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Chapter 16

Persistent Organic Pollutants and Adverse Health Effects in Humans in Singapore

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Abstract

Persistent organic pollutants (POPs) are man-made chemicals that have an intrinsic resistance to natural degradation processes, and are therefore environmentally persistent. The introduction of POPs into the environment from anthropogenic activities has resulted in their widespread dispersal and accumulation in soils and water bodies, as well as human and ecological food chains where they are known to induce toxic effects. Due to their ubiquity in the environment and lipophilic properties, there is mounting concern over the potential risks of human exposure to POPs. This has led to the establishment of a worldwide research program to determine prevailing levels of POPs in the population and investigate the health risks associated with background exposure. This paper reviews the state of knowledge regarding residual levels of POPs in human adipose tissue worldwide, and provides preliminary data on the levels of key POPs in female adipose tissues collected in Singapore. Organochlorine pesticides (OCPs) were found to be comparable to levels reported for Poland in 2001, with a mean of $0.98\,\mu g\,g^{-1}$ (or $0.84\,\mu g\,g^{-1}$ on a lipid weight basis). For total polychlorinated biphenyl (PCB) congeners, the mean concentration of $34\,ng\,g^{-1}$ (lipid weight basis) is lower than values reported from Japan in 1980 and Belgium in 2000. Polybrominated diphenyl ethers (PBDEs) are present at similar levels to Belgium in 2000, at 3.7 (or $4.7\,ng\,g^{-1}$ lipid weight basis).

16.1. Introduction

Environmental xenobiotic compounds that are both persistent and bioaccumulative have the potential to induce adverse effects on human health. Persistent organic pollutants (POPs) are a group of compounds

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that are prone to long-range atmospheric transport and deposition, and readily undergo biomagnifications in food chains. The global ubiquity of POPs became apparent following their detection in even remote regions of the Earth, including polar regions at levels posing risks to both wildlife (Barrie et al., 1992) and humans (Mulvad et al., 1996).

Of the numerous POPs that are prevalent in our environment, a ''black list'' (www.pops.int) of POPs has been designated under the diplomatic signing of the Stockholm Convention in 2001. The compounds include: pesticides, namely: aldrin, DDT, dieldrin, endrin, heptachlor, chlordane, mirex, and toxaphene; industrial compounds, namely: hexachlorobenzene and polychlorinated biphenyls (PCBs); and other chemical byproducts, namely: polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs)—a general name ''dioxins'' is used for PCDDs and PCDFs. These POPs are known to be particularly toxic with a strong propensity for foodchain biomagnification, and have been associated with both carcinogenic and endocrine disrupting effects in a range of biota. Although the effects on human health from environmental exposure to these POPs remain unclear, there is a growing concern over elevated concentrations of a broad spectrum of POPs in a range of human tissues, including blood, adipose tissue, and breast milk. Due to their lipophilic properties, POPs readily accumulate in human adipose tissue following ingestion of contaminated foodstuffs, and serves as a useful matrix for comparing accumulated levels in different countries.

The ecotoxicological effects of POPs in the aquatic environment have been of great concern in recent years. Based on the global distillation theory, POPs are volatilized into the atmosphere at tropical and temperate latitudes and are subjected to long-range transport resulting in widespread environmental contamination (Iwata et al., 1993). The importance of tropical regions as a source of POPs in the global circulation of these toxic chemicals led us to conduct investigations into the prevalence of these contaminants in the highly urbanized environment of Singapore. It was observed that concentrations of polycyclic aromatic hydrocarbons were considerably higher in Singapore's coastal marine environment, whereas OCPs and PCBs were generally lower than the reported levels for other Asian nations (Bayen et al., 2003, 2004a, b, c; Chanbasha et al., 2003a, b; Gong et al., 2003; Karups et al., 2004a, b; Wurl and Obbard 2004; Li et al., 2005). POPs have also been detected in beach sediments (Chanbasha et al., 2003a), the sea surface microlayer (Chanbasha et al., 2003b), and in marine organisms including the green mussel, *Perna viridis* (Bayen et al., 2003, 2004a, b, c), barnacles (Karups et al., 2004a), and odontocete species (Karups et al., 2004b). POPs with endocrine disrupting activity on androgen and estrogen receptors have been identified in

marine water samples from Singapore's coastal water (Gong et al., 2003). Enclosed coastal areas with weak tidal flushing were found to possess higher levels of endocrine disrupting chemicals in seawater extracts (Gong et al., 2003).

Since Singapore's marine environment is used for leisure, fishing, aquaculture, and desalination, bioaccumulation of POPs and their profound long-term effects on human health and reproductive function is a cause for concern. POPs may be introduced into the environment from a variety of emission sources and anthropogenic activities. Point, area, and line sources include releases from industrial installations, domestic premises, traffic, waste disposal operations such as incinerators and landfills, and activities such as crop stubble burning and the spreading of sewage sludge on land. Area sources can also include the release of POPs from diffuse sources such as contaminated land masses or water bodies that have accumulated POPs from both historical and ongoing deposition. The release of POPs into the environment may be subject to regulatory control (industrial installations, waste disposal operations), while others are unregulated (emissions from domestic premises, unmanaged releases from landfills) or adventitious (forest fires, spillages, and accidental releases).

The potential of a substance to bioaccumulate in an organism can be expressed by the bioaccumulation factor (BAF), the bioconcentration factor (BCF) or the octanol–water partition coefficient (K_{ow}). The BAF and the BCF measure the concentration of a particular substance in a living organism relative to its concentration in the surrounding medium. The criteria to determine the propensity of an organic chemical to undergo bioaccumulation were specified in the Toxic Substance Management Policy (TSMP) under the Canadian Environmental Protection Act of 1999 (www.ec.gc.ca). A BCF or BAF that exceeds 5000, or a logarithm of the octanol-water partition coefficient (Log K_{ow}) of the chemical substance that exceeds 5, indicates a propensity to accumulate in lipids. As a result of elevated hydrophobicity ($K_{ow} > 5$), POPs are readily concentrated and retained in the lipid tissues of biota. Humans, at the top of the food chain, typically accumulate the highest concentrations of these hydrophobic and persistent compounds.

For many years, residues of POPs have been readily detected in the human adipose tissue of individuals in a number of countries, including those living in Europe, Asia, Africa, and North America. In this chapter, we review available data from the scientific literature for levels of POPs in human adipose tissue from various countries around the world. We also report the first such data from Singapore and evaluate levels in the global context (Li et al., 2005) and the specific exposure risk to humans in Singapore via the consumption of seafood containing POPs.

16.2. Pesticides

Toxicity and persistence of pesticides are useful properties for killing their target organisms, but these qualities also cause problems for humans and the environment. The occurrence of organochlorine pesticides (OCPs) in the environment and subsequently in the food chain of humans and wildlife has been noted since the early sixties (Frederick, 1991). Toxicological investigations have shown that several pesticides are carcinogenic in animals, thus raising concern over human exposure (Krieger et al., 1994). To date, this concern has led to several OCPs being restricted or banned for agricultural and/or disease vector control under the joint sponsorship of the United Nations Environment Programme, the International Labour Organisation, and the World Health Organization. (www.inchem.org). However, secondary emissions of pesticides that have accumulated in various environmental compartments, including soils, sediments, and water bodies are expected to persist long into the future. Subsequent transport via the atmospheric pathway ensures that pesticides are widely dispersed from their source, regionally and even globally, thereby representing a ubiquitous global risk to human health and wildlife.

Dichlorodiphenyltrichloroethane (DDT) and its derivatives have now been found in most environmental media, and constitute the dominant OCPs found in human tissues, most notably adipose tissue. DDT is organochlorine compound that was first synthesized in Germany in 1874. DDT's insecticidal properties were realized in 1939, and subsequent commercial use began in 1945. In the human body, DDT is first dechlorinated to tetrachlorodiphenylethane (DDD) that is water soluble and less toxic to human health. Another group of DDT derivatives includes dichlorodiphenyldichloroethanes (DDEs), which readily accumulate in human adipose tissue, and represent a significant health threat due to its long half-life. DDE may be accumulated via the metabolism of DDT in the organism itself, or upon ingestion of DDE tainted foodstuffs (Frederick, 1991). Amongst the isomers in the DDE group, anti-androgenic 4,4′-DDE is the most abundant, with concentration levels in human adipose tissues ranging from 3.5 to 3229 $ng\,g^{-1}$ on a lipid basis in Greenland (Smeds and Saukko, 2001). In contrast, other pesticides, including aldrin, dieldrin, lindane, heptachlor, and heptachlor epoxide, are generally present at low concentrations or below analytical detection limits at the $ng\,g^{-1}$ level (lipid weight basis) (Dewailly et al., 1999).

Geographically, no major differences in the levels of DDT have been reported for human adipose tissues. Based on available data (Table 16.1), the mean concentration of ΣDDTs in the Asian region is $\sim 2.8\,\mu g\,g^{-1}$, and

Table 16.1. Levels of pesticides in human adipose tissue from various countries

Country and sampling time	*p*,*p*′-DDE	*p*,*p*′-DDD	*p*,*p*′-DDT	ΣDDTs	α-HCH	β-HCH	γ-HCH	ΣHCHs	References
Japan, 1986–1987	√	√	√	2.4	√	√	√	1.8	Kashimoto et al. (1989)
Greenland	3.1	–	0.9	4.0	–	–	–	–	Dewailly et al. (1999)
Denmark	1.8	–	0.3	2.1	–	–	–	–	Dewailly et al. (1999)
Finland	2.150	–	–	3.476	–	–	–	–	Hattula et al. (1976)
Finland	0.360	–	–	0.383	–	–	–	–	Mussalo-Rauhamaa et al. (1984)
Finland	0.535	–	–	0.557	–	–	–	–	Mussalo-Rauhamaa (1991)
Finland	0.567	0.006	0.011	0.584	–	–	0.178	0.204	Smeds and Saukko (2001)
Mexico, 1988	–	–	–	17.45	–	–	–	–	Waliszewski et al. (1996)
Mexico, 1991	–	–	–	14.60	–	–	–	–	Waliszewski et al. (1996)
Northern Italy, 1989	0.395	–	0.064	0.459	–	0.213	0.104	0.317	Gallelli et al. (1995)
Poland, 1990	√	√	√	15	√	√	√	0.25	Tanabe et al. (1993)
Poland, 1997–2001	0.770	–	0.072	0.842	–	0.064	–	–	Strucinski (2002)
Spain, 1991	√	–	√	4.4	–	√	–	1.53	Gómez-Catalán et al. (1995)
South Vietnam, 1991	√	√	√	4.9	√	√	√	0.03	Nakamura et al. (1994)
Korea, 1994–1995	1.0	0.007	0.056	1.1	0.002	0.18	0.0003	0.182	Kang et al. (1997)
Belgium, 1996–1998	0.471	0.019	0.097	0.587	–	0.004	0.0052	0.0045	Pauwels et al. (2000)
Singapore, 2003–2004	0.789	0.045	0.143	0.977	0.033	0.214	0.025	0.272	Li et al. (2005)

Notse: – = not target compounds; √ = no value reported; mean concentrations ($\mu g\, g^{-1}$ on a lipid weight basis).

for Europe and Greenland ~3.3 $\mu g\,g^{-1}$ and 4.0 $mg\,kg^{-1}$, respectively. The highest levels of ΣDDT, with a mean of 84.3 $\mu g\,g^{-1}$ in 1989 and 25.7 $\mu g\,g^{-1}$ in 1991 respectively, were found in persons living in a Veracruz city, Mexico (Waliszewski et al., 1996). Statistical analysis showed that age was positively correlated to levels of several OCPs in adipose tissues of persons living in British Columbia (Gallelli et al., 1995), but no statistically significant differences were found between sexes (Teschke et al., 1993).

Reporting on the storage of the separate isomers in fat tissue among the hexachlorocyclohexanes (HCHs) group of pesticides, investigators found that beta-HCH was the predominant isomer. Due to its greater stability and lipophilic properties, beta-HCH is metabolized only slowly and thus eliminated from the human body at a lesser rate than other HCH isomers (Greve and van Zoonen, 1990). Eighteen years of monitoring data from Holland between 1968 and 1986 showed that beta-HCH concentrations were persistent over time, with little evidence of a marked decline (Greve and van Zoonen, 1990).

Hexachlorobenzene (HCB) is another important pesticide contaminant in human tissues. HCB enters the environment via its agricultural use as a fungicide, and as an industrial byproduct. It is chemically stable and highly recalcitrant in the environment. Due to its persistence, specifically in the tissues of animals of a higher trophic status, HCB continues to pose a threat to public health. In the early seventies, in the Netherlands, an increase of HCB in human adipose tissues was associated with an increase in HCB concentrations in consumer products of animal origin (Greve and van Zoonen, 1990). Following a ban by some EU countries in the 1970s, there has been a gradual decrease of HCB concentrations in biota. In the adipose tissues of Dutch citizens, the concentration of HCB (0.7 $mg\,kg^{-1}$) in 1986 was half (1.3 $mg\,kg^{-1}$) of that reported in 1968–69 (Greve and van Zoonen, 1990).

16.3. Dioxins, furans, and polychlorinated biphenyls

Dioxins (PCDDs) and furans (PCDFs) are polyhalogenated aromatic hydrocarbons of high toxicity. There are a total of 210 different congeners; 75 dioxin congeners and 135 furan congeners, of which 17 are potentially toxic. Dioxins and furans are now found prevalent in air, water, and soil in almost all natural environments. PCDD/Fs are strongly bound to organic matter, where half-life in soil has been estimated at 10–20 years (Ryan et al., 1987). PCDD/Fs enter the environment primarily as unintentional byproducts of combustion and chemical processes.

Waste incinerators have been identified as one of the major sources in the urban environment, and others include uncontrolled combustion. Car-exhaust emissions, especially from cars using leaded gasoline with halogenated scavengers also contain considerable levels of PCDD/Fs.

From studies conducted in the 1980s, persons inhabiting industrial areas were found to have higher tissues dioxin levels in adipose tissue (Schecter et al., 1994). For example in rural China, values of 0.142 ng g^{-1} of dioxin were reported in 1984—almost 10 times less than levels reported for the industrialized areas of Japan, Canada, and New York, USA (Table 16.2). PCDD/F levels at 1.75 ng g^{-1} were measured in adipose tissues samples from certain areas of South Vietnam in 1984—10 times higher than concentrations measured in the north of the country. Over 170 kg of 2,3,7,8-TCDD, a congener of PCDD, was sprayed onto vegetation as a defoliant from fixed-wing airplanes or helicopters to certain areas in the south of Vietnam during the war between 1962 and 1970 and is the direct cause of the high PCDD/F levels found in human adipose tissues (Schecter et al., 1986).

The National Human Adipose Tissue Survey (NHATS) was first conducted by the United States Environmental Protection Agency in 1987. An average concentration of 5.38 pg g^{-1} (wet weight) of 2,3,7,8-TCDD was reported in the adipose tissue of the US population, increasing from 1.98 pg g^{-1} in children under 14 years of age to 9.40 pg g^{-1} in adults over 45 years. Due to the different health risks associated with individual PCDD/F congeners, the International Equivalency (I-TEQ) factor was established by the North Atlantic Treaty Organization (NATO) in 1989. More recently, in 1998, the World Health Organization specified Toxic Equivalency Factor (WHO-TEQ) values with slightly different weighting coefficient for PCDD/Fs toxicity compared with those from I-TEQ, and as such is not directly comparable for risk evaluation in human health.

A mean level of PCDD/PCDFs has been reported at 35.6 pg I-TEQg^{-1} lipids in adipose tissue of citizens of France. Samples analyzed in 1999 were found to have similar levels to those reported for other European countries and the USA. No relation to sex or age of the tissue donor was apparent, and levels can be considered as representative of prevailing concentrations of these compounds in most industrialized countries (Schecter et al., 1994). There was no obvious trend of PCDD/F levels found in adipose tissue in citizens of Japan between 1970 and 1999 (Choi et al., 2002; Takenaka et al., 2002) (Table 16.2). However, in 2000, the concentration of PCDD/Fs was reported as only one-third of the amount in 1999 (Choi et al., 2002). This magnitude of decline seems unlikely over such a short time interval, but a longer term reduction is supported by the downward trend in emission levels of PCDDs and PCDFs following

Table 16.2. Levels of PCDDs, PCDFs, and PCBs in human adipose tissue from various countries

Country and sampling time	PCDD/Fs	PCBs	References
Korea, 1994–1995	19[a] (males) 16.5[a] (females)	– –	Kang et al. (1997)
France, 1999	35.6[a]	–	Arfi et al. (2001)
Japan, 1970–1971	31.6[a]	–	Choi et al. (2002)
Japan, 1994–1996	31.5[a]	35.4[a]	Choi et al. (2002)
Japan, 1998–1999	49[b]	17[b]	Takenaka et al. (2002)
Japan 2000	11.9[a]	15.3[a]	Choi et al. (2002)
India	14–46[b] (males) 16–56[b] (females)	– –	Kumar et al. (2001)
Spain, 1997–1998	31[a], 36.3[b]	25.2[b]	Schuhmacher et al. (1999); Wingfors et al. (2000)
Spain, 2002	9.2[a], 11[b]	10.8[b]	Agramunt et al. (2005)
Italy	2.81–13.2	–	Baldassarri et al. (2002)
North of Vietnam, 1984	0.142	–	Schecter et al. (1986)
South of Vietnam, 1984	1.749	–	Schecter et al. (1986)
China, 1984	0.113	–	Ryan et al. (1987)
Japan, 1984	1.667	–	Ryan et al. (1987)
Canada, 1976	1.017	–	Schecter et al. (1986)
Canada, 1980	0.915	–	Schecter et al. (1986)
USA, New York, 1982–1983	1.047	–	Schecter et al. (1986)
Finland, 1984	<0.002–7.70	–	Koistinen et al. (1995)
Belgium, 1996–1998	–	373.1 (7 congeners)	Pauwels et al. (2000)
Belgium, 2000	–	879.7 (35 congeners)	Covaci et al. (2002)
Poland, 1979	–	1200[c]	Tanabe et al. (1993)
Poland, 1990	–	1500[c]	Tanabe et al. (1993)
Japan, 1980	–	3000[c]	Loganathan et al. (1990)
Japan, 1981	–	3100[c]	Mori et al. (1983)
Japan, 1986–1987		775[c]	Kashimoto et al. (1989)
Singapore, 2003–2004	–	34 (40 congeners)	Li et al. (2005)

Note: Mean concentrations ($ng\,g^{-1}$ on a lipid weight basis).
[a]PCDD/F and PCB toxic equivalents (TEQ) are given as I-TEQ.
[b]PCDD/F and PCB toxic equivalents (TEQ) are given as WHO-TEQ.
[c]Numbers of congeners not specified.

implementation of strict emission regulations and associated abatement technologies (Papke, 1998).

Polychlorinated biphenyls (PCBs) are a family of 209 congeners for which there are no known natural sources. PCBs were widely used as coolants and lubricants in electrical components and paint additives until they were widely banned in the 1970s by most developed nations. During

their manufacture and use, PCBs were released into the atmosphere via industrial emissions, weathering of PCB containing materials, and the incineration of PCB-containing products. However, PCBs continue to be released into the environment from leakage of defunct equipment, leaching from landfills, and from previously contaminated soils and sediments. Recent reports have shown that oily fish and Scottish-farmed salmon contain particularly high levels of PCBs (The Sunday Times Magazine, 2004). PCBs have the potential to affect thyroid hormone functions, thereby impairing mental development (The Sunday Times Magazine, 2004).

In a survey of Dutch citizens, the levels of PCBs did not change over a 10-year period between 1973 and 1983, where the persistence of PCBs in fatty tissues remained at a level of $3\,\mu g\,g^{-1}$ (lipid weight basis) (Ryan et al., 1987). With mean levels of PCBs among Poland's population measured at 1200 and $1500\,ng\,g^{-1}$ (lipid weight basis) in 1979 and 1990 respectively, the data indicates the potential of PCBs to persist in human adipose tissue over time (Shinsuke et al., 1993). Although a ban ended the use of PCBs in Japan in 1976, high levels of PCBs of up to $3000\,ng\,g^{-1}$ (lipid weight basis) were detected in human adipose tissue in 1981. However, the concentration of PCBs was reduced to a mean level of $775\,ng\,g^{-1}$ (lipid weight basis) within 36 human adipose tissue samples analyzed in Japan in 1987 (Kashimoto et al., 1989).

Due to similar mechanisms of toxicity to dioxins for some PCB congeners, including non-*ortho*, mono-*ortho* chlorine substituted biphenyls, these compounds are also referred to as "dioxin-like" compounds and rated in terms of toxic equivalency I-TEQ or WHO-TEQ. As shown in Table 16.2, recent studies conducted in Japan between 1994 and 2000 reported dioxin-like PCB congener I-TEQ or WHO-TEQ values which can be used to compare toxicity profiles with PCDD/Fs and assess exposure risk to human health (Choi et al., 2002; Schuhmacher et al., 2004). An earlier study, conducted in 1994, showed similar I-TEQ values of $31.5\,ng\,g^{-1}$ (or $35.4\,ng\,g^{-1}$ on a lipid weight basis) in human adipose tissues for PCDD/Fs and PCBs, which indicates a similar level of toxicity for PCDD/Fs and PCBs to human health, respectively. In 2000, the levels of PCDD/Fs and PCBs with I-TEQ values were also comparable, but reduced to the levels of $11.9\,ng\,g^{-1}$ (or $15.3\,ng\,g^{-1}$ on a lipid weight basis) in human adipose tissues from Japan, which indicates a declining toxic effect on human health over a period of 6 years. A separate study, conducted from 1982 to 1989, on human adipose tissues obtained from donors in Atlanta, Georgia, showed that dioxin-like PCB congeners in adipose tissue varied greatly between samples whereas PCDD and PCDF profiles were more consistent. Age was positively

correlated with the concentrations of PCDDs and PCBs in adipose tissue (She et al., 2002).

16.4. Polybrominated diphenyl ethers (PBDEs)

Rarely has there been so much discussion about a group of chemical products as the flame retardants (www.bsef.com). PBDEs are a specific group of flame retardants widely used in plastics, textiles, electronic circuitry, and other materials to suppress combustion. Three industrial formulations of PBDEs are used widely as flame retardants (BSEF, 2001). Deca-BDE (consisting almost completely of BDE-209) is used mainly in thermoplastics and textiles; octa-BDE (a mixture of hexa- to octa-BDE congeners) is used in acrylonitrile/butadiene/styrene plastics; and penta-BDE (a mixture of tetra-and penta-BDE congeners) is used mainly in polyurethane foam. In 2001, total worldwide demand for these three PBDEs reached 67 MT. Production of all PBDEs has escalated greatly over the last 20 years, and this has been accompanied by their emergence in a diversity of environmental and biological matrices (Cynthia and de Wit, 2002). To date, the toxicology of PBDEs is still under investigation, but it has been established that PBDEs are environmentally persistent, bioaccumulative, and toxic to human health (Watanabe and Tatsukawa, 1990). The critical effects of penta-BDE are associated with neurobehavioral development at low doses (from a dose of $0.6\,\mathrm{mg\,kg^{-1}}$ body weight) and, at higher doses, effects on thyroid hormone levels in rats and mice (Domingo, 2004). Due to concerns over rising levels of contamination in human breast milk and wildlife, as well as associations with thyroid dysfunction, penta- and octa-BDE were banned in the EU in mid-2003 (www.bsef.com). Deca-BDE has been classified as a possible human carcinogen and is known to interfere with brain development in rats (The Sunday Times Magazine, 2004).

PBDE congeners 2,2′,4,4′-tetra-BDE (BDE-47), 2,2′,4,4′,5-penta-BDE (BDE-99), and 2,2′,4,4′,5,5′-hexa-BDE (BDE-153) were detected at high levels in human adipose tissue samples from Sweden in 1999 and, combined, constituted 87–96% and 84–94% of the total sum of PBDEs in liver and adipose tissue, respectively (Meironyté Guvenius and Norén, 1999). BDE-47 has been identified as the predominant congener in the adipose tissue of contemporary California women, where concentrations ranged between 5.2 and $196\,\mathrm{ng\,g^{-1}}$ of lipid, with a median value of $28.9\,\mathrm{ng\,g^{-1}}$ of lipid among 32 breast adipose tissue samples analyzed in 1995. These levels were between 3 and 10 times higher than those measured in similar samples from Japan, Sweden, Germany, and Norway

Table 16.3. Mean levels of PBDEs ($ng\,g^{-1}$ on lipid weight basis) in human adipose tissue from various countries

Country and sampling time	PBDEs	References
USA, California	28.9 (PBDE47)	Washam (2003)
USA, San Francisco Bay	85.7 (PBDE47, 99, 153, 154)	Petreas et al. (2003)
Belgium, 2000	4.75 (PBDE28, 47, 99, 100, 153)	Covaci et al. (2002)
Sweden	5.36 (PBDE17, 28, 47, 66, 100, 99, 85, 154, 153)	Meironyté Guvenius et al. (2001)
Sweden	11.7 (PBDE47, 99, 100)	Haglund et al. (1997)
Sweden	3.8–16 (PBDE47)	Lindström (1998)
Sweden	5.0 (PBDE28, 47, 85, 99, 100, 153, 154)	Meironyté Guvenius and Norén (1999)
Finland	6.3–22 (PBDE47, 99, 153)	Strandman et al. (1999)
Spain	25.1 (PBDE47, 99, 153)	Meneses et al. (1999)
Singapore, 2003–2004	3.7 (PBDE47, 99, 100, 153, 154)	Li et al. (2005)

Note: Mean concentrations ($ng\,g^{-1}$ on a lipid weight basis).

(Lindström, 1998; Strandman et al., 1999; Covaci et al., 2002; She et al., 2002; Washam, 2003). The authors speculated that high levels may be due to California's flammability regulations that demand the use of treated polyurethane foam and textiles used in furnishings. A relatively high concentration of BDE-47 in the adipose tissue of a 74-year-old Swedish male was reported at $8.8\,ng\,g^{-1}$ lipid on a weight basis in 1994 (Haglund et al., 1997), which indicates the strong propensity of PBDE bioaccumulation in the human body over a lifetime. In four Swedish reports, the sum concentrations of PBDEs did not show any clear change in adipose tissue samples between 1997 and 2001 (Haglund et al., 1997; Lindström, 1998; Meironyté Guvenius and Norén, 1999; Meironyte Guvenius et al., 2001).

Reported data on the levels of PBDE congeners in human adipose tissues from various countries are given in Table 16.3.

16.5. POPs in human disease

Diet plays an important role as a source of exposure to many synthetic organic chemicals used in industry, agriculture, or accidentally released to the environment. Among these, OCPs, PCBs, hexachlorobenzene (HCB), and the pesticides including DDT and lindane (γ-hexachlorocyclohexane, HCH) have been classified as possible carcinogens to humans by the International Agency for Research on Cancer (IARC, 1987, 1991). Although initial data for these classifications have been derived from animal experimentation, the evidence for carcinogenesis in humans is

accumulating from both non-occupational and occupational exposure studies (ATSDR, 2000, 2002). It is noteworthy that these POPs remain one of the most important groups of persistent pollutants to which humans are exposed to, primarily via dietary intake of dairy products, meat, and fish. Being lipophilic, they are not readily metabolized or excreted and readily accumulate in adipose tissue. Many are semivolatile, capable of traveling in air for thousands of miles before they settle. These compounds resist photolytic, biological, and chemical degradation and persist in the environment, taking as long as a century to degrade.

Concerns over the detrimental effects of exposure to POPs initially arose from investigations of wildlife communities that showed reproductive, developmental, endocrine, and carcinogenic effects. High rates of malformed genitalia, aberrant mating behavior, sterility, cancer, immune, and thyroid dysfunction have been reported for a range of species (Colborn et al., 1993; Johnson, 1998; Tryphonas, 1998; Fisher, 1999). Moreover, hematological, neurodevelopmental, and reproductive effects have been noted in animals at levels of exposure that overlap the range of concentrations and body burdens found in humans (Johnson, 1998). Available epidemiological evidence in humans suggests that they are similar to those in animals affecting neurodevelopment (Lai et al., 1994; Lai, 2001; Ribas-Fito, 2001), thyroid (Brouwer et al., 1998), estrogen (Wade, 2000), and immune functions (Jacobson, 1990a). The developing brain and nervous system in the fetus appears to be most vulnerable. A landmark longitudinal study from Lake Michigan indicated that babies born to mothers who consumed large amounts of fish contaminated with PCBs had lower birth weights, smaller head circumference, and a shorter attention span than those from mothers who did not eat fish (Jacobson, 1990a, b). Follow-up of these children over a time span of 11 years showed that they continued to perform poorly in a range of skills and developmental tests, including deficits in intellectual functions (Jacobson and Jacobson, 1996). Although postnatal exposure to PCBs through breast-feeding was reported to be not clearly related to any effect on neurological development in these children, current understanding of such adverse health effects following exposure to POPs is limited by available research methods and measures of exposure and outcome. Twelve POPs including nine pesticides designated as the ''Dirty dozen'' have been identified by the United Nations Environment Programme as potential threats to the health of humans and wildlife and have been targeted for elimination (Ritter, 1995).

In addition, residence in close proximity to hazardous waste sites containing POPs has shown to be a risk factor for several cancers (Ozonoff et al., 1994), low birth weights (Baibergenova et al., 2003), and congenital

malformations (Geshwind et al., 1992; Marshall et al., 1997) in susceptible individuals. Besides, persons living near the three areas of concern in New York had significant elevations in rates of hospitalization for thyroid and genital cancers (Carpenter, 2001). Marked elevations in respiratory diseases also correlated well with the concentrations of PCBs measured in blood (Nakanishi et al., 1985). Although conventional wisdom dictates that ingestion is the major source of exposure to POPs, the above studies clearly demostrate that living in proximity to POP-contaminated sites could lead to widespread exposure of the population via air transport of contaminants. Either particulates or vapor phase POPs could have been ingested or inhaled.

More recently, it had been reported that prenatal exposure to PCBs and their heat degradation product dibenzofurans resulted in babies born with retarded growth with dysmorphic physical findings, delayed cognitive development, increased otitis media, and more behavioral problems than in unexposed children (Guo et al., 2004b).

Ingestion of rice oil contaminated with PCBs resulted in the "Yucheng" disease in Taiwan in which the patients developed chloracne, hyperpigmentation, and peripheral neuropathy amongst other signs and symptoms. The acneform eruptions were predominantly in the form of open comedones, papules, and pustules which were distinct from acne vulgaris in that they had dark heads and were found not only in the classified sites for acne but also on extremities, axillae, and external genitalia (Guo et al., 2004a). The offspring of Yucheng women after exposure to PCBs and dinitrofurans constitute one of the largest groups (~2000) of humans prenatally contaminated with high levels of these pollutants (Guo et al., 2004b). A more meticulous longitudinal study of this group of toxicants in the second and third generation offspring of affected women could provide more information on dose-response relationship, teratogenicity, and endocrine-disrupting mechanisms of these hazardous chemicals.

16.6. POPs as endocrine disrupters

Recently, Harrison and Sharp and Irvine summarized the hypothesis and status of evidence implicating endocrine disruption and their adverse impacts on human health (Sharp and Skakkebaek, 1993; Harrison, 2001; Sharp and Irvine, 2004). They outlined that exposure of the fetal/developing male to environmental pollutants resulted in hypospadias, cryptorchidism, prostrate cancer, testicular cancer, a global decrease in sperm counts, and decreased male reproductive capacity. Detrimental effects in women include breast cancer, cystic ovaries, and endometriosis.

The hallmark of Harrison and Sharp's proposition is the so-called Sharp-Skakkebaek hypothesis for reproductive abnormalities in men that can be attributed to an increased exposure to estrogens in utero (Sharp and Skakkebaek, 1993), whereby synthetic chemicals in the environment are the prime source of excessive estrogenic stimulation of the male fetus. However, in many instances it is difficult to assign causality because of the complexity of environmental contaminants and the lack of adequate analytical data that document contaminant levels during critical windows of exposure. Nevertheless, there have been several adverse effects in wildlife populations that strongly correlate with POP exposure (Brunstrom and Halldin, 2000). As noted by the recent assessment by the World Health Organization, the clearest available evidence is derived from animal studies including reproductive and immunological effects in marine organisms living in environments contaminated by OCPs PCBs, DDE (WHO, 1978). Egg-shell thinning and embryonic abnormalities in various avian species exposed to DDT and PCBs have been reported (Vos, 2000), as well as the induction of vitellogenin in fish and marine mollusks living near effluent sewage treatment plants (Damstra, 2005). Alligators exposed to organochlorine pesticides spilled into lakes display a host of morphologic and hormonally related abnormalities of the male and female reproductive tracts (Guillette et al., 1994, 1996). Conversely, lower concentrations of POPs in the Great Lakes region have correlated with dramatic improvements in reproductive capacities with significant increases in the populations of cormorants, gulls, terns, herons, and other predatory birds in this region (Tremblay and Gilman, 1995).

In humans, adipose tissue levels of POPs are 200–1000 times higher than in the corresponding concentrations in serum (Kutz and Wood, 1991; Mussalo-Rauhamaa, 1991; Toppari, 1996). Therefore, even small samples of adipose tissue have organochlorine compounds in the detectable range and are more suitable for congener-specific analysis (Wolff, 1983). This tissue also provides a good measure of cumulative internal exposure to these environmental agents. In order to evaluate the abdominal adipose tissue concentrations of POPs, a hospital based control study of pregnant women undergoing elective caesarean section was conducted in Singapore (Li et al., 2005). Our investigations on adipose tissue levels of POPs revealed that DDT and its derivatives were the dominant pollutants present in adipose tissue with a concentration range from 7 to 2928 $ng\,g^{-1}$, and a mean of 977 $ng\,g^{-1}$ on a lipid weight basis which is comparable to the value from Finland (Mussalo-Rauhamaa et al., 1984) and Poland 2001 (Teschke et al., 1993). A total of 41 congeners of PCBs were analyzed and ranged in concentration from 7 to 71 $ng\,g^{-1}$ with a mean of 34 $ng\,g^{-1}$ on a lipid weight basis. These levels are much lower

than reported in Japan in 1980 Mori et al., 1983; (Loganathan, 1990), and Belgium in 2000 (Covaci et al., 2002). A sum of five PBDE congeners (BDE-47, 99, 100, 153, 154) were present at a concentration range of 0.5–12.3 ng g^{-1}, and a mean of 3.6 ng g^{-1} (lipid weight basis) in adipose tissue which is comparable to values obtained from Belgium citizens in 2000 (Covaci et al., 2002).

The concern that organochlorine pollutants in the environment or ingested foods may cause cancer in humans is widespread. Evidence has been accumulating that PCBs are carcinogenic in nature due to the fact that they are weakly estrogenic and some organochlorines have been tested almost exclusively in epidemiological studies in breast (Aronson et al., 2000; Hoyer et al., 2001), prostrate (Hoyer et al., 2001), colorectal, endometrial cancers (Howsam et al., 2004), and in non-Hodgkin's lymphoma (Quintana et al., 2004). Moreover, critical periods of urogenital tract and nervous system development in utero and during early postnatal life are especially sensitive to hormonal disruption with a potential impact on sexual maturation (Krstevska-Konstantinova, 2001) and neuro-developmental retardation (Jacobson and Jacobson, 1996). A delay in pubertal breast development has been linked with dioxin-like compounds in serum samples in girls (Staessen et al., 2001), while DDT/DDE exposure resulted in precocious puberty (Staessen et al., 2001). Early onset of menarche has also been associated with PBB exposure, but no such effect was observed in pubertal girls with regard to PCB exposure (Den Hond, 2002; Vasiliu et al., 2004). It is likely that a different endocrine activity could be ascribed to the different halogenated POPs. Further research is warranted to elucidate the mosaic of putative endocrine disruption effects induced by specific POPs in relation to aberrant sexual maturation and carcinogenicity in humans.

16.7. POPs in typical seafood consumed in Singapore

The routes of exposure to POPs for human beings include inhalation, dermal exposure, and ingestion of water or food contaminated by POPs. Cases of occupational or accidental exposure to POPs have been recorded, particularly during the application of pesticides (Vallack et al., 1998). However, food consumption is one of the most important pathways of exposure for the general population (Dougherty et al., 2000; Stefanelli et al., 2004).

Seafood consumption is a rich source of vitamins, minerals, proteins, and omega-3 polyunsaturated fatty acids that have a wide range of beneficial effects for human health (Sidhu, 2003). However, among food

items, seafood has been identified as a major source of POPs (Bocio et al., 2003), particularly in Asian countries, where fish and shellfish account for a significant component of the dietary intake (Kannan et al., 1997; Ohta et al., 2002; Simmonds et al., 2002). As a consequence of high consumption rates, a strong correlation exists between the levels of PBDEs in human breast milk and fish consumption (Meironyté et al., 1999; Ohta et al., 2002). Most controversially, in the USA, it has been shown that 11-year-old children born to women who had consumed contaminated fish from the Great Lakes, i.e., exposed to polychlorinated biphenyls, had memory and attention problems (Jacobson and Jacobson, 1996). Reports of POPs in seafood include xenobiotics such as PCBs (Bjerregaard et al., 2001), organochlorine pesticides (Smith and Gandolli, 2002), PBDEs (Bocio et al., 2003), dioxins, and furans (Schecter et al., 2003). In particular, the impact of consuming farmed fish has recently raised health concerns as elevated levels of POPs have been measured in edible tissues relative to wild fish (Easton et al., 2002; Antunes and Gil, 2004; Hites et al., 2004).

However, although the importance of seafood in the human exposure to POPs, no information is available regarding the human intake of contaminants via seafood consumption in Asian countries, including Singapore. Seafood consumption in Singapore averaged 46.3 and 49.9 g/day for women and men, respectively, in 1998 (Ministry of Health, Singapore, 2001), which is comparable to typical seafood consumption rates in Taiwan (Chien et al., 2003), but more than twice the intake in San Francisco, USA (Greenfield et al., 2003). Most seafood consumed in Singapore is imported—principally from elsewhere in Asia (e.g., prawns from Thailand), but also from Europe (e.g., salmon from Norway) and the Americas (e.g., scallops from USA), (Singapore Trade Development Board, 2001). In this study, the levels of POPs (i.e., PCBs, PBDEs, and organochlorine pesticides) were measured in the edible portions of 20 different seafood types commonly consumed in Singapore and a contaminant risk assessment was conducted based on seafood consumption (Bayen et al., 2005b).

Twenty types of seafood were collected from local supermarkets between June 2002 and June 2003. Details of the samples analyzed are presented in Table 16.4. The selection of the seafood types was based on supermarket sales figures and represents typical consumption patterns in Singapore (personal communication). Levels of α- and γ-chlordane, DDTs (sum of *p,p′*-DDT, *p,p′*-DDD and *p,p′*-DDE), PCBs (sum of congeners 17, 18, 28–31, 33, 44, 49, 52, 70, 74, 82, 87, 90–101, 105, 110, 118, 128, 132, 138, 149, 151, 153, 156, 169, 170, 171, 177, 180, 183, 187, 194, 199, 201, 205, 206, 208, and 209), and PBDEs (sum of congeners 47, 99,

Table 16.4. Sample types and characteristics of seafood

Seafood type	Species	Sample size
Shark steak	Species not identifiable	4 fillets
Shark fin	Species not identifiable	2×100 g
Cod fillet	Species not identifiable	2 fillets
Stingray fillet	*Dasyatis kuhlii*	2 fillets
Tuna steak	Species not identifiable	4 fillets
Canned tuna	Species not identifiable	2 cans
Silver pomfret fillet	*Pampus argenteus*	3 fish
Selar fillet	*Selar crumenophtalamus*	3 fish[a]
Kuning fillet	*Selaroides leptolepis*	10 fish[a]
Conger eel fillet	Species not identifiable	2 fillets
Greasy grouper fillet	*Epinephelus coioides*	3 fish
Sea bass fillet	*Lates calcarifer*	3 fish
Song fish fillet	*Aristichthys nobilis*	2 fillets[a]
Salmon fillet	*Salmo salar*	3 fillets
Squid ring	Species not identifiable	140 g[a]
Grey prawn	Species not identifiable	12 prawns[a]
Giant tiger prawn	*Penaeus monodon*	12 prawns[a]
Flower crab	*Portunus pelagicus*	3 crabs[a]
Green mussel	*Perna viridis*	9 mussels[a]
Scallop	*Pectinidae spp.*	10 scallops[a]

[a]Refers to a pooled sample.

and 100) in the edible parts of the 20 types of seafood analyzed are presented in Figs. 16.1a–d. The average concentrations of POPs in seafood, as well as their limit of detection and relative occurrence amongst the seafood types are reported in Table 16.5. Chlordanes, DDTs, and PCBs were detected in 75, 90, and 100% of the seafood types, respectively. On the contrary, mirex and pentachloronitrobenzene (PCNB) were only detected in 10 and 15% of the seafood types, at a concentration two orders of magnitude less than for DDTs and PCBs, respectively. Levels of chlordanes were below 1 ng g^{-1} wet weight (ww) in all seafood types, except for green mussels (14.4 ± 2.0 ng g^{-1} ww) and salmon fillets (2.9 ± 1.7 ng g^{-1} ww). Levels of DDTs were below 5 ng g^{-1} ww, except for green mussels (37.8 ± 0.7 ng g^{-1} ww) and salmon fillets (14.4 ± 9.1 ng g^{-1} ww). Levels of PCBs were below 5 ng g^{-1} ww in all seafood types except for green mussels (8.2 ± 2.6 ng g^{-1} ww) and salmon fillets (28.5 ± 14.4 ng g^{-1} ww). Levels of PBDEs were below 0.1 ng g^{-1} ww for all seafood types, except salmon fillets (2.8 ± 1.8 ng g^{-1} ww). Amongst the 20 seafood types analyzed, green mussel and salmon fillets samples contained the highest levels of POPs. Green mussels are filter-feeders and therefore readily accumulate POPs. The levels found in market samples were in the upper range of concentrations found in wild mussels in

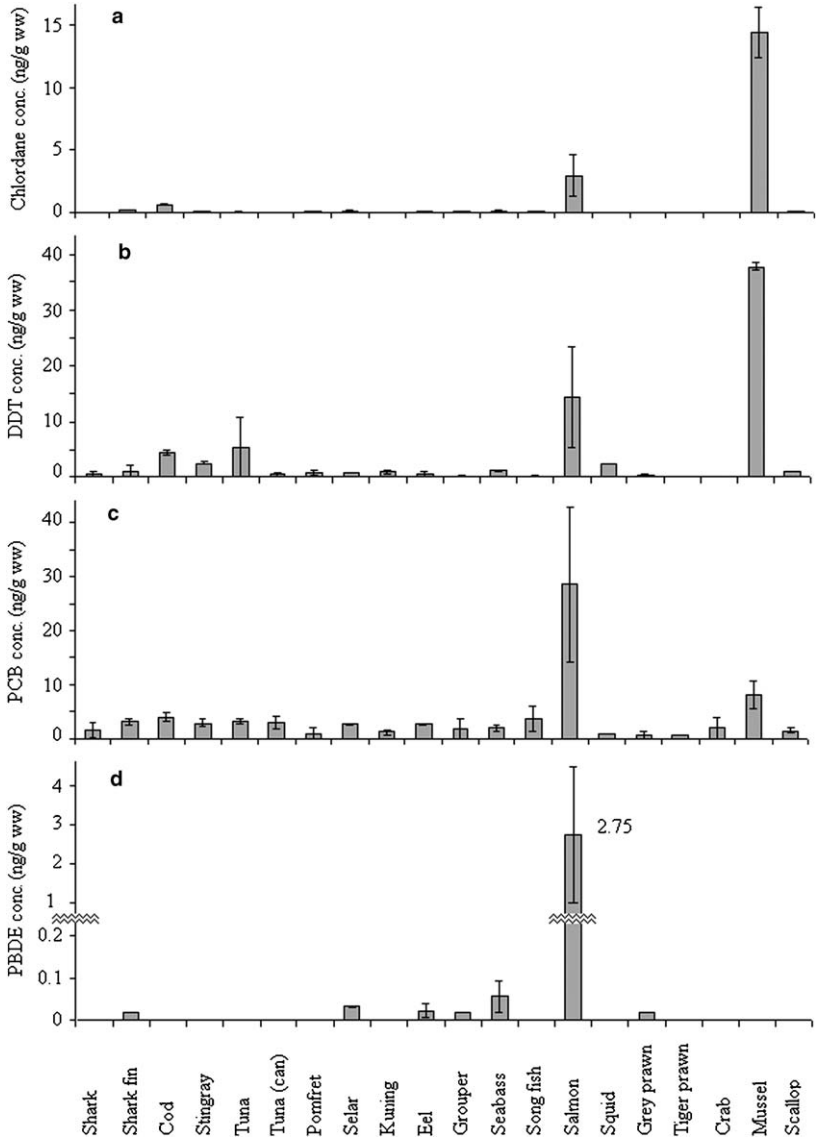

Figure 16.1. Total levels in ng g^{-1} wet weight (ww) of chlordanes (a), DDTs (b), PCBs (c), and PBDEs (d) in the major seafood types commonly consumed in Singapore (mean level $\pm$ SD).

Table 16.5. Mean level, occurrence, and mean daily intake of POPs from seafood for a 60 kg person in Singapore

Contaminant	Method limit of detection in $ng\,g^{-1}$ ww	Mean level (range)[d] in $ng\,g^{-1}$ ww	Percentage of seafood types with levels above MDL (%)	Mean daily intake ($\mu g\,kg^{-1}$ body weight/day)		Oral RfD[e]	Cancer benchmark concentration[f]
				Hypothesis no detect = 0	Hypothesis no detect = 0.5 DL		
Chlordane[a]	0.04	0.95 (BLD–14.39)	75	0.75	0.76	500	1
DDTs[b]	0.04–0.09[c]	3.76 (BLD–37.84)	90	3.00	3.01	500 as *p,p′*-DDT	3
PCNB	0.04	0.03 (BLD–0.13)	15	0.01	0.02	3000	
Heptachlor	0.08	0.39 (BLD–6.52)	35	0.29	0.31	500	0.22
Heptachlor epoxide	0.04	0.16 (BLD–2.13)	55	0.123	0.130	13	
Mirex	0.02	0.01 (BLD–0.04)	10	0.003	0.012	200	
PCBs	0.01–0.2[c]	3.72 (0.61–28.47)	100	2.99	2.99	20 as Aroclor 1254	0.13
PBDEs	0.01–0.03[c]	0.17 (BLD–2.75)	30	0.117	0.134	2000	

[a]Sum of α and γ-chlordanes.
[b]Sum of *p,p′*-DDT, *p,p′*-DDD, and *p,p′*-DDE.
[c]Range for all congeners.
[d]Mean concentration (range between brackets) amongst the various types of seafood for the hypothesis "Nondetect = 0.5 detection limit"—BLD: below limit of detection.
[e]Data obtained from USEPA' Integrated Risk Information System (http://www.epa.gov/iris).
[f]Represents the exposure concentration at which lifetime cancer risk is one in one million.

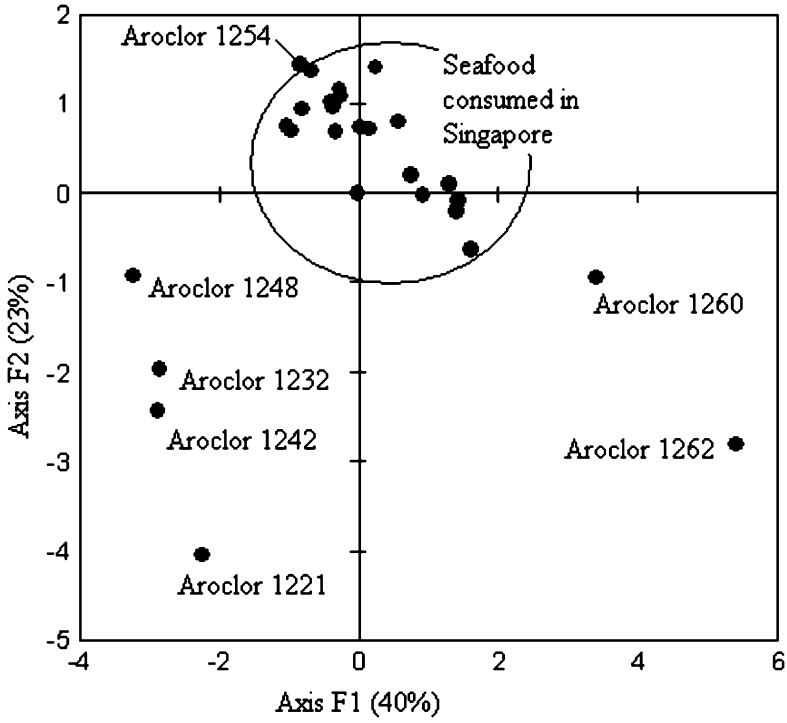

Figure 16.2. Bi-plots showing the first two principal components of relative individual polychlorinated biphenyl (PCB) congener profiles in seafood in relation to congener profiles for Aroclor mixtures 1221, 1232, 1242, 1248, 1254, 1260, and 1262.

Southeast Asia. Several recent studies have revealed that farmed salmon contains higher levels of POPs including PCBs and PBDEs compared to wild specimens (Easton et al., 2002; Ohta et al., 2002; Hites et al., 2004). Concentrations of POPs in typical salmon fillets consumed in Singapore are in the range of values reported for farmed salmon.

Principal component analysis of PCB congener profiles of seafood relative to commercial Aroclor mixtures is shown in Fig. 16.2. PCB congener profiles for Aroclor mixtures 1221, 1232, 1242, 1248, 1254, 1260, and 1262 were determined by Frame (1997). The first principal component, F1, has a positive loading on hepta-CBs and octa-CBs and a negative loading on tri-CBs and tetra-CBs. The second principal component, F2, has a positive loading on penta-CBs and hexa-CBs and negative loading on tri-CBs, hepta-CBs, and octa-CBs. The PCB congener profile of seafood types analyzed reflects the presence of a mixture of Aroclor 1254 and Aroclor 1260 congeners, with the majority of the 20 types

closely matching the PCB congener profile of Aroclor 1254. A similar match was observed for a variety of marine organisms elsewhere (Miao et al., 2000). The congener BDE-47 (2,2′,4,4′ tetra-BDE) was proposed as an indicator for PBDE contamination in marine fish (Akutsu et al., 2001). In addition to salmon fillets, BDE-47 has also been detected in selar, seabass, and grey prawn samples in this study—seafood that originates from within Southeast Asia (MFRD, 1996). Little is known about the occurrence of brominated flame retardants in the environment of Asia (Kemmlein et al., 2003). The present data suggest that BDE-47 is present in the marine environment of Southeast Asia and is accumulating in the foodchain.

To undertake a risk assessment on the consumption of food, the first step is to compare the levels with the maximum residue limits (MRLs). MRLs for POPs in Singapore (Government of Singapore, 1990) and the United States (USFDA, 2001) are presented in Table 16.6. Samples were lower than respective MRLs for all POPs.

Periodically, the Ministry of Health in Singapore conducts a survey on the dietary habits of the population. Latest available seafood consumption figures, from 1998 (Ministry of Health, Singapore, 2001), were used for risk assessment calculations using analytical data obtained from this study. In this survey, the mean daily intake (MDI) of fish/seafood for the general population of Singapore is similar for both females and males and approximated to 46 and 50 g/day, respectively. Therefore, an average MDI for seafood of 48 g/day was used in dietary exposure calculations, together with an average body weight of 60 kg. The estimated MDI of contaminants from seafood was calculated as the MDI of seafood multiplied by the mean concentration of contaminants in the 20 seafood types. Using the methodology specified by the US Environmental Protection Agency (Dougherty et al., 2000), the mean concentration of contaminants was calculated according to two hypotheses: "non-detect samples are equal to zero" i.e., contaminant values below the limit of

Table 16.6. Maximum residue limits for POPs in seafood in Singapore and the United States

Contaminant	Singapore ($\mu g\,g^{-1}$)	US FDA ($\mu g\,g^{-1}$)
Chlordane	0.05	0.3
Heptachlor		0.3
Heptachlor epoxide		0.3
Mirex		0.1
DDTs	5.0	5.0
PCBs		2

analytical detection are ascribed a value of zero; and "non-detect samples are equal to half of the limit of detection" i.e., contaminant values below the limit of analytical detection are ascribed a value of 50% of the limit of analytical detection. The first hypothesis tends to underestimate the concentration, and therefore the intake of contaminants. MDIs of POPs in seafood types typically consumed in Singapore are presented in Table 16.5. MDIs were similar (difference $<10\%$) regardless of the value attributed to the non-detect samples (i.e., zero or half of the detection limit), except for PBDEs, PCNB, and mirex. Differences of greater than 10% can be attributed to the low occurrence of contaminants amongst food types (Dougherty et al., 2000). The mean daily intake of DDTs, PCBs, and PBDEs from seafood for a 60 kg person in Singapore reaches 3.0, 3.0 and 0.1 ngkg body weight per day, respectively. Data comparison with human consumption studies conducted elsewhere shows that mean daily intake of PCBs from seafood in Singapore represents only 6% of the total daily intake of a whole diet in Italy (Zuccato et al., 1999). The MDI of DDT from seafood in Singapore is 2.5 times higher than for a seafood diet in Italy, according to a study conducted in 1997 (Stefanelli et al., 2004). Contributions of specific types or groups of seafood to the MDI of POPs were calculated for the hypothesis of "non-detect samples are equal to zero" (See Fig. 16.3). On this basis, it can be concluded that salmon consumption accounts for 19, 38, and 94% of the mean calculated intake of DDTs, PCBs, and PBDEs, respectively, and green mussel consumption accounts for 51, 11, and 77% of the mean calculated intake of DDTs, PCBs, and chlordanes, respectively.

The "oral reference dose" (Oral RfD) is an estimate of the daily exposure of a person to a contaminant that is likely to be without appreciable risk of a deleterious non-carcinogenic effect during a lifetime (USEPA; http://www.epa.gov/iris/). Oral RfD values for POP concentrations in seafood types are presented in Table 16.5, together with the daily intake of POPs from seafood consumed in Singapore. Daily intakes of POPs from seafood are below the oral RfD. The "cancer benchmark concentration" (Dougherty et al., 2000) represents the exposure concentration at which a lifetime cancer risk equates to one excess cancer death in one million persons. This level is defined as the public health protective concentration in the Congressional House Report to the Food Quality Protection Act of 1996 in the USA. Cancer benchmark concentrations were exceeded for DDTs, heptachlor, and PCBs (See Table 16.5). The "cancer hazard ratio" is the ratio of the MDI for a specific contaminant relative to the cancer benchmark concentration. The cancer hazard ratio represents the extent to which average daily exposure exceeds the benchmark concentration. The cancer hazard ratio of seafood consumption

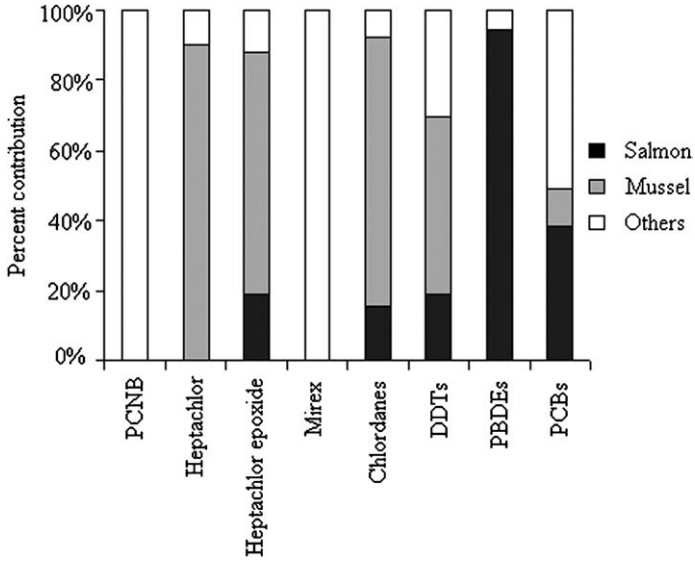

Figure 16.3. Percentage contribution of salmon, green mussels, and other types of seafood to the mean daily intake of POPs via seafood consumption in Singapore.

was equal to 1, 1, and 23 for DDTs, heptachlor, and PCBs, respectively, meaning that, according to Dougherty et al., (2000), a significant number of people are potentially at risk in Singapore over a lifetime of seafood consumption. However, it is important to note that the standard deviation for per capita seafood consumption data in Singapore is large (i.e., 46.3 ± 36.9 and 49.9 ± 40.0 g/day for women and men, respectively), where significant levels of variability exist between ethnic subgroups in the population. As an example, a Malay male adult in Singapore consumes 64.9 g/day of seafood versus 42.8 g/day for an Indian and 48.2 g/day for a Chinese male adult, respectively (Ministry of Health, Singapore, 2001). The risk associated with seafood consumption is therefore up to 51% higher for a male adult in the Malay community.

It is also worth noting that these calculations are derived from raw tissue analysis and more information is required on the effect of seafood cooking in Singapore on the final load of contaminants ingested. Cooking processes, such as baking, frying, or boiling are known to reduce the burden of POPs in fish (Schecter et al., 1998; Zabik and Zabik, 1999). In another study on the effect of cooking on the loss of POPs from salmon (see Bayen et al., 2005a), the initial burden of POPs in the fish

steaks decreased by $26 \pm 15\%$ after cooking with an additional loss of $9 \pm 3\%$ when the skin was removed from the cooked steak. Human health effects associated with the consumption of cooked salmon can be considered to decrease proportionally, and it can be concluded that cooking has a beneficial reduction on the burden of POPs in salmon steaks. Therefore, based on the present results, an average reduction of the POPs load by 26% following cooking would decrease the mean daily intake of POPs proportionally. The consequence is that the mean daily intake of DDT would then be below the cancer benchmark concentration; however, the cancer hazard ratio for PCBs (17) would be still significantly greater than one—even when taking into account the cooking of the seafood.

16.8. Conclusions

Among the pesiticides, DDT group of OCPs are the most prevalent contaminants in human adipose tissues worldwide. Typically, levels of PBDEs are still the lowest amongst the major groups of POPs prevailing in the environment, but levels in biological matrices, including human adipose tissues are rising around the world due to large scale production and usage of these compounds. Clearly, it will remain a challenge for the scientific community to continue to monitor and evaluate the risk of current and emerging POPs in our environment, and ultimately to eliminate the most harmful compounds and identify safer alternatives. Historical data on the known effects of POPs on wildlife and human health clearly advocates the adoption of a precautionary principle prior to the intentional or incidental emission of novel POPs into the environment.

Analysis of local samples in Singapore has demonstrated the prevalence of POPs in human adipose tissues. Our investigations have also shown the ubiquity of POPs in seafood commonly consumed in Singapore. As a result of the use of contaminated fish meals, levels of POPs in farmed salmon were relatively higher than in any other fish commonly consumed in Singapore. Human health risks associated with the consumption of contaminated seafood exist and maximum exposure criteria are exceeded when considering seafood consumption alone. Further investigation should be undertaken to consider exposure through the whole diet for the Asian population. Although concentrations are comparable to those observed elsewhere, longer term monitoring of a larger cross section of the popluation is warranted in order to establish temporal trends and potential long-term risks to human health.

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