), 1.2 µg/m^3; network repair ($n = 6$), 0.5 µg/m^3; topping-up transformer oil ($n = 3$), 0.5 µg/m^3; explosion spill clean-up ($n = 16$), 1.7 µg/m^3; and filtering transformer oil ($n = 6$), 6.1 µg/m^3 (Lees *et al.*, 1987). Transformer-repair activities included handling transformer parts that were wet with transformer fluid without protective gloves, resulting in extensive dermal exposure. In one case, a maintenance transformer worker involved in cleaning up transformer fluid spills daily had a plasma PCB concentration of 250 µg/L (Tröster *et al.*, 1991). [This value is comparable to highly exposed capacitor-manufacturing workers.]

1.5.4 Waste incineration of PCB materials

Ten studies from seven countries (USA, Germany, Spain, Japan, the Republic of Korea, Belgium, and Poland) reported PCB exposures during waste incineration of PCB materials (Colucci *et al.*, 1973; Angerer *et al.*, 1992; Wrbitzky *et al.*, 1995; Gonzalez *et al.*, 2000; Kitamura *et al.*, 2000; Domingo *et al.*, 2001; Raemdonck *et al.*, 2006; Mari *et al.*, 2009; Park *et al.*, 2009). The PCB congeners frequently reported in this industry were PCB-28, PCB-138, PCB-153, and PCB-180. The distribution of PCB congeners in plasma depended on the type of waste material, the furnace (age and type), and the workers' activities. During burning of waste in a waste-incinerating plant, heat from combustion gases is recuperated in a cauldron to produce electricity. PCBs are, together with dioxins, produced by synthesis from organic substances and chorine during this and subsequent cooling-down processes. PCBs (with dioxins) precipitate onto particulate matter (fly ash) and are trapped in the filter (Raemdonck *et al.*, 2006).

Exposed refuse workers ($n = 37$) in the USA had a median plasma PCB concentration of 2.6 ppb (maximum, 14.1 ppb) (Colucci *et al.*, 1973). [No methods were reported.] Hazardous-waste workers ($n = 53$) in Germany had a mean plasma PCB concentration of 6.33 µg/L calculated as the sum of PCB congeners PCB-138 (1.86 µg/L) + PCB-153 (2.83 µg/L) + PCB-180 (1.65 µg/L), which was not significantly different from controls (6.22 µg/L) in the same study (Angerer *et al.*, 1992).

Another study in Germany (Wrbitzky *et al.*, 1995) reported mean plasma PCB concentrations in waste-incineration workers (total PCBs, 3.10 µg/L; range, 1.59–6.89 µg/L) that were approximately half those in the previously described study (Angerer *et al.*, 1992). The total PCBs were the sum of the same PCB congeners as previously reported: PCB-138, 0.95 µg/L (range, 0.49–2.60 µg/L); PCB-153, 1.38 µg/L (range, 0.97–3.10 µg/L); PCB-180, 0.79 µg/L (range, 0.32–1.63 µg/L). Concentrations of PCB-28, PCB-52, and PCB-101 were below the limit of detection (< 0.2 µg/L). These workers operated the incinerator, control panels, electronics, waste gas and transfer stations, and maintained and cleaned boilers and furnaces. Workers employed in the central laboratory, incoming control and sampling, chemical-sorting station, waste-water purification, and mechanical workshop among other periphery jobs had blood PCB concentrations similar to those of workers in management. Concentrations in exposed workers were: total PCBs, 2.82 µg/L; range, 1.21–7.03 µg/L, and this was the sum of PCB-138 (0.87 µg/L; range, 0.24–2.35 µg/L), PCB-153 (1.22 µg/L; range, 0.27–2.83 µg/L), and PCB-180 (0.72 µg/L; range, 0.32–3.48 µg/L). Concentrations in workers in management were: total PCBs, 3.19 µg/L (1.59–7.53 µg/L); PCB-138, 0.98 µg/L (0.49–1.98 µg/L); PCB-153, 1.42 µg/L (0.67–3.37 µg/L); PCB-180, 0.80 µg/L (0.43–2.18 µg/L). [Of the six PCB congeners analysed, only these three were detected.]

Waste-incinerator workers in a plant in Spain were reported to have mean total PCB concentrations of 1.47 µg/L: as in the German study, this was the sum of congeners PCB-138 (0.36 µg/L) + PCB-153 (0.49 µg/L) + PCB-180 (0.57 µg/L) (Gonzalez *et al.*, 2000). Congeners PCB-28 and PCB-52 were not detected, and the concentration of PCB-101 was very low (0.02 µg/L). In another study in Spain, congener-specific concentrations were reported as means (and geometric means): PCB-28, 18.5 (12.9) µg/kg lipid; PCB-52, 10.4 (7.5) µg/kg lipid; PCB-101, 9.0 (7.1) µg/kg lipid; PCB-138, 151 (129) µg/kg lipid; PCB-153, 213 (182) µg/kg lipid; and PCB-180, 209 (158) µg/kg lipid (Domingo *et al.*, 2001). [Although the distribution of congeners differed between the two studies, the PCB concentrations could not be directly compared as the latter values were lipid-adjusted.]

Kitamura *et al.* (2000) reported blood PCB concentrations in Japanese waste workers ($n = 94$) for other PCB congeners: mean (median) PCB-77, 148.59 (149.07) pg/g lipid; PCB-126, 131.81 (98.60) pg/g lipid; and PCB-169, 104.55 (90.45) pg/g lipid. [None of these congeners were measured in the other studies.]

Workers ($n = 15$) employed as operators for incinerators, boiler-maintenance, furnace maintenance, control panel, and waste-gas washing had a mean concentration of total PCBs of 115.7 µg/kg lipid (PCB-28, 0.7 µg/kg lipid; PCB-138, 17.5 µg/kg lipid; PCB-153, 45.5 µg/kg lipid; PCB-180, 52 µg/kg lipid) (Mari *et al.*, 2009). The sum of congeners PCB-138 + PCB-153 + PCB-180 in this study resulted in a total concentration of 115 µg/kg lipid, which was five times lower than that reported in the workers in Spain (573 µg/kg lipid) (Domingo *et al.*, 2001).

In 26 waste-incineration workers from the Republic of Korea, Park *et al.* (2009) found a mean concentration of total PCBs of 214.93 ng/g lipid (median, 161.13 ng/g lipid), of which hexachloro- and heptachloro-congeners accounted for 70% (congeners measured, PCB-77, PCB-81, PCB-105, PCB-114, PCB-118, PCB-123, PCB-126, PCB-156, PCB-157, PCB-167, PCB-169, and PCB-189). [Co-exposures to dioxins, furans, and other combustible products found in fly-ash are common for waste-incineration workers.] The waste-incinerator workers did not have statistically significantly higher PCB concentrations than control subjects ($n = 7$) (mean PCB concentration, 19.13 ng/g lipid; median, 94.63 ng/g lipid).

1.5.5 Electronic-waste recycling and scrap-metal dealers

One study reported PCB exposures of workers in e-waste recycling in China (Wen *et al.*, 2008). However, they did not report air or serum PCB concentrations, but PCB concentrations in hair samples collected from 94 workers. The PCB concentration range was 55.4–7200 ng/g.

In 17 scrap-metal dealers in two plants in the USA, mean serum PCB concentrations were 7.5 ppb (range, 1–65.3 ppb) (Malkin, 1995). Serum PCB concentrations were significantly related to eating lunch outside the lunchroom [suggesting hand-to-mouth contact as a source of exposure]. The gas-chromatography peak pattern resembled that of Aroclor 1260. [Both waste recycling and scrap handling result in coexposures to dioxins and metals.]

1.5.6 Locomotive-repair workers

Locomotive-repair workers (*n* = 120) in the USA were found to have elevated serum concentrations of PCBs, which was attributable to exposure to transformer fluids (Pyranol, Inerteen, Aroclor) (Chase *et al.*, 1982). Workers were divided into three exposure groups: "exposed" workers who had frequent opportunity for direct contact with PCB-containing transformer fluids; "nominally exposed" workers in the facility did not have opportunity for contact with PCBs; and "non-exposed" workers whose work environment did not involve any PCB fluids. Workers' plasma PCB concentrations were: exposed workers, 33.4 ppm (10–312 ppm); nominally exposed workers, 14.2 ppm (10–30 ppm); and non-exposed workers, 12.0 ppm (10–27 ppm).

1.5.7 Miscellaneous use of PCB oil

PCBs can be emitted by several other sources, including light ballasts and microscopic immersion oil, which contains 30–45% PCBs. Fluorescent light ballasts emit PCBs during burnout (IARC, 1978) and air concentrations depend on the distance from the source. Since the previous *IARC Monograph* on PCBs (IARC, 1978), no new studies regarding PCB exposures during work with carbonless copy paper, microscopic immersion oil, or after a fluorescent light ballast burnout have been published.

One study reported a PCB air measurement from a carbonless copy paper stockroom of 0.07 mg/m^3 (Tatsukawa, 1976). Hasegawa *et al.* (1973) reported that blood PCB concentrations in workers in carbonless paper producing plants (0.01–0.02 µg/g) 2 years after exposure were 10% those found during the period when the PCBs were used. No air or biological monitoring data have been published to assess the extent of PCB exposures during the use of microscope oil (Bennett & Albro, 1973). Four and a half hours after burn-out of a ballast, the concentration of PCBs was the highest (166 µg/m^3) 1 m below the burned-out ballast, while the lowest concentration (12 µg/m^3) was found at a distance of 4.5 m from the fixture (Staiff *et al.*, 1974).

In 1958–1978 in Canada, areas around transformers mounted outdoors were treated with phenoxy herbicides (2,4-D and 2,4,5-D) to reduce foliage (Hay & Tarrel, 1997). To increase adherence of the herbicides to the plant leaves, herbicide sprayers (*n* = 225) would mix 4 pounds [1.8 kg] of phenoxy herbicide with 10 gallons [37.9 L] of used transformer fluid and 90 gallons [340.7 L] of water before spraying. PCB exposures were not measured during this operation.

Use of Aroclor 1254 was reported in a petrochemical plant in the USA during the 1950s, where 31 men had been "heavily exposed" (Bahn *et al.*, 1976). No information regarding how PCB was used was given [but could have been PCBs used as fluids for hydraulic and heat-transfer systems]. No air or blood concentrations of PCBs were reported.

United States navy vessels built between 1946 and 1977 commonly contained PCBs in insulation material, electrical cable, and ventilation gaskets (Still *et al.*, 2003). In nuclear submarines, PCBs were also used in soundproofing material, missile-launch tubes, electrical cables, banding and sheet rubber, heat-resistant paints, hull coatings, and electrical transformers. Activities associated with PCB exposure during dismantling of these vessels were transformer clean-up and removal; cutting/crushing of PCB-contaminated steel, steel-shot blasting of PCB-contaminated surfaces, chiselling/hand-chipping of PCB-contaminated surfaces, and shoveling/sweeping of PCB-contaminated debris. Surface-wipe sampling showed PCB amounts ranging from non-detects to 11 000 µg/100 cm^2. [Information for PCB exposure in the military is scarce.]

Cumulative lifetime exposure to PCBs among Mohawk men at Akwesasne (a Native American community of more than 10 000 persons located along the St Lawrence River in New York, Ontario, and Quebec) who had been occupationally exposed to PCBs was positively associated with serum total PCB concentration ($P = 0.03$) (other non-occupational sources such as fish consumption and living close to hazardous waste sites discussed in the article are not referenced here). The congener profile was most similar to that of Aroclor 1248, the commercial mixture used at local industrial facilities (Fitzgerald *et al.*, 2007) as a hydraulic fluid in a foundry's die-casting machines from 1959 to 1974, and as a component in aluminium-processing heat-transfer equipment. The occupational exposure of Mohawk men was independently assessed by two occupational hygienists as the probability of exposure to PCBs for all jobs of more than 6 months duration with the following qualitative ratings: (1) definitely not exposed; (2) possibly exposed; (3) probably exposed; and (4) definitely exposed. These ratings were assigned weights of zero, 0.25, 0.5, and 1.0, respectively. The weights for each job were then multiplied by duration of employment in that job, and the results were summed over all jobs to estimate cumulative lifetime occupational exposure to PCBs for each man. [The Working Group noted that this population was also exposed environmentally (see Section 1.4.1(a)).]

A qualitative PCB exposure assessment among welders in Sri Lanka was performed recently (Lankatilake *et al.*, 2012). PCB oil extracted from discarded transformers was widely used as coolant oil in small-scale welding facilities in Sri Lanka to facilitate heat transmission and thereby assist in the cooling process. Exposure to coolant oil occurs during replacement of the coolant and while repairing machinery. The amount of coolant oil used in a welding machine depends on the type of machine, but on average is about 5 L. During repairs, there is a high risk of exposure to PCBs in the transformer oil.

1.5.8 Occupations with exposure to PCB by-products

PCBs have also been reported as a by-product in an electric arc furnace steelmaking plant in the United Kingdom (Aries *et al.*, 2008). Air PCB concentrations in decreasing order by department were: melting shop, 586 pg/m^3 (range, 144–1313 pg/m^3); casting area, 187 pg/m^3 (range, 73–272 pg/m^3); control cabin, 99 pg/m^3 (range, 57–129 pg/m^3). The most prominent congeners were PCB-118 (100–500 pg/m^3), PCB-105 (10–80 pg/m^3), and PCB-77 (5–35 pg/m^3).

Using static high-volume samplers (0.2 m^3/min for 12 hours or 24 hours) in a basic oxygen steelmaking (BOS) and iron ore sintering plant, Jackson *et al.* (2012) calculated mean TEQ pg/m^3 for the by-products PCDD/F and PCBs. The BOS process involves the transfer, desulfurization, and refining of hot metal in a steel converter, and secondary steelmaking treatments. Sintering is a process for blending and fusing iron-ore fines, fluxes, coke, and recycled materials (grit and dusts from other processes). Air concentration ranges were: sinter plant, 0.19–3.72 TEQ pg/m^3 ($n = 12$); and BOS plant, 0.08–0.71 TEQ pg/m^3 ($n = 24$). In all instances, concentrations of PCBs were much higher than of PCDD/Fs. PCB-126 contributed significantly to the total TEQ (5–20%).

PCBs have been reported as a by-product in penta- and trichlorophenol wood-preservation pesticide manufacturing with PCDFs and PCDDs (Hryhorczuk *et al.*, 1998; Collins *et al.*, 2008). Serum PCB concentrations (sum of PCB-77, PCB-81, PCB-126, and PCB-169) were measured by the company in these workers (Collins *et al.*, 2008): for pentachlorophenol workers only (n = 26; period exposed, 1944–1980) 73.6 pg/g lipid; trichlorophenol workers only (n = 12; period exposed, 1954–1979): 75.9 pg/g lipid; pentachlorophenol and trichlorophenol workers (n = 14; period exposed, 1961–1980): 86.3 pg/g lipid; tradesmen (n = 10): 121.1 pg/g lipid. These PCB concentrations were not much different from those of a selected reference population (n = 36; 75.0 pg/g lipid).

1.5.9 Removal of PCB-containing sealants

PCB-containing sealants were used in building construction before PCBs were banned in that country. For example, sealant used in Sweden contained 4.7–8.1% Clophen A40 (Sundahl *et al.*, 1999). Air PCB concentrations of 10–120 μg/m^3 were reported after removal of the sealant by a variety of methods: cutting the elastic sealant with an oscillating knife; grinding the concrete with a mechanical machine; sawing the concrete with a mechanical saw; or cutting the concrete with a mechanical chisel. The removal methods were changed by equipping the tool with suction, which reduced air PCB concentrations to non-detects to 3.1 μg/m^3 (Kontsas *et al.*, 2004). Serum PCB concentrations in sealant-remover workers were 0.6–17.8 μg/L (mean, 3.9 μg/L; and median, 1.9 μg/L). For highly chlorinated PCBs, the mean was 3.5 μg/L (median, 1.6 μg/L), and for less chlorinated PCBs, the mean was 0.4 μg/L (median, 0.2 μg/L). Correlation between concentrations in air and serum was only noted for PCB-28 and PCB-52.

During sealant removal in Finland, total PCB concentration in dust samples was 0.026 mg/m^3 (Priha *et al.*, 2005). Congeners determined in the sealant were: PCB-28, 82 mg/kg; PCB-52, 3030 mg/kg; PCB-77, 37 mg/kg; PCB-101, 10 325 mg/kg; PCB-118, 6145 mg/kg; PCB-126, 42 mg/kg; PCB-138, 11 765 mg/kg; PCB-153, 11 185 mg/kg; PCB-169, 32 mg/kg; and PCB-180, 7254 mg/kg (Priha *et al.*, 2005).

Swedish construction workers removing PCB-containing sealants had serum PCB concentrations (sum of 19 congeners) of 575 mg/g lipid, while controls (construction workers not involved in PCB abatement work) had levels of 267 mg/g lipid (Seldén *et al.*, 2008; Wingfors *et al.*, 2006). Concentrations of PCB-180 were not significantly different between groups, while concentrations of many less chlorinated PCBs (especially PCB-66 and PCB-56/PCB-60, but also PCB-28, PCB-44, PCB-52, PCB-74, PCB-101, and PCB-105) were much higher in the exposed workers than in the controls.

1.5.10 People working in contaminated buildings

People working in contaminated buildings (office workers, teachers) are exposed to PCBs (Wiesner *et al.*, 2000); PCB concentrations have been surveyed in workers' air (Gabrio *et al.*, 2000; Schwenk *et al.*, 2002; Peper *et al.*, 2005; Schettgen *et al.*, 2012) and blood (Gabrio *et al.*, 2000; Schwenk *et al.*, 2002; Peper *et al.*, 2005; Herrick *et al.*, 2011; Schettgen *et al.*, 2012).

Mean indoor air concentrations of PCBs in three contaminated schools in Germany were reported to be between 77 and 10 125 ng/m^3; 90% of the total PCBs were either PCB-28 or PCB-52 (Gabrio *et al.*, 2000). These congeners were also reported to be found at high concentrations (> 4000 ng/m^3) in other studies in Germany (Schwenk *et al.*, 2002; Peper *et al.*, 2005; Schettgen *et al.*, 2012). The teachers (n = 96) working in the three contaminated buildings had mean blood PCB-28 concentrations that differed by school (0.045 µg/L, 0.057 µg/L, and 0.098 µg/L, respectively), and that were significantly elevated compared with teachers (*n* = 55) not working in contaminated schools (range, not detected to 0.035 µg/L) (Gabrio *et al.*, 2000).

Median indoor air concentrations were measured over 2 years in schools in Germany for congeners PCB-28 (33 ng/m^3), PCB-52 (293 ng/m^3), and PCB-101 (66 ng/m^3) (Liebl *et al.*, 2004). Concentrations of more highly chlorinated indicator congeners (PCB-153, PCB-138, and PCB-180) were all below 80 ng/m^3. The median sum of indicator congeners was 2.04 µg/m^3. Biomonitoring of teachers (*n* = 9) and cleaning personnel (*n* = 1) in schools in Germany showed that median blood PCB concentrations exceeded the German reference values after adjusting for age in 8 out of 10 workers for PCB-138, 7 out of 10 for PCB-153, and 8 out of 10 for PCB-180 (Neisel *et al.*, 1999).

In teachers in the USA, the relative contribution of lighter congeners (PCBs 6–74) (mean total serum PCB concentration, 1.86 ng/g; *n* = 18) was higher than in controls (Herrick *et al.*, 2011). This was also observed in other studies: mean concentration of PCB-28, 0.28 µg/L; PCB-101, 0.07 µg/L; PCB-138, 1.29 µg/L; PCB-153, 1.68 µg/L; and PCB-180, 1.14 µg/L in Peper *et al.*, (2005); median concentration of PCB-28, 0.087 µg/L; PCB-52, 0.024 µg/L; and PCB-101, 0.012 µg/L in Schettgen *et al.*, (2012); and mean concentration of PCB-28, 0.24 µg/L; PCB-52, 0.07 µg/L; PCB-101, 0.02 µg/L; PCB-153, 0.96 µg/L; PCB-138, 0.70 µg/L; and PCB-180, 0.62 µg/L in Schwenk *et al.*, (2002).

People working inside contaminated buildings other than schools may also be exposed to PCBs. In Germany, air PCB concentrations in contaminated commercial buildings were 1280 ng/m^3 (PCB-28, 110 ng/m^3; PCB-52, 125 ng/m^3; PCB-101, 11 ng/m^3; PCB-138, < 2 ng/m^3; PCB-153, < 2 ng/m^3; PCB-180, < 2 ng/m^3) (Broding *et al.*, 2007). The PCB contamination originated from insulation material and elastic sealing compounds. Serum PCB concentrations were determined in 2002 for 583 persons who had worked between 1 and 40 years in the contaminated commercial building. The median serum total PCB concentration was 2.32 µg/L (PCB-28, 0.09 µg/L; PCB-52, 0.01 µg/L; PCB-138, 0.55 µg/L; PCB-153, 0.9 µg/L; and PCB-180, 0.7 µg/L). People not working in the contaminated building (*n* = 205) had significantly lower serum concentrations of PCB-28 and PCB-52 (0.023 µg/L and 0.004 µg/L, respectively) (Broding *et al.*, 2008).

1.5.11 Clean-up of hazardous waste

Occupational exposure to PCBs has also been measured in workers who perform clean-up of hazardous waste. After an explosion and fire of unlabelled chemical waste drums at the former site of Chemical Control Corporation in Elizabeth, New Jersey, USA, the mean air PCB concentration was 0.11 µg/m^3 (*n* = 3) (Costello & King, 1982). In workers (*n* = 32) removing hazardous waste, including transformers, in the USA, plasma PCB mean concentration was 205 ng/g lipid (range, limit of detection to 527 ng/g lipid) (Horii *et al.*, 2010). Hexa and heptachlorinated biphenyls accounted for 60% of the PCB concentrations.

1.5.12 Firefighters and rescue workers

Firefighters and rescue workers have also been surveyed for PCB exposure in several recent studies, demonstrating a wide variability in serum PCB concentrations (Table 1.33; Kelly *et al.*, 2002; Schecter *et al.*, 2002; Dahlgren *et al.*, 2007; Chernyak *et al.*, 2009, 2012).

1.6 Exposure assessment of epidemiological studies

1.6.1 Studies of occupational exposure

Many epidemiological studies of occupational PCB exposure and cancer have been performed; the majority are among workers in capacitor-manufacture and transformer manufacture and repair. Duration of employment was used to assess exposure in most of these studies (Brown & Jones, 1981; Bertazzi *et al.*, 1982, 1987; Cammarano *et al.*, 1984; Brown, 1987; Nicholson & Selikoff, 1987; De Guire *et al.*, 1988; Taylor *et al.*, 1988; Liss, 1989; Petruska & Engelhard, 1991; Greenland *et al.*, 1994; Tynes *et al.*, 1994; Yassi *et al.*, 1994, 2003; Gustavsson *et al.*, 1986; Savitz & Loomis, 1995; Tironi *et al.*, 1996; Gustavsson & Hogstedt, 1997; Hay & Tarrel, 1997; Kimbrough *et al.*, 1999, 2003; Loomis *et al.*, 1997; Charles *et al.*, 2003; Mallin *et al.*, 2004; Caironi *et al.*, 2005; Prince *et al.*, 2006a, b; Ruder *et al.*, 2006; Ahrens *et al.*, 2007; Hopf *et al.*, 2009b, 2010, 2014; Silver *et al.*, 2009; Pesatori *et al.*, 2013). In the remaining studies, exposure to PCBs was assessed using a variety of approaches, including job-exposure matrices (JEM), development of worker's exposure zones, and measurement of serum PCB concentrations.

JEMs were used in several studies (Greenland *et al.*, 1994; Loomis *et al.*, 1997; Prince *et al.*, 2006a, b; Ruder *et al.*, 2006; Silver *et al.*, 2009).

Greenland *et al.* (1994) developed a JEM in a case–control study of cancer mortality at a transformer-assembly facility. Pyranol was used as the transformer oil from 1936 to 1976. Pyranol was composed of 50% PCBs (mainly hexachlorobiphenyls) and 50% trichlorobenzene, but the PCB content could vary from 45% to 80%. A combination of 1000 job titles in 50 departments in 100 buildings resulted in more than 5500 entries in the JEM. Each entry was rated for seven selected exposures from 1901 to 1984. A four-point categorical rating scale was used to rate the jobs. Former employees and experienced industrial hygienists rated each entry. For pyranol, benzene, and solvents, the analysis categories were: 0, no exposure; 1, indirect exposure, meaning that the chemical was found in the work area, but the worker did not perform tasks using it; 2, direct exposure. Cumulative exposures were calculated using these scores and individual job histories.

A cancer mortality study among electric-utility workers in five companies exposed to PCBs used job categories to estimate weekly exposures in hours for each job (Loomis *et al.*, 1997). PCBs were used in capacitors, transformers and switches. Capacitor fluids were 100% PCBs, while transformer fluids contained 70% PCBs and 30% chlorinated benzene solvents. Exposure assessments were performed by expert panels for each company. The panel members (industrial hygienists, safety personnel, managers, and long-term workers) recorded their individual exposure assessments for PCBs, and other exposures, which were later discussed to resolve differences. For each occupational category and decade, the frequency in times per week and duration in hours of exposure to insulating fluids during the average working week was indicated. This was used to construct company and calendar time-specific JEMs Industrial hygiene surveys of the plants were used to interpret the panel's exposure assessment. Each occupational category (in total, 28) was classified according to workers' potential exposure to PCBs.

Three plant-specific semiquantitative JEMs were used in a study of cancer of the breast in former capacitor-manufacturing workers (women) in Indiana, Massachusetts, and New York, USA (Silver *et al.*, 2009). Two of these JEMs had been used previously in a mortality study (Prince *et al.*, 2006a, b) of former workers at the Indiana and Massachusetts plants, and one in a mortality study of former workers at the Indiana plant (Ruder *et al.*, 2006). Two of the three JEMs have been described in detail in separate publications (Hopf *et al.*, 2009b,

Table 1.33 Serum PCB concentrations in firefighters

Country	Population	Activity	PCB congeners measured	Mean serum PCB concentration	Reference
USA	Firefighters (*n* = 58)	Extinguishing a transformer fire	NA	2.96 ppb (range, 1.9–9.6 ppb)	Kelly *et al.* (2002)
	Rescue workers (*n* = 7)	Working during the collapse of the World Trade Center, New York, September 2001	Non-*ortho* PCBs	43–328 pg/g lipid	Dahlgren *et al.* (2007)
			Mono-*ortho* PCBs	19–404 ng/g lipid	
			ΣDL-PCBs[a]	19–405 ng/g lipid	
Russian Federation	Firefighters, symptomatic (*n* = 8)	Participated in extinguishing a fire at a cable-manufacturing plant (no SCBA)	ΣDL-PCBs[a]	198.6 pg/g lipid	Chernyak *et al.* (2009, 2012), Schecter *et al.* (2002)
	Firefighters, asymptomatic (*n* = 5)	Participated in extinguishing a fire at a cable-manufacturing plant (no SCBA)	ΣDL-PCBs[a]	198.9 pg/g lipid	
	Firefighters (*n* = 7)	Other fires	ΣDL-PCBs[a]	231.7 pg/g lipid	
			Congener 77, 126, 169	Symptomatic firefighters > than the other groups	Schecter *et al.* (2002)

[a] DL-PCBs, dioxin-like PCBs, i.e. PCBs 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, 189

NA, not available; PCB, polychlorinated biphenyl; SCBA, self-contained breathing apparatus

2010). Exposure determinants or factors that influenced PCB exposures for each plant were assessed for all jobs listed in the work histories. Jobs with similar rating of the exposure determinants were grouped into exposure categories. Each job-exposure category, exposure intensity (high, medium, low, background) and frequency (continuous, intermittent) were qualitatively rated separately for inhalation and dermal exposure. The plant-specific JEMs used available air PCB concentrations (the same as in Sinks *et al.* (1992) for the Bloomington plant) to assign inhalation weightings. The product of intensity and frequency (fraction of day exposed) was calculated for each job-exposure category. Finally, the JEM was modified for eras with different conditions of PCB exposure (change in Aroclor use, ventilation-system improvements, lay-out changes etc).

[These historical reconstructions are better than using duration of employment alone in the epidemiological studies, since duration does not distinguish between jobs with higher or lower potential for PCB exposure. Most of these retrospective studies involved manufacturing plants that used limited amounts of other chemicals, or at least when other chemicals were used, these jobs were often indicated and could be excluded from the epidemiological analysis. Creating cohorts of today's working environment would include a very diverse industry with multitude of job activities, including an array of different chemicals. Therefore it would be difficult to draw definitive statements on the causations of a possible observed mortality excesses.]

In their retrospective study of mortality, Sinks *et al.* (1992) developed workplace exposure zones to classify worker exposure. The capacitor-manufacturing plant studied was divided into five zones of exposure by drawing consecutive circles (radius, approximately 69 m) centred upon the heaviest source of PCB exposure. The production area was thus divided into three zones by proximity to PCB source. Two other zones were defined: maintenance and office workers. Air sampling was conducted in these five zones, and means were assigned as the weight (1–5) of PCB exposure for the zone.

Serum PCB concentrations were used in one case–control study (Laden *et al.*, 2001b), and in a recent cross-sectional study (Persky *et al.*, 2012) (see Section 1.5.2).

1.6.2 Studies of environmental exposure

Cohort studies of environmental exposure have used many approaches to assess exposure to PCBs. Exposure approaches include interview, questionnaires, cumulative PCB exposure, dietary intake of fatty fish, PCB concentrations in biological media such as blood, adipose tissue, and breast milk, and in the environment such as carpet dust, or any combinations of these. Biological measures of body burden have been used extensively (see Table 1.34).

1.7 Regulations and guidelines

1.7.1 Global

For Parties to the Stockholm Convention on Persistent Organic Pollutants (POPs) (UNEP, 2001), presently 179 Member States, the production of PCBs is totally prohibited, although the presence of PCBs in equipment is allowed to continue until 2025. The environmentally sound management of waste containing or contaminated with PCBs at a content above 0.005% must be achieved by 2028.

Annex I of the Basel Convention on the Transboundary Movements of Hazardous Wastes and Their Disposal (UNEP, 2011) defines a category of hazardous waste specific to PCBs: “Y10 waste substances and articles containing or contaminated with PCBs and/or polychlorinated terphenyls (PCTs) and/or polybrominated biphenyls (PBBs).” Additionally, Annex VIII defines as “hazardous” any electrical waste containing or contaminated with PCBs at a concentration greater than 50 mg/kg. The Basel Convention is legally binding for 179 countries (status in 2013).

The Codex Alimentarius Commission, recognizing the importance of prevention of human exposure through source-directed measures (i.e. strict control of industrial and agricultural processes that may generate and release PCDDs, PCDFs, and PCBs), adopted the Code of Practice Concerning Source Directed Measures to Reduce Contamination of Food with Chemicals (CAC/RCP 49–2001) (Codex Alimentarius, 2001) and the Code of Practice for the Prevention and Reduction of Dioxin and Dioxin-like PCB Contamination in Foods and Feeds (CAC/RCP 62–2006) (Codex Alimentarius, 2006). No limits in foodstuffs were included, but management options were recommended.

(a) Provisional tolerable monthly intake

In 2002, the Joint Food and Agriculture Organization of the United Nations (FAO)/WHO Expert Committee on Food Additives (JECFA) established a provisional tolerable intake of 70 pg/kg bw per month for PCDDs, PCDFs, and DL-PCBs expressed as TEFs, based on reproductive end-points (JEFCA, 2002). The value was expressed “per month” to reflect that exposure is cumulative and chronic rather than acute.

(b) Drinking-water

No water quality guidelines have been set for these substances because of their low solubility in water.

(c) Air

Air quality guidelines for PCBs have not been established, because exposure by direct inhalation generally constitutes only a small proportion of total exposure, in the order of 1–2% of the daily intake from food. Although this air concentration is only a minor contributor to direct human exposure, it is a major contributor to contamination of the food-chain (WHO, 2000).

Table 1.34 Common measures of exposure to PCBs and design of the exposure assessment in epidemiological studies in non-occupational settings

Exposure measure	Exposure assessment	Examples of exposure categories reported
Cumulative PCB exposure	Regular jobs held	• Job-exposure schemes • Industry classifications • Potential exposure to PCBs as assessed by an occupational hygienist
Dietary intake of fatty fish containing PCBs	Standardized questionnaires or interviews	• Number of fish meals per day
High-level dietary intake of contaminated rice oil (mass poisoning)	Admission to hospital	• Area of residence
Environmental PCB concentrations	PCB concentrations in carpet dust PCBs in soil	• Amount of PCBs in dust • Amount of PCBs in soil
PCB concentrations in biological samples	Serum PCB concentration, non-lipid adjusted	• Sum of PCB congeners • High or low PCB body burden: 'high' exposure (higher than the median based on the control group) vs. 'low' exposure (lower than the median based on the control group)
	Serum PCB concentration, lipid adjusted	• Sum of PCB congeners measured • Single PCB congeners • Potentially estrogenic PCBs (PCB-44, PCB-54) and PCB-101, PCB-187 • Potentially anti-estrogenic, immunologic, dioxin-like, non-*ortho* substitution, mono-*ortho* substitution, moderately persistent (PCB-66, PCB-77, PCB-105, PCB-118, PCB-126) • Immunotoxic PCBs (PCB-66, PCB-74, PCB-105, PCB-118, PCB-138, PCB-153, PCB-156, PCB-167, PCB-180) • Di-*ortho* substitution, limited DL-PCBs and persistent PCBs (PCB-128, PCB-138, PCB-170) • Biologically persistent inducers of CYP1A and CYP2B • Environmentally relevant PCBs (PCB-195, PCB-206, PCB-209) • Neurotoxic PCBs (PCB-18, PCB-28) • Non-dioxin-like PCBs (PCB-74, PCB-99, PCB-118, PCBs 138–158, PCB-146) • Sum of DL-PCBs (PCB-105, PCB-118, PCB-156) • Sum of NDL-PCBs (PCB-28, PCB-99, PCB-138, PCB-153, PCB-170, PCB-183, PCB-187) • *BRCA1* inhibiting PCBs (PCB-101, PCB-138) • Pseudo-estrogen PCBs (PCB-28, PCB-52, PCB-153) • Phenobarbital inducers (PCB-101, PCB-153, PCB-180, PCB-194) • Most-represented congeners (PCB-118, PCB-138, PCB-153, PCB-180)
	Plasma PCB concentration	Sum of the four most prevalent PCB congeners (PCB-118, PCB-153, PCB-138, PCB-180)
	Adipose tissue PCB concentrations	Sum of 18 PCBs Sum of dioxin-like PCBs (PCB-77, PCB-126, PCB-169)
PCB concentrations in biological samples (cont.)	Tumour tissue PCB concentrations	Sum of PCB congeners (PCB-28, PCB-31, PCB-49, PCB-52, PCB-101, PCB-105, PCB-118, PCB-138, PCB-153, PCB-170, PCB-180), measured at the time of diagnosis

DL-PCB, dioxin-like polychlorinated biphenyl; NDL-PCB, non-dioxin-like polychlorinated biphenyl

1.7.2 Environmental regulations

(a) European Union and Member States

The Member States of the European Union have taken actions to eliminate the production, use, and release of PCBs since 1985. In 2004, to implement the Stockholm Convention on POPs, by regulation EC/850/2004 (EU 850/2004), the production, placing on the market, and use of PCBs were prohibited. Low POPs concentration limits were adopted through Council Regulation (EC) No. 1195/2006 (EU 1195/2006) amending Annex IV to Regulation (EC) No 850/2004. Within the European Union of 26 Member States, several measures have been adopted to reduce the presence of PCDDs, PCDDs, PCDFs, and PCBs in the environment, in food and in feed. These include:

- Commission Regulation (EC) No. 1883/2006 of 19 December 2006 laid down methods of sampling and analysis for the official control of levels of dioxins and DL-PCBs in certain foodstuffs;
- Commission Recommendation 2006/88/EC of 6 February 2006 concerning the reduction of the presence of dioxins, furans and PCBs in feedingstuffs and foodstuffs;
- Commission Recommendation 2006/794/EC of 16 November 2006 on the monitoring of background levels of dioxins, DL-PCBs and NDL-PCBs in foodstuffs.
- The most recent Commission Regulation (EU) No. 1259/2011 amended Regulation EU 1881/2006 as regards maximum levels for DL-PCBs and NDL-PCBs (EC, 2011a); it also changed the formerly used 1998 WHO TEFs to the scheme adopted in 2005 (referred to as WHO_{2005}-TEFs) (Van den Berg *et al.*, 2006) and includes maximum levels for NDL-PCBs in food.

  See Table 1.35

(b) North America

(i) USA

The United States Food and Drug Administration has established tolerance levels in various foods in an attempt to reduce human exposure to PCBs (FDA, 2013). [These limit values were set in 1971 and 1977, before any epidemiological and most experimental studies were conducted, and have not been revised since.] The temporary tolerance levels for PCB residues are as follows:

- 1.5 ppm in milk (fat basis);
- 1.5 ppm in manufactured dairy products (fat basis);
- 3 ppm in poultry (fat basis);
- 0.3 ppm in eggs;
- 0.2 ppm in finished animal feed for food-producing animals (except the following finished animal feeds: feed concentrates, feed supplements, and feed premixes);
- 2 ppm in animal feed components of animal origin, including fishmeal and other by-products of marine origin and in finished animal feed concentrates, supplements, and premixes intended for food-producing animals.
- 2 ppm in fish and shellfish (edible portion). The edible portion of fish excludes head, scales, viscera, and inedible bones;
- 0.2 ppm in infant and junior foods;
- 10 ppm in paper food-packaging material intended for or used with human food, finished animal feed and any components intended for animal feeds. The tolerance does not apply to paper food-packaging material separated from the food therein by a functional barrier that is impermeable to migration of PCB.

The United States Environmental Protection Agency (EPA) has set a maximum contaminant level for PCBs of 0.0005 mg/L (500 ppt) in drinking-water. The EPA requires that spills

Table 1.35 Maximum permitted levels for dioxin-like compounds and indicator PCBs in the European Food and Feed regulation

Foodstuffs	Maximum permitted levels[a]		
	Sum of PCDDs, PCDFs, DL-PCBs[b] (pg WHO_{2005}-TEQ per g fat)	DL-PCBs[b] (pg WHO_{2005}-TEQ per g fat)	Sum of PCB_6[c] (ng/g fat)
Meat and meat products (excluding edible offal) of the following animals):			
Bovine animals and sheep	4.0	1.75	40
Poultry	3.0	0.75	40
Pigs	1.25	0.5	40
Liver of terrestrial animals and derived products thereof	10.0		40
Muscle meat of fish and fishery products and products thereof (with the exemption of wild caught eel and wild-caught fresh water fish, with the exception of *diadromous* fish species caught in fresh water, fish liver and derived products, and marine oils)[a]	6.5 pg/g ww	2.5 pg/g ww	75 ng/g ww
Muscle meat of wild caught fresh water fish, with the exception of *diadromous* fish species caught in fresh water, and products thereof[d]	6.5 pg/g ww		125 ng/g ww
Muscle meat of wild caught eel (*Anguilla anguilla*) and products thereof	10.0 pg/g ww		300 ng/g ww
Fish liver and derived products thereof with the exception of marine oils referred to above	20.0 pg/g ww		200 ng/g ww
Marine oils (fish body oil, fish liver oil and oils of other marine organisms intended for human consumption)	6.0		200
Raw milk and dairy products, including butter fat	5.5	2.0	40
Hen eggs and egg products	5.0	1.75	40
Fat of the following animals:			
Bovine animals and sheep	4.0		40
Poultry	3.0		40
Pigs	1.25		40
Mixed animal fats	2.5	0.75	40
Vegetable oils and fats	1.25		40
Foods for infants and young children	0.2 pg/g ww		1.0 ng/g ww
Fruits, vegetables and cereals		0.1 pg/g ww	

[a] The maximum level expressed on fat is not applicable for foods containing < 2% fat (the maximum level expressed on product basis for foods containing < 2% fat = maximum level expressed on fat for that food × 0.02).

[b] The Commission Recommendation 2011/516/EU (EC, 2011b) replaces regulation 2006/88/EC and sets separate action levels for PCDD/PCDF (expressed as WHO_{2005}-TEQ) and DL-PCB (expressed as WHO_{2005}-TEQ).

[c] PCB_6 comprises PCB-28, PCB-52, PCB-101, PCB-138, PCB-153, and PCB-180

[d] The maximum level for crustaceans applies to muscle meat from appendages and abdomen. In the case of crabs and crab-like crustaceans (*Brachyura* and *Anomura*) it applies to muscle meat from appendages.

DL-PCB, dioxin-like polychlorinated biphenyl; PCDD, polychlorinated dibenzodioxins; PCDF, polychlorinated dibenzofurans; TEQ, toxic equivalent; ww, wet weight

Adapted from EC (2011a) and EC (2011b)

or accidental releases into the environment of 1 pound (0.45 kg) or more of PCBs be reported to the EPA (ATSDR, 1996).

(ii) Canada

The import, manufacture, and sale (for re-use) of PCBs were made illegal in Canada in 1977. Release of PCBs to the environment was made illegal in 1985. However, use of PCB-containing equipment is allowed until the end of its service life. The storage of PCBs has been regulated since 1988. Export has been regulated since 1997. These provisions are maintained in the Chlorobiphenyls Regulations, under the Canadian Environmental Protection Act, 1999 (CEPA, 2011).

The regulation of waste is consistent with the Basel Convention's "Technical guidelines for the environmentally sound management of wastes consisting of, containing, or contaminated with persistent organic pollutants" (Basel Convention, 2007, 2015).

(c) Australia and New Zealand

(i) Australia

The Industrial Chemicals (Notification and Assessment) Act 1989 was amended to give effect to the Stockholm Convention (NICNAS, 1989).

The National Strategy for The Management of Scheduled Waste was endorsed by the Australian and New Zealand Environment and Conservation Council in 2003 (ANZECC, 2003) and provides for the safe management and disposal of organochlorine pesticides, PCBs and hexachlorobenzene. The PCB Management Plan provides treatment provisions for different types of PCB waste including liquid residues and discharges, gaseous emissions, solid residues and disposal (Australian Government, 2006, 2007).

(ii) New Zealand

The Hazardous Substances and New Organisms (HSNO) Act 1996 (as amended by the HSNO [Stockholm Convention] Act Amendment 2003), prohibits the production, use and import of the chemicals listed in Annex A of the Convention, including PCBs. Exempted use of PCBs as per the Toxic Substances Regulations 1983 is permitted, but subject to phase-out no later than December 2016. The HSNO Act 1996 is administered by The New Zealand Environment Risk Management Authority (ERMA) by: assessing new chemicals, pesticides or industrial chemicals currently in use that exhibit POP characteristics (Articles 3.3 and 3.4); permitting the appropriate use of POPs for laboratory-scale research or as a reference standard (Article 3.5); managing the existing exempted use and storage of PCBs (Article 3.6); prohibiting import, manufacture, or use of POPs (Article 3.1 and 3.2). The Imports and Exports (Restrictions) Act 1988, via the Imports and Exports (Restrictions) Prohibition Order (No. 2) 2004, prohibits export of POPs (except as conditionally provided under Article 3.2). Import and export are regulated under The Imports and Exports (Restrictions) Act 1988.

(d) Asia

(i) China

China implements an import and export registration system, included under its Regulations on Environmental Management of Chemicals and the Import and Export of Toxic Chemicals of 1994. In 2005, PCBs were included in the List of Toxic Chemicals Strictly Prohibited from Import and Export, by No. 116 Notice on the List of Goods Prohibited from Import (the Sixth Group). The National Implementation Plan under the Stockholm Convention entered into force for China in 2004, and also applied to the Special Administrative Regions of Hong Kong and Macao (NIP China, 2007). This plan aims to prohibit and prevent the production and import of PCBs, and to achieve the environmentally sound management of currently used equipment containing PCBs. China used to produce PCBs, but production was stopped in the 1970s. The plan called for establishing a system for the declaration, registration, and environmentally sound management of equipment in use containing PCBs by 2010. Identification of high-risk equipment currently in use across the country is to be achieved by 2015, with uses of PCBs eliminated by 2025.

Furthermore, China has also stipulated special administrative regulations and standards with regard to PCBs. The Notice on the Issues Concerning Prevention of Pollution Caused by Hazardous Polychlorinated Biphenyls was promulgated in 1979 to ban future imports of power equipment containing PCBs. The Notice on Enhancement of the Management over Waste Polychlorinated Biphenyl Power Capacitors was issued in 1990 to forbid trading and dismantling downstream capacitors containing PCBs. The Provisions on the Pollution Caused by Power Installations Containing Polychlorinated Biphenyls and Related Wastes of 1991 addresses the declaration, transfer, transport, import, treatment, disposal, sealing-up and storage of PCB wastes and other sources. The Control Standard on Polychlorinated Biphenyls for Wastes (GB13015–91) was implemented in 1991, in which the value of the control standard on PCBs wastes and the treatment methods for wastes containing PCBs are stipulated (NIP China, 2007).

(ii) Taiwan, China

Importation of PCBs was prohibited in 1980. The Environmental Protection Administration of Taiwan, China, banned the manufacture, sale, and use of PCBs in 1988. An extensive investigation of electrical devices in 1990–1991 indicated that more than 80 000 PCB-containing electrical devices were still in use, mainly capacitors and transformers. A full-scale ban on the use of PCBs, with the exception of experimental, research, and educational purposes, took effect in January 2001. This prohibited use of any electrical devices containing PCBs by the end of 2000, mandating immediate disposal at end of use of capacitors and transformers containing PCBs (Environmental Protection Administration, 1988). Furthermore, PCBs may not be detectable in effluents from business, sewage systems and building sewage-treatment facilities.

(iii) India

According to Schedule VI of the Hazardous Waste (Management, Handling and Transboundary Movement) Rules 2008, the import and export of hazardous wastes, substances and articles containing or consisting of or contaminated with PCBs are prohibited (Ministry of Environment and Forests, 2008).

(e) Africa

United Republic of Tanzania

The Industrial and Consumer Chemicals (Management and Control) Act of 2003 provides for the management and control of PCBs under the list of severely restricted/banned/eliminated chemicals in Schedule 8. The government of the United Republic of Tanzania issued an Environmental Management Act (Government of the United Republic of Tanzania, 2004) that specifically provides for the control and management of current and future POPs, requiring submission of an annual report on implementation.

1.7.3 Occupational exposure limits

(a) USA

The manufacture of PCBs ended in the USA in 1977. Standards for occupational exposures (permissible exposure limits; PELs) in the USA are set by the Occupational Safety and Health Administration (OSHA) (29CFR1910.1000 Table Z-1 Limits for air contaminants). The PELs are 8-hour TWAs unless otherwise noted, and are determined from breathing-zone air samples. The PELs established by OSHA are 1000 μg/m^3 for PCB mixtures containing 42% chlorine, and 500 μg/m^3 for PCB mixtures containing 54%

chlorine (set in 1971 and not revised after this time). Both standards encompass all physical forms of these compounds: aerosols, vapour, mist, sprays, and PCB-laden dust particles. OSHA recognizes that PCBs are absorbed through intact skin; therefore, routes for dermal and inhalation exposure should be evaluated by an industrial hygienist. The National Institute for Occupational Safety and Health (NIOSH) recommends a 10-hour TWA of 1 µg/m^3 based on minimum reliable detectable concentration and the potential carcinogenicity of PCBs. NIOSH also recommends that all workplace exposures be reduced to the lowest feasible level.

(b) Europe

The maximum allowable airborne concentrations for PCBs containing 42% and 54% chlorine in the Federal Republic of Germany [before reunification] were 1.0 and 0.5 mg/m^3, respectively; and in Sweden, 0.5 mg/m^3 (IARC, 1978).

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