Aquatic Ecosystem Contamination and Riverine Flux of Persistent Organochlorine Pollutants (POPs) to Coastal Seas in the Asia-Pacific Region

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Preface

This report documents the occurrence of selected POPs [persistent organochlorine pollutants] in the freshwater aquatic systems of the Asia-Pacific region. The region is defined herein to include Asian coastal mainland countries from Iraq to China, and the major islands of the Eastern Indian and Western Pacific Oceans. The POPs elected for study included the industrial chemical PCBs [polychlorinated biphenyls]; the insecticides DDT, HCH [hexachlorocyclohexane], chlordane, heptachlor, *drins* [aldrin, dieldrin and occasionally endrin]; and HCB [hexachlorobenzene]. All, except HCH, are members of UNEP's *dirty dozen POPs* list. HCH was added because it has been heavily used in the Asia-Pacific region since the early 1950s.

The thrust of the current study was two fold: (A) to assess the relative local contamination from reported POPs concentrations in waters, sediments and biota; and (B) to estimate exports of POPs by surface water drainage systems into territorial coastal seas. The concentration data alone gauge the intensity of local contamination, while the river export estimates indicate the territorial capacity to deliver POPs to coastal waters.

As POPs data in the GEMS/Water data base were limited to a few chemicals in a few countries, most POPs data summarized in the present study were obtained from published scientific literature. The numerous published studies conducted in the region by Professor Shinsuke Tanabe and associates of Ehime University, Japan, gave a nucleus of high quality, high resolution POPs data from which other data sources could be evaluated and incorporated into the construction of a broad image of POPs contamination across the Asia-Pacific region. The four core study POPs with widest data availability were PCBs, DDT, HCH and chlordane, while data for *drins* (chiefly aldrin and dieldrin), heptachlor and HCB were of more limited availability.

The results of this investigation, which are too numerous to list in a few paragraphs, are summarized in Chapter 2, and the country-by-country details are presented in Chapters 3–14. While data are often incomplete and uncertain, there was sufficient information to compile reasonable partial regional totals of riverine exports by territory for the four core POPs. Most major sources are represented, but exports from the Indus River, Malaysian Borneo, Philippines, Indonesia exclusive of Java, the Koreas and China are presently indeterminable for some POPs. The current regional totals could rise perceptibly if these were included, but it is unlikely that the relative ranking of the predominant sources of PCBs, DDT and HCH would change appreciably. More complete estimates demand the collection of more extensive data, not only aquatic measurements, but also production, importation and consumption statistics for POPs insecticides, and inventories of PCBs stocks. Until then, the current regional partial totals are useful figures for policy development, planning and management purposes.

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Units and Reporting Conventions

Time

s	second									
d	day									
a	annum	=	1	year						

Distance

mm	millimetre	=	10^{-3}	m						
cm	centimetre	=	10^{-2}	m	=	10	mm			
m	metre									
km	kilometre	=	10^{3}	m						

Area

m^2	square metre									
ha	hectare	=	10^{4}	m^2						
km^2	square kilometre	=	10^{6}	m^2	=	10^{2}	ha			

Volume

L	liter	=	10^{-3}	m^3						
m^3	cubic metre									
km^3	cubic kilometre	=	10^{9}	m^3	=	1	teralitre			

Mass

fg	femtogram	=	10^{-15}	g								
pg	picogram	=	10^{-12}	g								
ng	nanogram	=	10^{-9}	g								
µg	microgram	=	10^{-6}	g								
mg	milligram	=	10^{-3}	g								
g	gram											
kg	kilogram	=	10^{3}	g								
t	tonne	=	10^{3}	kg	=	10^{6}	g					
Kt	kilotonne	=	10^{3}	t	=	10^{6}	kg	=	10^{9}	g		
Mt	megatonne	=	10^{6}	t	=	10^{9}	kg	=	10^{12}	g		

Chapter 1 Introduction

1.1 Motivation

Global contamination by persistent organochlorine pollutants (POPs) such as DDT and polychlorinated biphenyls (PCBs) is now well documented. After release into the environment, POPs disperse widely across environmental compartments, and the world's oceans gradually become the largest reservoirs of the most persistent chemicals. Their resistance to degradation and the high bioaccumulative capacity jeopardize the health of biological organisms incapable of metabolizing or excreting these synthetic compounds. High-ranking members of food chains such as fish-eating birds and marine mammals are most seriously threatened.

While the developed world has made great strides in controlling organochlorines that were produced mainly between the end of World War II and 1970, there has been growing concern about organochlorines emanating from the developing nations of the Southern Hemisphere and the tropics. Twenty years ago, Goldberg (1975) foresaw the *"southward tilt"* in organochlorine production to developing countries needing cheap biocides for public health (disease vector control) and expanding agricultural production. Recent reports suggest that the trend has occurred, but the global implications remain unclear (Iwata et al., 1994; Simonich and Hites, 1995). It is also claimed that 15% of global PCB stocks reside in developing countries, mainly in old electrical equipment that is expected to end active service in the latter 1990s (Cummins, 1988). Failure to contain and destroy these PCBs will result in significant environmental contamination.

In late 1995, the POPs threat to the marine environment was formally recognized by more than 100 countries in the **Washington Declaration on Protecting the Marine Environment from Land-based Activities** that included a call for "the development of a global, legally-binding instrument on persistent organic pollutants (POPs)"[1]. Under UNEP, a process has been initiated for scientific assessment of 12 specific POPs known as the *"dirty dozen"*. The Washington Declaration also recognized "the need for regular reviews of the state of the world's marine and freshwater environments". The present study has been undertaken in support of the POPs initiative.

[1] UNEP press release, November 7, 1995. See also UNEP/IRTPC www pages regarding IFCS meetings on POPs, Manila, Philippines, 17- 22 June 1996, http://irptc.unep.ch/pops/manila/manexp10.html.

1.2 Study Objectives

1. To estimate global riverine POPs discharges into the world's oceans where sufficient water quality data existed.

2. To qualitatively assess potential POPs discharges from river systems where water quality measurements were unavailable from ancillary data on regional chemical usage and chemical occurrence in other environmental compartments.

3. To summarize the results and identify dominant riverine POPs sources at regional and global scales.

4. To identify monitoring and survey strategies for filling data gaps, and to identify salient technical issues for implementing these strategies.

1.3 Work Plan

Initially, the intention was to rely on POPs data contained in the GEMS fresh surface water quality data base maintained at the UNEP GEMS Collaborating Centre for Freshwater Monitoring and Assessment at the Canada Centre for Inland Waters, Burlington, Ontario, Canada. However, review of GEMS data files found only sparse POPs data outside western Europe. Next an extensive survey was conducted for POPs data reported in readily available scientific and technical literature. The survey identified POPs data for numerous major rivers, and voluminous secondary reports of POPs usage and environmental occurrence in otherwise unrepresented river systems.

Between GEMS data files and literature sources, there was sufficient information to undertake a freshwater assessment that could deliver (a) hard numerical results for numerous major global river systems, (b) plausible qualitative assessments of potential POPs contamination in many other watershed systems, and (c) a list of unrepresented drainage systems and data gaps. Collectively, the results provide a basis for establishing provisional policies and the design of future POPs water monitoring and survey activities in the region.

Following the initial appraisal, work proceeded into the detailed assembly, analysis and interpretation of POPs data from GEMS and literature sources. It was decided to approach the task on a regional basis that would produce regional reports that could stand alone or appear as technical appendices to the final summary report. The Asia-Pacific region was arbitrarily selected for the initial regional study. Several recent reports have suggested that the Asia-Pacific region has become a major global emitter of POPs thanks to rapid development over the past two decades (Iwata et al., 1994; Tatsukawa et al., 1990).

1.4 Selection of Project POPs

Selection of study contaminants was guided by the UNEP *dirty dozen* POPs of the November 1995 Washington Declaration given in Table 1.1. This was modified to the current study POPs list given in Table 1.2. Dioxins and furans were excluded from study because current data availability is limited to certain developed countries and virtually nonexistent elsewhere. The widely used organochlorine pesticide HCH was added because it remains an ongoing concern as a ubiquitous environmental contaminant and a potential human carcinogen. Moreover crude HCH usage has been especially high in east and south Asia where cumulative production and consumption between 1950 and 1990 may have exceed the combined usage of other POPs listed in Table 1.2. Several members of the original UNEP POPs list have been combined into two familial subclasses of cyclodiene insecticides: (1) *drins* [*aldrin, dieldrin, endrin*], and (2) *heptachlor and chlordane*.

The characteristics of these POPs are briefly reviewed in Appendix A, except for HCB, mirex and toxaphene. There are currently insufficient HCB data for Asia-Pacific region to derive meaningful conclusions. Other than limited regional usage data in (Voldner and Li, 1995), there are no Asia-Pacific environmental concentration data for mirex and toxaphene.

Table 1.1 UNEP *Dirty Dozen* POPs list.

dioxins (PCDDs)	aldrin
furans (PCDFs)	dieldrin
PCBs	endrin
DDT	hexachlorobenzene (HCB)
chlordane	toxaphene
heptachlor	mirex

Table 1.2 Current study POPs list.

PCBs	heptachlor and chlordane
DDT	HCB (hexachlorobenzene)
HCH (including lindane)	toxaphene (campheclor) †
drins (aldrin, dieldrin, endrin)	mirex †

† no GEMS or literature environmental data available for Asia-Pacific region.

1.5 Geographic Scope

For present study, the Asia-Pacific region is defined by territories of mainland Asia from Iraq to Russia with direct surface discharge to sea, and includes the major island territories of the western Pacific. Nations and territories are listed in Table 1.3 in three classes. Group A are territories for which quantitative POPs fluxes from rivers could be estimated. Group B are territories with inadequate data for river flux estimation, but with enough information to permit limited qualitative assessment. Group C are territories with no readily available data of any kind.

Table 1.3 Territories of Asia-Pacific region with direct surface discharge to the sea.

A. Data sufficient to estimate POPs river exports		
Iraq	Taiwan	Indonesia (Java only)
India	China	Malaysia (peninsula only)
Thailand	Japan	Australia
Vietnam		
B. Qualitative assessment		
Pakistan	Philippines	Solomon Islands
Bangladesh	Papua New Guinea	New Zealand
Hong Kong		
C. No Data		
Iran	Singapore	Russia
Sri Lanka	Cambodia	North Korea
Myanmar (Burma)	Brunei	South Korea

1.6 POPs Data Sources

The GEMS/Water data base contained useful records only for HCH and DDT in China, and some fragmentary HCH and DDT data for Japan. All other data were culled from scientific literature. The most important literature data set was provided by the regional synoptic surveys of air, surface waters and sediments conducted over December 1989 to December 1991 by Iwata et al. (1994). Geographic coverage included India, Thailand, Vietnam, Indonesia, Taiwan, Australia, Papua New Guinea, and Solomon Islands. Although spatial and temporal sampling coverage was limited and not necessarily representative, the survey provided a nucleus of high resolution

data for four core POPs (PCBs, HCH, DDT and chlordane) that made it feasible to undertake this project.

As surface water POPs concentration data were often meagre or non-existent, reports of POPs usage statistics and POPs occurrence in air, sediment, soil, and biota, both wildlife and human, were examined as indirect information sources. While such data do not permit direct estimation of river exports into coastal waters, they often yield valuable background information on POPs usage patterns, government restrictions and a broad sense of the degree of contamination within and between regions. For India which has a profusion of fragmentary and inconsistent data, it was possible to identify dubious water quality data by cross comparison with POPs concentrations reported in other environmental compartments.

Though data availability appears to be relatively good, much POPs data available for the region is spatially and temporally sparse, temporally dated, of indeterminate representativeness (e.g., potentially biased to agricultural or urban drainage canals, or to the low discharge season), or far upstream from river outlets. Despite data deficiencies, this report presents a reasonably comprehensive image of recent POPs contamination of freshwater ecosystems across the region. Except for developed countries such as Japan that may have abundant data on file, this report summarizes the best data readily available at this time. As such, it also indicates where water quality monitoring and survey infrastructure are weak and need to be strengthened.

1.7 Hydrologic Data

The primary source for river discharge data required for river flux estimation was the Global River Index [GLORI] mean annual discharge volumes and watershed drainage areas for world rivers compiled by (Milliman et al., 1995). However, experience revealed that GLORI data quality is highly variable with numerous errors and omissions. Amongst the problems encountered were:

- GLORI is only a partial compilation of individual rivers. Coverage in many areas is inadequate, particularly for small to medium drainage which in certain areas is the dominant source of surface runoff. A global hydrologic watershed atlas similar to those compiled by India (Rao, 1975) and Australia (AWRC, 1987) is required.

- Some countries' hydrologic data are very poor quality. The worst data in GLORI are the Indonesian river discharges and drainage areas. Coverage is largely limited to Java where surface runoff depth estimates derived by normalizing reported discharges to reported drainage areas are generally 100–1,000% of mean annual rainfall. Using these figures to estimate surface discharge from the entire island of Java leads to gross overestimation of Java water and pollutant discharges. The problem appears to originate with the Indonesian source data.

- Coverage across the Indonesian archipelago, Borneo and New Guinea is almost nil. Thanks to very wet climates, these islands generate huge surface water discharges. As there were no POPs water quality data for the these islands except Java, the problem was not too critical for the present report. Development is accelerating across the islands and the lack of good hydrologic data will be a serious impediment to future water pollution investigations.

Though the most serious deficiencies in GLORI were overcome, attempting to verify GLORI data and searching for alternatives entailed considerable extra effort.

1.8 Report Organization

Chapter 2 presents a summary of regional POPs river fluxes by country, and regional intercomparisons of water, sediment and fish POPs concentrations. Chapter 3 and upwards give the detailed reports for each country.

Chapter 3 on India amounts to a report in itself. As the world's leading producer and consumer of DDT and HCH, data are often at levels not seen or reported in the developed world since 1970. Limited resources, weak field and lab capability, and an often weak sense of how, what, where and when to sample have produced a mass of difficult to digest data that are rarely available where they are most useful for estimating river flux into the coastal margins. Resolving the numerous fragmentary, and at times, contradictory reports of POPs occurrence to make plausible inferences about POPs exports by India's rivers, required detailed study-by-study examination of data, analysis across environmental compartments, and examination of the complex and subtle issues lurking behind high DDT and HCH usage.

In contrast to India, Chapters 4–13 dealing with China, Japan, Iraq, Malaysia, Thailand, Vietnam, Indonesia, Taiwan, Australia and New Zealand are all much shorter. Several remaining countries for which only qualitative data exist are reviewed briefly in Chapter 14.

Chapter 2 Asia-Pacific Region Summary

2.0 Preliminaries

In this chapter, the main project findings are summarized for ***PCBs in Section 2.1***, and ***POPs insecticides in Section 2.2***. PCBs are treated separately because they are industrial contaminants. The summaries include an overview for the POPs class and separate country synopses. By necessity, the country summaries are terse. Detailed information and references are found in the chapters assigned to each country. For major POPs source countries like India and Japan, there is too much important information to be reduced a few paragraphs.

The conclusions presented here are based on:

1. river exports to coastal waters (Tables 2.1–2.2)
2. concentrations in surface waters (Figures 2.1–2.2)
3. concentrations in bottom sediments (Figures 2.1–2.2)
4. concentrations in fish (Figures 2.1 and 2.3)

Site codes and other information needed to interpret Figures 2.1–2.3 have been placed at the end of the Chapter in Sections 2.3–2.4.

The ***river export estimates quantify mass flow capacity to coastal seas*** on national or other geographic bases, while ***POPs concentration data indicate the local intensity of contamination***. Water concentration data are perhaps the most directly informative of local contamination, but indications can be fleeting in dynamic aquatic systems.

Sediments are subject to physical transport, sorting and focusing mechanisms that can lead to over or under indication of local contamination. In appropriate conditions, sediments can better indicate *in-place pollutants* that may exert long term effects.

Fish can: (1) indicate general ecosystem health, (2) facilitate potential human intake estimation which is important in a region where fish are major dietary staples in many countries, and (3) function as time-integrating sensors that accumulate and amplify persistent lipid soluble chemicals that may be difficult to observe in primary media such as air, water and sediments.

Consistent POPs contamination in water, sediment and fish from the same locality can be interpreted as reasonable evidence of the relative strength or weakness of local POPs contamination. Inconsistent evidence may indicate weaknesses in the sampling program, field or lab methods, and other confounding effects.

2.0.1 Caveats

The indications of the available data sources should be interpreted cautiously. Spatial coverage is sparse and sample sizes are mostly small. Some surveys have been flagrantly biased to specific conditions, e.g., wet season, dry season, headwaters far upstream from river outlets, near-bank contaminant plumes, and urban drainage and sewage canals. Most sampling has been *opportunistic* and it is impossible to determine how well sample data represent general local conditions. Often, only incomplete chemical species data are available. For soil, sediment and biotic compartments, investigators often fail to report the measurement basis, i.e., wet, dry or lipid weight, thereby introducing an additional order of uncertainty into the analysis. Several reports present implausible data that mostly seem to result from misreported units (e.g., mg/L instead of ng/L), but occasionally flawed field or laboratory methods. Wherever possible, attempts were made to assess questionable data reliability by cross comparisons between sites, contaminants, and environmental compartments.

For the four core POPs [PCBs, HCH, DDT, chlordane], the river flux estimates in Tables 2.1–2.2 have been summed to give ***cumulative calculable regional river exports*** that only partially account for true cumulative river exports from across the region. Most major sources are represented, but presently indeterminable exports from the Indus River, Malaysian Borneo, Philippines, Indonesia exclusive of Java, the Koreas and China could increase totals perceptibly for some POPs. More complete estimates demand the collection of more extensive data. Until that time, the regional partial totals are useful figures for planning and management purposes.

2.1 PCBs

Table 2.1 summarizes the numerical PCBs river flux estimates by country, and Figure 2.1 presents regional intercomparisons of water, sediment and fish contaminant data that give a broader assessment of relative POPs contamination across the region. Information required to interpret Figure 2.1 is given in Sections 2.3 and 2.4.

2.1.1 PCBs — Synopsis

The cumulative calculable regional river exports of PCBs are estimated at 5 t/a within a plausible range of 3–7 t/a. About 55% of this load originates from Japan, 24% from India, 12% from Vietnam, 3.5% from Java, and the rest from elsewhere.

As a major producer, consumer and repository of existing stocks, ***Japan is the major regional source of PCBs exports (2.8 t/a; plausible range 1.8–3.9 t/a) by surface drainage.*** PCBs levels in water, sediment and fish of urban-industrial Japan (J-Y on Fig. 2.1; J-N on Fig. 2.2; wet weight fish data not shown) rank as the highest or near highest reported in the region. PCBs levels in Japan's rivers have decreased appreciably since 1970, but the decline slowed or stabilized in the 1980s.

Table 2.1 Summary: Annual POPs mass flux (t) to the sea.

	Water Discharge km^3	PCBs flux kg	flux range kg
Japan	422	2,810	1,780 – 3,940
India	1,656	1,200	800 – 2,400
China	1,414	<1,000	–
Vietnam	759	620	470 – 790
Java	146	180	100 – 310
Taiwan	70	71	–
Australia	375	70	60 – 80
Thailand	102	59	–
Peninsular Malaysia	122	55	–
Regional partial total	5,066	† 5,065	3,000 – 7,000

† does not include a flux estimate for China.

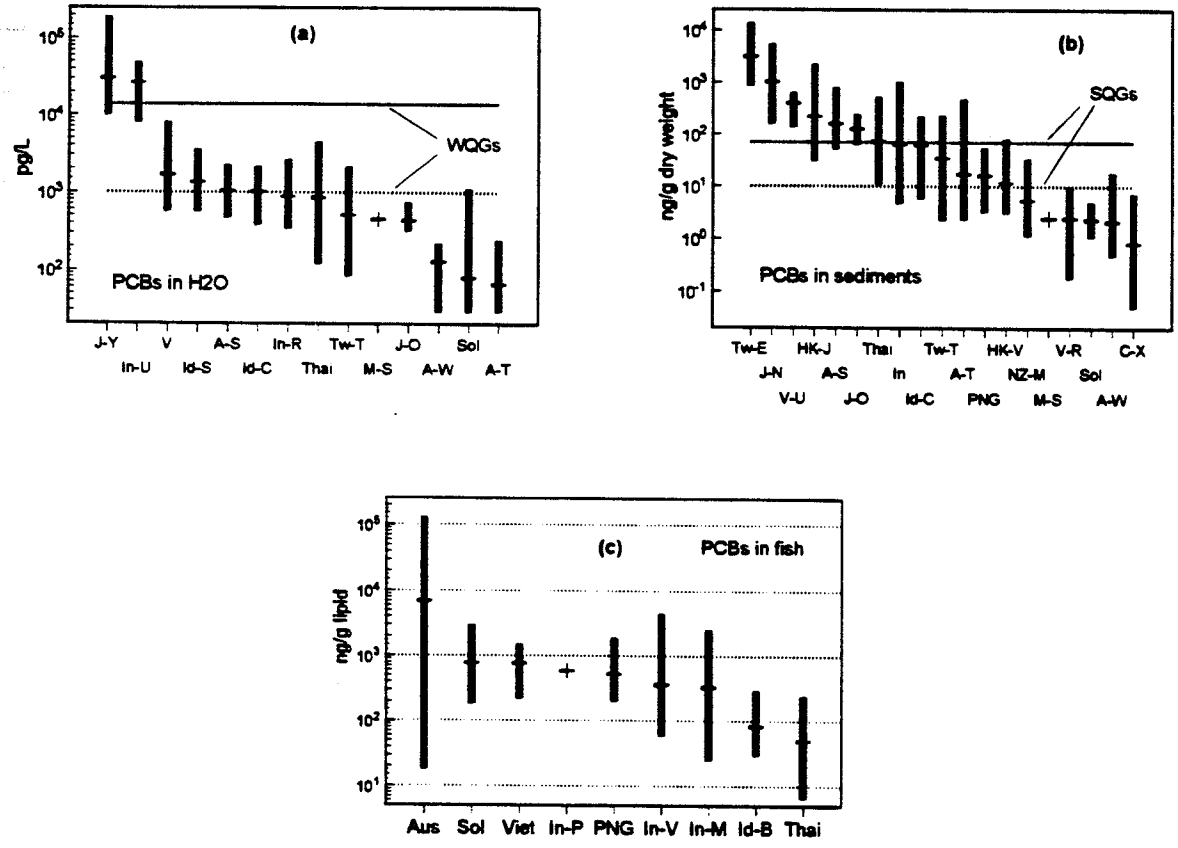

Figure 2.1 Regional variations of PCBs in water and sediments; vertical bars indicate range and cross-bars indicate geometric means; see Sections 2.2–2.3 for site codes and data sources; WQG = water quality guideline; SQG = sediment quality guideline; see Section 2.3.

After Japan, ***India (1.2 t/a; plausible range 0.8–2.4 t/a) and Vietnam (620 kg/a; plausible range 470–790 kg/a) are the next most important regional sources of PCBs.*** It is remarkable that these countries contribute so much. Exports from both countries may be overestimated because available water concentration data are few and do not represent wet season discharges. Nonetheless, ***low to modest PCB contamination is widely present in the environments of India and Vietnam which, by virtue of large water volume discharges, can generate significant riverine exports***. The available water concentration data (Figure 2.1a) show that waters of urban India (In-U) have PCB levels rivaling those of urban-industrial Japan (J-Y), waters of Vietnam (V) rank third, and waters of the rest of India (In-R) rank lower but are statistically indistinguishable from waters of Vietnam. PCB exports for India and Vietnam given in Table 2.1 were developed from the respective concentration means of data sets V and In-R [2]. Using the lowest PCB water concentration reported for the two countries, 340 pg/L, gives total annual PCB river exports as only 430 kg, but this conservative estimate virtually dismisses the influence of major urban complexes including Bombay, Delhi, Calcutta, Madras, Ho Chi Minh, Hanoi and others that are likely the primary PCB source areas.

Collectively, water, sediment and fish data reveal that ***high PCB contamination is found mainly in localities of concentrated urban, industrial and waste disposal activity***. The most contaminated waters are those of the Yodo River (Osaka) [J-Y] and the major cities of India (Bombay, Calcutta, Delhi and Madras) [In-U]. The highest PCBs in sediments are seen in the Er-Jen River, Taiwan [Tw-E] and the Neya River, Japan [J-N], both downstream of waste incineration sites; followed by the sewage canals of Ho Chi Minh, Vietnam [V-U]; Junk Bay, Hong Kong [HK-J] where industrial activity and a large landfill have been cited as possible PCB sources; and Sydney, Australia [A-S] where urban storm runoff and sewage discharges have been implicated.

Otherwise, ***PCB contamination is present at all locations. Even the cleanest sites in Western Australia, PNG, Solomon Islands and Xiamen, China, have some samples with marginally high PCBs that likely indicate small scale local sources.***

2.1.2 Japan

Japan leads PCBs exports with 2.8 t/a (range 1.8–3.9 t/a) that originate mainly in the Osaka-Tokyo urban industrial region of south central Honshu. ***Rivers including the Yodo, Nagaragawa, and Tokyo Bay tributaries are highly contaminated.*** In regional comparisons, *Yodo River water concentrations (J-Y) rank highest amongst all available data sets,* while *sediments of adjacent Neya River (J-N) rank second highest.*

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[2] For both countries, estimated PCB exports originate mainly from territories within national boundaries with annual water volumes of about 910 km^3 for India and 345 km^3 for Vietnam for a total of about 1,255 km^3 for the two. Large water volumes that originate in territories with negligible PCB exposure, i.e., discharges of the Brahmaputra, Meghna and upstream Mekong River inflows were explicitly assigned minimal PCB concentrations. See Chapters 3 and 7 for details.

PCB levels have declined significantly since the early 1970s, but remain high on a regional and global scale. The decline of PCB concentrations in Yodo River water and River Nagaragawa fish appeared to slow or stabilize in the 1980s. These rivers may have many small, difficult to control sources. In the late 1980s, about 29 Kt PCBs remained in service in older electrical equipment. It is not known what measures Japan has implemented to manage the destruction of its PCB stocks.

2.1.3 India

India tentatively ranks second in PCB exports with river flux of 0.8–2.4 t/a. Indian PCBs flux may be overestimated from limited sample data, but current estimates do not adequately account for PCB losses from India's leading urban-industrial centres. ***Four samples from drains and streams of Bombay, Calcutta, Delhi and Madras*** (In-U on Figure 2.1a) ***ranked with Japan's Yodo River as most contaminated in regional water concentration comparisons***. PCB transport into river systems and coastal waters would occur mostly during the wet monsoon. PCBs in storm runoff of coastal cities like Bombay and Madras would be flushed directly to the sea. PCBs in runoff from inland cities like Delhi and Calcutta may be diluted to the point of being difficult to detect in the great wet monsoon discharges of the Ganges and other large rivers.

The four lower level samples used to calculate Indian PCB flux estimates (In-R on Figure 2.1a), ranked about mid range in regional comparison. Though the water concentration data are meagre, ***biotic data also indicate widespread, low level PCB contamination of the Indian environment***. By virtue of size, even at low to modest levels of contamination, India can become a major regional emitter of PCBs. India's PCB stocks are unknown. There may be PCB laden electrical equipment scattered across the country. It is not known what, if any, plans India has for management and destruction of its PCB holdings.

2.1.4 China

The fate of 10 Kt PCBs manufactured by China from 1966-76 is unknown. If these PCBs remain in service, inappropriate disposal could lead to perceptible environmental contamination. PCBs have yet to be measured in China's rivers. On meagre evidence, China's annual riverine PCB exports are thought to be <1 t, possibly <500 kg. Sediments of Xiamen Harbour at the mouth of the Jiulong River showed modest contamination on a few samples, but generally Xiamen sediments had the lowest PCB levels in a regional comparison of 19 localities. An indirect report cites modest contamination of mussels from the coastal waters of Dalian City in the north.

2.1.5 Vietnam

Vietnam is third in terms of estimated annual PCB exports with 470-790 kg/a. ***PCBs in Vietnamese waters, and sediments of Ho Chi Minh sewage canals ranked third in broad regional comparisons.*** Outside Ho Chi Minh, there is sporadic evidence of elevated PCBs concentrations in water, sediments and soils from near Hue, Hanoi and elsewhere, but sample data are limited. Thao et al. (1993) have theorized that elevated PCB levels in the Vietnam environment may be partly attributable to PCBs in munitions, weaponry and defoliants deployed during the Indochina War of 1961-71, and contamination in the vicinity of former U.S. military installations. Otherwise, PCBs were likely imported from USA, Australia, USSR, and possibly France. Vietnam's current PCB stocks are unknown.

2.1.6 Indonesia — Java

After Vietnam, the Indonesian island of Java generates about 100–300 kg annually. PCB contamination in the Jakarta area is not unexpected, but similar levels were seen in the primarily agricultural Solo and Brantas Rivers in 1984. In regional comparisons, Javanese waters and sediments show modest levels of contamination, but sample data are meagre. Indonesian PCB stocks are not known. Better data are required.

2.1.7 Taiwan

Taiwan's rivers are estimated to carry about 70 kg/a PCBs. Limited sampling of Taipei waters and sediments reveals modest PCB contamination that ranks in the mid range seen across the region. However, ***Taipei and area soils had about 5 fold higher PCBs concentrations than seen in Vietnam. Er-Jen River sediments downstream of waste incineration sites had the highest PCB levels in a regional comparison*** (Tw-E on Figure 2.1). Data are inadequate to estimate Er-Jen River PCB load that is likely localized to lands near incinerator sites in the lower reaches of the river. The Er-Jen watershed is acknowledged to be the most polluted in Taiwan. Taiwan's PCB holdings are not known, but could be significant.

2.1.8 Australia

Australia's PCB flux, estimated at 70 kg/a, emanates mainly from the Southeast in drainage from New South Wales and Victoria States. In regional comparisons, PCBs in Sydney area waters and sediments rank as moderately contaminated. ***The Australian aggregate fish sample, which is dominated by Sydney area fish, had the highest PCB concentrations in a limited regional comparison. On wet weight basis, PCB levels in Sydney fish are comparable to fish of River Nagaragawa Japan.*** Contamination is attributed to sewage discharges and storm runoff. The Melbourne

area also has a documented history of PCB contamination. Sediments of the River Derwent on Tasmania show sporadically high PCBs. Elsewhere, PCB contamination is very low. ***Australia is currently formulating a PCB management plan for securely destroying existing PCB stocks.*** Recently, a plant for destroying PCBs and other organochlorines was commissioned for Western Australia.

2.1.9 Thailand

Thai rivers export only about 60 kg PCBs annually. Water and sediment samples from Bangkok area canals (Thai on Figure 2.1) show moderate PCB contamination. ***In comparison to Vietnam and Taiwan, Thai soils were the least contaminated*** averaging only 3 ng/g (dry weight) in contrast to 20 ng/g in Vietnam and 100 ng/g in Taiwan. ***In a limited comparison, Bangkok fresh market fish were the least contaminated of all countries including Papua New Guinea and Solomon Islands.***

Thai PCB stocks are precisely known in terms of the numbers of transformers and capacitors imported by the country's three leading electrical authorities. One third of this equipment is currently in storage pending the development of a disposal strategy.

2.1.10 Malaysia

Peninsular Malaysia's PCB exports were estimated at 55 kg/a on the basis of only a single water sample from an agricultural river. In the late 1980s, fish and shellfish from coastal waters showed modest contamination (20–44 ng/g wet weight). The PCBs inventory is unknown, but may be enough to cause perceptible contamination if not disposed securely. Water samples are needed in the vicinity of Kuala Lumpur and other urban centres. Contamination on Sarawak and Sabah is likely very low.

2.1.11 Hong Kong

There is a known PCB problem in Junk Bay (HK-J on Figure 2.1b) where local industry or a large landfill has produced significant contamination of sediments. Victoria and Tolo Harbours are only modestly contaminated ranking after Port Moresby, PNG area mangroves, and ahead of Manukau Harbour, New Zealand. Hong Kong's PCB inventory is unavailable, but could be significant. **Hong Kong was expected to have introduced a *Code of Practice for Handling, Transport and Disposal of PCBs* in 1988.** The status of the proposed code is not known.

2.1.12 New Zealand

New Zealand's PCBs holdings are low. There has been modest contamination by surface runoff of sediments and biota in Manukau and Tauranga Harbours on the North Island. *Under the New Zealand Ministry for the Environment's Organochlorines Programme* [3]*, comprehensive surveys of air, soils, river water, river biota, estuarine sediments and estuarine shellfish are under way.* The river surveys involved the collection of 3 samples from 16 sites representing about 13% of the country's surface drainage. Sampling was conducted from January to March 1996. River exports can be estimated when data become available.

2.1.13 Papua New Guinea [PNG] and Solomon Islands

Sediment and market fish samples from the Port Moresby area show modest PCB contamination in regional comparisons. Sediment concentrations are comparable to River Derwent, Tasmania and Hong Kong's Victoria Harbour. ***Fresh market fish from Port Moresby have PCB contamination similar to fish of India and Vietnam.***

In regional comparisons, Solomon Islands waters and sediments (Sol on Figure 2.1) had low PCB contamination except for an urban sample from the Mataniko River, Honiara, Guadalcanal that exceeded 1 ng/L. ***Guadalcanal market fish had PCB concentrations slightly higher than Vietnamese market fish.***

Though the PCB stocks of PNG and the Solomons are likely small, PCBs are entering the aquatic environment most likely from insecure handling of old electrical equipment.

2.1.14 Other Countries

Other countries of the region likely to have some PCBs include Iraq, Pakistan, Sri Lanka, Bangladesh, Singapore, Philippines and South Korea. ***Countries with strong western associations, i.e., Singapore, South Korea and the Philippines may have significant PCB holdings.*** Large population centres such as Seoul, Manila, Jakarta, Singapore, Kuala Lumpur, Karachi, Lahore, Baghdad and many cities in India, are more likely to be have PCB stocks or waste disposal sites that serve as significant local sources.

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[3] Organochlorines Programme, Ministry for the Environment, PO Box 10-362, Wellington, Phone (04) 4734 090, Fax (04) 4710 195; see also: http://www.mfe.govt.nz/organo.htm

2.2 POPs Insecticides

River flux estimates are summarized for the three core POPs insecticides (HCH, DDT, and chlordane) in Table 2.2a, and for three other POPs insecticides with limited data in Table 2.2b. Regional comparisons of POPs insecticides' concentrations in water and sediments are shown in Figure 2.2, and in fish in Figure 2.3. Information required for interpreting the figures are given in Sections 2.3 and 2.4.

2.2.1 HCH — Synopsis

The cumulative calculable regional river exports of total HCH isomers are estimated at 282 t/a within a plausible range of 159–456 t/a. About 81% of this load is attributed to India, 15% to China, 2.2% to Japan, 1.2% to Vietnam, and the rest to other countries.

The highest HCH river exports (230 t/a; 120–380 t/a plausible range) originate from India, and the highest HCH levels are found in the waters, sediments, and fish of India (In, In-G, In-V on Figure 2.2, In-V, In-P, In-M on Figure 2.3), the major crude HCH producer and user in the region. It has been reported recently that ***India has now committed to reducing HCH consumption by 30 Kt/a*** from levels that reached 35–45 Kt/a in the late 1980s.

China generates the second highest HCH river exports (42 t/a; 31–55 t/a plausible range). China banned crude HCH in agriculture about late 1983. River exports have declined significantly from a peak of over 1,000 t/a reached in the early 1980s. Nonetheless, appreciable quantities of HCH isomers continue to leach from soils in 1992, and if Japan is a reliable indicator, perceptible HCH discharges will continue for at least another decade.

Japan generates third highest regional HCH river exports (6 t/a; 4–9 t/a plausible range). In the Yodo River, β-HCH isomer now comprises about 73% of the total HCH composition (versus 8–12% in the crude technical mixture). Remarkably, nearly 20 years after prohibition, the most persistent, bioaccumulative and potentially ecologically hazardous HCH isomer is still leaching in appreciable quantity from Japanese soils. This may foretell what can be expected in the waters of other present and past crude HCH users, namely China and India.

Vietnam generates fourth highest regional HCH river exports (3 t/a; 2.6–5.0 t/a plausible range). Vietnamese water concentrations averaging near 10 ng/L coupled with high water discharge produces sizable river exports. The isomer composition indicates that crude HCH is being used. Sediments of Ho Chi Minh sewage canals have the highest reported HCH concentrations. It is not known whether Vietnam manufactures or imports the HCH.

Table 2.2a Summary: Annual POPs insecticide mass flux to the sea.

	Water discharge km^3	ΣHCH t	ΣDDT t	ΣCHL t
India	1,656	† 229 ‡ 98–380	180 66–548	0.2 <0.1–0.5
Iraq	46	—	4.4 2.1–6.2	3.0 2.0–3.7
Peninsular Malaysia	122	0.44 0.056–4.84	4.2 3.4–7.8	0.26 —
China	1,414	42.5 31-56	5 1–2	—
Japan	422 400–443	6.3 4–9	0.4 0.2–0.6	1.4 1.2–2.9
Vietnam	759	3.3 2.6–5.0	0.56 0.38–1.75	0.08 0.05–0.14
Java	146 102–205	0.33 0.10–0.76	0.06 0.04–0.09	0.04 0.01–0.05
Thailand	102	0.031	0.032	0.045
Australia	375	0.07 0.07–0.08	0.02 0.02–0.03	0.04 0.03–0.04
Taiwan	70 66–73	0.014 —	0.006 —	0.003 —
Regional partial totals	5,112	282 159–456	195 73–575	5.1 3.6–7.6

† *medium* estimate; ‡ plausible range on second line where available.
ΣHCH = α-HCH + β-HCH + γ-HCH; sometimes includes δ-HCH.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD; some cases missing *o,p'*-DDT or *p,p'*-DDD.
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

Long term surface water sites monitoring α-, β- and γ-HCH isomers are recommended for countries that have used appreciable quantities of crude HCH.

Though data may not represent current conditions, in the mid 1980s, the Diyala River near Baghdad (Ir-D), Iraq was contaminated with γ-HCH alone at levels that rivaled concentrations of total HCH in Indian waters. Iraq may have been using lindane (γ-HCH) rather than crude HCH. Diyala River fish had γ-HCH levels comparable to the

Table 2.2b Summary: Annual POPs insecticide flux to the sea.

	heptachlor (t)	aldrin+dieldrin (t)	endrin (kg)
India	27 13–62	31 19–45	—
Iraq	2.3 1.5–2.7	2.8 1.9–3.6	45 39–52
Peninsular Malaysia	0.58 0.17–1.42	0.40 0.01–0.99	62 12–122
Thailand	— —	0.23 0.13–0.36	— —
Regional partial totals	29.64 14–66	34.37 21–50	108 51–174

highest total HCH levels in Indian fish. In 1986, little HCH seems to have reached the Shatt al-Arab River where no HCH isomers were detected.

Sporadic water concentrations exceeding 10 ng/L indicate continuing HCH usage in Malaysia (M-W), Vietnam (V), Indonesia (Id-C) and Thailand (Thai). Sediment data from urban Ho Chi Minh (V-U) sewage canals suggest considerable HCH usage in the city. Vietnamese market fish were perceptibly contaminated with HCH.

2.2.2 DDT — Synopsis

The cumulative calculable regional river exports of total DDT isomers are estimated at 190 t/a (plausible range 73–575 t/a). About 92% of this load originates from India, and from 4–5 t/a (2–2.6%) each from China, Iraq, and peninsular Malaysia. Collectively these four countries account for 99.5% of the total calculable regional river exports.

The greatest DDT river exports (180 t/a; plausible range 66–550 t/a) originate from India where the highest regional levels of DDT in water were observed in the Ganges River (In-G, In). Uncertainties in estimates reflect uncertainties in the data. If the low estimate of 66 t/a were valid, India's contribution would still be 90% of the total calculable DDT load. India is the world's leading DDT producer and consumer. Since October 1989, DDT usage is limited to 10 Kt/a for public health purposes, chiefly anti-malarial operations. Most DDT is deployed on the Gangetic plain where the highest malaria rates occur; hence, River Ganga accounts for most (77–87%) of India's DDT river exports.

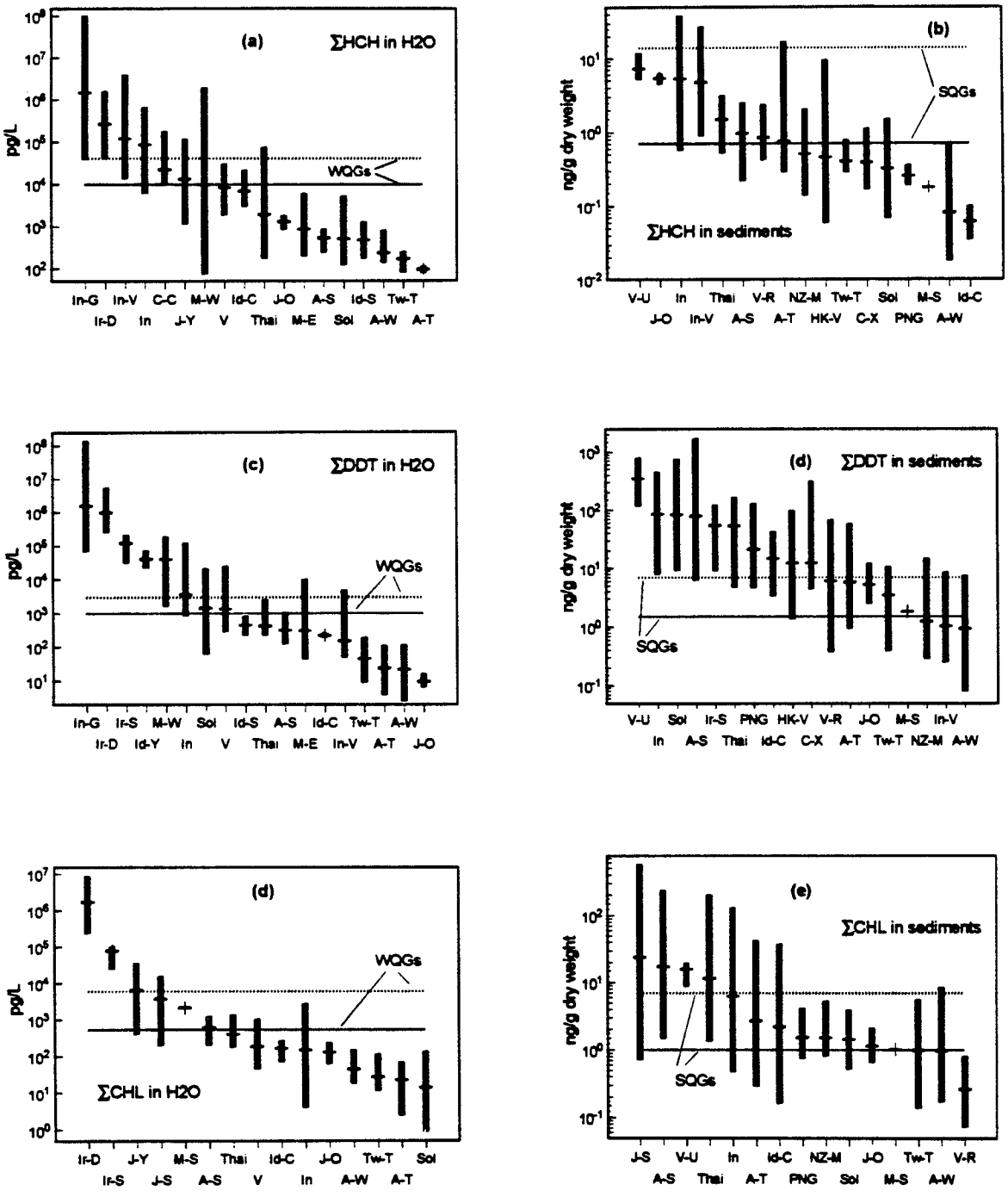


Figure 2.2 Regional variations of HCH, DDT and chlordane (ΣCHL) in water and sediments; vertical bars indicate range and cross-bars indicate geometric means; see Tables 2.3–2.5 for site codes and data sources; WQG = water quality guideline; SQG = sediment quality guideline; see Section 2.3.

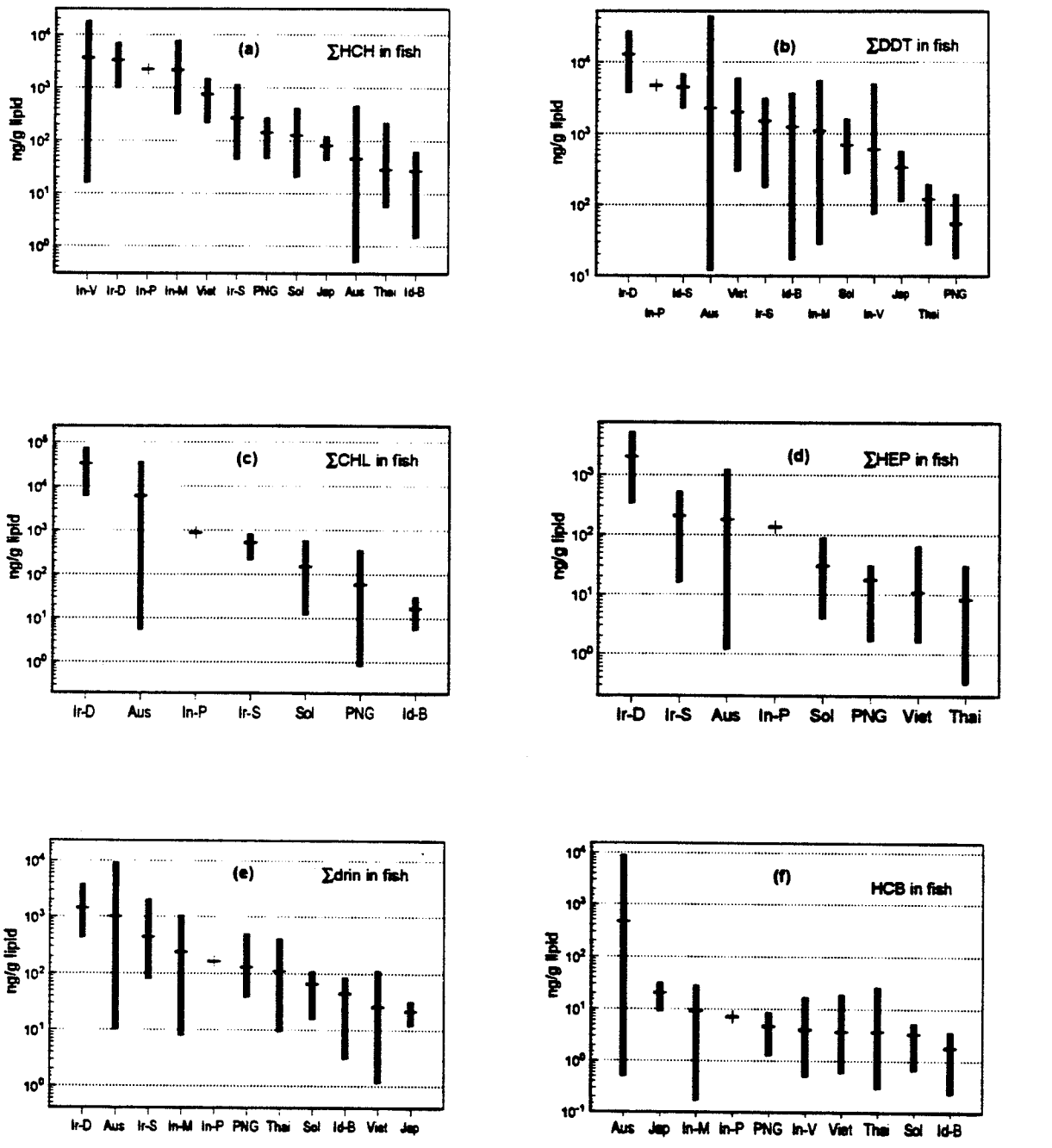


Figure 2.3 Regional variations of POPs insecticides in fish; see Table 2.6 for figure key, source references, and notes. Vertical bars show concentration ranges; cross-bars show arithmetic means; ΣHCH = α-HCH + β-HCH + γ-HCH; ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD; ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor (except Iraq Ir-D and Ir-S which lack nonachlors); ΣHEP = heptachlor + heptachlor epoxide; Σdrin = aldrin + dieldrin.

For practical purposes, China, Iraq and peninsular Malaysia are tied as the second most significant DDT sources with 4–5 t/a each, or collectively about 7% of the cumulative calculable regional river exports.

China is estimated to yield 5 t/a DDT (plausible range 1–11 t/a). Since prohibiting DDT usage in agriculture in 1983, water concentrations in China's rivers have fallen below China's measurement capability. Wet weight food fish concentrations from 1990 rank with fishes from the Ganges (India) and Diyala (Iraq) Rivers. Per capita average daily food intake has fallen only 50% since 1973-78. Thus, significant river exports may persist for some time.

Iraq generated 4.4 t/a DDT (plausible range 2–6 t/a) during the mid 1980s. DDT levels in the Diyala River (Ir-D) near Baghdad rivaled the highest levels seen in the Ganges. In 1984, the Baghdad sewage treatment plant alone may have discharged up to 64 t DDT to the Diyala River. Downstream in the Shatt al-Arab River (Ir-S), DDT concentrations were an order of magnitude lower, but still higher than most other waterbodies across the region. Recent measurements are unavailable.

Peninsular Malaysia, particularly the west coast, ***yielded about 4.2 t/a DDT (plausible range 3.4–7.8 t/a) in 1991. If there is significant DDT use in Sarawak and Sabah, Malaysia may be the second most important DDT source after India.***

Headwater fish ponds near Yogyakarta, Central Java (Id-Y) had higher DDT water concentrations than Malaysia, but it is unclear that these translate into significant river exports of DDT. Vietnam is likely the next most significant DDT source. Current river export estimates of about 600 kg/a may be underestimated if Thailand is using appreciable quantities of DDT for antimalarial operations in its portion of the Mekong River basin. Japanese river exports of 400 kg/a may be overestimated due to lack of recent data. Direct Thai river exports of DDT to the sea may be somewhat higher than the estimated 32 kg/a because exports from streams of peninsular Thailand may be significantly underestimated. Solomon Islands also have high water concentrations suggestive of recent DDT use, but river loads would be low.

Sediments from many locations including Vietnam (V-U, V-R), Sydney (A-S), Solomon Islands (Sol), Iraq (Ir-S), India (In), Bangkok (Thai), Port Moresby (PNG), Hong Kong (HK-V), Bogor/Jakarta (Id-C), Xiamen Harbour China (C-X), and Tasmania (A-T) show elevated DDT levels that may indicate present or past DDT use.

Fish from Iraq's Diyala River (Ir-D) and the mid reaches of the Ganges River (In-P) had the highest DDT burdens. On wet weight basis, 1990 Chinese food fish data have comparable DDT concentrations. Fish from Sydney Australia environs that define the high range of the aggregate Australia sample (Aus), may be equally contaminated. Vietnamese market fish, and Indonesian fish taken from the Solo and Brantas River estuaries of Java (Id-S) in 1984, also had relatively high DDT concentrations.

DDT's value for public health usage, especially in malaria prophylaxis, accords it special status amongst POPs insecticides. Where they are conducted, anti-malarial operations are typically described as *"residual spraying"* on indoor surfaces of domiciles and other structures such as cattle sheds. The present review finds ample empirical evidence that ***in countries where DDT is used for antimalarial operations, DDT residues abound in soils, sediments, surface waters and drinking water supplies.*** *This raises the question:* ***"How does spraying DDT on interior surfaces lead to such high contamination of exterior soils and surface waters?". Ubiquitous environmental data would suggest that spraying and dusting of outdoor surfaces including waters are routine and widespread.***

2.2.3 Chlordane and Heptachlor — Synopsis

The total calculable regional river exports of chlordane are estimated at 5 t/a within a plausible range of 3.6–7.6 t/a. About 59% of this load is attributed to Iraq, 28% to Japan, 5% to peninsular Malaysia, 4% to India, 1.6% to Vietnam, and the rest to other countries.

The total calculable regional river exports of heptachlor are estimated at 30 t/a within a plausible range of 14–66 t/a. About 90% of this load is attributed to India, 8% to Iraq, and 2% to peninsular Malaysia. The heptachlor export estimates are weak. While India made high use of heptachlor in the early 1990s, water data are available only at sites far upstream on the Ganges. Heptachlor can degrade quickly in water; thus, India river export estimates are very uncertain. The heptachlor load attributed to Iraq is also doubtful as the report citing the field data claimed that heptachlor had never been used in Iraq. Oddities in the Iraqi data suggest that field or laboratory problems may lurk behind the unusually high heptachlor reported in the Shatt al-Arab River.

Tentatively, the Shatt al-Arab River, specifically the discharges originating in Iraq, generate the highest chlordane river exports of (3 t/a; plausible range 2–3.7 t/a)*, if current chlordane levels are similar to those seen in 1984-86.* The Diyala (Ir-D) and Shatt al-Arab (Ir-S) rivers circa 1984-86 had extraordinarily high chlordane water concentrations that exceed all other reported data. Diyala River fish had the highest reported chlordane content despite incomplete chlordane isomer measurements.

As indicated in Table 2.2b, the Shatt al-Arab River exported 2.3 t/a heptachlor in 1986, but as cited above, the heptachlor estimate is suspect. An alternative estimate based on total chlordane and heptachlor species concentrations averaged across the two Shatt al-Arab survey sites gives the combined river load as 3.6 t/a within a range of 2–5 t/a. This implies that ***the Shatt al-Arab River exported about 600 kg/a heptachlor in 1986***, and may be a more reliable estimate than the 2.3 t/a obtained by direct calculation. Better data are needed to clarify chlordane and heptachlor exports from the Shatt al-Arab River.

Japan's rivers yielded the second highest river exports of chlordane (1.4 t/a; plausible range 1.2–2.9 t/a). Japanese waters of the Yodo River (J-Y) and the streams of Saga City, Kyushu (J-S) had the highest water concentrations after Iraq, and sediments of Saga City streams had the highest reported chlordane levels. On wet weight basis, fish from River Nagaragawa had chlordane concentrations comparable to fish from the Shatt al-Arab River.

The estimate of peninsular Malaysian chlordane river exports (256 kg/a) is based on a single water sample and should be regarded with due caution. However, in the early 1990s, peninsular Malaysia rivers were exporting about 575 kg/a heptachlor within a plausible range of 170–1,400 kg/a; so that, chlordane exports associated with high heptachlor usage could approach the 256 kg/a estimate. ***Had complete measurements of all relevant chlordane and heptachlor species been extensively available, it is possible that the combined chlordane and heptachlor river exports from peninsular Malaysia in the early 1990s could have approached 1 t/a.*** If heptachlor and chlordane are being used on Sarawak and Sabah, Malaysia's total heptachlor and chlordane river exports could be appreciably higher.

In the early 1990s, Indian's rivers were estimated to deliver 27 t/a heptachlor (plausible range 13–62 t/a), but only 200 kg/a chlordane (plausible range 100–500 kg/a). As mentioned above, the heptachlor export estimates are weak. Nonetheless, data from the mid upper reaches of the Ganges clearly show that India was using heptachlor there at high rates, most likely as a soil insecticide. The low chlordane exports may have originated from heptachlor usage. ***India banned both heptachlor and chlordane in the early 1990s.***

Australian river exports of chlordane are quite low, but ***sediments and fish of Sydney, Australia (A-S), have chlordane concentrations that rank closely with Japan and Iraq***. Australia has used appreciable chlordane and heptachlor as termiticides. Significant contamination may enter local fresh and nearshore coastal waters of southeast Australia (New South Wales and Victoria) via urban storm runoff and sewage effluents that cannot be estimated from available data. ***Termiticidal use of chlordane and heptachlor was prohibited in Australia in mid 1995.***

Otherwise, high chlordane concentrations, which may indicate past or present use, have been observed sporadically in Bangkok canals (Thai), Vietnamese (V) surface drainage, sewage canals of Ho Chi Minh (V-U), River Derwent, Tasmania (A-T) and Jakarta (ID-C, Ciliwung River).

2.2.4 Drins — Synopsis

The total calculable regional river exports of ∑drin [aldrin+dieldrin] in the early 1990s were estimated at 34 t/a within a plausible range of 21–50 t/a. About 90% of this load is attributed to India, 8% to Iraq, 1% to peninsular Malaysia, and the rest to Thailand.

India had the greatest ∑drin exports at 31t/a (plausible range 18–49 t/a), but estimates are uncertain because water concentration data are only available at sites in the mid upper Ganges basin. The ∑drin river load originated primarily from usage of aldrin as a soil insecticide. ***India banned aldrin in the early 1990s***, so river exports should decline quickly. Perceptible dieldrin exports may persist for some time. The aggregate Indian fish sample (Figure 2.3e) shows that Indian fish generally are less contaminated than Shatt al-Arab River fish. However, an independent 1988-89 sample of Delhi (River Jamuna) fish [see Chapter 3, Fig. 3.6D], had wet weight concentrations approaching those in River Diyala fish.

Iraq had the second greatest ∑drin exports at 2.8 t/a (plausible range 1.9–3.5 t/a) via the Shatt al-Arab River as indicated by 1986 data. Diyala River fish had the highest ∑drin levels. Contamination was lower, but nonetheless significant in Shatt al-Arab River fish.

Peninsular Malaysia had the third greatest ∑drin exports at 400 kg/a (plausible range 12–990 kg/a). If aldrin and dieldrin were being used on Sarawak and Sabah, the total Malaysian river exports could be somewhat higher.

Thailand had the fourth greatest ∑drin exports at 230 kg/a (plausible range 12–122 kg/a) as estimated for 1991. Before prohibition in 1988, Thai river exports were estimated at 1.4 t/a. Thus exports seem to have declined rapidly in three years to about 1/7 of 1988 exports. Perceptible dieldrin residues may persist for some time. Bangkok area fish had perceptible dieldrin contamination. Between 1989 and 1991, ∑drin concentrations in mussels from 9 nearshore and estuarine sites along the Gulf of Thailand coast declined by 35-40%,

Despite low ∑drin river exports, Australian fish, specifically from the Sydney area, had ∑drin concentrations rivaling those in fish from the Diyala River, Iraq. Contamination seems to originate from persistent residues of past high dieldrin usage in southeast Australia.

Endrin was reported for only two countries: Iraq during 1984-86, and peninsular Malaysia in 1989. The combined river exports are estimated at only 108 kg/a. However, Iraqi data did indicate high levels of endrin contamination in upstream waters and biota of the Euphrates and Diyala Rivers. Evidently the endrin either degrades or volatilizes rapidly before reaching the Shatt al-Arab River.

2.2.5 India

India was the leading source of DDT, HCH, aldrin+dieldrin and heptachlor in the early 1990s. India is the region's leading producer and user of DDT and crude HCH. During the 1980s, DDT usage averaged about 8–10 Kt, while HCH usage averaged about 35 Kt/a. Since October 1989, annual DDT usage has been limited to 10 Kt for public health purposes. HCH production rose steadily over the past decades to approach 40–45 Kt by the late 1980s. About 7–8 Kt HCH were used for public health, while the remainder was widely used in rice cropping or as general purpose insecticide.

DDT and HCH are used for control of vector borne diseases including malaria, Japanese encephalitis, dengue, filaria and leishmaniasis (Kala Azar). Anti-malarial operations consume most insecticide. India's public health usage of POPs insecticides involves complex issues of water management, public sanitation, insect resistance and other factors that are elaborated in more detail in Chapter 3.

Up to 80% of the 66–548 t/a DDT riverine exports are transported by the Ganges River. The greatest incidence of rural malaria is in the upper Gangetic plain; hence, most DDT is used there. Because the few data available for the lower reaches of the Ganges are uncertain, potential DDT exports from the Ganges alone could range from 8 to >1,000 t/a, more widely than the subjectively selected *most plausible* range given in Table 2.2. To resolve uncertainties, ***monthly monitoring stations are needed at well-mixed cross-sections on the Ganga main branch either at Farraka or the Hardinge bridge upstream of the Bangladesh frontier, and on the Hooghly River distributary below Calcutta.***

HCH usage is more evenly distributed across India; hence, peninsular India contributes a greater share (25–45%) of the 98–380 t/a total riverine exports.

In the early 1990s, annual river exports of heptachlor and aldrin+dieldrin were each about 30 t. The primary source was usage of heptachlor and aldrin as soil insecticides in the upper and mid Gangetic plain. ***Aldrin, heptachlor and chlordane were banned for agriculture during 1992-94***; thus, river exports should now be declining significantly. Dieldrin residues may persist for some time.

DDT and HCH usage raises other environmental concerns beyond river exports to coastal seas. There is widespread contamination of surface waters, drinking water supplies, biota, foodstuffs and humans across the country. Though it is formally limited to public health operations, ***DDT is found in abundance in surface waters and drinking water supplies. Nominally, anti-malarial operations are restricted to indoor residual spraying; however, environmental data suggest that spraying and dusting of outdoor surfaces including water are routine and widespread.***

2.2.6 Iraq — Shatt al-Arab River

Iraq was the leading source of chlordane, and after India, the second leading source of DDT, aldrin+dieldrin and heptachlor. This is remarkable because the Iraqi portion of Shatt al-Arab water discharge (29 of 46 km^3) is the least of all the territories considered in Table 2.2. ***For almost all POPs insecticides for which environmental data exist, Iraqi water, sediments and fish have amongst the highest concentrations reported for the region.*** POPs concentrations upstream in the Diyala River near Baghdad and the Hor al-Hammar marsh in the lower Euphrates were much higher than in the Shatt al-Arab River, suggesting that fluxes from the latter, which are based on dry season concentration data, may be too low. However, slow moving conditions in the marshy lower reaches of the Tigris and Euphrates Rivers may favour high volatilization losses before waters reach the Shatt al-Arab River. That possibility is supported by an analysis of marine biota from Kuwaiti coastal waters near the Shatt al-Arab outlet that showed very low DDT concentrations.

Iraq was one of two countries where perceptible endrin fluxes were observed. Though the annual flux from the Shatt al-Arab River was only 40–50 kg, there appeared to be discharges approaching 11 t/a through the Baghdad sewage treatment plant to the Diyala River tributary of the Tigris in 1984, and high contamination of biota in the Hor al-Hammar lake of the lower Euphrates River.

Iraq used organochlorine insecticides extensively against malaria and in agriculture for 25 years [beginning about 1950] before nominal banning in 1976. However, the ***environmental data unequivocally indicate that high POPs insecticide usage continued into the mid 1980s***, and may well continue today. Iraq's political and economic isolation, oil resources and petrochemical industry, may favour the continued production and consumption of cheap organochlorine biocides.

2.2.7 China

China banned HCH and DDT use in agriculture about 1983 after years of usage that was particularly high in the case of HCH. Recent data suggest that some usage of DDT continues, possibly as a public health insecticide. It would be helpful knowing the precise status of DDT and HCH, and, if some restricted usage continues, the quantities used. Other than toxaphene, which was banned in 1975, it is not known if China has used other POPs insecticides.

After India, China was the leading of source of HCH at 30–52 t/a over 1991-92. This represents lingering HCH residues from the years of intensive HCH usage before 1984. From available records that begin in 1980, ***HCH river exports peaked from April 1982 to March 1983 at 780 (1020) t*** where the figure in parentheses applies if the Chinese ΣHCH data in GEMS files did not include β-HCH isomer. By 1991-92, exports had fallen to 39 (52) t and were declining slowly. Considering Japan's

experience, it is possible that appreciable HCH residues may continue to leach from Chinese soils for decades, and that β-HCH may comprise an increasing fraction of the total HCH content. In 1991-92, the Changjiang (Yangtze) River accounted for about 59%, and the Xijiang (Pearl) River for about 33% of total China HCH flux.

The Chinese DDT flux for 1991-92 was estimated crudely by assuming that exports of 10–20 t estimated for 1980 [the only year with sufficient data above detection limits to calculate a direct estimate] declined by from 1/10 [approximately the decline in HCH river loads] to 1/2 [approximately the decline in average daily per capita food intake]. The average among alternative estimates of total China river flux is 5 t/a within a plausible range of 1-11 t/a. Recent food fish data show very high DDT levels that suggest significant river loads may persist for some time. At probable current levels, detection capability down to 100 pg/L or less is needed to quantify the major DDT species in Chinese river waters.

2.2.8 Japan

Japan ranks second to Iraq in chlordane exports with 1.2–2.9 t/a, and third after India and China in HCH exports with 4–8 t/a. The chlordane exports are expected as Japan imported 17.5 Kt, mostly in the decade preceding prohibition in 1986. Chlordane was used primarily for termiticidal timber treatments. DDT exports estimated at 200–600 kg/a may be high as recent data were unavailable.

The relatively high HCH river flux is noteworthy because Japan banned HCH in 1971-72. Japan produced about 400 Kt HCH before prohibition, and annual usage reached about 45 Kt during the late 1960s. The intensive usage during those years appears to be responsible for continued leaching of HCH residues twenty years after prohibition. With reference to the Yodo River that had HCH concentrations of 20–25 ng/L in 1988, Tanabe et al. (1989) argued that because HCH is generally more mobile than other POPs, it should have largely disappeared by the mid 1980s, and they go on to suggest that HCH levels were sustained by long range atmospheric inputs from China, India and elsewhere in southeast Asia where HCH may have still been in heavy use. However more ***recent Yodo River data have a unique composition dominated by 73% β-HCH*** which is the HCH isomer that is most persistent, least volatile and most likely to accumulate in soils, sediments and biota. This suggests that ***the continuing presence of HCH in Japanese waters is more likely due to persistent domestic HCH residues in soils and sediments rather than external sources.***

2.2.9 Malaysia

Annual DDT exports by Peninsular Malaysian rivers of 3–8 t rank about second with China and Iraq behind India. With development accelerating in Sarawak and Sabah, the Malaysian total could be significantly larger.

Malaysia appears to be using most leading POPs insecticides including HCH, heptachlor, chlordane, aldrin, dieldrin and endrin. Peninsular river exports of HCH, heptachlor and aldrin+dieldrin are each about 400–600 kg/a, those of chlordane about 250 kg/a, and those of endrin about 60 kg/a. Export estimates are uncertain. Seasonal usage patterns and quantities are not known, nor were sampling dates given for the available data. The true HCH exports could approach 5 t if the available data represent the off-usage, low water concentration season. Continuous monitoring data are needed. If these chemicals are being used in Sarawak and Sabah, the total Malaysia exports could be much higher.

2.2.10 Vietnam

Vietnam has been using HCH and DDT in perceptible quantities, and lower amounts of other POPs insecticides. River exports were greatest for HCH with 2.6–5 t/a. HCH has likely been used mainly in rice cropping. DDT exports were next with about 400–1,750 kg/a. DDT is most likely used for public health, but may also be used in agriculture. DDT inflows from upstream Mekong River countries [assumed negligible in present work], may raise the estimated total Vietnamese river exports to 1 t/a or more. Chlordane exports were relatively low at under 100 kg/a.

In regional comparisons, Vietnamese water data for HCH, DDT, and chlordane generally ranked in the mid range showing moderate contamination. Vietnamese sediments showed a distinct urban-rural dichotomy. ***Sediments from Ho Chi Minh sewage canals had amongst the highest concentrations of HCH, DDT and chlordane in regional comparisons***, while sediments from outside Ho Chi Minh had modest to low contamination. Sediments of other major cities were not represented. Regional comparisons of POPs concentrations in market fish show that chlordane, heptachlor, and aldrin+dieldrin levels are generally low.

2.2.11 Thailand

Thailand has progressively banned most POPs insecticides for agricultural uses; hence, the river exports are all low, generally under 100 kg/a. ***Thailand retains DDT and lindane (γ-HCH) for selected uses.*** From 1987-90, 90 t/a DDT were used for malaria control in Thai provinces with drainage to the Gulf of Thailand. DDT usage increased in peninsular Thailand during that time, and the increase was observed in the DDT concentrations seen in mussels from coastal waters. Current river exports

figures may have significantly underestimated DDT exports from streams of peninsular Thailand. If DDT were used at comparable rates in Thailand's portion of the Mekong River basin, there may have been significant DDT exports via the Mekong River that have not been factored into current Vietnamese DDT river export estimates.

Dieldrin, which was prohibited in 1988, still has river exports of about 250 kg/a. Aldrin+dieldrin concentrations in Chao Phraya waters and mussels from Gulf of Thailand coastal waters have been declining. In regional comparisons, Thai water sample data for HCH, DDT and chlordane rank in the mid range showing modest levels of contamination; however, the sediment data, which are skewed toward Bangkok canals, rank moderately high. POPs insecticide concentrations in Bangkok market fish generally rank very low in regional comparisons except for aldrin+dieldrin.

2.2.12 Indonesia — Java

The status of POPs insecticide usage in Indonesia is unclear. In 1986, after severe rice crop failures in the mid 1980s due to pest resistance to heavily used insecticides, Indonesia prohibited 57 pesticides in rice cultivation and nationally promoted an *integrated pest management* [IPM] policy. While a recent report claims that there are now 10,000 IPM field schools in the country and that chemical pesticide use in rice agriculture has declined, other reports claim that Indonesia's pesticide consumption jumped from 28 Kt in 1991 to 41 Kt in 1993, and that organochlorine use has increased in rice cultivation. The extent of public health usage of POPs insecticides in Indonesia is not known. With development accelerating in Sumatra, Kalimantan, Irian Jaya, and elsewhere across the archipelago, the potential for expansion of POPs insecticide use in agriculture and public health is considerable.

POPs environmental concentration data are available only for Java. Estimated Java river exports based on these limited data are very low. In regional comparisons, the available water concentration data showed modest to low levels except for DDT in fish pond waters near Yogyakarta, central Java. There, in 3 of 5 ponds in villages that had been sprayed with DDT in anti-malarial operations 8 and 25 years previously, DDT concentrations of pond waters exceeded 20 ng/L, and the isomer profile indicated predominantly unmetabolized forms. Either there was recent unreported DDT use, or there are unusual soil conditions that inhibit DDT degradation.

With contradictory claims regarding POPs insecticide usage in Indonesia, and only meagre, divergent environmental data, ***much better environmental surveys are needed to characterize current conditions on Java and across Indonesia.*** The poor quality of Indonesian hydrologic data is a serious impediment to the study of water pollution. Most reported annual river discharges when normalized to reported drainage areas, yield mean annual runoff estimates grossly in excess of mean annual rainfall. In present work, the reported Indonesian hydrologic data were discarded, and replaced by

crude runoff estimates derived from precipitation data. ***Good quality hydrologic data are needed on Java and across the Indonesian archipelago.***

2.2.13 Taiwan

Taiwan prohibited most organochlorine insecticides about 1975, and is not a major regional source of these. In regional comparisons, waters and sediments of Taipei canals showed low POPs insecticide contamination. A recent soil survey showed somewhat high HCH levels in soils of Taipei and adjacent areas. This may indicate some persistent, possibly illicit usage of crude HCH.

2.2.14 Australia

Historically, Australia has used most POPs insecticides including DDT, aldrin, dieldrin, heptachlor, chlordane and HCH. DDT appears to have been widely used in the 1950s; then dieldrin became popular in the 1960s. In the 1970s, aldrin, heptachlor, chlordane, HCH, and lindane (γ-HCH) usage became more prevalent. ***High usage of most POPs insecticides, except chlordane and heptachlor, appears to have ended by 1985.*** DDT, endrin, HCB, and crude HCH had no permitted uses circa 1994, while selected usage of lindane, aldrin, dieldrin, heptachlor, chlordane and mirex continued in some states. ***In mid 1995, heptachlor and chlordane were prohibited except in the Northern Territory which was given to June 30, 1997, to phase out stocks.*** Currently (October 1996) only lindane and mirex remain registered for limited uses in Queensland, Western Australia and Northern Territory.

Estimated river exports of POPs insecticides from Australia are very low. For DDT and chlordane, the little flux that does occur originates mainly from the southeast drainage division (New South Wales and Victoria), while the HCH flux seems to be more evenly distributed. Internal comparisons of available water concentration data from Sydney area streams, River Derwent, Tasmania, and the environs of Perth, Western Australia, show Sydney area waters and sediments to be most contaminated with DDT and chlordane. ***In regional comparisons, Sydney area sediments are second only to Saga City, Japan sediments in chlordane contamination, and are tied with Indian and Solomon Island sediments for second place in DDT contamination, marginally lower than Ho Chi Minh sewage canal sediments.***

A survey of locally-caught fresh market fish from 6 urban centres shows Sydney area fish to be the most contaminated with chlordane, DDT, aldrin+ dieldrin and HCH. Hobart and Brisbane vie for second place, while fish from Perth, Western Australia and two small northern Queensland towns have very low contamination. ***In regional comparisons, the Australian aggregate sample that is dominated by the Sydney fish, vies with Iraqi and Indian fish for highest levels of chlordane, heptachlor, DDT, and aldrin+dieldrin.***

Most notably, ***Australian fish, have 1-2 orders of magnitude higher levels of HCB than fish from 6 other countries***. HCB data were not given for the 6 city Australian survey, but it is likely that again, Sydney area fish are responsible for the high HCB levels. ***Industrial sources rather than fungicide usage are suspected.***

2.2.14.1 Organochlorine Management Plans

Since 1994, Australia has been in the process of developing management plans aimed at the ultimate destruction of existing stocks certain hazardous materials and wastes. Recent bulletins [4], have announced ***specific plans are under development for organochlorine pesticides, HCB and PCBs.*** Recently, Australia commissioned a plant near Perth to destroy stockpiles of organochlorine pesticides (predominantly DDT) and PCBs [5].

2.2.15 Papua New Guinea and Solomon Islands

No river flux exports could be calculated, but water, sediment and market fish data for PNG and the Solomons indicate low level usage of HCH, heptachlor, chlordane, aldrin and dieldrin. The exception is DDT in the Solomon Islands where ***Guadalcanal sediments that are tied for second highest DDT levels*** with sediments of India and Sydney Australia after Ho Chi Minh sewage canal sediments. PNG sediments of Port Moresby area mangroves rank lower in DDT after Bangkok canals and ahead of sediments of Jakarta/Bogor, Java. ***Port Moresby area fish have aldrin+dieldrin levels approaching those of Ganges River fish.*** In regional comparisons, Solomon Islands water and fish samples exhibit mid range levels of DDT.

2.2.16 New Zealand

Most POPs insecticides were likely used in New Zealand. Available sources suggest that DDT, dieldrin and chlordane saw most usage, but data are inadequate to estimate river exports. DDT and dieldrin appear to have been long prohibited, but ***chlordane has been used until recently as a termiticidal timber treatment and in plywood adhesives.*** New Zealand had over 400 timber treatment yards that may be sources of chlordane runoff, but it is unknown how many actually used chlordane.

Comprehensive environmental sampling of POPs in air, soils, river water, river biota, estuarine sediments and estuarine shellfish is now under way by New Zealand's Ministry for the Environment[1]. River surveys of 16 sites representing

[4] Summer 1996, Australian Environmental Protection Agency; see http://www.erin.gov.au/portfolio/epa/publications.html.
[5] See http://www.mfe.govt.nz/organo.htm.

about 13% of the country's surface drainage were conducted from January to March 1996. Results can be expected in the forthcoming months.

2.2.17 Other Countries

Most other countries in the region have likely used POPs insecticides at some time present or past. Some like South Korea and Sri Lanka nominally prohibited the major organochlorine insecticides long ago. Other countries like Myanmar, and upstream Mekong River countries, i.e., Laos and Cambodia, remain relatively undeveloped with little pesticide use. The situation in North Korea is unknown.

Iran, whose Karun River accounts for 40% of the total Shatt al-Arab River water discharge, may have been a past contributor of several *dirty dozen* POPs insecticides. It is reported that since 1985, only lindane (γ-HCH) remains in use.

The Philippines appears to have been a significant past user of DDT, dieldrin and endrin. Informal reports claim that, as of 1995 or earlier, DDT, crude HCH, aldrin, dieldrin, endrin, heptachlor and toxaphene were banned, while restricted usage of chlordane and lindane was permitted.

In the mid 1980s, Pakistan was producing about 2.6 Kt each of DDT and HCH. Endrin was reported banned in 1993, while informal reports suggest DDT and toxaphene have also been banned in recent years. The status of HCH, aldrin, dieldrin, heptachlor and chlordane is not known. The Indus River, in addition to contamination generated by Pakistan, also carries POPs insecticides, particularly HCH and DDT, that originate in India. Due to prohibitions in Pakistan, the POPs insecticide flux from the Indus may not be large. ***POPs survey data are needed for the lower Indus River.***

Though it remains a minor user, Bangladesh pesticide consumption increased steadily through the 1980s. In 1989, Bangladesh used about 400 t of unspecified organochlorines but these may have comprised mostly endosulfan and other low priority POPs insecticides. While DDT has been long banned there is evidence of some black market usage. Crude HCH, lindane and endrin have been banned for some time. The status of other high priority POPs insecticides is not known.

2.3 Water and Sediment POPs Data: Figures 2.1–2.2

Data for most sites in the regional comparison were taken from the synoptic survey conducted by Iwata et al. (1994). Water bodies ranging from small to large streams, urban drainage and sewage canals, paddy fields, mangroves and estuaries were sampled. Data for additional sites were culled from the sources reviewed in this report.

Data were reduced to geometric means (except for a few cases where only arithmetic means were given) and ranges by country or regional groups as given in Table 2.3.

Often, raw data contained numerous results reported as non-detections. These were treated by fitting log-normal probability models to estimate geometric means. Where possible, models were fitted to local data subsets, or if necessary, the whole data set. Generally, models were fitted independently to specific isomers before totals (e.g., ΣDDT) were determined. Model estimates of the expected value of non-detect data were taken as quantitative estimates of minima for plotting purposes.

Generally, the additional data either possessed the standard chemical species measurements employed by Iwata et al. (1994), or were adjustable to approximate equivalence with the standard definitions by empirical relations derived from the Iwata data set. Exceptions are noted in Table 2.4. In some cases, data from more than one source were combined to reduce graphical clutter. For example, ΣDDT and ΣHCH measurements for Iwata et al.'s single water sample for the Selangor River of Malaysia [M-S] were merged with western peninsular Malaysian stream data [M-W] of Tan and Vijayaletchumy (1994). For Manukau Harbour, New Zealand [NZ-M], ΣDDT observations from the studies of Fox et al. (1988) and Holland et al. (1993) were combined, but otherwise data from Fox et al. (1988) were used.

For countries with multiple water bodies represented by multiple data sources, additional data were selected to represent broadly the range of conditions across the country as best as possible. For example, in India, data for Ganges River at Varanasi were chosen to represent upper extremes observed in the Ganges under full monsoonal flooding conditions, while the Vellar River was chosen to represent humid subtropical India where HCH usage predominates and DDT is little used.

River Neya, Japan [J-N], and Er-Jen River, Taiwan [Tw-E] sites were included to show the extreme levels of localized PCB contamination that exist in industrialized countries.

2.3.1 Water and Sediment Quality Criteria

Selected water and sediment quality guidelines from a compendium compiled by MacDonald (1994) were superimposed on presentation graphics as an approximate qualitative aid to interpretation. These are listed in Table 2.5. The selections are not intended as an endorsement. The criteria were chosen somewhat arbitrarily to reflect the typical range quality guidelines developed for aquatic life protection in temperate North American waters. These criteria may be inappropriate for tropical and sub-tropical conditions, but in the absence of local guidelines, they may tentatively indicate where significant contamination problems exist.

Table 2.3 Site codes for Figures 2.1 and 2.2; see Table 2.4 for Sources.

Codes	Country	Locality	Source
A-S	Australia	Sydney and environs	1
A-T		Tasmania, River Derwent	1
A-W		Western Australia, Perth and environs	1
C-X	China	Xiamen Harbour	2
C-C		Changjiang R 1991-92 GEMS data	3
HK-J	Hong Kong	Junk Bay	4
HK-V		Victoria Harbour	2
Id-C	Indonesia	Ciliwung R, Java; Jakarta / Bogor	1
Id-S		Surabaya area, Java; Solo / Porong Rivers, Surabaya-Madura Strait	5
Id-Y		Yogyakarta, Java area fish pond waters	6
In	India	synoptic; 8 water, 7 sediment samples	1
In-U		urban waters; 4 samples, PCBs in water only	1
In-R		rest of India; 4 samples; PCBs in water only	1
In-G		Ganges R at Varanasi; Aug 1992	7
In-V		Vellar R, Tamil Nadu	8,9
Ir-D	Iraq	Diyala R near Baghdad	10
Ir-S		Shatt al-Arab R	11
J-O	Japan	Osaka Bay	1
J-N		Neya R, Osaka City	12
J-Y		Yodo R, Osaka City	13,14
J-S		Saga City, Kyushu	15
M-E	Malaysia	peninsular east coast streams	16
M-W		peninsular west coast streams	1,16
M-S		Selangor R (included in M-W for HCH and DDT)	1
NZ-M	New Zealand	Manukau Harbour, Auckland, North Island	17,18
PNG	Papua New Guinea	Port Moresby area	1
Sol	Solomon Islands	Honiara area, Guadalcanal	1
Thai	Thailand	Bangkok canals / lower Chao Phraya R	1
Tw-T	Taiwan	Taipei canals	1
Tw-E		Er-Jen R	19
V	Vietnam	synoptic	1
V-U		Ho Chi Minh sewage canals	1
V-R		rest of Vietnam	1

Table 2.4 Data sources for Table 2.3 (W = Water, S= Sediment).

Source No.	Medium	Date	POPs	Reference
1	W,S	1989-91	PCBs, DDT, HCH, CHL †	Iwata et al., 1994
2	S	1992-94	PCBs, DDT [a], HCH	Hong et al., 1995
3	W	1991-92	HCH	GEMS data
4	S	?	PCBs	Kannan et al., 1989
5	W	1984	PCBs, DDT [a], HCH	Hillebrand et al., 1989
6	W	1989	DDT	Noegrohati et al., 1992
7	W	Aug 1992	DDT [b], HCH [c]	Nayak et al., 1995
8	W	1988-89	DDT, HCH [c]	Ramesh et al., 1990
9	S	1988-89	DDT, HCH [c]	Ramesh et al., 1991
10	W	1984	DDT [d], HCH [e], CHL [f]	Al-Omar et al., 1989
11	W,S	1986	DDT [d], HCH [g], CHL [f]	DouAbul et al., 1988
12	S	1985-86	PCBs [h]	Miyata et al., 1988
13	W	1986-88	PCBs	Tanabe et al., 1989
14	W	1990	HCH, CHL [j]	Yamaguchi et al., 1992
15	W,S	1987-88	CHL [k]	Hirai and Tomokuni, 1989
16	W	1990-91	DDT [l], HCH	Tan and Vijayaletchumy, 1994
17	S	?	PCBs [m], DDT [n], HCH, CHL [o]	Fox et al., 1988
18	S	1989	DDT [p]	Holland et al., 1993
19	S	?	PCBs	Ling et al., 1995

† standard measurements by Iwata et al. were:
ΣHCH = α-HCH + β-HCH + γ-HCH;
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD;
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

[a] ΣDDT = *p,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.
[b] ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE.
[c] ΣHCH = α-HCH + β-HCH + γ-HCH + δ-HCH.
[d] ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD.
[e] ΣHCH = γ-HCH.
[f] ΣCHL = 1.43 × (*trans*-chlordane + *cis*-chlordane).
[g] no HCH species detected.
[h] PCBs data read from "Polluted Area" subplot of Figure 2, Tanabe et al.
[j] HCH & CHL data read from Figure 2, Yamaguchi et al.
[k] CHL data read from Figure 2, Hirai and Tomokuni.
[l] ΣDDT = *p,p'*-DDT + *p,p'*-DDE.
[m] total PCBs estimated as 2.35 total reported for 51 PCB congeners.
[n] ΣDDT = *p,p'*-DDT + *p,p'*-DDE + *o,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD.
[o] ΣCHL = 1.165 × (*trans*-chlordane + *cis*-chlordane + *trans*-nonachlor).
[p] ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD.

Table 2.5 Selected water and sediment quality criteria (W = Water, S= Sediment).

Contaminant	Medium	Criterion	Source
PCBs	W	1 ng/L	Provincial Water Quality Guideline, OMEE (1994)
	W	14 ng/L	Chronic criterion, USA, USEPA (1990)
	S	10 ng/g	No Effect Level, (Persaud et al., 1992)
	S	70 ng/g	Lowest Effect Level, Persaud et al. (1992)
ΣHCH	W	10 ng/L	γ-HCH Provincial Water Quality Guideline, OMEE (1994)
	W	41.4 ng/L	technical HCH, USEPA (1990)
	S	0.7 ng/g	Sum of no effects levels / thresholds for α-, β- and γ-HCH; Persaud et al. (1992); Environment Canada (1992)
	S	14 ng/g	Sum of lowest / minimal effects levels / thresholds for α-, β- and γ-HCH; Persaud et al. (1992); Environment Canada (1992)
ΣDDT	W	1 ng/L	Chronic criterion, USA, USEPA (1990)
	W	3 ng/L	Provincial Water Quality Guideline, OMEE (1994)
	S	1.5 ng/g	Chronic Marine EqP Threshold, Bolton et al. (1985)
	S	7 ng/g	Lowest Effect Level, Persaud et al. (1992)
ΣCHL	W	0.53 ng/L	Cancer risk value, Michigan, MDNR (1989)
	W	6 ng/L	Chronic criterion, USA, USEPA (1990)
	S	1 ng/g	No effect threshold, Environment Canada (1992)
	S	7 ng/g	Lowest Effect Level, Persaud et al. (1992)

2.4 POPs in Fish

Fish are a major dietary staple of many Asian countries, and thus a significant ingestion route back to humans for bioaccumulative POPs that enter aquatic ecosystems. Surveys of fresh market fish for human consumption add valuable perspective on the occurrence of POPs in aquatic systems of Asia, Australia and Oceania. Sufficient data were found to permit intercomparison of HCH, DDT, chlordane, heptachlor + heptachlor epoxide, aldrin + dieldrin, PCBs and HCB for 5-9 countries on a lipid weight basis. The lipid weight concentrations are shown in Figure 2.1c and 2.3. Data sources are listed in Table 2.6. Most data represent baskets of freshwater and near shore food fishes obtained fresh from markets as indicated. Other fish data from India, Iraq and Indonesia were included to broaden the comparison.

Table 2.6 Figure key, source references and notes for Figures 2.1c and 2.3 POPs concentrations in fish of Asia, Australia and Oceania.

Australia	
Aus	37 sample of freshwater and near shore fishes from markets of Sydney, Hobart, and Perth [1990]; and Brisbane, Townsville and Atherton [1992] (Kannan et al., 1994b).
India	
In-P	India – Ganges River near Patna, single composite of four fish species from the gut of 4 Ganges River dolphins collected over 1988-92 (Kannan et al., 1994a).
In-V	India – Vellar River near Porto Novo, Tamil Nadu; 28 samples (muscle) of 6 estuarine, mangrove and near shore species collected from 1987–91(Ramesh et al., 1992).
In-M	India – market fish; 42 samples of freshwater and nearshore fishes, and prawn, from fresh fish markets in New Delhi, Bombay, Calcutta, Madras, Porto Novo, and Chidambaram collected December 1989 (Kannan et al., 1992b).
Indonesia	
Id-B	Indonesia – Bogor area; 5 samples, 199? (Kannan et al., 1995); lipid weights approximated from mean and range of fat % and wet weights.
Id-S	Indonesia – Surabaya area; 6 samples whole teleost fish from Solo, Wonokromo, and Porong (Brantas) estuaries, east Java, 1984, (Boon et al., 1989); only DDT consistently present above detection; mean γ-HCH and HCB < 1 ng/g lipid weight.
Iraq	
Ir-D	Iraq – Diyala River near Baghdad; 17 samples mixed freshwater species from 1984 (Al-Omar et al., 1986); ΣCHL = *trans-* + *cis-* + oxychlordane.
Ir-S	Iraq – Shatt al-Arab River; 16 samples *Tenualosa ilisha* (muscle tissue) from 1984 (DouAbul et al., 1987); ΣCHL = *trans-* + *cis-* + oxychlordane; ΣHCH = sum of measured β-and γ-HCH, and estimated α-HCH.

2.4.1 Note on Lipid versus Wet Weight Concentration Data

For comparative purposes, lipid normalized concentration data are preferable because POPs partition into fats; hence, variations in wet weight concentrations may reflect variations in fat content more than true variations in POPs contamination. Wet weight POPs concentration data were available for a broader selection of countries and localities, but were used only selectively because conversions to lipid weight basis were not possible. Either fat % was not reported or was given only as mean and range summaries. ***Lipid weight normalization is a nonlinear transformation. The mean of individual lipid weight concentrations is not equal to mean wet weight concentration divided by mean lipid concentration.*** Where fat % range is narrow, biases are usually small, but for some wet weight fish data, fat % ranged as much as 0.1–20%; such that, normalizing wet weights by mean fat % risked introducing great bias into the derived lipid weight concentrations. Consequently, wet weight data were evaluated cautiously and are cited only on a case-by-case basis where comparisons are unambiguous.

Table 2.6 cont'd. Figure key, source references and notes for Figures 2.1c and 2.3 POPs concentrations in fish of Asia, Australia and Oceania.

Japan	
Jap	1 sample each of 2 species of cultured fish from markets of Fukuoka, Kyushu 1992-93 (Nakagawa et al., 1995).
Papua New Guinea	
PNG	9 samples of 2 species from Port Moresby markets 1990 (Kannan et al., 1994b).
Solomon Islands	
Sol	10 samples of 3 species from Honiara markets Aug – Sep 1990 (Kannan et al., 1994b).
Thailand	
Thai	15 samples of 5 species from Bangkok markets Dec 1989 – Jan 1990 (Tanabe et al., 1991a); due to numerous typographical errors in Table 2 Tanabe et al., fat weights were approximated as necessary from wet weights and fat % in Table 1 Tanabe et al.
Vietnam	
Viet	11 samples of freshwater and nearshore fishes from Hanoi, Phu Loc Lake [Hue area], and Duyen Hai [Ho Chi Minh] 1990-91 (Kannan et al., 1992a).

ΣHCH = α-HCH + β-HCH + γ-HCH.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.
Σdrin = aldrin + dieldrin.
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.
ΣHEP = heptachlor + heptachlor epoxide.

Chapter 3 India

Numerous reports are now available documenting POPs occurrence in India's surface water systems. However, most data are for mid to upper reach sites on the Ganges system, and there are no water concentration data for rivers of peninsular India other than for two minor streams in the south. Because available data are rife with potential biases and contradictions, it was necessary to undertake a meticulous, time intensive review of available data sources in order to select plausible POPs water concentrations for river flux estimation. Ultimately the process produced acceptable first order results; however, appreciable uncertainty remains concerning riverine POPs exports until more spatially and temporally representative data are obtained. This chapter reviews all readily available reports of POPs presence in Indian surface waters, and gives estimates of POPs exports to coastal seas by Indian river systems.

The POPs flux to the sea from the vast Ganges–Brahmaputra–Meghna river system through Bangladesh originates mainly in India. For practical purposes Bangladesh contributions can be ignored. POPs usage in Bangladesh is discussed in Chapter 14.

3.1 India: Pesticide Usage

3.1.1 POPs Insecticide Production and Usage

Informal reports ranked India ninth amongst world producers and consumers of pesticides circa 1990 (PANNA, 1991b). India is further reported to produce about 95% of domestic pesticide consumption that has been variably given as 60–90 Kt/a. FAO reports annual usage at 72 Kt active ingredient over 1990-93 (FAO, 1996). Total domestic production for fiscal 1988-89 was reported as 59.6 Kt of which 51.77 Kt (87%) were insecticides (Srivastava and Patel, 1990). Most recently, an informal report citing the trade publication *Agrow World Crop Protection News* claimed that in fiscal 1994-95, India produced 85.8 Kt of technical grade active pesticide ingredients (PANNA, 1995b). That is an increase of 34 Kt in 7 years.

Organochlorines HCH and DDT have been the leading insecticides accounting for almost 70% of total production in the 1980s. Both have been used in public health applications for malaria control mainly, in agriculture where HCH is heavily used in rice cultivation, and as general purpose insecticides in domestic, urban, and industrial settings. India began importing both DDT and HCH for public health purposes following WW II, and agricultural usage began in the late 1940s. Domestic production began in 1952 [HCH] and 1955 [DDT] (Gupta, 1986).

Various production and usage statistics have been reported. Srivastava and Patel (1990) gave the 1986-87 combined DDT and HCH production as 35.05 Kt. The split

between HCH and DDT production was likely similar to that in the 1982-83 accounting year — 32,389 t HCH and 6,453 t DDT (Gupta, 1986). The actual consumption figures are likely underestimated as both HCH and DDT may have been imported some years to meet demands. Mid to late 1980s' usage figures reported in environmental monitoring and assessment literature include: 36 Kt HCH and 14 Kt DDT (Singh et al., 1988); and 45 Kt HCH and 19 Kt DDT (Ramesh et al., 1992). The higher, often quoted figures may be demand forecasts generated in the early to mid 1980s rather than actual usage statistics. Malaria control usage over 1988-89 was 8.68 Kt DDT and 7.25 Kt HCH (Sharma, 1991). Dua et al. (1994) gave the 1992 HCH usage for public health at approximately 8.25 Kt. Gupta (1986) gave cumulative DDT usage to 1965 for public health as 126,147 t DDT, while an informal report citing the trade publication *Europa Chemie*, gave India's cumulative DDT usage to 1989 as 280 Kt DDT for malaria control and 50 Kt in agriculture (PANNA, 1990). This suggests that annual average DDT usage for 1966–1989 was about 8.5 Kt. Assuming usage of 9 Kt/a for public health, cumulative DDT production to 1995 would be almost 400 Kt. Actual usage may have been higher to due to imports. The report edited by (David, 1992) may have more recent information, but was unavailable for review.

Cumulative HCH usage figures are not readily available. Usage through the 1970s may have averaged about 15 Kt/a and increased to 35 Kt through the 1980s (Gupta, 1986; Srivastava and Patel, 1990); so that, cumulative Indian production from 1952–1995 likely exceeds 500 Kt. Kannan et al. (1995) estimated cumulative HCH production to 1995 at 1,000 Kt which seems to be a significant overestimate.

Technical HCH used in India seems to have variable composition. Agarwal et al. (1987) reported that HCH used in the Delhi area over 1980-82 comprised 82% α-HCH, 13% γ-HCH, and 5% β- and δ- isomers. Recently, (Dua et al., 1994) reported 1992 usage in Hardwar of a *fortified* HCH comprising 38% α-HCH, 37% γ-HCH, 24% δ-HCH with β-HCH virtually absent. India may be moving to boost production and consumption of more purified HCH products including lindane.

3.1.2 Role of DDT and HCH in Malaria and Vector-Borne Disease Control

DDT and HCH have played and continue to play a central role in India's efforts to control malaria, dengue, filaria and Japanese encephalitis which are transmitted by different families of mosquitoes. There has also been a resurgence of Kala Azar (leishmaniasis) which is transmitted by a sand fly against which DDT has previously proven effective (Verghese, 1990). While each disease poses serious threats, malaria has been the main concern. Following WW II, broad scale spraying of DDT almost eradicated malaria by 1960; however, cases surged from 50,000 in 1961 to more than 6.45 million in 1976 [(Sharma, 1991), also see Figure 3 in (Bradley, 1991)] despite continued mosquito control spraying. In 1977, anti-malaria operations were modified to focus attention on high transmission areas with the result that cases reduced gradually to under 2 million by 1986; however, there has been no further improvement since then.

The dramatic resurgence of malaria over 1961–1976 has been attributed to several causes. Firstly, Sharma (1991) concedes that the designated anti-malarial insecticides (DDT, HCH and malathion) have gradually lost effectiveness due to resistance that has developed by the mosquitoes. Two mosquito vectors responsible for 80% of malaria cases, and 14 public health insect pests are now resistant to the major public health insecticides (Mehrota, 1993). Previous high agricultural usage of DDT may have significantly hastened the process (Chapin and Wasserstrom, 1981). Environmental data suggest that, despite their apparent ineffectiveness, DDT and HCH use in rural and urban public health operations has continued unabated into the early 1990s.

The rise and spread of malaria also closely followed the tremendous expansion of India's irrigation network that saw irrigated lands double from 29 million ha in 1960 to 60 million ha in 1980 (Verghese, 1990). Expansion continues steadily toward the government target of 113 million ha irrigated lands by 2010. Citing Sharma, Verghese observes it is not irrigation per se, but rather "defective and untidy irrigation" that promotes mosquito survival and reproduction, and advises that reduction of malaria incidence and organochlorine pesticide usage should depend more on "integrated vector control systems" that couple improved irrigation practice with "bio-enviromental" controls being explored by India's public health authorities. Related strategies can also control the encephalitis vector which favours paddies and other deep irrigated areas. Good water management is also central to controlling the filaria and dengue mosquito vectors. The former favours unsanitary waters, while the latter favours stagnant urban waters including cisterns and unenclosed overhead water tanks.

Given the central role of water as a breeding medium for the transmission vectors, it should not be too surprising if most insecticide were applied directly onto outdoor soil and water surfaces including ponds, ditches, irrigation channels, sewage canals, rice paddies and other slow moving or stagnant water bodies that serve as mosquito breeding sites. The empirical evidence strongly suggests this; however, there has been little open discussion of the actual practices. The usual accounts of anti-malarial operations emphasize the periodic "residual spraying" of indoor walls and ceilings of dwellings, cattle sheds, and other rural buildings with DDT and HCH (e.g., Battu et al., 1989; Singh et al., 1988). To the contrary, (Ramesh et al., 1990a) suggested that direct application to water surfaces was required to explain the apparent supersaturation of observed water concentrations relative to air measurements in the Vellar River watershed of Tamil Nadu state. Dua et al. (1994) describe spraying of 3 t HCH on plants, soil and buildings to control of mosquitoes and flies during a religious festival on a 130 km^2 site, while high levels of HCH in air and rainwater of the Delhi Town Hall district have been attributed to civic practice of frequently dusting district roadsides [presumably roadside drains] with HCH to control flies and mosquitoes (Agarwal et al., 1987; Kaushik et al., 1987). Saxena et al. (1987) referred to DDT usage in Delhi area anti-malarial programs as "uncontrolled".

Knowing the high DDT usage areas and application rates would tremendously assist the interpretation of available aquatic data. The malaria-irrigation linkage suggests that most DDT is being applied on the upper Gangetic plain in western Uttar Pradesh state which has India's most extensive irrigation network. DDT use is also likely heavy in Punjab and Haryana states where malaria surged with irrigation expansion that promoted water intensive relay-cropping of wheat and rice (Verghese, 1990). Punjab surface waters drain westward to the Indus system, while Haryana is in the Ganges watershed. As rural malaria incidence is lower in the mid and lower reaches of the Ganges basin, DDT usage is probably less. To the south, in peninsular India, malaria and DDT usage may be more concentrated in urban areas. Regional public health insecticide usage data would considerably assist future aquatic and other environmental monitoring activities.

3.1.3 Other Organochlorine Pesticides

Environmental studies have shown the occurrence of aldrin, dieldrin, endosulfan, heptachlor and chlordane. Dieldrin seems to have been restricted some years ago, but is present as the degradation product of aldrin. Through the latter 1980s, aldrin and heptachlor appear to have been the most heavily used organochlorine insecticides after HCH and DDT. Reports suggest both were used mainly as soil insecticides. Heptachlor, chlordane, and most likely aldrin, would have been imported. Data indicate relatively low environmental contamination by chlordane; hence, imports have not likely been great or the chlordane presence is due to heptachlor usage. Heptachlor levels rival aldrin and endosulfan at some sites; thus, imports may have been somewhat greater. Import and consumption statistics are not readily available.

3.1.4 Recent Prohibitions on POPs Insecticides

In October 1989, India banned agricultural usage of DDT, limited annual public health usage to 10 Kt, and banned DDT importation (Srivastava and Patel, 1990).

Recently (early 1990s), HCH usage has been restricted on certain food crops (David et al., 1993).

At the UNEP/IRPTC sponsored IFCS meetings on POPs in Manila, Philippines, 17-22 June 1996, it was reported that "recently in India the Government has taken a decision to phase out a production of 30,000 tonnes of BHC [HCH] per annum" (Sugavanam and Kim, 1996). The authors also gave data showing that between 1988-89 and 1993-94 accounting years, combined Indian DDT, HCH and endosulfan production declined by about 5 Kt from 40.4 Kt to 35.3 Kt.

India banned heptachlor and chlordane effective February 17, 1992, and aldrin effective January 1, 1994, (David et al., 1993). Ganges River data showed continuing

presence of aldrin and heptachlor in 1992 (Agnihotri et al., 1994; Gajbhiye et al., 1995). Levels should have begun declining as residual stocks were consumed.

3.2 India–Bangladesh Hydrology

3.2.1 The Southwest Monsoon

The dominant hydroclimatic feature of the Indian subcontinent is the southwest monsoon rains from June to September that account for 80-90% of annual rainfall and river discharge over much India and Bangladesh. Furthermore, these rainy months generally induce major use of insecticides for both public health and agricultural purposes. In particular, public health application of DDT and HCH for mosquito control may be heaviest and most prone to surface runoff losses during the monsoon rains.

Figure 3.1 shows the distinctive monthly rainfall pattern at sites on the mid and upper Gangetic plain. Seasonal rainfall patterns vary somewhat in the Himalayas, northeastern hill states, and on the southern coasts; however, the rainfall pattern over the mid and upper Gangetic plain is most important as the high malaria transmission and DDT usage areas are located here.

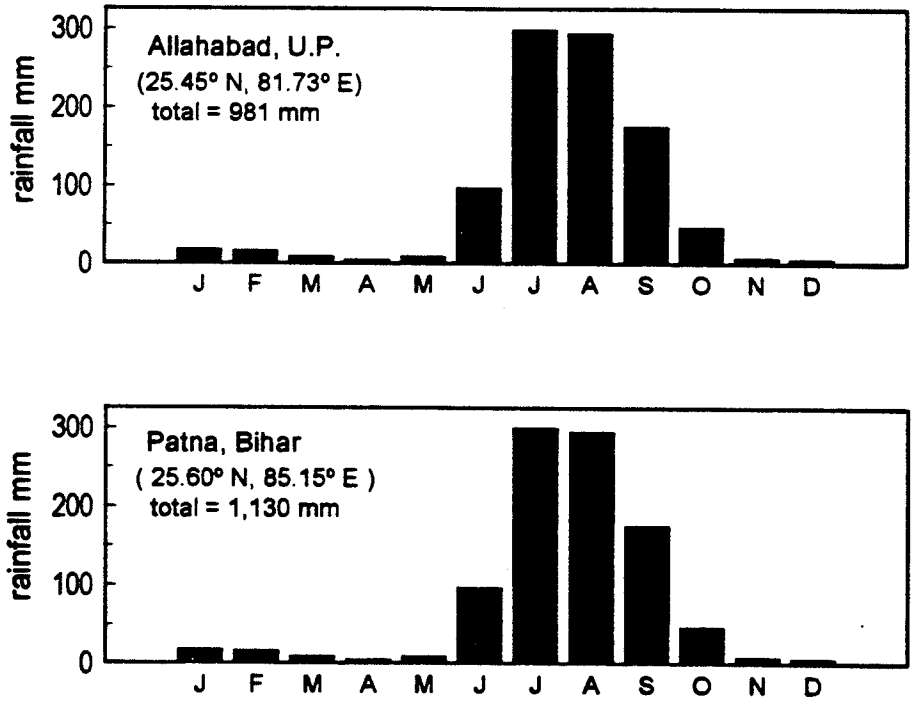


Figure 3.1 Mean monthly rainfall at Allahabad and Patna.

3.2.2 River Discharge

Rivers debouching to the sea from India and Bangladesh can be classified into two groups: (1) the Ganges–Brahmaputra–Meghna system, and (2) the peninsular rivers. The latter can be subdivided further into those discharging to the Arabian Sea or to the Bay of Bengal. Discharge and area data are given in Table 3.1.

The vast Ganges–Brahmaputra–Meghna system generates about 77% of total surface water discharges from India and Bangladesh. The combined mean annual total discharge of about 1,270 km^3 ranks second only to the Amazon. The delta complex through which these rivers course to the Bay of Bengal is the largest in the world and occupies much of the territory of Bangladesh. Figure 3.2 shows the mean monthly discharges of the Ganges River at Farakka on the Bangladesh frontier and the Brahmaputra River at Pandu (Gauhati) India. The annual discharge of both rivers, but especially the Ganges, is clearly driven by the monsoon rains.

Though the peninsular rivers contribute only 23% of total discharge, these rivers are important sources of contaminants to the local coastal waters of the peninsula. The names and locations of the 40 minor rivers were given by Rao (1975). The 19 small rivers on the west coast yield have an annual discharge greater than the 4 large west coast rivers. These small rivers mostly rise in the escarpment of the Western Ghats

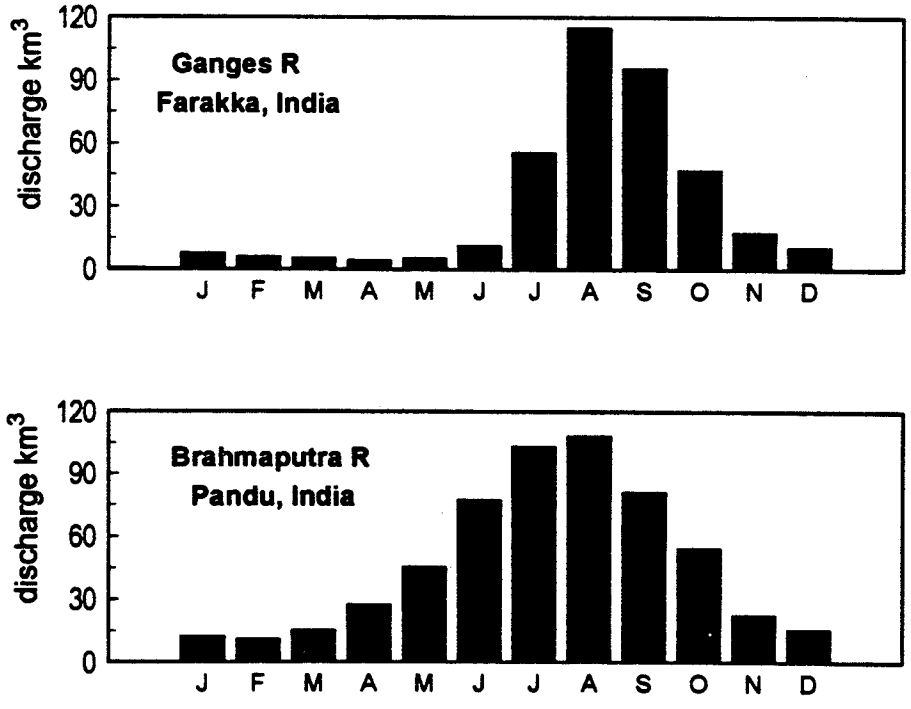


Figure 3.2 Mean monthly discharge of the Ganges and Brahmaputra rivers.

Table 3.1 River discharge to the sea from India and Bangladesh.

	Area 10^3 km^2	Discharge km^3	%	Runoff mm
Northern Rivers				
Ganges	[a] 1,060	[b] 554	33.5	523
Brahmaputra [c]	580	606	36.6	1,045
Meghna [d]	80	111	6.7	1,383
Northern Total	1,720	1,271	76.7	739
Peninsular Rivers [†]				
A. to Bay of Bengal				
Subarnarekha	19	7.9	0.5	411
Brahmani	28	16.3	1.0	579
Mahanadi	88	54.5	3.3	617
Godavari	313	92.2	5.6	295
Krishna	259	33.4	2.0	129
Penner	49	5.2	0.3	107
Cauvery	66	11.5	0.7	174
21 minor rivers [‡]	134	20.6	1.2	154
Subtotal A	956	241.7	14.6	253
B. to Arabian Sea				
Sabarmati	14	1.5	0.1	102
Mahi	26	10.8	0.7	424
Narmada	88	46.7	2.9	531
Tapti	49	9.7	0.6	198
19 minor rivers [‡]	80	75.0	4.5	937
Subtotal B	257	143.6	8.7	560
Peninsular Subtotal	1,213	385.3	23.3	318
India-Bangladesh total	2,933	1,656.3	100	565

[a] Ganges drainage area from (Rao, 1975). Separate estimates of Ganges drainage area in India (860,404 km^2), Nepal (140,797 km^2), and Bangladesh (38,800 km^2) sum to 1.04 million km^2 implying that ca. 20,000 km^2 lies in China which seems high.

[b] Total discharge is estimated as Ganges basin discharge from India (525 km^3) plus 0.4 times the mean annual total rainfall of 1.8 m falling on the 38,800 km^2 Ganges drainage area in Bangladesh.

[c] Brahmaputra area from (Milliman et al., 1995) and discharge from (Chowdhury and Bhuiya, 1990).

[d] Meghna drainage area and discharge from (Chowdhury and Bhuiya, 1990).

[†] data for main peninsular rivers are mainly from (Milliman et al., 1995) with modifications after (Ramesh and Subramanian, 1988, 1993; Vaithiyanathan et al., 1992).

[‡] 40 minor peninsular rivers given by (Rao, 1975).

and drain the narrow coastal strip from Bombay south to Cape Comorin. The strip enjoys abundant annual rainfall of 2–4 m. A rain shadow effect in the lee of the Western Ghats explains the low runoff yield of the eastern peninsular rivers from the Godavari southwards.

According to Verghese (1990), India's estimated total annual surface discharges [including those to the Indus, closed desert rivers, and small streams flowing to Burma] were revised upwards in 1988 by the Central Water Commission to 1,880 km^3 from the 1973 estimate of 1,645 km^3 given by Rao (1975). Of the 235 km^3 increase, about 150 km^3 seem to be accounted for within the Ganges, Brahmaputra and Meghna discharge estimates given in Table 3.1. How the remaining 84 km^3 of the 235 km^3 increase has been distributed is not known. The peninsular rivers listed in Table 3.1 include more recent data for several large rivers (Ramesh and Subramanian, 1988; Ramesh and Subramanian, 1993; Vaithiyanathan et al., 1992), but some of the medium rivers and all the minor rivers are as originally estimated by Rao (1975). Many of these may have been revised.

3.2.3 River Names of the Ganges–Brahmaputra–Meghna System

The varying river names in the delta complex are a source of much confusion.

- The Ganges or Ganga retains its name until it merges with the Brahmaputra (Jamuna) at Goalando in Bangladesh where the combined stream takes the name Padma for about 110 km until it further merges with the Meghna under which name the combined stream debouches into the Bay of Bengal.
- Several distributaries and spill channels branch off southward from the Ganga and Padma. The westernmost distributaries that lead off in India, are known initially as the Bhagirathi and the Jangari. These unite above Calcutta to form the Hooghly River. Since the main Ganga flow shifted eastward in the 1700s, the Bhagirathi-Hooghly has been a dying distributary; however, the Hooghly may be a significant source of contaminants emanating from the Calcutta urban-industrial complex including the coal-steel belt drained by the Damodar River that joins the Hooghly south of Calcutta.
- The Brahmaputra originates in Tibet as the Yarlung Zangbo (Yalu Tsangpo in Chinese), becomes the Dihang on entering India in Arunachal Pradesh state, becomes the Brahmaputra on the 720 km passage through Assam state, and finally becomes the Jamuna on entering Bangladesh until it merges with the Ganga to form the Padma. A former main channel, the Old Brahmaputra, leads off southeastward inside the Bangladesh frontier to join the Meghna above Dhaka.
- The Meghna originates as the Barak River in the northeastern hill states of India to the south of the Brahmaputra. Near the Bangladesh border, the Barak bifurcates into the Surma and Kushyara that reunite in Bangladesh at Markulia to form the Meghna. The watershed is amongst the wettest in world receiving up to 4 m or more rainfall annually in the uplands. Thus, despite its relatively small area, the Meghna generates appreciable water volume.

3.3 POPs Contamination of India's Aquatic Ecosystems

3.3.1 Brahmaputra and Meghna Rivers

No POPs measurements were found for the Brahmaputra River proper; however, Pathak et al. (1992) have reported HCH, DDT, aldrin and endosulfan data from a synoptic survey of Brahmaputra tributary streams in India's northeastern hill states. Over 1986-89, single samples were collected at 27 sites distributed by states as: Sikkim (3); Meghalaya (4); Arunachal Pradesh (3); Tripura (3) and Mizoram (14). Mean concentrations by state are given in Table 3.2. It is unclear whether pesticide analyses were performed on samples from the three streams in Arunachal Pradesh, or whether analyses were performed and yielded non detections.

Table 3.2 POPs insecticides (ng/L) in streams of India's northeastern states.

	ΣHCH	ΣDDT	Endosulfan	Aldrin
Sikkim	25	218	7.1	—
Meghalaya	22	21	—	—
Tripura	214	67	—	—
Mizoram	116	13	6.8	6.5

3.3.2 Ganges River Basin

Most POPs data for Indian surface waters are available in the Ganges basin; however, nearly all readily available data represent sites in the upper and mid reaches that are remote from the outlets to the Bay of Bengal. In order to select plausible mean water concentrations suitable for estimating mass discharges of the Ganges to the Bay of Bengal, all readily available data sources were thoroughly reviewed. The results are presented below.

3.3.2.1 River Jamuna tributary

The River Jamuna (alternatively Yamuna) is the largest tributary of the Ganga with a basin area of 366,233 km^2 [~ 42.5% of the Ganges basin]. Gopal and Sah (1993) give a general account of Yamuna watershed geography, ecology and water pollution status. The basin is heavily agricultural and has numerous large urban centres, notably Delhi which is located in the upper reaches. Monthly mean rainfall patterns in Delhi and Jaipur (Figure 3.3) have the typical monsoonal pattern. Pesticide transport

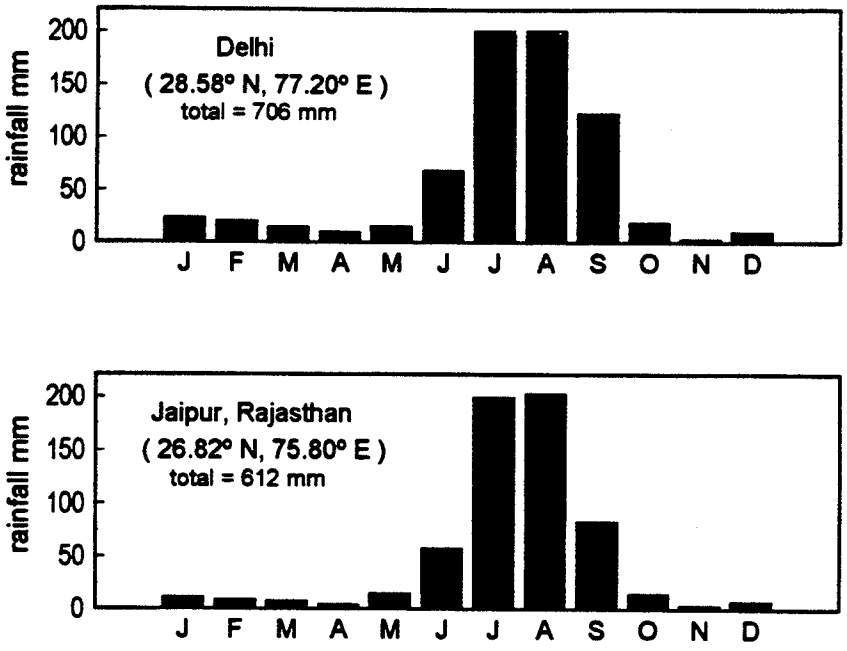


Figure 3.3 Mean monthly rainfall at Delhi and Jaipur.

by area streams and rivers will be much greater in the months following the onset of the rainy season even if the water concentrations do not change appreciably.

3.3.2.1.1 River Jamuna tributary – Delhi area

Both HCH and DDT have been used heavily in the Delhi area for malaria control particularly during the wet monsoon season from July to September. HCH dust has been frequently applied to roadsides in some Delhi districts to control houseflies and mosquitoes. The Moti Nagar district in the city's northwest is home to a DDT manufacturing plant that discharges effluents to a drain that enters River Yamuna upstream in Wazirabad on Delhi's northeastern fringe (Agarwal et al., 1987; Agarwal et al., 1986).

Numerous studies have been conducted of organochlorine pesticide pollution in the Delhi area environment (Agarwal et al., 1987; Agarwal et al., 1986; Dureja et al., 1991; Kaushik et al., 1987; Nair et al., 1991; Nair et al., 1996; Nair and Pillai, 1989; Nair and Pillai, 1992; Pillai, 1986; Saxena et al., 1987a; Saxena et al., 1987b; Zaidi et al., 1989). Reconciling these data is difficult because it is unclear how representative the sampling has been, and investigators rarely state the reporting basis for soil, sediment and biotic samples.

To assess the degree of contamination, available data were examined across surveys and across environmental compartments after converting to approximate equivalency on a pg/g basis. Distributions of ΣHCH, ΣDDT, Σdrin = aldrin+dieldrin, and HCB in Delhi environmental compartments as indicated by available literature data are shown in Figures 3.4–3.6. The figure keys and data sources are listed respectively in Tables 3.3–3.5. The 1976-78 and 1988-89 River Yamuna water, sediment, clam and fish data are gross aggregates of all Delhi area sites in (Agarwal et al., 1986) and (Nair and Pillai, 1992), respectively. The chief purpose of the diagrams and tables is to assess the internal and intercompartmental consistency of available Delhi area data, and to establish the general range of environmental concentrations in the Delhi area for cross-comparison with Indian data from other locations. Apparent time trends should be regarded cautiously.

In Figures 3.4–3.5, the broad trend expected in chemical concentrations across the aquatic ecosystem components is *river water* < *sediment* ≤ *soil* ≤ *biota*. Among soil, sediment and biotic compartments, ordering may be masked by inconsistent reporting bases and sample representativeness across surveys. Figure 3.6 shows the relative cross-media concentrations which for soil, sediment and biotic compartments are expected to be ordered approximately as ΣDDT > ΣHCH > Σdrin ≥ HCB.

3.3.2.1.1.1 Suspect Water Quality Data — Delhi Area

Studies of the diagrams suggest the following aquatic data are suspect:

1. 1988-89 River Yamuna ΣDDT and ΣHCH concentration data from survey W2 (Nair and Pillai, 1992) would appear to be too low by 10–1,000 fold (Figure 3.4).

2. aldrin and dieldrin water data given by Nair et al. (1991) are likely too high by 1–3 orders of magnitude (Figures 3.5A and 3.6A).

The source of the discrepancy in the suspect 1988-89 ΣDDT and ΣHCH concentration data reported by Nair and Pillai (1992) is not obvious and these data should be disregarded. As there is no conclusive evidence that DDT and HCH usage had declined appreciably since 1976-78, the results of the single sample obtained by Iwata et al. (1994) [survey W3] are probably on the correct order.

The suspect aldrin and dieldrin data may result from transcription errors. Generally for these organochlorines we would expect *water* ≤ *soil* < *human*; however, on Figure 3.5, Delhi data show *water* ≥ *soil* > *human*. As judged from Figure 3.6, the soil, clam and fish data seem consistent. This suggests that the water and human milk data are in error. The water concentrations of aldrin and dieldrin, 14 and 20 μg/L respectively, stand out as extraordinarily high, while the human milk data may be 1,000 fold too low.

Table 3.3 ΣHCH in Delhi environmental compartments.

Key	Medium / unit †	Date	Mean	Min.	Max.	Source
A1	air ng/m^3	1980-82	20	1	21,800	1
A2	air ng/m^3	Dec 1989	46			2
R	rain water µg/L	1980-82	5.28	0.08	43.0	3
★ W2	river water ng/L	1988-89	★ 1.6	★ <1.0	★ 8.0	4
W3	river water ng/L	Dec 1989	660.0			2
S2	soil ng/g	1988-89	40	1	230	4
C2	clam ng/g	1988-89	330	20	710	4
F2	fish ng/g	1988-89	55	<1	470	4
F3	fish ng/g	Dec 1989	5.5	2.8	8.9	6
B2	human blood µg/L	1988-89	1,600	430	2,790	4
M2	human milk µg/L	1988-89	750	400	4,090	4
M3	human milk µg/L	199?	327	20	4,160	5
L2	human fat ng/g	1988-89	570	100	1,940	4

† Soil, sediment, clam, fish, and human data basis not given —wet weight basis assumed.
Sources: 1. Kaushik et al., 1987; 2. Iwata et al., 1994; 3. Agarwal et al., 1987a; 4. Nair and Pillai, 1992; 5. Nair et al., 1996; Kannan et al., 1995. ★ Dubious data.

3.3.2.1.1.2 Trends and Discrepancies in Delhi Area Fish Data

On Figure 3.6D, there is a probable downtrend in ΣDDT between surveys F1 and F2 that agrees with other water and soil data. There are also obvious differences between surveys F2 and F3 for ΣDDT, ΣHCH and Σdrin. Survey F2 involved 19-20 samples collected over 1988-89 from River Jamuna on its course through greater Delhi, while survey F3 involved 7 samples of fresh, *nominally locally-caught* market fish obtained in Delhi about December 1989. Apparent declines from F2 to F3 are not necessarily due to time trends. Other factors that may explain the differences include:

- artefacts of small sample size, particularly for survey F3,
- significant differences in lipid content coupled with failure to normalize results on a lipid weight basis,
- market fish in survey F3 may have come from less contaminated waters near Delhi, rather than the River Jamuna in greater Delhi,
- seasonal differences, e.g., F2 fish may have been collected during wet monsoon months coincident with heavy insecticide usage, while F3 fish were collected in December at the onset of the low discharge season.

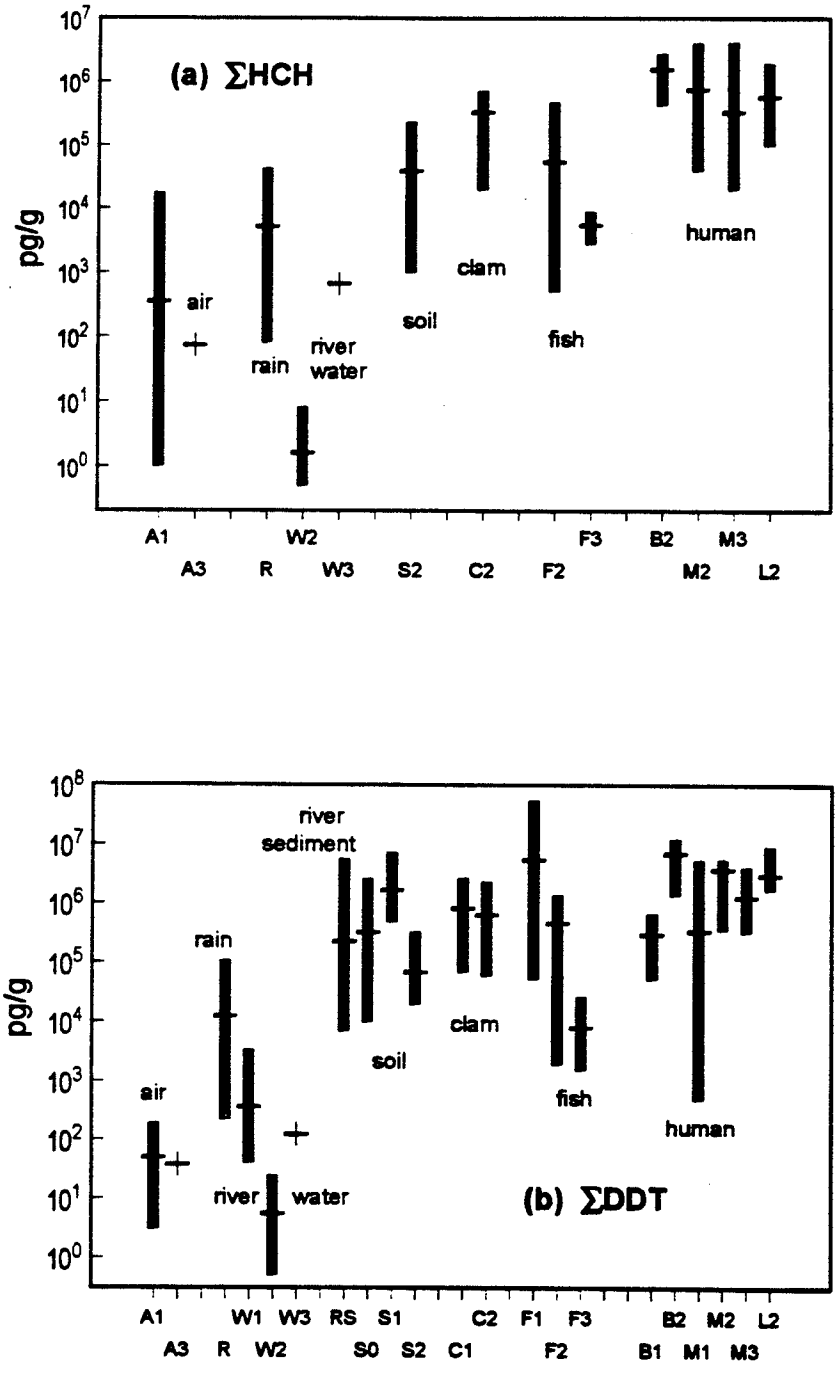


Figure 3.4 ΣHCH and ΣDDT distribution in Delhi environment; see Tables 3.3–3.4 for legend and data sources; vertical bars show range; cross bar shows arithmetic mean.

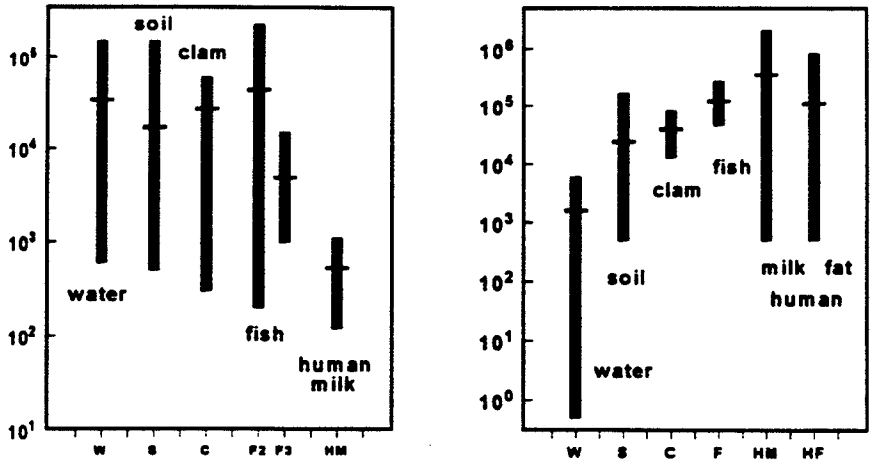


Figure 3.5 Σdrin (aldrin+dieldrin) and HCB (pg/g) distribution in Delhi environment; see Table 3.5 for data sources; vertical bars show range; cross bar shows arithmetic mean.

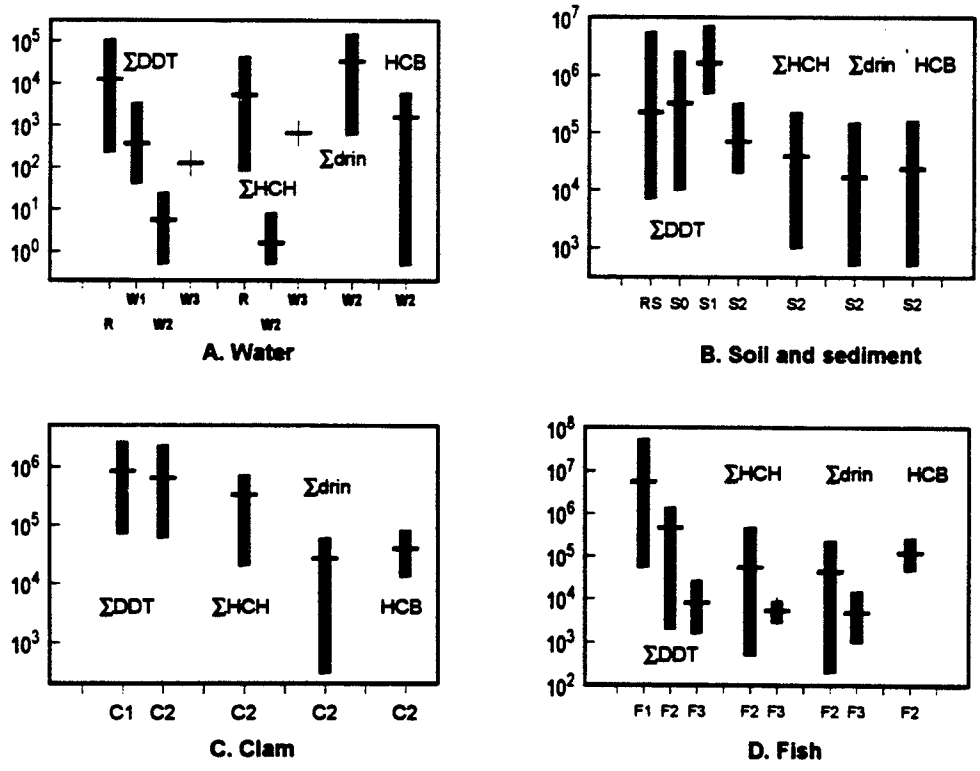


Figure 3.6 POPs insecticide (pg/g) cross-comparisons by environmental compartment.

Table 3.4 ΣDDT in Delhi environmental compartments.

Key	Medium / unit †	Date	Mean	Min.	Max.	Source ‡
A1	air ng/m^3	1980-82	60	4	232	1
A2	air ng/m^3	Dec 1989	46			2
R	rain water µg/L	1980-82	12.5	0.22	108	3
W1	river water ng/L	1976-78	368	40	3,420	5
★ W2	river water ng/L	1988-89	★ 5.5	★ <1	★ 24	4
W3	river water ng/L	Dec 1989	124			2
RS	river sediment ng/g	1976-78	230	7	5,630	5
S0	soil ng/g	1974	330	10	2,630	6
S1	soil ng/g	1983	1,670	490	7,270	7
S2	soil ng/g	1988-89	70	20	330	4
C1	clam ng/g	1976-78	840	70	3,860	5
C2	clam ng/g	1988-89	640	60	2,340	4
F1	fish ng/g	1976-78	5,500	549	56,000	5
F2	fish ng/g	1988-89	470	2	1,380	4
F3	fish ng/g	Dec 1989	8	1.6	27	11
B1	human blood µg/L	*?	301	53	663	8
B2	human blood µg/L	1988-89	7,170	1,380	12,210	4
M1	human milk µg/L	1985-86	344	<1	5,488	9
M2	human milk µg/L	1988-89	3,740	360	5,500	4
M3	human milk µg/L	**?	1,270	330	4,100	10
L2	human fat ng/g	1988-89	3,030	170	9,120	4

† Basis of soil, sediment, clam, fish, and human data generally unstated in source references, assumed to be on wet weight basis.

‡ Sources: 1. Kaushik et al. (1987); 5. Iwata et al. (1994); 3. Agarwal et al. (1987); 4. Nair and Pillai (1992); 5. Agarwal et al. (1986); 6. Yadav et al. (1981); 7. Saxena et al. (1987a); 8. Saxena et al. (1987b); 9. Zaidi et al. (1989); 10. Nair et al. (1996); Kannan et al. (1995).

★ Dubious data; * most likely sometime over 1980-1985. **most likely sometime over 1990-1995.

Table 3.5 Aldrin, dieldrin and HCB in Delhi environmental compartments.

Key	Medium / unit	aldrin	dieldrin	Σdrin	HCB
W2	**water** µg/L	★ 14 0.5–50	★ 20 0.1–100	★ 34 0.6–150	1.60 <.001–5.97
S2	**soil** ng/g	13 0.3–120	4 0.2–30	17 0.5–150	24 <1–165
C2	**clams** ng/g	17 0.2–10	10 0.1–50	27 0.3–60	40 13–82
F2	**fish** ng/g	14 0.1–30	30 0.1–200	44 0.2–230	122 47–267
F3	**fish** ng/g	—	—	4.9 1–15	—
M2	**human milk** ng/g	★ 0.4 0.1–1.0	★ 0.13 0.02–?	★ 0.530 0.12–1.1	350 <1–2102
L2	**human fat** ng/g	—	—	—	110 <1–830

Mean on 1st line, range below; ★ dubious data (see text).

Sources: 1. aldrin and dieldrin in water, soil, clam and fish: (Nair et al., 1991); except F3 fish Σdrin from (Kannan et al., 1995); 2. aldrin and dieldrin in human milk: (Dureja et al., 1991); 3. HCB in all compartments: (Nair and Pillai, 1989).

3.3.2.1.2 Jaipur Lakes

Two small lakes, Mahala Reservoir / Mahalon Lake and Jalmahal Lake in the Jaipur area south of Delhi in Rajasthan state, were sampled for POPs insecticides from September 1985 to October 1987. Jaipur lies just inside the greater Ganges watershed in the headwaters of the River Chambal tributary of the River Yamuna that ultimately discharges to the Ganges. The two lakes appear to be located on headwater tributaries of River Chambal draining lands to the southwest of Jaipur. The mostly agricultural upstream drainage basin of Mahala Reservoir is about 57,000 ha. Watershed area was not given for Jalmahal Lake. The sketch map suggests it has a surface area of about 125 ha. For 1987-88, Bakre et al. (1990a) have reported annual usage of 4.02 t ai HCH and 895 litres of aldrin 30% ai solution in the upstream Mahala Reservoir watershed. No data were given for DDT or heptachlor usage, nor was it specified whether DDT was used for agriculture or vector control. Rainfall patterns for the area were shown in Figure 3.3.

Initially, Kumar et al. (1988) reported on Mahalon and Jalmahal Lakes from June 1985 to July 1986, and Bakre et al. (1990a) reported on Mahala Reservoir from September 1985 to October 1987. Mahalon Lake and Mahala Reservoir appear to be

Table 3.6 POPs insecticides (μg/L) in Jaipur area Lakes.

Pesticide	n	A-mean [a]	G-mean [b]	Min.	Max.	% detected
γ-HCH	42	2.30	0.96	<0.01	26.4	95
ΣHCH [c]	42	5.15	2.02	0.02	46.5	—
p,p'-DDT	42	2.58	0.23	<0.01	12.1	71
p,p'-DDE	42	2.69	0.38	<.004	31.6	83
p,p'-DDD	42	2.97	0.58	<0.01	22.6	90
ΣDDT	42	8.21	3.46	0.08	49.7	[d] 100
aldrin	42	1.76	0.57	<0.01	25.0	98
heptachlor	33	0.54	0.12	<.004	2.1	76

[a] arithmetic mean, [b] geometric mean,
[c] ΣHCH = α-HCH + β-HCH + γ-HCH prorated to all 42 samples from complete isomer data on final 9 samples from Mahala Reservoir (Bakre et al., 1990a),
[d] one or more DDT species were detected in all sample records.

the same water body. Data for overlapping months differ slightly between the two reports; however, temporal patterns are similar and the two data sets are statistically indistinguishable. Jalmahal Lake had only 7 sampled months and observations were not statistically different from the other two data sets. POPs concentrations in fish (Bakre et al., 1990b) and birds (Misra and Bakre, 1994) are available for Mahala Reservoir.

Table 3.6 gives the gross statistical summary of all POPs data reported for Mahala Reservoir and Jalmahal Lake. The general order of contamination is ΣDDT > ΣHCH > aldrin > heptachlor at levels that are strikingly high. Arithmetic mean concentrations are especially high due to a few extreme observations. The geometric mean concentrations are more broadly representative of the majority of observations for each variable. These very high concentrations indicate heavy pesticide usage. Peak concentrations approach or exceed nominal water solubility limits for these chemicals. Bakre et al. (1990a) suggest that the organic carbon (TOC ~ 6.3 mg/L) and suspended solids (20–2190 mg/L) content of these samples accounts for the extraordinarily high concentrations observed.

Complete HCH isomer data, which were only available for 1987 samples by Bakre et al. (1990a), had a mean profile of α-HCH (34%), β-HCH (21%), γ-HCH (46%). The mean composition of DDT was roughly equal among species; however, there was considerable variability over the two years of record. At several times [Apr-May 1986, Mar-May 1987, and Aug-Oct 1987] the percentage of unmetabolized DDT [*p,p'*-DDT] ranged from 50–99 % indicating fresh inputs. On roughly 33% of samples, ΣDDT comprised 50–99% *p,p'*-DDD indicative of reducing conditions. Otherwise, there were no obvious seasonal patterns evident in the HCH, DDT and aldrin data.

3.3.2.2 Ganges River — Main Branch

3.3.2.2.1 HCH in Rainwater at Hardwar 1992

Hardwar (also Haridwar) is located in the Himalayan foothills not far below where the Ganga leaves the mountains and before it descends onto the Gangetic plain. Other than for malaria control, few pesticides are used in the area (Gajbhiye et al., 1995).

Dua et al. (1994) describe the use of HCH to control mosquitoes and flies during a major religious congregation from January to April 1992 on a 130 km^2 site, as well as for routine anti-malarial operations within the greater 1962 km^2 Hardwar district. About 3 t HCH ai were used on the festival site and 2.8 t ai were used within the greater Hardwar district. A *fortified* HCH comprising 38% α-HCH, 37% γ-HCH, 24% δ-HCH was used during 1992.

Monthly rainfall, mean rainwater ∑HCH concentrations and ∑HCH wet deposition over Hardwar district are given in Table 3.7. The distribution of HCH isomers observed in rainwater was similar to that in the applied technical HCH. It is expected that at least 75% of the applied HCH volatilized, and that the true volatilization losses may well have been 90% or more. Even if only 50% volatilized, it is clear that very little of the volatilized HCH returned to the district in rainfall.

Several important points are illustrated by the Hardwar district case:

- Wet deposition of HCH is only a small fraction of the insecticide applied, and most likely only a small fraction of the insecticide lost to volatilization.
- The highly bioaccumulative β-HCH isomer was absent in the *fortified* HCH applied at Hardwar. It is unclear whether this is merely indicative of typical variability in Indian HCH composition, or a trend towards using technical HCH formulations that release less non-insecticidal organochlorine isomers.
- Areally, HCH usage for the religious festival was 16 fold more intense than routine anti-malarial operations. Sacred sites and extended religious congregations are frequent along the Ganges. Cumulative insecticide usage at religious sites and activities could be significant.

3.3.2.2.2 Ganges River at Farrukhabad

Farrukhabad is on the Ganges about 250 km downstream of Haridwar and upstream of the confluence with the Ramganga tributary. The Farrukhabad administrative district on the south side of the Ganges comprises about 428,883 ha under intensive cultivation of maize, potato, wheat and vegetables. According to Gajbhiye et al. (1995), pesticide usage jumped dramatically from about 8 t in 1990 to 42.5 t in 1992. They estimate 50% of the pesticides were insecticides of which 60% were organochlorines.

Table 3.7 ΣHCH in rainwater and wet deposition on Hardwar district 1992.

Month	Rainfall mm	ΣHCH ng/L	ΣHCH deposition kg
Jan	35.1	113	7.8
Feb	43.6	60	5.1
Mar	2.5	‡ —	0.4
Apr	0	—	0
May	23.5	—	3.5
Jun	31.9	152	9.5
Jul	215.0	95	40.1
Aug	686.9	32	43.8
Sep	249.7	8	3.9
Oct	1.4	—	0.2
Nov	5.4	—	0.8
Dec	0	—	0
Total	1,295.0		115.2

‡ average concentration (77 ng/L) used for missing months.

Thus organochlorines constituted about 30% of the total pesticide consumption — about 2.4 t in 1990 and 12.8 t in 1992. Insecticide concentrations in agricultural drain water, groundwater, and Ganges River waters are available in two studies: (Agnihotri et al., 1994; Gajbhiye et al., 1995).

In 1991-92, Agnihotri et al. (1994), collected bimonthly water samples at 10 sites along an 8 km reach of the Ganges past Farrukhabad district. Samples were obtained for the months of September, November, January, March, May and July. Samples were collected "mostly from points where [agricultural] drains met the river", and thus may represent near-bank contaminant plumes of agricultural drains rather than well-mixed Ganges waters.

Farrukhabad data are summarized in Table 3.8. The two dominant insecticides are ΣHCH and ΣDDT. The *p,p'*-DDD metabolite was not measured; so that, true ΣDDT would be somewhat higher. Because DDT usage was restricted to public health in October 1989, the DDT seen in the Ganges over 1991-92 should ostensibly represent mainly runoff from anti-malarial operations. Unmetabolized DDT always exceeded DDE with a mean DDT/DDE ratio of about 5 suggesting fresh DDT inputs. The mean HCH composition profile in water was 44% α-HCH; 26% β-HCH; and 30% γ-HCH indicating usage of standard crude HCH.

The drins, endosulfan and heptachlor are present at nearly identical concentrations. Total drin composition averaged 65% aldrin and 35% dieldrin. Aldrin was apparently used as a soil insecticide in potato cultivation (Agnihotri et al., 1994). Heptachlor likely had similar usage. Endosulfan composition averaged 79% α-endosulfan and 21% β-endosulfan. Endosulfan sulphate was never detected.

Table 3.8 POPs insecticides (ng/L) in the Ganges River near Farrukhabad, 1991-92.

	Means			Samples	
	†Grand mean	Min. mean	Max. mean	Min.	Max.
ΣHCH	245	154	374	22	1,119
ΣDDT	252	159	362	‡nd	832
Σdrin	38	28	56	nd	120
Σendo	36	15	60	nd	232
heptachlor	37	16	78	nd	412

† mean of 10 site means; ‡nd = not detected;
ΣHCH = α-HCH + β-HCH + γ-HCH;
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *o,p'*-DDE;
Σdrin = aldrin + dieldrin; Σendo = α-endosulfan + β-endosulfan + endosulfan sulphate.

3.3.2.2.2.1 Agricultural Drain Waters and Groundwaters

Gajbhiye et al. (1995) undertook further studies in an attempt to quantify insecticide losses from the adjacent agricultural lands into the Ganges River. They collected additional samples from the agricultural surface drains and groundwater samples from agricultural fields. These data should be regarded cautiously as the numbers of samples taken are unknown, but they are helpful for assessing insecticide water concentration data from other sources. Sampling appears to have been conducted from September 1991 to August 1992, but the year is never stated clearly.

Figure 3.7 compares the concentration means and ranges observed in agricultural drain water, groundwater and river water. Generally, we would expect the nearshore river water samples to have undergone some dilution and be lower than drain water. During the high river flow months, Ganges waters flow into the alluvial aquifers of the adjacent flood plains. The groundwater concentrations shown in Figure 3.7 are for "dry" months when the aquifer was thought to be discharging to the river.

It seems clear that HCH concentrations are highest in drain waters. As HCH is the most water soluble of these insecticides, it should be more prone to surface runoff. HCH usage patterns in the district were not given. Direct spraying onto agricultural drains during monsoon months for mosquito control could explain its high concentration in agricultural drain water.

DDT concentrations were comparable in both drain waters and groundwaters at levels about 90-100 ng/L higher than observed in river waters. Mosquito control spraying could explain its presence in drain waters; however, its abundance in

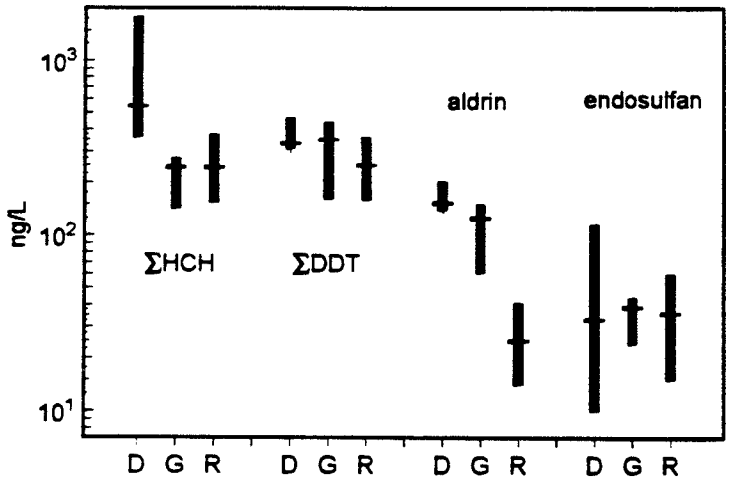


Figure 3.7 POPs insecticides in drain, ground and near bank river water near Farrukhabad, 1991-92; D ≡ drain water; G ≡ groundwater, R ≡ nearshore river water (see Table 3.8); minimum endosulfan concentration is < 1 ng/L.; vertical bars show range; cross bar shows arithmetic mean.

groundwaters suggests continuing usage in agriculture despite the ban of October 1989. Legislative provisions for consumption of existing stocks might explain why DDT usage in agriculture appears to have persisted into 1991-92.

Aldrin concentrations in drain waters and groundwaters were remarkably high (125-150 ng/L) relative to river waters. Micro organisms may be metabolizing the aldrin rapidly to dieldrin that evidently was not measured in these samples. Aldrin usage appears to have been appreciable in Farrukhabad district during the sampling period. For endosulfan there appears to be no difference among the three waters.

3.3.2.2.3 Ganges River Sediments at Allahabad

In April and June of 1988, DDT, HCH, aldrin and endosulfan were determined at 4 sites on a reach of the Ganges in Allahabad district (Sinha, 1991). It is not known if the reach is above or below the confluence with River Yamuna, the Ganges' largest tributary. At each site, samples were collected near the left and right banks, and at midstream. Problems with this data set include the following:

- Data were reported in implausible units of mg/g at which levels the insecticide content of the sediments approaches the active ingredient concentrations in some commercial formulations. Cross referencing HCH and DDT against other soil and sediment data for India, suggested that the true units were most likely ng/g.

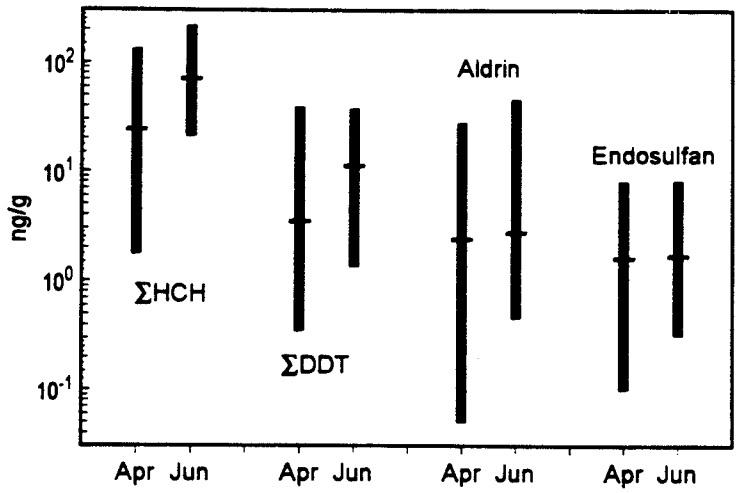


Figure 3.8 POPs insecticides in Ganges River bottom sediments near Allahabad, 1988; vertical bars show range; cross bar shows arithmetic mean.

Table 3.9 POPs concentrations in Ganges River sediments near Allahabad, 1988; data adapted from (Sinha, 1991); assumed units are ng/g wet weight (see text).

	ΣHCH			ΣDDT		
	A-mean	G-mean	Range	A-mean	G-mean	Range
April	43.6	24.3	1.8–133	8.50	3.49	0.35–38.7
June	85.4	71.2	21–216	17.40	11.26	1.36–37.5
all	64.5	41.6	1.8–216	13.00	6.27	0.35–38.7
	Aldrin			Endosulfan		
	A-mean	G-mean	Range	A-mean	G-mean	Range
April	7.52	2.39	0.05–27.9	3.04	1.60	<0.1–8.01
June	9.23	2.73	0.46–45.5	2.43	1.68	0.32–8.14
all	8.37	2.56	0.05–45.5	2.73	1.64	<0.1–8.14

A-mean = arithmetic mean; G-mean = geometric mean.

- The measurement basis is unspecified. Wet weight concentrations are assumed. Dry weight equivalents would be higher.

- Data are reported as total measurements, i.e., ΣDDT, ΣHCH and endosulfan; however, precisely which components were actually measured was not stated.

Data were summarized various ways. Samples from near banks generally had higher concentrations, but patterns were often inconsistent from site to site, across insecticides, or from April to June. Lumped data do exhibit rising HCH and DDT levels from April to June (Table 3.9, Figure 3.8). Mean concentrations of ΣHCH and ΣDDT approximately double from April to June. Evidently, the beginning of the wet season in Allahabad district (Figure 3.1) instigates spraying presumably for mosquito control.

There was no appreciable rise in either aldrin or endosulfan levels at the onset of the rainy season. Aldrin usage in the area was remarkably high in 1988. Had the degradation product dieldrin been measured, the Σdrin concentration may well have exceeded that of ΣDDT during April.

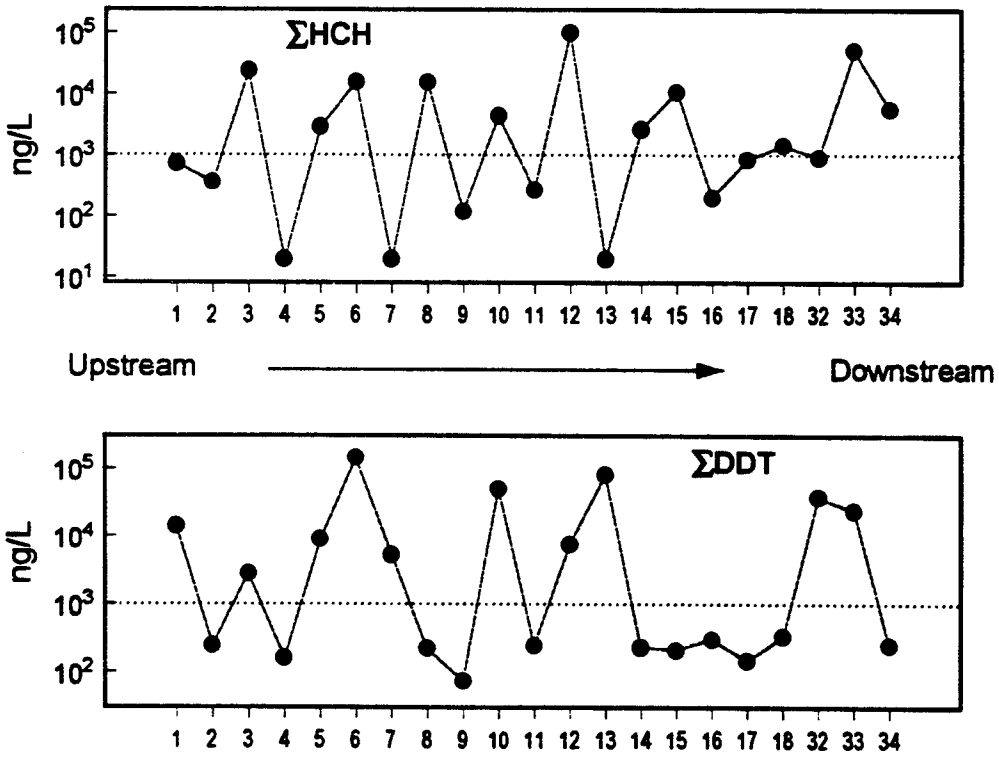


Figure 3.9 ΣHCH and ΣDDT in Ganges River along Varanasi waterfront, August 1992; site numbers as in Nayak et al. (1995).

Table 3.10 POPs insecticides in Ganges River at Varanasi (μg/L).

Pesticide	n	A-mean [a]	G-mean [b]	L-mean [c]	Min.	Max.
ΣHCH	34	11.6	1.49	0.39	<0.04	99.8
ΣDDT	34	12.9	1.58	0.29	0.07	143.0
Σendo	34	10.6	1.96	0.44	0.10	66.5

[a] arithmetic mean; [b] geometric mean; [c] mean of lower (<1 μg/L reference line) observations; see text.
ΣHCH = α-HCH + β-HCH + γ-HCH + δ-HCH; ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE,
Σendo = α-endosulfan + β-endosulfan + endosulfan sulphate.

3.3.2.2.4 Ganges River at Varanasi

In August 1992, during peak monsoonal rains, Nayak et al. (1995) sampled the Ganges at Varanasi at 34 sites off the city ghats [stairways descending to the water] on one side, and in the nearshore of the agricultural lands on the opposite bank. These data bear the following caveats:

- Samples were all collected from the banks and may be most representative of near shore contaminant plumes from urban and agricultural drains.
- *p,p'*-DDD was not measured; hence, total DDT is underestimated.
- In standard technical DDT, the ratio $\lambda = p,p'\text{-DDT} / o,p'\text{-DDT} \approx 5$, and usually for water samples, $\lambda > 1$. The Varanasi samples have mean $\lambda \approx 0.4$ and only 3 of 34 samples have $\lambda > 1$. If the unusual composition is not the result of inadvertent transposition of the DDT isomer labels, the analytical methods employed by the investigators may be suspect.

Summary data for total pesticide concentrations are given in Table 3.10. Remarkably, all three insecticides are at virtually the same mean level. ΣDDT would likely be higher if *p,p'*-DDD had been measured. ΣDDT and endosulfan were strongly correlated and would yield good regression predictor equations on a log-log scale. Evidently, DDT and endosulfan are being used either jointly, or concurrently in close proximity, while ΣHCH was used independently.

Varanasi data show tremendous variability from site to site. Figure 3.9 shows the site-to-site variation in concentrations along the Varanasi City bank of the Ganges in upstream to downstream order where the site numbers are those used by Nayak et al. (1995). The plots suggest that the sporadic low concentrations, generally those below the 1 μg/L reference lines, are likely more representative of the background insecticide concentrations in the main Ganges water volume passing Varanasi. The nominal *background* mean concentrations calculated from data below the 1 μg/L reference lines are: 390 ng/L ΣHCH, 290 ng/L ΣDDT, and 440 ng/L Σendo. These are similar to mean

concentrations reported upstream at Farrukhabad. What cannot be determined is how much the strong, high concentration influents entering at Varanasi would raise the concentration of Ganges after thorough mixing.

3.3.2.2.5 Hooghly (Ganges) River near Calcutta

Haldar et al. (1989) collected 14 samples in a 20 km reach of the lower Hooghly River through the district known as *24 Parganas*. Samples were analyzed for DDT and endosulfan isomers. Data are difficult to interpret. Samples were collected off the banks (ghats), from mid stream, and from canals discharging to the Hooghly; but, the locations of several samples were not given, nor were sampling dates, year of sampling, nor analytical detection limits. DDD isomers were not determined.

DDT data are summarized in Table 3.11 as best as could be interpreted given the deficiencies. Samples 1 and 2 were from a tributary industrial canal, and thus deleted. The origin of sample number 3 with a high concentration of about 4 µg/L was unclear. Two samples (13 and 14), clearly identified as mid stream, were grouped with another (sample 5) that was taken at a bridge (presumably over the Hooghly).

Taken together, the data suggest that DDT concentrations in the Hooghly River can range widely from ng/L to µg/L levels over short distances and times. Reasonably good temporally and cross-sectionally representative sampling is essential to obtaining reliable estimates of Hooghly River POPs concentrations and mass discharges.

Table 3.11 ΣDDT (ng/L) in lower Hooghly River.

	n	Mean	G-mean	Min.	Max.
samples 3–14	12	1,340	488	6	4,070
samples 4–14	11	1,090	402	6	2,660
mid-stream (samples 5,13,14)	3	683	190	6	1,180

ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *o,p'*-DDE; G-mean = geometric mean.

3.3.2.2.6 Ganges River Dolphins and Fish

POPs burdens in 4 river dolphins accidentally netted near Patna were reviewed by (Kannan et al., 1993a; Kannan et al., 1994a). Mean concentrations are given in Table 3.12 for dolphin blubber and a pooled sample of 4 bottom feeding fish species from the guts of the dolphins. For contrast, mean POPs concentrations are also given for Bay of

Bengal dolphins and St. Lawrence River beluga. Ganges fish are contrasted against Japanese river fish in Table 3.13.

The order of contamination in Ganges dolphin is ΣDDT > PCBs ≥ ΣHCH > Σdrin ≥ ΣCHL > HCB, while in fish, ΣDDT > ΣHCH > ΣCHL ≥ PCBs > Σdrin > HCB. From fish to dolphin, DDT is the most bioaccumulative contaminant followed by PCBs and HCB.

After eliminating variations due to differences in fat content, comparison of Ganges dolphins to Bay of Bengal dolphins from south India shows no real differences in DDT, PCBs or HCB burdens. For DDT, which is more heavily used in the Ganges basin, these are reassuring results. Superficially HCH contamination of Ganges dolphins appears higher, but the sample size is too small to claim significant differences exist.

In comparison with St. Lawrence beluga on a fat weight basis, Ganges dolphins are about 3 fold more contaminated with HCH, but 10–80 fold less contaminated with PCBs, HCB and chlordane. DDT contamination in Ganges dolphins is about 1/3 that in St. Lawrence beluga. This is noteworthy given that DDT water concentrations are so high in Ganges waters.

On a wet weight basis, Ganges fish have 14-15 times higher levels of DDT and HCH than Japanese fish. Surprisingly, chlordane levels are about equal. Chlordane in Ganges fish may originate mostly from heavy usage of heptachlor as a soil insecticide rather than chlordane usage per se. PCBs in Ganges fish are only about 15% of those in Japanese fish.

Table 3.12 POPs (ng/g wet weight) in Ganges River dolphin blubber and fish near Patna.

	Ganges dolphin	Ganges fish	†BCF Ganges dolphin / fish	Bay of Bengal dolphin [a]	St. Lawrence beluga [b]
fat %	45	3.4	—	58	87
ΣDDT	9,700	160	61	12,774	57,700
PCBs	453	20	23	699	54,400
ΣHCH	425	77	6	316	298
dieldrin	42	2.9	15	—	724
Σdrin	56	5.6	10	—	—
ΣCHL	46	35	1	—	[c] 5,270
HCB	5.6	0.2	23	7.0	929

† BCF = bioconcentration factor.
[a] mean of 12 specimens of 3 species given by (Tanabe et al., 1993).
[b] mean of 4 males and 5 females given by (Béland et al., 1993).
[c] may not include heptachlor, heptachlor epoxide, and oxychlordane.
Σdrin = aldrin + dieldrin; ΣCHL = heptachlor + heptachlor epoxide + *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor + oxychlordane.

Table 3.13 POPs (ng/g wet weight) in Ganges fish versus River Nagaragawa, Japan lizard goby; 1982-86; data from (Loganathan et al., 1989).

	ΣDDT	ΣHCH	PCBs	ΣCHL
Ganges fish	160	77	20	[a] 35
Japanese fish	11	5.5	128	[b] 30
ratio	14.5	14	0.16	~1.0

[a] ΣCHL = heptachlor + heptachlor epoxide + *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor + oxychlordane.
[b] ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

3.3.2.2.7 Ganges River: Other Data Sources

Other reports on River Ganga that were not available for review include (Haldar et al., 1990; Thakkar, 1986). There now seems to be some regular, possibly annual, pesticide monitoring conducted in the mid reaches of the Ganges (Anon, 1992; Ray, 1989; Ray, 1992). Kulshrestha (1991) is reported to have reviewed pesticide surveys of Indian surface waters and sediments to about 1990.

3.3.3 South India Marine Environment

3.3.3.1 POPs Residues in Seawater

Some concentration data for the surrounding seawaters of India are listed in Table 3.14. The first data are from a Japanese global seawater survey conducted by Iwata et al. (1993) over 1989-90. Data represent the mean of 7 surface samples from the Arabian Sea and the Bay of Bengal analyzed by the most modern high resolution methods. Sites appear to represent open seas at least 100 km offshore.

The second data set (Sarkar and Sen Gupta, 1989) was obtained in the Arabian Sea off the central west Indian coast in January 1987. Samples were obtained at 20 m depth at 6 sites between 16° N and 21° N. The 6 sites appear to represent nearshore seas of about 2–20 km off the coast. Each site was represented by 3 replicates and the means of the 6 sub-means are given in Table 3.14.

The differences between the inshore and offshore samples for γ-HCH and ΣDDT are extraordinary. The respective ratios of inshore concentrations to offshore concentrations are about 25 for γ-HCH, and 5,860 for ΣDDT. The difference in mean ΣDDT concentrations between the two data sets defies ready explanation. ***The ΣDDT data reported by Sarkar and Sen Gupta (1989) are higher than observed in certain***

Indian freshwaters. These data should be regarded with suspicion until corroborated by further sampling.

3.3.3.2 POPs in Coastal Marine Sediments

Contamination by organochlorine pesticides of the offshore marine environment is evident in several limited marine sediment surveys including:

A. West Coast — Arabian Sea

1. HCH, DDT, aldrin and dieldrin measurements on 14 samples obtained from September 11-30, 1986 on the central west coast (15N –19N) roughly between Bombay and Marmagao (Sarkar and Sen Gupta, 1987),
2. HCH, DDT, aldrin and dieldrin measurements on 12 samples obtained January-February 1988 from the central west coast (15N –22N) roughly between Dwarka and Marmagao (Sarkar and Sen Gupta, 1991),

B. East Coast — Bay of Bengal

3. γ-HCH, ΣDDT, aldrin and dieldrin measurements on 29 samples from the east coast (12N –22N) roughly from south of Madras to the mouth of the Hooghly River (Sarkar and Sen Gupta, 1988a) on cruises in 1984 [Sep 27 – Oct 6] and 1985 [May 26 – Jun 10; Sep 30 – Oct 3],
4. DDT species measurements on 29 samples from the east coast (12N –22N) roughly from south of Madras to the mouth of the Hooghly River (Sarkar and Sen Gupta, 1988b) from the same cruises in 1984 and 1985 as for data set 3.

As judged from sketch maps, most sites appear to be within 1–20 km of the coast, while some extend out to 100 km or more. The east coast data sets 3 and 4 are of particular interest as most surface drainage from India discharges to the east coast.

To summarize measurements reported in sources 1 to 4, data were entered in spreadsheets for statistical analysis. Data sets 1 and 4 were read directly from source tables; however, data sets 2 and 3 had to be scaled from smallish bar graphs which introduced some errors. Cross-checks on sum ΣDDT where it was given both in graphs and tables revealed certain discrepancies. DDT data from the tables in (Sarkar and Sen Gupta, 1987) and (Sarkar and Sen Gupta, 1988b) were given precedence.

For the east coast surveys, γ-HCH is the only HCH isomer given. East coast ΣHCH measurements were estimated by pro-rating from γ-HCH measurements according to the mean HCH isomer composition profile for the 12 samples in west coast data set 2 [44% α-HCH, 18% β-HCH, 38% γ-HCH]. West coast data set 1 reported *BHC* [interpreted here as ΣHCH], but no individual HCH isomer data were given.

Table 3.14 POPs insecticides in seawaters surrounding India.

	Sea water concentrations pg/L		
	Mean	Min.	Max.
Arabian Sea / Bay of Bengal — offshore [a]			
γ-HCH	110	27	190
ΣHCH [b]	720	130	1,300
ΣDDT [c]	10	1.6	24
PCBs	21	13	46
ΣChlordane [d]	9.5	3.4	17
Arabian Sea — nearshore [e]			
γ-HCH	2,720	260	9,400
ΣHCH [f]	17,800	1,700	61,500
ΣDDT [c]	58,600	[h] nd	307,000
ΣDDT [g]	96,000	nd	444,000
Σdrin [j]	14,070	2,580	51,000

[a] from (Iwata et al., 1993); [b] ΣHCH = α-HCH + γ-HCH;
[c] ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE.
[d] Σchlordane = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor.
[e] from (Sarkar and Sen Gupta, 1989); most sites about 1–20 km offshore.
[f] pro-rated estimate as ΣHCH / γ-HCH ratio observed by Iwata et al. (1993) for open Arabian Sea and Bay of Bengal samples.
[g] ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *o,p'*-DDE + *o,p'*-DDD.
[h] nd = not detected. [j] Σdrin = aldrin + dieldrin.

The available data and summary statistics must be interpreted cautiously as coarse, exploratory survey results. Sampling sites are sparsely and non-systematically distributed. Useful ancillary data including sample particle size composition, organic carbon content, analytical detection limits, precise site locations, and depositional regimes were not given. Thus, sites having low organochlorine levels may indicate coarse, inorganic sediments, rather than contaminant discharges emanating from adjacent coastal watersheds. Conversely, sites with high concentrations may be indicative of depositional regimes that focus fine, organic rich sediments.

3.3.3.2.1 Marine Sediment Data Summary — East / West Coast Comparison

Data summaries for ΣHCH, Σdrin = aldrin + dieldrin, and ΣDDT are given in Table 3.15 and shown on Figure 3.10. Differences between west and east coasts are dramatic and consistent across the three contaminants. For all three, contamination of

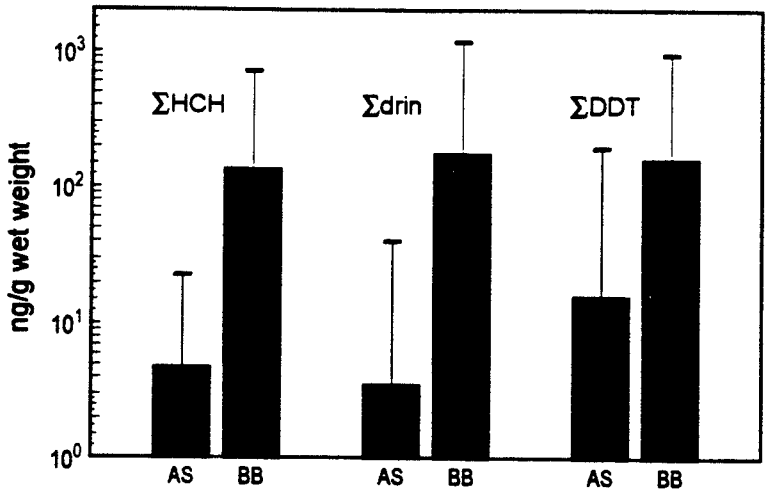


Figure 3.10 ΣHCH, Σdrin (aldrin + dieldrin) and ΣDDT in marine sediments from west and east coasts of India. AS ≡ Arabian Sea; BB ≡ Bay of Bengal.

Table 3.15 POPs insecticides (ng/g wet weight) in Indian coastal marine sediments.

Pesticide	Water body	N	[a] % detected	Mean	Min.	Max.
ΣHCH [b]	Bay of Bengal	29	97	135.0	nd	584.0
	Arabian Sea	26	92	4.7	nd	17.9
Ratio of east / west means				28.4		
Σdrin	Bay of Bengal	29	86	175.2	nd	1,004.0
	Arabian Sea	26	62	3.5	nd	36.6
Ratio of east / west means				284.9		
ΣDDT	Bay of Bengal	29	93	157.7	nd	792.0
	Arabian Sea	26	56	15.6	nd	179.2
Ratio of east / west means				10.1		

[a] percentage of samples with at least one isomer or metabolite above detection limits.
[b] ΣHCH estimated by pro-rating γ-HCH according to HCH isomer composition profile for Arabian Sea.
nd = not detected.
Σdrin = aldrin + dieldrin.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *o,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD.

Bay of Bengal sediments was at least one order of magnitude greater than contamination of Arabian Sea sediments in the mid 1980s.

While contamination by HCH and DDT is not unexpected, the severity of aldrin and dieldrin pollution is startling as the drins are rarely mentioned in discussions of Indian pesticide production and usage.

3.3.3.2.2 Comparisons With Other Data

As systematic marine bottom sediment survey data are not widely available, the extent and degree of Indian coastal marine sediment contamination were assessed vis-à-vis systematic bottom sediment survey data for the Great Lakes circa 1970 (Frank et al., 1977; Frank et al., 1981; Frank et al., 1979). The Great Lakes surveys were conducted about the time when both Canada and the U.S. began to severely restrict DDT after almost two decades of uncontrolled heavy usage. Having the most agricultural drainage, Lake Erie showed the highest levels of DDT contamination.

The Indian marine sediment data pose another problem in comparisons with external data as the Indian east coast [and most likely also west coast] sediment concentrations have been determined on wet weight basis. Wet weight concentrations will be lower than sediment data from elsewhere that are usually reported on a dry weight basis to eliminate variations due to water content. Wet weight concentrations cannot be readily adjusted to dry weight basis without knowledge of the water content of the sample and the specific gravity of the solid phase (Mudroch and MacKnight, 1991). Under worst case scenarios, the equivalent dry weight concentrations could be an order of magnitude higher than the wet weight concentrations; however, it is more likely that the differences are within a factor of two.

3.3.3.2.3 ΣDDT

DDT survey data were summarized in Table 3.15. Figure 3.11 shows that Arabian Sea sediments off India's central west coast are modestly contaminated with ΣDDT, while sediments of the Bay of Bengal off India's east coast have higher DDT levels than bottom sediments from Great Lakes Erie and Huron ca. 1970. Non-depositional zones of Lake Huron were lightly contaminated at a mean concentration of 3.8 ng/g, while those of Lake Erie showed modest contamination of 15.5 ng/g dry weight comparable to the mean ΣDDT concentration of 17.9 ng/g wet weight observed at 26 sites in the Arabian Sea.

The 29 Bay of Bengal sites off India's east coast show the highest ΣDDT levels, and would be higher yet had concentrations been standardized to dry weight basis. In the 1970 Great Lakes surveys, western Lake Erie was the Great Lakes zone most contaminated by ΣDDT with a mean concentration of 70 ng/g dry weight that originated

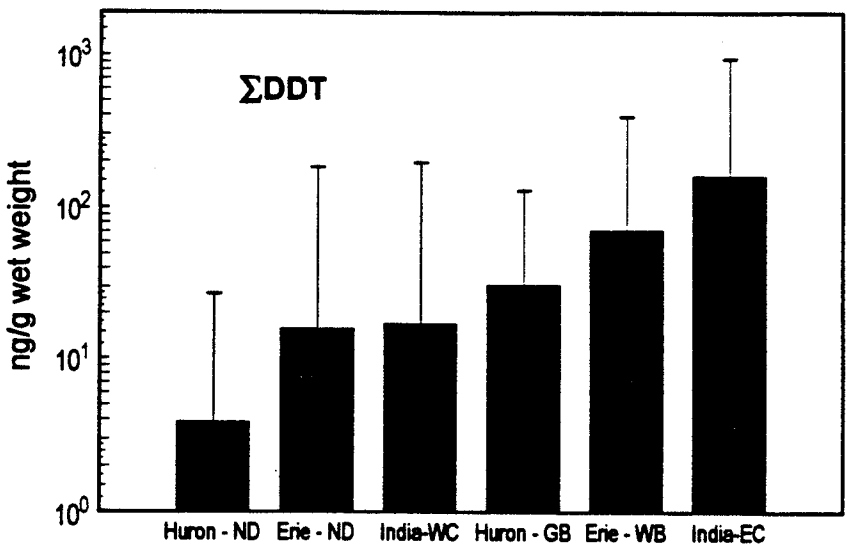


Figure 3.11 ΣDDT in sediments of Indian coastal waters versus Great Lakes sediments; ND ≡ non-depositional zone; WC ≡ west coast; GB ≡ Goderich basin; WB ≡ western basin; EC ≡ east coast; Indian concentrations on wet weight basis; Lake Huron and Erie concentrations on dry weight basis; crossed bars show concentration maxima; Great Lakes data from (Frank et al., 1977; Frank et al., 1979).

from intensive agriculture. Mean ΣDDT levels of 31 ng/g in the Goderich Basin of southern Lake Huron reflected lower intensity agriculture on the adjacent Canadian side. ΣDDT in Bay of Bengal marine sediments have a mean concentration (158 ng/g wet weight) more than two fold higher than 1970 Lake Erie western basin sediments.

3.3.3.3 POPs Residues in Mussels

Table 3.16 lists 9 coastal sites of southern peninsular India where mussels were surveyed by Ramesh et al. (1990). HCH, DDT and PCBs data are given in Table 3.17 which includes oyster data from U.S. sites on the Gulf of Mexico coast for contrast. Because variations in fat content mask certain trends in the mussel data, summaries of lipid normalized data are also given. West coastal mussels were much fatter than east coast mussels (mean 3.1% fat versus mean 0.6% fat respectively).

On fat weight basis, the east coast sites show unambiguously higher levels of all three contaminants, and the two Madras sites have the highest DDT and PCB levels. Higher PCB contamination is expected in the vicinity of India's fourth largest city, and the high DDT levels are likely attributable to anti-malarial operations in the Madras

Table 3.16 Mussel survey sites, south India.

Location	Coast	Date
Ennore estuary mouth — Madras	E	21-Aug-88
Kasimedu-Royapuram fishing harbour — Madras	E	30-Aug-88
Pondicherry harbour	E	21-Aug-88
Cuddalore harbour	E	21-Aug-88
Vellar River estuary — Porto Novo	E	17-Feb-89
Nagapatti-nam harbour	E	28-Aug-88
Calicut	W	1-Dec-88
Suratkal	W	6-May-89
Goa harbour	W	25-Feb-89

Table 3.17 POPs in mussels from south India coastal sites (Ramesh et al., 1990b).

Concentration ng/g	Fat %	ΣDDT	ΣHCH	PCBs
A. wet weight basis				
Mean all sites	1.46	18.8	7.3	3.7
Range	0.29–3.80	3.0–39.6	4.3–15.6	<1.0–7.1
East Coast – Madras	0.37	24.7	5.0	6.5
East Coast – other	0.75	7.2	9.2	1.6
West Coast	3.13	30.4	6.3	4.7
Gulf of Mexico oysters [a]		44	1.0	170
B. fat weight basis				
Mean all sites		2,430	1,100	588
Range		249–8,130	147–2,550	111–2,150
East Coast – Madras		6,500	1,390	1,783
East Coast – other		1,400	1,620	314
West Coast		1,090	206	157

[a] U.S. Mussel Watch program data for 1980 from (Wade et al., 1988).
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD; ΣHCH = α-HCH + β-HCH + γ-HCH.

area (Sharma, 1991). In south India, malaria is evidently more concentrated in urban than in rural areas.

From broad global comparison of ΣHCH concentrations reported in clams, mussels and oysters, Ramesh and colleagues concluded that India's coastal marine environment ranked amongst the most highly contaminated with HCH in the world. DDT contamination was considered moderate, while PCB contamination was low. U.S. Gulf of Mexico oysters circa 1986 had significantly higher PCBs, slightly higher ΣDDT and much lower ΣHCH.

Table 3.18 POPs in blubber of dolphins from Porto Novo coastal seas.

wet weight basis	n	Fat %	ΣDDT μg/g	ΣHCH ng/g	PCBs ng/g
Spinner dolphin					
Mean	5	46	17.4	345	530
Range		22–69	4.6–35.4	60–860	240–950
Bottlenose dolphin					
Mean	4	57	7.3	132	530
Range		40–69	2.1–14.2	60–860	63–192
Humpback dolphin					
Mean	3	79	12.3	514	1,230
Range		52–93	10.8–14.4	200–1,120	920–1,800
Ganges R dolphin [a]					
Mean	4	45	9.7	425	453
Range		31–74	4.7–13.0	190–610	360–620
St. Lawrence R beluga [b]					
Mean (male)	4	87	101	370	75,800
Range		83–90	52–123	280–510	54,000–89,000
Mean (female)	5	87	23	240	37,300
Range		80–91	4–43	120–320	14,500–68,700

[a] Ganges dolphin data from (Kannan et al., 1994a).
[b] St. Lawrence R beluga data from (Béland et al., 1993).

3.3.3.4 POPs Residues in Dolphins

POPs concentrations in dolphin blubber from carcasses netted accidentally by fishermen over 1990–91 in the seas off Porto Novo, Tamil Nadu (Tanabe et al., 1993) are given in Table 3.18. HCB was also determined, but the levels were very low and have not been shown. For reference, contaminant burdens from Ganges River dolphins (Kannan et al., 1994a) and St. Lawrence River beluga (Béland et al., 1993) are included. The latter rank amongst the world's most contaminated small cetaceans.

In Bay of Bengal dolphins, the order of contamination is DDT > PCBs > HCH > HCB. Contrasted against mussel data of Table 3.18, PCBs accumulation generally exceeds that of HCH. In global comparison of DDT in small cetaceans, the Bay of Bengal dolphins rank toward the lower range of concentrations observed in animals from the north Atlantic and north Pacific oceans, but are notably more contaminated than cetaceans from the south Atlantic (Tanabe et al., 1993).

HCH levels in marine mammals have rarely been reported. HCH in Bay of Bengal dolphins does exceed concentrations observed in St. Lawrence beluga; however, HCH usage in North America is restricted to lindane which is used in much lower quantities than crude HCH in India. Considering the level of DDT and HCH used in India, the investigators construed the relatively low degree of contamination as evidence that most of the HCH and DDT applied is volatilized and transported out of the region by atmospheric processes. Nonetheless, DDT and HCH are reaching the Indian coastal marine environment and contaminating wildlife. PCB burdens in Bay of Bengal dolphins are at the low end of reported global concentrations.

3.3.4 South India Surface Waters

3.3.4.1 HCH in the Periyar River at Cochin, Kerala State

The Cochin estuary (also known as the Cochin backwater system) is located in Kerala state on the southwest coast of India [9.67–10.17° N; 76.17–76.50° E]. Two small rivers, the Periyar and Muvattupuzha, discharge to the lagoon system which is also influenced by tidal action through the Cochin barmouth. The Periyar has drainage area of 5,243 km^2 and unit runoff of 2.3 m giving it an annual runoff volume of roughly 12.3 km^3 (Rao, 1975), the largest from the small west coast rivers originating in the Western Ghats.

Results for one year of surface water sampling for HCH were reported by (Sujatha et al., 1993) at 6 sites extending from above the outlet of River Periyar through the lagoon system to the barmouth exit to the open sea. The year in which sampling was conducted was not stated, but was likely between 1985 and 1993. Three samples were collected during each of three *seasons*:

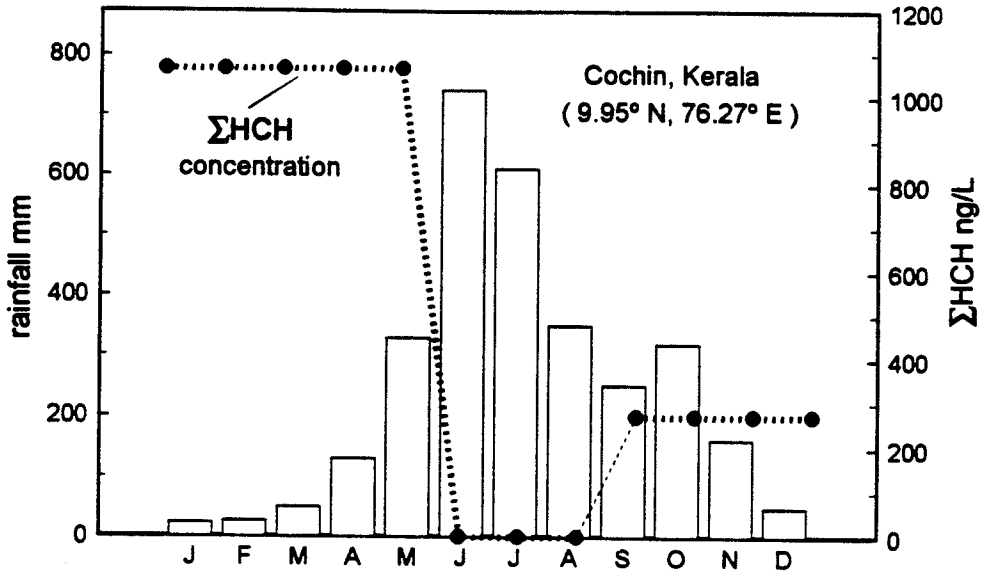


Figure 3.12 ΣHCH in Periyar River and mean monthly rainfall at Cochin, Kerala.

Table 3.19 ΣHCH in Periyar River, Cochin, Kerala.

A. Mean Cochin Rainfall

	Jan-May	Jun-Aug	Sep-Dec	Total
rainfall mm	551	1,700	773	3,025
%	18	56	26	

B. Periyar River ΣHCH concentrations ng/L

Site	Jan-May	Jun-Aug	Sep-Dec	mean [a]	tw-mean [b]	pw-mean [c]
1	1,067	[d] 1	272	447	536	265
2	1,125	1	447	524	618	320

[a] arithmetic mean, [b] time-weighted mean, [c] precipitation-weighted mean,
[d] 1 ng/L assumed for concentrations reported as not detected.

1. pre-monsoon (January – May)
2. monsoon (June – August)
3. post-monsoon (September – December)

for a total of 9 samples at each of the 6 sites.

Mean ΣHCH concentrations in the Periyar River (site 1 in Sujatha et al., 1993) for the 3 periods are shown on Figure 3.12 superimposed over the mean monthly rainfall at Cochin. Both the usual wet season (May-November) and the mean annual total rainfall of about 3.0 m differ significantly from the Vellar watershed [next section] of the east coast. The figure suggests a seasonal cycle where HCH levels are negligible during the heavy summer monsoon rains, and begin rising during the post-monsoon season toward peak concentrations that occur from January to May. It is remarkable that ΣHCH virtually disappears from June to August, but it is possible that if no new HCH is applied, concentrations become very low due to streamflow dilution as 56% of annual rainfall occurs during these months.

ΣHCH concentrations generally decline from the river outlet to the estuary barmouth, and range from non-detectable to about 500 ng/L depending on location and season. Concurrent salinity measurements were not taken; hence, it is difficult to interpret the data vis-à-vis sea water dilution effects. Broadly similar, but muted seasonal patterns are evident across lagoon sites.

Table 3.19 summarizes data at sites 1 and 2 on River Periyar, and shows attempts to find annual mean concentrations suitable for determining annual river exports. Site 1 is about 2 km upstream of the outlet and represents mainly agricultural inputs. Site 2 is at the Periyar River outlet to the lagoons, and includes industrial discharges from fertilizer, insecticide and other manufacturing facilities. The arithmetic mean is the simple average of concentrations. The time-weighted mean is the average of seasonal concentrations weighted by the length of the season. Assuming that monthly River Periyar discharges follow the monthly rainfall pattern, the precipitation-weighted mean approximates river flow-weighted mean concentration that should be best for estimating annual HCH mass exports from River Periyar. Accordingly, 3.3–3.9 t ΣHCH are discharged annually with about 0.68 t originating from the urban manufacturing area near the outlet, and 3.3 t from upstream basin area.

3.3.4.2 Vellar River Watershed, Tamil Nadu state

Vellar River is a small coastal watershed in the humid tropics of Tamil Nadu state about 200 km south of Madras. The Vellar River discharges to the sea at Porto Novo [Parangipettai] (11.48° N, 79.77° E). A collaboration of Indian and Japanese investigators intensively studied the occurrence and fate of HCH in Vellar River air, soils, sediments, surface waters and biota during the late 1980s (Ramesh et al., 1990a; Ramesh et al., 1992; Ramesh et al., 1991; Ramesh et al., 1989; Takeoka et al., 1991; Tanabe et al., 1991b). The watershed is dominated by rice paddy culture that relies on HCH as its principal pesticide. Annual HCH usage was about 42 t ai in the late 1980s. Concurrent DDT usage in the area was minor.

Collectively, the studies yield strong evidence that most HCH volatilizes readily due to heat, humidity and strong solar radiation of the humid tropics, and that airborne HCH

is transported away via prevailing atmospheric patterns. Takeoka et al. (1991) went so far as to suggest that only 0.4% of HCH applied was actually transported to the sea by stream water, and that 75% of the little HCH that reaches coastal seas likely volatilizes readily. There is little doubt that volatilization losses are great, but the degree of the claims is questionable as wet season runoff losses may well have been underestimated by indirect methods employed in lieu of continuous discharge measurements.

3.3.4.2.1 Vellar River Hydrology

Local climate is dominated by the monsoon rains which occur later in the year than in the Ganges basin. Long term monthly rainfall patterns in the Porto Novo area will lie between to the two southeast Indian coastal sites about 200 km to the north and south respectively (Figure 3.13) as suggested by the monthly rainfall observed at Porto Novo during the December 1987 – January 1989 study period (Figure 3.14). In 1988, total Porto Novo rainfall was 1.12 m, almost exactly the average of the long term mean annual total rainfalls expected in Madras and Pamban. In Vellar watershed, rice cultivation and intensive pesticide usage begin with the first rains of the wet season that may arrive as early as August as occurred in 1988.

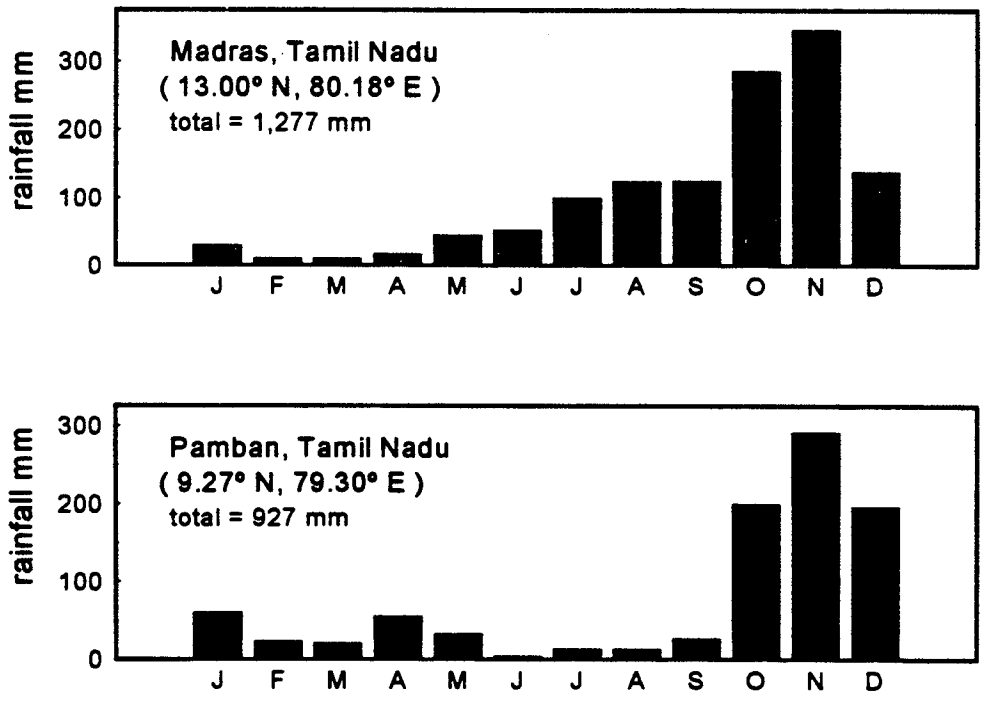


Figure 3.13 Mean monthly rainfall at Tamil Nadu coastal sites.

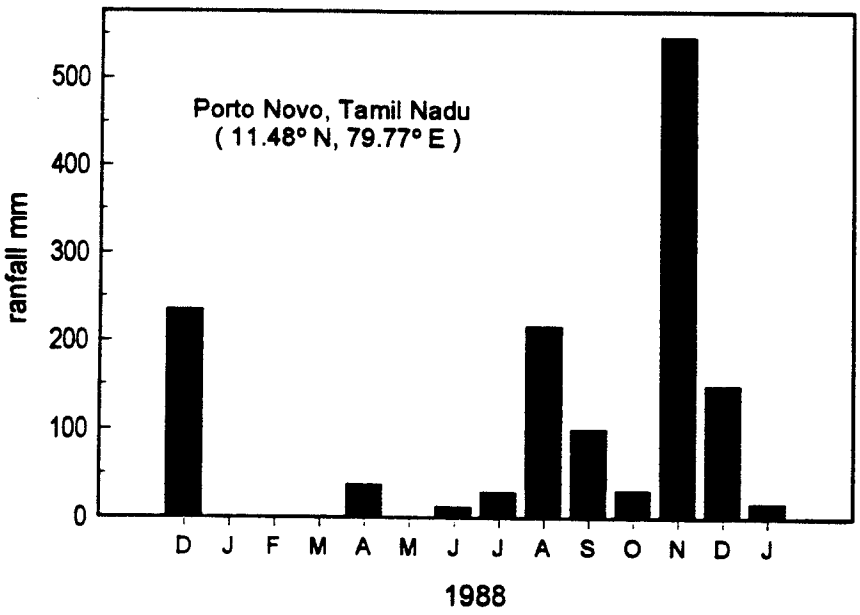


Figure 3.14 Monthly rainfall at Porto Novo, Dec 1987 – Jan 1989; after (Ramesh et al., 1989).

Vellar River watershed has an area of about 8,600 km^2 and mean annual discharge estimated roughly at 0.85 km^3 (Rao, 1975). Construction of reservoirs 20–30 years ago has significantly altered seasonal discharge patterns. Takeoka et al. (1991) observed only tidally driven estuary-sea water exchange during the dry sea season (July). Estuarine outflows are further complicated by flux from a coastal mangrove lagoon system [Pichavaram Mangrove] that also connects to the Coleroon River to the south. The mangrove receives freshwater inputs directly from irrigation channels and rice paddy drains, and salt water inputs via the Coleroon River estuary. Dry season water discharge from the Vellar River estuary to the sea for 1988 was estimated to be 1.05×10^6 m^3/day or 0.38 km^3/a. About 10% of this was estimated to originate from the mangrove channels.

3.3.4.2.2 HCH and DDT in Vellar River Water

Typical seasonal variation of ΣHCH concentrations in Vellar River waters is illustrated in Figure 3.15 for a stream site [site 5 in (Ramesh et al., 1990a)] about 4-6 km upstream from the estuary outlet. From concentrations of about 20 ng/L in the driest months (ca. June 15 – August 15), ΣHCH levels rose steadily to peaks of >1 µg/L at the culmination of the wet season in December. ΣDDT in river waters showed only random variations over the same period at the same site.

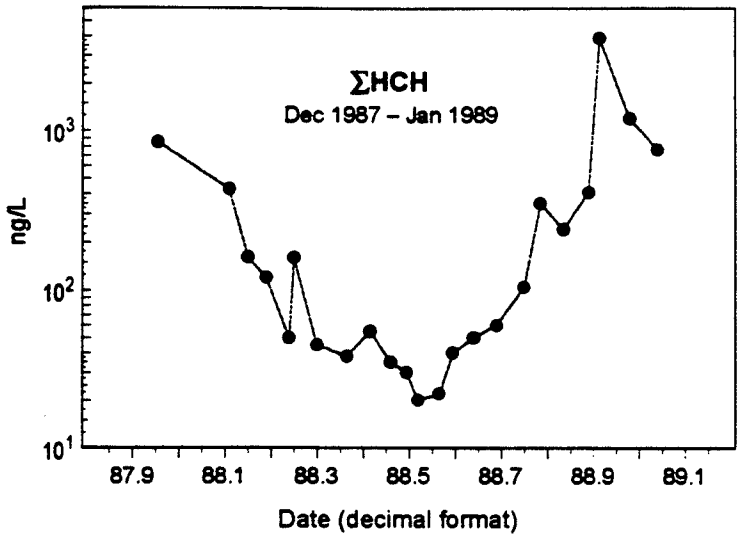


Figure 3.15 Seasonal variation of ΣHCH in Vellar River; data read from Fig. 10 Ramesh et al. (1990) for Site 5; dates are in decimal format (e.g., Jan 1 00:00 1988 ≡ 88.0; July 1 1988 ≡ 88.5).

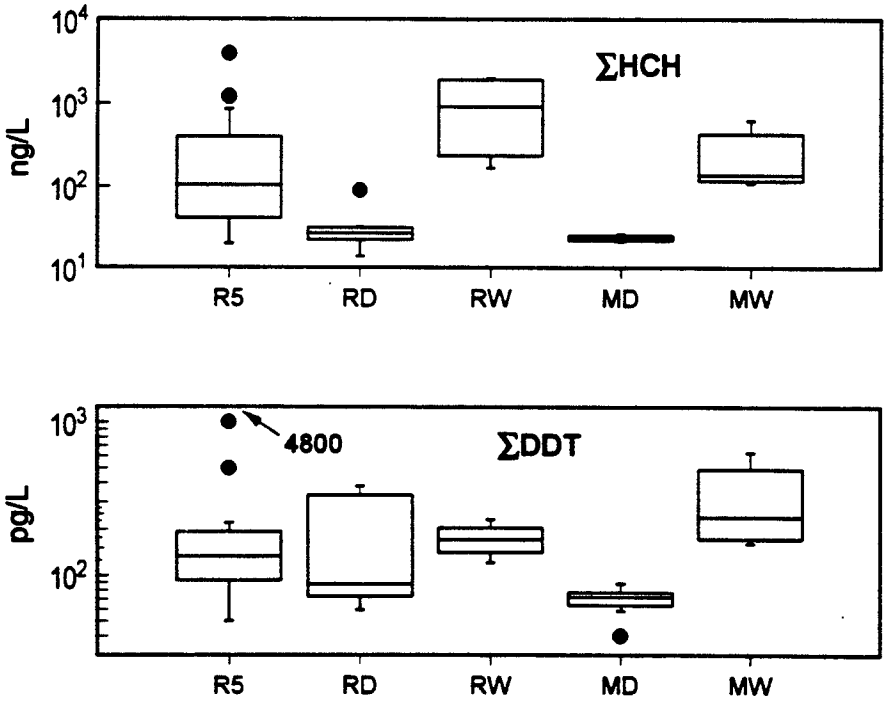


Figure 3.16 Boxplots: ΣHCH and ΣDDT in Vellar River and Pichavaram mangroves; R ≡ river; M ≡ mangrove; R5 ≡ river site 5; D ≡ dry season (July); W ≡ wet season (January); sample size ≡ RD (8); RW (6); MD (8); MW (4); data from Ramesh et al. (1990).

Figure 3.16 summarizes dry (July) and wet (January) season ΣHCH and ΣDDT concentrations at sites distributed along the lower 50 km of the Vellar River channel, and across the Pichavaram mangrove channels that extend south about 25 km between the Vellar and Coleroon River estuaries. River site 5 study period data are included for reference. For HCH, the strong seasonal pattern was evident in both river and mangrove waters; however, the wet season increase approached two orders of magnitude in river waters, but only about one order of magnitude in mangrove waters.

Although small quantities of DDT were applied to crops (Ramesh et al., 1990a), no significant seasonal variation was observed in river waters. However, a perceptible increase in ΣDDT was evident in mangrove waters, apparently due to mosquito control operations conducted in the marshes, swamps and wetlands of the coastal mangrove during the monsoon (Ramesh et al., 1991). The quantity of DDT used is not known.

3.3.4.2.3 HCH and DDT in Vellar River Soils and Sediments

Dry and wet season pesticide concentrations for Vellar watershed soils and sediments are summarized in Figure 3.17. As expected, paddy soils showed the highest ΣHCH levels which increased significantly from dry to wet season. Despite very small sample sizes, both river and mangrove sediments had perceptibly higher wet season ΣHCH concentrations. Generally, ΣDDT soil and sediment concentrations were modestly higher during the wet season due to small applications to crops and anti-mosquito spraying. Evidence was strongest in paddy soils; however, river and mangrove sample sizes are too small to demonstrate clear statistical significance.

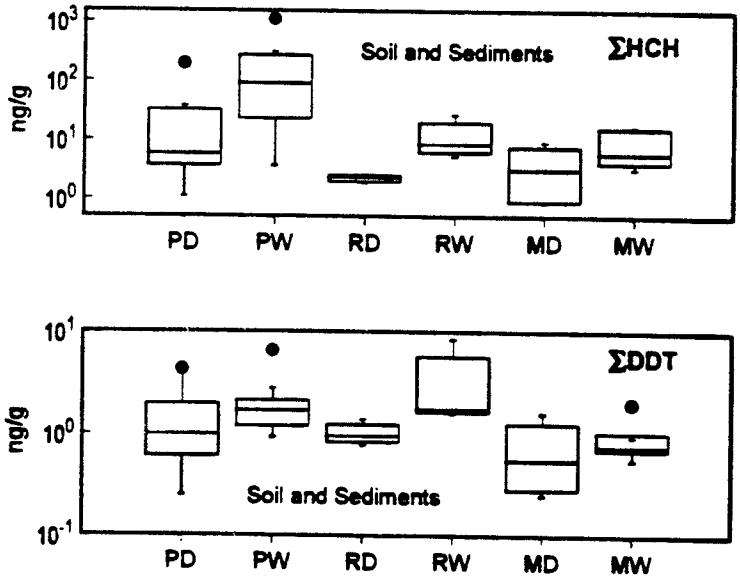


Figure 3.17 Boxplots: ΣHCH and ΣDDT in Vellar River soils and sediments; P ≡ paddy field soil; R ≡ river sediment; M ≡ mangrove sediment; D ≡ dry season (July); W ≡ wet season (January); sample sizes — PD (10); PW(9); RD, RW, MD (4); MW (6); data from Ramesh et al. (1991).

3.3.4.2.4 POPs in Vellar River Biota

A survey of Vellar watershed wildlife (Ramesh et al., 1992) further reveals the fate of POPs insecticides in tropical India. The study examined HCH, DDT, PCB and HCB residues in fishes, crabs, monitor lizards, turtles, and birds. All species were contaminated in the general order ΣHCH > ΣDDT > PCBs > HCB. In crab, turtle, monitor lizard and birds, HCH and DDT bioaccumulated mainly as β-HCH and *p,p'*-DDE respectively. Fishes exhibited less contamination. HCB levels were very low.

Top predator birds — (1) inland piscivores and scavengers, and (2) coastal piscivores — bioaccumulated appreciable levels of HCH that rival or exceed the highest levels reported elsewhere. Likewise HCH contamination of fishes is amongst the highest reported. Evidence notwithstanding, the authors mysteriously conclude that "the transport of residues from agricultural areas to the aquatic environment and their bioavailability to resident organisms is low in tropical areas like South India".

DDT and PCBs were not compared with international data; however, it is thought that the extent of contamination is relatively modest. In top predator birds, DDT concentrations were 1/3–1/2 those of HCH, while PCBs were about 20–50 fold lower. However, PCBs were found in all species tested indicating that low level contamination is occurring in rural south India.

3.3.4.2.5 HCH and DDT transport from Vellar watershed

A rice paddy field experiment (Tanabe et al., 1991b), air and water monitoring (Ramesh et al., 1990a; Ramesh et al., 1989), and a mass flow / fate modelling exercise (Takeoka et al., 1991) yield reinforcing evidence that most HCH volatilizes readily and is removed from the Vellar watershed system by atmospheric transport. The 1988 air monitoring data (Ramesh et al., 1989) showed appreciable rise in airborne HCH beginning in August well before significant increases were observed in river waters. Evidently, HCH application to paddy fields begins with the onset of the rainy season when higher summer temperatures and storage of early rains in dry reservoirs and paddy fields limit surface runoff and promote volatilization. The rice paddy experiment showed that >50% of HCH applied was lost after 10 days, and that after 40 days, HCH in air, water and soils returned to pre-application levels. Loss rates appeared to be about two fold or more higher than observed in northern Indian field studies.

In the study of river concentrations (Ramesh et al., 1990a), examination of theoretical air-water partition coefficients for HCH and DDT species vis-à-vis the measured air and water concentrations, showed that all species were present in surface waters at concentrations that exceeded theoretical equilibrium conditions. Furthermore these conditions prevailed throughout the year. Thus measured air-water distribution of HCH and DDT species appeared to favour mass transfer from water to atmosphere. The investigators suggest this occurs in Vellar watershed because both

HCH and DDT are applied directly to water surfaces in rice fields, and in anti-malarial operations, DDT is applied to irrigation channels, sewage canals, ditches and other stagnant water bodies.

Mass flux modelling gave much stronger indications that 99.4% of 42 t HCH applied in 1988 volatilized before reaching the sea (Takeoka et al., 1991), and that 75% of the 170 kg estimated to have reached the Vellar estuary evaporated in the estuary before reaching the sea. The figures are striking and taken with other evidence on global contaminant movement, imply that India is a major global emitter of certain persistent organochlorine contaminants that are prone to long range atmospheric transport.

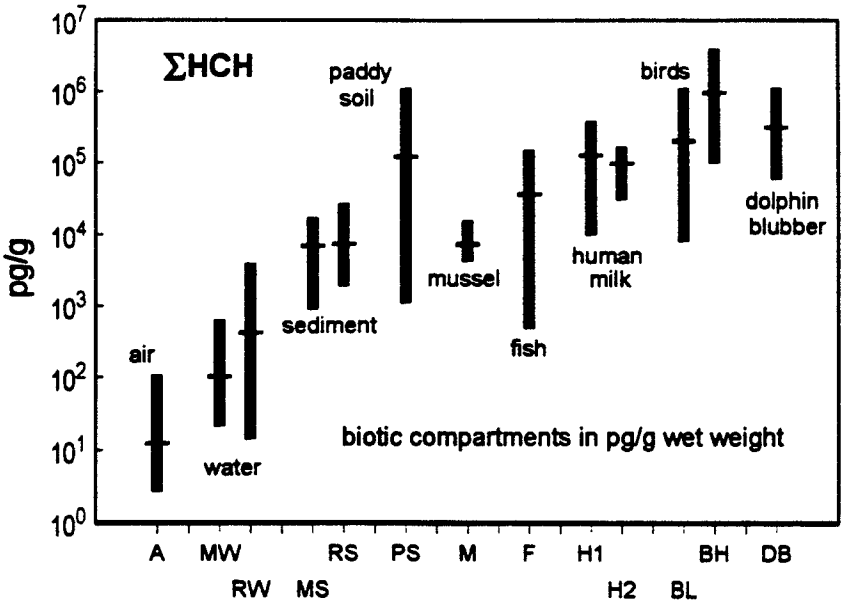
While there is little doubt that the HCH mass flows indicated by the modelling exercise are approximately correct, there is good reason to believe that (1) HCH mass flux to the sea via Vellar River flow may have been underestimated, and (2) conclusions derived from the Vellar River case study may not represent less hydrologically regulated or otherwise altered watershed regimes. As direct discharge measurements were unavailable, Vellar River water flux was inferred by methods that should yield reasonable dry season baseflows, but would be prone to potentially significant errors during the peak runoff months (October – January) when HCH concentrations in surface waters also reach their maximum. Also, the assumed HCH concentrations in the water volume reaching the sea may have been somewhat conservative during the wet season.

The Vellar River watershed is unrepresentative of natural and less regulated river systems. On the basis of wet season flow estimates derived from pre-regulation 1967 current measurements at the estuary outlet, Takeoka et al. estimate that HCH flux to the sea from Vellar watershed would likely have been much larger in the past. Thus alterations to the Vellar River seem to have been considerable. The extent that the modern Vellar River hydrologic regime represents other south Indian watersheds is unknown. The investigators conclude that *control works that temporally and spatially re-distribute water, retard natural flows, increase surface areas and raise water temperatures, should enhance volatilization losses relative to natural conditions.*

3.3.4.2.6 Vellar River Cross-compartmental Summary

Available concentration data for Vellar River environmental compartments were converted to a common pg/g basis and summarized in Figures 3.18–3.19. Because the Vellar River has the highest quality data available for India, these diagrams were used to evaluate data quality and consistency elsewhere in India.

All biotic data (mussel [M], fish [F], human [H1,H2], bird [BL,BH] and dolphin [D]) were expressed separately on wet and fat weight bases. Fish and bird data were from (Ramesh et al., 1992). Expressing concentrations on lipid weight basis reduces scatter between biotic components and generally raises biotic compartments above abiotic



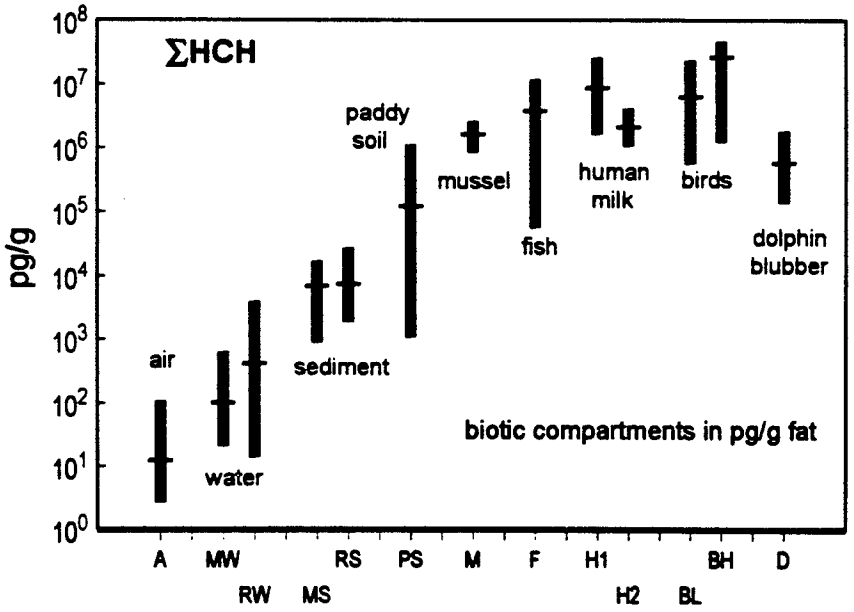
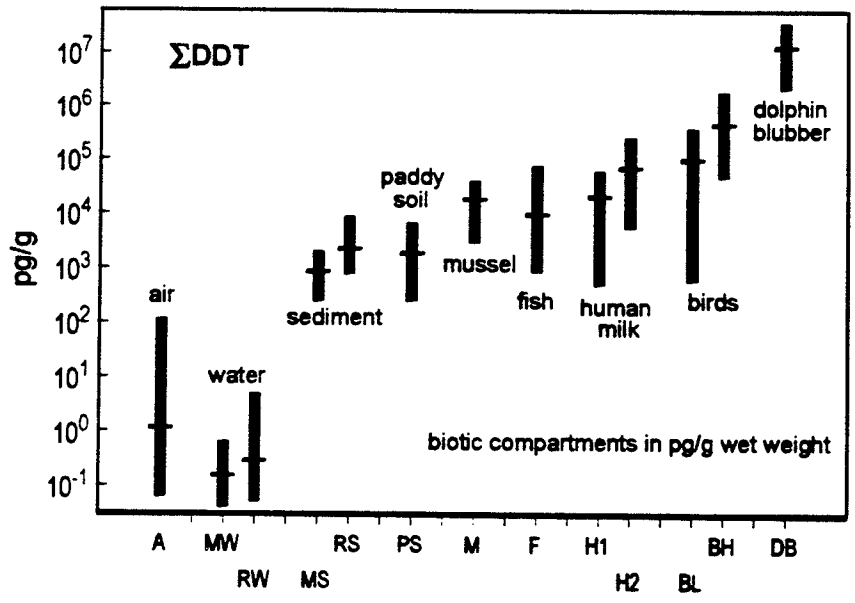


Figure 3.18 ΣHCH distribution in Vellar River environment; vertical bars show range; cross-bars show mean; A ≡ air; MW ≡ mangrove water; RW ≡ river water; MS ≡ mangrove sediment; RS ≡ river sediment; PS ≡ paddy soil; M ≡ mussel; F≡ fish; H1,H2 ≡ human milk; BL,BH ≡ birds, D ≡ dolphin.



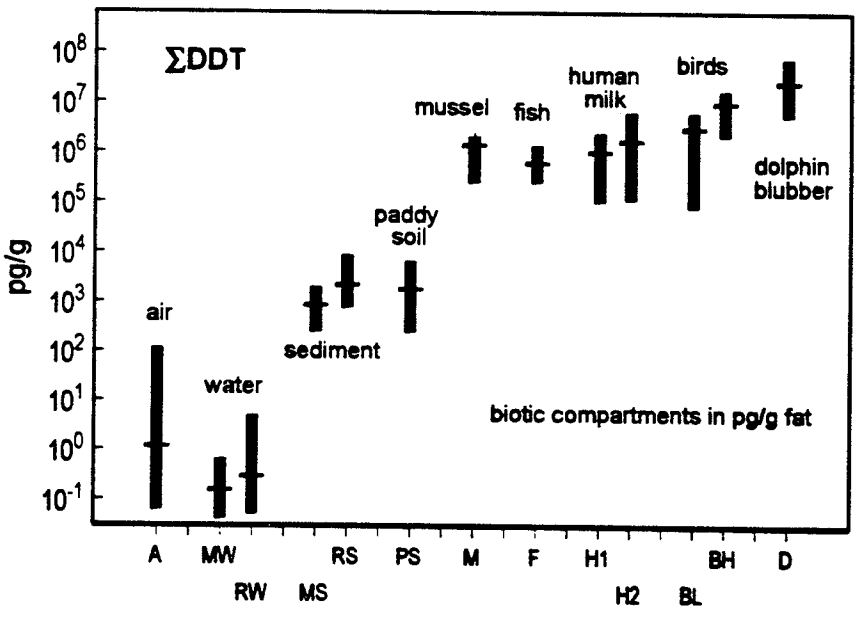


Figure 3.19 ΣDDT distribution in Vellar River environment; vertical bars show range; cross-bars show mean; A ≡ air; MW ≡ mangrove water; RW ≡ river water; MS ≡ mangrove sediment; RS ≡ river sediment; PS ≡ paddy soil; M ≡ mussel; F≡ fish; H1,H2 ≡ human milk; BL,BH ≡ birds, D ≡ dolphin.

compartments. Birds were in two classes: BH representing 6 species in the top two piscivorous / predator classes, and BL representing 9 species in the lower three non-predator classes as defined by Ramesh et al. (1992). South Indian mussel and dolphin data were taken from (Ramesh et al., 1990b; Tanabe et al., 1993). Human milk data (Tanabe et al., 1990) are for two villages in Vellar watershed area: Chidambaram [data set H1 with 11 samples] a rural "semiurban" community, and Chinnoor Parangipettai [data set H2 with 5 samples] a fishing village 30 km to the north.

3.3.5 Drinking Water Surveys

Drinking water surveys for the cities of Ahmedabad and Bhopal give information for areas of northwest and north central India that otherwise lack data. Ahmedabad is India's 6th largest city with a population exceeding 2.5 million. It is situated on the lower reaches of the Sabarmati River that drains areas of Rajasthan and Gujarat states. Bhopal is on the southern fringe of the Ganges Basin.

3.3.5.1 Ahmedabad Drinking Water

Jani et al. (1991) collected 57 samples from residential and industrial sites across the city from May to July 1987. Samples appear to have come from both the municipal system and other wells. Samples were analyzed for DDTs, HCHs, and 18 PAHs. No significant differences were found between water sources; hence, data were lumped for statistical purposes. Aldrin, dieldrin, endrin, chlordane, heptachlor, and PCBs were also analyzed, but were not found at operational detection limits that would appear to be 0.5–1.0 ng/L as judged from the reported HCH and DDT results.

ΣHCH and ΣDDT are given in Table 3.20. Concentrations are high for drinking water by most western standards. The composition profile for ΣHCH is 25% α-HCH, 13% β-HCH, and 62% γ-HCH. The high γ-HCH percentage suggests that either lindane or *fortified HCH* usage may be common in the Ahmedabad area.

The ΣDDT composition profile is dominated by 88% unmetabolized DDT [*p,p'*-DDT + *o,p'*-DDT] suggesting that newly applied DDT is entering the drinking water distribution system and wells. However, *p,p'*-DDD was not measured and could be present at significant levels.

Table 3.20 POPs insecticides (ng/L) in Ahmedabad drinking water.

Pesticide	n	Mean	median	Min.	Max.
ΣHCH	57	257	87	24	2,490
ΣDDT	57	47	24	11	314

ΣHCH = α-HCH + β-HCH + γ-HCH; ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE.

3.3.5.2 Bhopal Drinking Water

HCH and DDT data were reported by Dikshith et al. (1990) for 60 samples from wells, "hand pumps" [presumably hand-pumped tube wells], and ponds used to supply the Bhopal area. The survey dates were not given; however, sampling would have to have been conducted sometime before September 1989.

The survey results summarized in Table 3.21, show levels of ΣHCH and ΣDDT that defy credibility, and most likely involve misreported units. ΣDDT water concentrations in the ppm range are virtually impossible to achieve, particularly in potable water supplies that generally contain minimal suspended solids and organic carbon. Even in turbid ponds with high DDT inputs, ΣDDT levels exceeding 50 μg/L (ppb) would be unlikely. Technically, ΣHCH levels of 5-10 mg/L are possible, but unlikely to occur in surface or groundwaters without significant direct inputs.

There are few clues as to what the true HCH and DDT levels might be. Considering that unenclosed ponds may be subject to insecticidal spraying as potential breeding sites for malarial mosquitoes, ΣDDT and ΣHCH concentrations could well occur in the ppb (μg/L) range, particularly in turbid ponds. Such levels would be marginally higher than reported for the Jaipur Lakes. If the true units were μg/L, the ΣDDT and ΣHCH concentrations observed in the "wells" and "hand-pumps" would be disturbingly high, and most likely indicative of poor well head maintenance and protection against contaminated surface runoff entering the wells.

Table 3.21 HCH and DDT (mean and range) in Bhopal drinking water ppm (mg/L)‡.

	Wells	Hand-pumps	Ponds
ΣHCH	4.65	6.13	9.41
	1.58–7.75	4.32–8.72	6.22–15.6
ΣDDT	5.79	14.5	16.1
	3.15–9.36	10.4–22.3	3.67–34.8

‡ as reported by Dikshith et al. (1990); true units are likely ppb (μg/L) or less; see text.
ΣHCH = α-HCH + β-HCH + γ-HCH; ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.

3.3.6 North India — Indus River Drainage

In a survey of northern streams, Pathak et al. (1992) sampled the Indus River in Jammu and Kashmir, and the Beas River in Himachal Pradesh. Respective ΣHCH concentrations were 31.2 ng/L for the Indus, and 5.7 ng/L for the Beas which also had 25.3 ng/L ΣDDT. It is unclear whether DDT and metabolites were not detected in the Indus, or DDT analyses were simply not performed.

3.4 Pesticide Residues in Soil

Despite a bias towards southern India, a 1988-89 survey (Kawano et al., 1992) shows generally widespread DDT and HCH contamination of Indian soils. Cotton fields appear to have the most heavily contaminated soils, followed by tea, fruit and vegetable fields. Rice paddies show high HCH but relatively low DDT contamination.

In addition, there are numerous studies on the fate of pesticides applied to Indian soils (Kaushik, 1989; Ramesh et al., 1991; Samuel and Pillai, 1989; Samuel and Pillai, 1990; Samuel and Pillai, 1991; Singh and Agarwal, 1995; Singh et al., 1991; Singh et al., 1989; Tanabe et al., 1991b). Mainly these show that the humid, tropical conditions of India promote more rapid volatilization losses and degradation relative to temperate climates. HCH in particular exhibits extraordinarily rapid dissipation rates (Ramesh et al., 1991; Takeoka et al., 1991); such that, up to 99% of losses from south Indian rice paddies may be via atmosphere. Accordingly, relatively much less pesticide will be lost to surface runoff and transported to the sea by rivers and streams. Relatively low contamination of biotic compartments despite high pesticide usage rates lends support to the premise that volatilization is the major loss route (Kannan et al., 1994a; Pillai, 1986; Ramesh et al., 1992; Ramesh et al., 1990b; Tanabe et al., 1993).

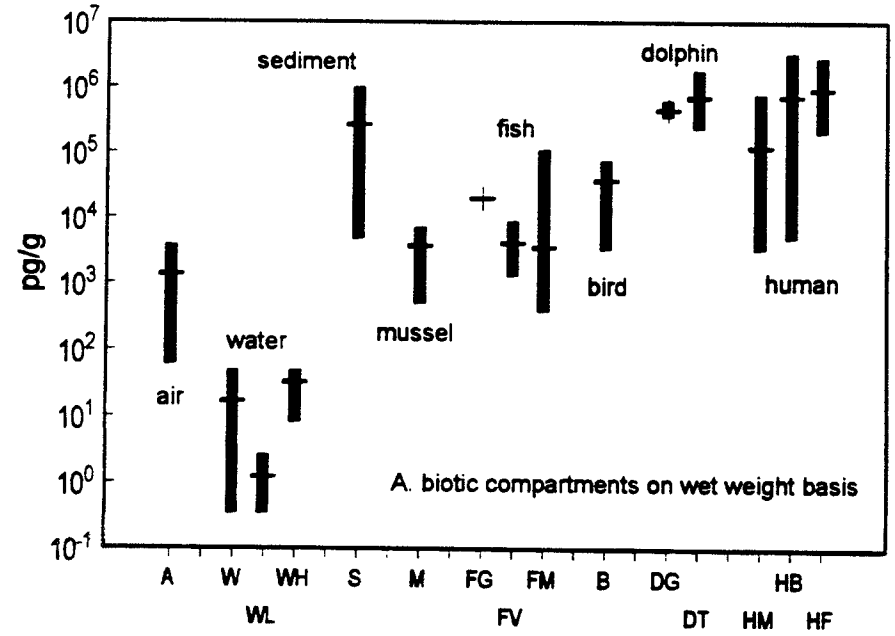
3.5 POPs Summaries

In preparation for river flux estimation, this section summarizes POPs data reviewed in Section 3.3.

3.5.1 PCBs in India

Currently, PCB concentration data for India's environmental compartments are meagre. Water concentration data are limited to the 8 samples from the 1989 synoptic survey by Iwata et al. (1994). Fortunately, additional data from biotic compartments scattered across India permits first order generalizations about the PCB presence in Indian surface waters. Available data are summarized in Table 3.22 and Figure 3.20.

Biotic data have been summarized on both wet and fat weight bases, except for the Bombay human blood and fat data for which the basis was not reported. Southern India is represented specifically by the Vellar River fish and birds, the Tamil Nadu dolphins, human milk samples, and within the aggregate market fish sample. The Ganges River is represented by the Ganges dolphins, a pooled fish sample and 2 water samples. The market fish sample represents edible portions of pooled freshwater and marine fish and prawns obtained in the markets of New Delhi, Bombay, Calcutta, Madras, Porto Novo (Parangipettai) and Chidambaram (a rural community in Vellar River watershed). See Section 3.5.6 for regional analysis of these data.



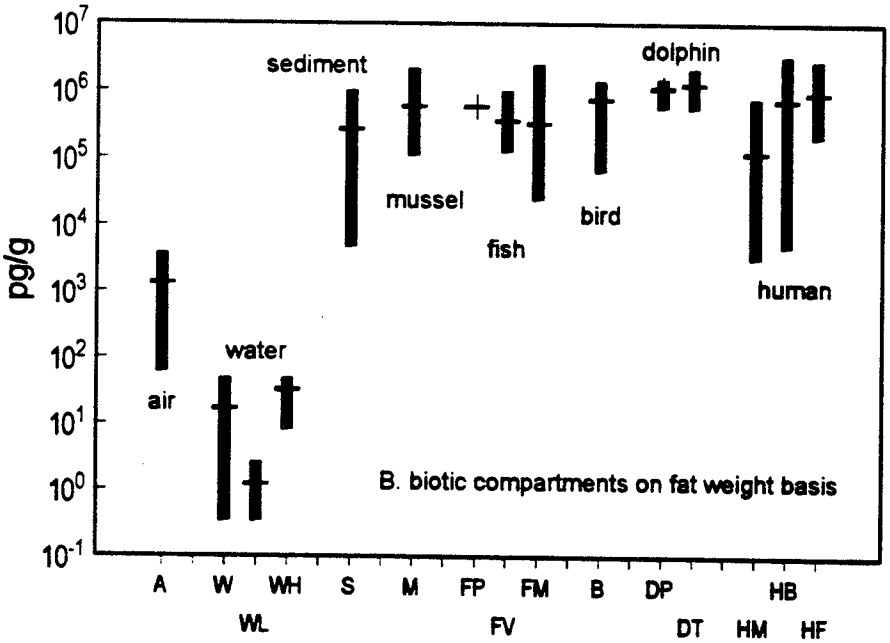


Figure 3.20 PCBs in the Indian environment; vertical bars show range; cross-bars show mean.

Table 3.22 PCBs in environmental compartments of India.

Key	Medium / unit	Date	n	Mean	Min.	Max.	Mean	Min.	Max.	Source
A	air ng/m^3	Dec 1989	7	1,641	74					1
W	surface water ng/L	Dec 1989	8	16.7	0.34	48				1
WL	surface water – low ng/L	Dec 1989	4	1.22	0.34	2.6				1
WH	surface water – high ng/L	Dec 1989	4	32.3	8.0	48				1
S	sediment ng/g dry	Dec 1989	6	272	4.8	1,000				1
	Biota			Wet weight basis			Lipid weight basis			
M	mussel ng/g	1988-89	9	3.7	0.5	7.1	588	111	2,152	2
FG	Ganges fish ng/g	1988-89	1	20			588			3
FV	Vellar R fish ng/g	1987-91	28	4.1	1.3	8.9	363	127	989	4
FM	market fish ng/g	Dec 1989	42	3.5	0.38	110	330	25	2,500	5
B	Vellar R bird ng/g	1987-91	58	37.7	3.3	76	754	65	1,407	4
DG	Ganges dolphin ng/g	1988-92	4	453	360	620	1,115	568	1,512	3
DT	Tamil Nadu dolphin ng/g	1990-91	12	699	240	1,800	1,262	558	2,136	6
HM	S. India human milk mg/L	1988	25	3.16	0.013	21.78	120	3.3	770	7
HB	Bombay human blood mg/L	198?	80	† 727	5	3,330				8
HF	Bombay human fat ng/g	198?	17	† 920	206	2,843				9

Sources: 1. Iwata et al., 1994; 2. Ramesh et al., 1990b; 3. Kannan et al., 1994a; 4. Ramesh et al., 1992; 5. Kannan et al., 1992b; 6. Tanabe et al., 1993; 7. Tanabe et al., 1990; 8. Rao and Banerji, 1989; 9. Rao and Banerji, 1988.
† weight basis not specified; wet weight assumed.

The table and graphics show that PCB levels in the Indian environment are generally low, but widely present. On fat weight basis, PCBs are fairly evenly distributed across the biotic compartments. Detailed inspection of individual data sets shows sporadically high levels that occurred mainly in samples from the major urban centres, namely Bombay, Delhi, Calcutta and Madras. Some relatively high values were observed for smaller urban and rural centres in Tamil Nadu state suggesting that old electrical equipment containing PCBs may be scattered across India.

For determining river PCB fluxes, data are limited to the 8 water samples (denoted W on Figure 3.20). These were separated into 4 high samples (denoted WH on Figure 3.20) from water courses draining major urban complexes (Bombay, Delhi, Calcutta, Madras), and 4 low samples (denoted WL on Figure 3.20) from other areas. Subset WH shows that PCB levels in Indian urban waters can rival those in developed countries. Subset WL is more appropriate for estimating river flux to the sea, but may not adequately represent PCB fluxes from major coastal cities via urban drainage and sewage canals.

More aqueous PCB data are required for India's rivers. Detection capability down to 100 pg/L or less is essential.

3.5.2 ΣHCH in Freshwater Water Systems

HCH data for freshwater systems (Table 3.23) show that HCH is widely used across India. Superficially, sites in the Ganges watershed seem to have the highest HCH concentrations. This may be the artefact of more extensive sampling in the Ganges Basin, sampling near headwater sources (Jaipur lakes), and sampling near-bank waters (Ganges at Farrukhabad and Varanasi) rather than well-mixed cross-sections. With the exception of Vellar River in Tamil Nadu, the remaining non-Ganges sites have not been well sampled over the annual hydrologic and usage cycles.

3.5.3 ΣDDT in Freshwater Water Systems

DDT data for freshwater systems (Table 3.24) vary widely. The highest levels are generally observed in the Ganges basin. From the perspective of Ganga DDT flux estimation, the sites at Farrukhabad, Varanasi, and *24 Parganas* are the most important. However, data from these sites must be used cautiously as only partial DDT species measurements are available.

3.5.4 Cyclodienes in Freshwater Water Systems

Available data for aldrin, dieldrin, heptachlor and chlordane are summarized in Table 3.25. Data sources are limited, particularly for heptachlor and chlordane.

Table 3.23 ΣHCH (ng/L) in Indian aquatic systems.

Site	Date	Mean	[a] G-mean	other	Min.	Max.	Source
India — 8 samples	Dec 1989	225	87		6	660	4
Ganges River Basin							
Hardwar, rain	1992	77	54	[b] 44	8	152	1
Delhi, rain	1980-82	5,280			80	43,000	2
Delhi, Jamuna River ★★	1988-89	1.6			<1	8	3
Delhi, Jamuna River	Dec 1989	660					4
Jaipur, lakes	1985-87	5,150	2,020		20	46,500	5,6
Farrukhabad, Ganges R.	1991-92	245			22	1,119	7,8
Farrukhabad, drainwater	1991-92	544			360	1,784	7,8
Farrukhabad, groundwater	1991-92	245			142	274	7,8
Varanasi, Ganges R.	Aug 1992	11,600	1,490	[c] 390	<40	99,800	9
Peninsular India							
Cochin, Periyar R.	198?	524		[b] 320	<1	1,125	10
Tamil Nadu, Vellar R.	1987-89	421	121		14	3,900	11
Brahmaputra / Meghna							
Sikkim, streams	1986-89	25					12
Meghalaya, streams	1986-89	22					12
Tripura, streams	1986-89	214					12
Mizoram, streams	1986-89	116					12
Indus River Basin							
Indus River	1986-89	13					12
Beas River	1986-89	6					12
Drinking Water							
Ahmedabad	1987	257		[d] 87	24	2,488	13
Bhopal ★★	198?	6,908,000			1,576,000	9,941,000	14

Sources: 1. Dua et al., 1994; 2. Agarwal et al., 1986; 3. Nair and Pillai, 1992; 4. Iwata et al., 1994; 5. Kumar et al., 1988; 6. Bakre et al., 1990a; 7. Agnihotri et al., 1994; 8. Gajbhiye et al., 1995; 9. Nayak et al., 1995; 10. Sujatha et al., 1993; 11. Ramesh et al., 1990; 12. Pathak et al., 1992; 13. Jani et al., 1991; 14. Dikshith et al., 1990.

[a] geometric mean; [b] precipitation-weighted mean; [c] mean of concentration below 1 μg/L reference line; [d] median; ★★dubious data.

Table 3.24 ΣDDT (ng/L) in Indian aquatic systems.

site	Date	Mean	[a] G-mean	other	Min.	Max.	Source
India — 8 samples	Dec 1989	18	3.6	[b] 2.3	0.9	124	4
Ganges River Basin							
Delhi, rain	1980-82	12,470	7,700		220	108,000	1
Delhi, Jamuna River	1976-78	368			40	3,420	2
Delhi, Jamuna River ✳	1988-89	6			<1	24	3
Delhi, Jamuna River	Dec 1989	124					4
Jaipur lakes	1985-87	8,210	3,460		80	47,970	5,6
Farrukhabad, Ganges R ★	1991-92	252			1	832	7,8
Farrukhabad, drain water ★	1991-92	334			313	467	7,8
Farrukhabad, groundwater ★	1991-92	352			162	441	7,8
Varanasi, Ganges R ★	Aug 1992	12,900	1,580	[c] 290	70	143,000	9
24 Parganas, Hooghly R, near banks ★	198?	1,090	402		6	2,660	12
24 Parganas, Hooghly R, mid stream ★	198?	683	190		6	1,180	12
Calcutta, Hooghly R	Dec 1989	1.5					4
Peninsular India							
Tamil Nadu, Vellar R	1987-89	0.29	0.15		0.05	4.8	10
Brahmaputra / Meghna							
Sikkim, streams	1986-89	218					11
Meghalaya, streams	1986-89	21					11
Tripura, streams	1986-89	67					11
Mizoram, streams	1986-89	13					11
Indus River Basin							
Beas River	1986-89	25					11
Drinking Water							
Ahmedabad	1987	47		[d] 24	11	315	13
Bhopal ✳	198?	12,134,000			3,153,000	10,392,000	14

Sources: 1. Agarwal et al., 1987; 2. Agarwal et al., 1986; 3. Nair and Pillai, 1992; 4. Iwata et al., 1994; 5. Kumar et al., 1988; 6. Bakre et al., 1990a; 7. Agnihotri et al., 1994; 8. Gajbhiye et al., 1995; 9. Nayak et al., 1995; 10. Ramesh et al., 1990; 11. Pathak et al., 1992. 12. Haldar, et al., 1989; 13. Jani et al., 1991; 14. Dikshith et al., 1990.

[a] geometric mean; [b] precipitation-weighted mean; [c] mean of concentration below 1 μg/L reference line; [d] median.

✳ dubious data; ★ does not include *p,p'*-DDD.

Table 3.25 Cyclodiene insecticides (ng/L) in Indian aquatic systems.

Site	Date		Mean	[a] G-mean	Min.	Max.	Source
drins							
Delhi, Jamuna River ★★	1988-89	aldrin	14,000		500	50,000	1
Delhi, Jamuna River ★★	1988-89	dieldrin	20,000		100	100,000	1
Delhi, Jamuna River ★★	1988-89	Σdrin	34,000		600	150,000	1
Jaipur lakes	1985-87	aldrin	1,700	570	10	25,000	2,3
Farrukhabad, Ganges R.	1991-92	aldrin	25		<1	99	4
Farrukhabad, Ganges R.	1991-92	dieldrin	13		<1	49	4
Farrukhabad, Ganges R.	1991-92	Σdrin	38		<1	120	4
Farrukhabad, drain water	1991-92	aldrin	164		139	203	5
Farrukhabad, groundwater	1991-92	aldrin	100		61	150	5
Mizoram, streams	1986-89	aldrin	6.5				6
heptachlor & chlordane							
Jaipur lakes	1985-87	heptachlor	542	116	<2	2,100	2,3
Farrukhabad, Ganges R.	1991-92	heptachlor	37		<1	412	4
India — 8 samples	Dec 1989	[b] ΣCHL	0.864	0.268	<0.004	2.75	7

Sources: 1. Nair et al. (1991); 2. Kumar et al. (1988); 3. Bakre et al. (1990a); 4. Agnihotri et al. (1994); 5. Gajbhiye et al. (1995); 6. Pathak et al. (1992); 7. Iwata et al. (1994).
★★ dubious data; [a] geometric mean; [b] ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

3.5.5 HCB in India

HCB data are insufficient to attempt river flux estimation, but do suggest that HCB is likely widespread in the Indian environment at very low levels. Where HCB has been measured — in Vellar River fish and birds (Ramesh et al., 1992); Tamil Nadu dolphins (Tanabe et al., 1993); Ganges River fish and dolphins (Kannan et al., 1994a); and market fish from Delhi, Bombay, Calcutta, Madras and Tamil Nadu (Kannan et al., 1992b) — the concentrations were generally the lowest amongst the contaminants examined. Some limited 1987 data for Delhi area environmental compartments (Nair and Pillai, 1989) show water concentrations with a mean of 1-2 μg/L on 6 samples, and perceptible levels in fish and humans. Delhi HCB likely originates from industrial processes. Other industrial areas in India may present similar localized emission sources. HCB will volatilize more rapidly in hot, humid conditions than the other POPs considered here.

3.5.6 POPs in Indian Fish

POPs levels in fish from various localities in India were included by (Kannan et al., 1995) in a broad regional comparison. In addition to data given earlier for Patna (Kannan et al., 1993a) and Vellar River (Ramesh et al., 1992), wet weight data were presented for Delhi, Calcutta, and Bombay. Summary statistics (Kannan et al., 1995) suggest that the national data are identical to the fish contaminant summary data included with an earlier Indian foodstuff survey (Kannan et al., 1992b). Wet weight concentration data for the 5 localities are summarized in Figure 3.21.

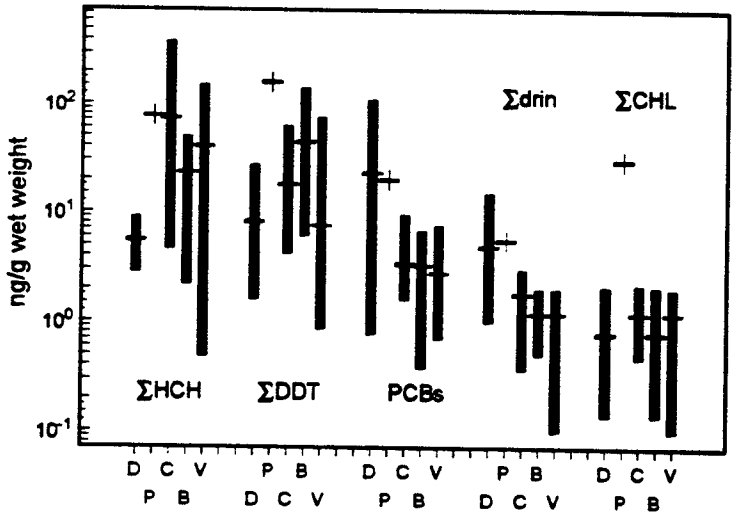


Figure 3.21 POPs in fish from across India; vertical bars show range; cross-bars show mean; D ≡ Delhi, P ≡ Patna, C ≡ Calcutta, B ≡ Bombay, V ≡ Vellar River; ΣHCH = α-HCH + β-HCH + γ-HCH + δ-HCH; ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD; ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor + oxychlordane; Σdrin = aldrin + dieldrin.

Apparent regional differences may be the artefacts of small fish sample sizes, not normalizing measurements to lipid weight basis, and other factors. The general order of contamination is HCH > DDT > PCBs > drins > chlordane. HCH and DDT are the most prominent contaminants. It is notable that PCBs are higher than drins and chlordane given that aldrin and heptachlor were being heavily used at the time. The Delhi fish seem to have remarkably high PCBs for India; however, the sample of 7 fish is dominated by one fish with 110 ng/g that raises the mean to 23 ng/g. The mean PCB content of the remaining 6 fish is only 8.5 ng/g. It is surprising that fish from Bombay and Calcutta waters have mean PCB contents similar to Vellar River fish given that water samples from the surface drainage systems of Bombay, Calcutta, Delhi and Madras had PCB levels rivaling the Yodo River in Japan. The Delhi sample is also suspiciously low in DDT, HCH and drins relative to a larger sample of 19-20 fish obtained over 1988-89. In Section 3.3.2.1.1.2, potential confounding factors were suggested that might explain the differences.

3.6 POPs Flux to the Sea from India's Rivers

Flux estimates for DDT, HCH, drins, heptachlor, chlordane and PCBs are given at the end of this section, in Table 3.34 of Sub-section 3.6.10. The rationale behind these estimates is developed in Sub-sections 3.6.1–3.6.9.

3.6.1 Hydrologic Considerations for Mass-Flux Estimation

Because Bangladesh has long banned the major organochlorine insecticides (DDT, HCH), it is simplest to calculate contaminant fluxes carried by the Ganga, Brahmaputra and Meghna rivers as they exit India and assume that contaminants are transported through Bangladesh to the sea without losses. River discharges entering Bangladesh from India are shown in Table 3.26. As the River Meghna is confined to India and Bangladesh, the flux from India is easily obtained.

Table 3.26 Annual river discharge (km^3) entering Bangladesh from India ‡.

River	Discharge
Ganges (including Hooghly River discharge)	525.0
Brahmaputra	537.4
Meghna	59.8
total	1,122.2

‡ April 1988 revised estimates of discharges from India by Indian Central Water Commission as reported in (Verghese, 1990).

3.6.1.1 Ganges River

Except for DDT which has some data for the lower Hooghly River, estimating Ganga River POPs fluxes from India requires combining chemical concentration data from the mid reaches of the Ganges main branch with discharges at the Bangladesh frontier. About 15% of the Ganges drainage area above Bangladesh rises in Himalayan headwater tributaries of Nepal, China and India whose waters likely carry negligible POPs loads. Between Allahabad (after confluence with the Yamuna) and Farakka at the Bangladesh border, the northern tributaries contribute 85% of inflows to the Ganges. Thus, high POPs insecticide loads from the upper-mid Gangetic plain may be diluted as the Ganges progresses downstream through the mid-lower reaches, and using high concentration data from the upper–mid reaches may significantly overestimate delivery to the Bay of Bengal.

Presently, there is no choice but to estimate Ganga exports of industrial organic chemicals such as PCBs in the same manner as insecticide fluxes. If the Calcutta urban industrial complex and the coal-steel belt in the Damodar River basin to the northwest of Calcutta were generating significant industrial POPs contamination, specific surveys would have to be conducted in the lower Hooghly to characterize this aspect of the river's chemical loads.

3.6.1.2 Brahmaputra and Meghna Rivers

POPs insecticide exports in Meghna waters originating in India were obtained by applying water concentration data obtained by Pathak et al. (1992) to the Meghna discharge volume given in Table 3.26.

The Brahmaputra presents more complex circumstances as only about 187,110 km^2 (32%) [Rao, 1975] of the basin lies in India, and little direct information is available to guide flux estimation. As explained below, POPs flux from the Brahmaputra was ultimately estimated by applying water concentration data for Brahmaputra tributaries in the Indian sub-basin to a crude estimate of the water volume discharged by these tributaries, and assuming other inputs to be zero.

As the influent mean annual Brahmaputra discharge entering India was unavailable, the annual water volume generated within Indian territory was crudely estimated by areal prorating to be almost 200 km^3. POPs load in Brahmaputra inflows from China is likely nil. The Indian sub-basin comprises most of Arunachal Pradesh, Assam and Sikkim states, 2/3 of Nagaland, and 1/2 of Meghalaya. Limited data in Rao (1975) suggested that most arable land in Assam and Meghalaya appeared to be under cultivation. For Assam this represented 27% of the state. The cultivated fraction of watershed lands in Meghalaya and Nagaland is likely higher. Relatively large Arunachal Pradesh state had little agriculture or other development. From these considerations, a very rough working estimate is that 25% of the Indian sub-basin of

the Brahmaputra contributes POPs insecticides, or equivalently, about 49 km^3 of annual water discharge carried pesticides at the concentration levels observed in the small tributary streams by Pathak et al. (1992) in a late 1980s stream survey.

3.6.2 Assigning Chemical Concentrations

It is assumed that Bangladesh and upstream nations on the Ganges-Brahmaputra (Nepal, China, Bhutan) generate no chemical inputs, and that contaminant flux from India is transported through Bangladesh without losses. Then, calculating contaminant exports by India's rivers reduces to separate estimation of:

1. Ganges River discharges to the Bangladesh frontier and down the Hooghly River,
2. discharges originating within the Indian sub-basin of the Brahmaputra River,
3. headwater discharges of the Meghna River at Bangladesh frontier,
4. discharges from peninsular India.

This four component scheme acknowledges fundamental geographic differences in POPs occurrence that emerged from the available environmental data and other information reviewed in this Chapter. Given the highly variable POPs data assembled in Tables 3.22-3.25, it is prudent to further consider for each component a range of river flux scenarios defined by an array of plausible water concentrations. Depending on data availability, from one to four concentration scenarios were specified. The particular choices and the underlying rationale for each contaminant are presented in the following sections. The river flux estimates are given last in subsection 3.6.10.

3.6.3 Ganga River Flux: Alternative Estimates

Alternative insecticide flux estimates can be developed for the Ganges River. Two options are given below that provide a partially independent basis for evaluating the direct estimates based on concentration and discharge data given in subsection 3.6.5.

3.6.3.1 Alternative A: Ganges Flux from Farrukhabad Unit Area Loads

Gajbhiye et al. (1995) used river concentration data [Table 3.8] to estimate insecticide mass flux in the Ganges River past Farrukhabad, and by assuming that the insecticide flux derived mainly from the 40,000 km^2 influent drainage area between Haridwar and Farrukhabad, estimated unit area loads (Table 3.27) for that intervening drainage area. ΣDDT river load estimates are 70% higher than ΣHCH loads despite similar mean concentrations because the maximum ΣDDT concentrations coincide with maximum river discharges while the maximum ΣHCH levels occur at lower discharges.

Table 3.27 includes estimates of the river insecticide flux that would be generated by about 600,000 km^2 of Ganges basin land under agriculture [6] if that land were similar to the 40,000 km^2 between Haridwar and Farrukhabad. On this basis, the annual total Ganges River fluxes to the Bay of Bengal would be 104 t ΣDDT, 64 t ΣHCH, 29 t Σdrin, 17 t endosulfan, plus 17-29 t heptachlor that was estimated to be in the same range as Σdrin and endosulfan estimates. These are very rough estimates. This area of the upper Gangetic plain containing the Farrukhabad district is the most intensively irrigated in India and may not represent the cropping and insecticide usage patterns of the equally large mid and lower plains of the Ganges valley.

3.6.3.2 Alternative B: Ganges DDT Flux from Suspended Sediment Data

Simple flux estimates can be derived for the Ganges by applying insecticide concentrations in sediment to estimated annual sediment flux by the Ganges. The mean concentration of sediment in Ganges water is about 1 g/L based on data reported by Milliman et al. (1995). If ϕ is the mean concentration in ng/g of contaminant in the sediment, the annual contaminant flux by the Ganges from India is 0.525ϕ tonne after conversions are taken into account. Because this assumes all contaminant is attached to sediment, the method works best for DDT which has the highest sorptive tendency, and would be least effective for the much more water soluble HCH isomers.

ΣDDT concentrations for sediment of the Ganges' Hooghly-Bhagirathi distributary were reported as 17–89 ng/g [David et al. (1993) attributed to Joshi (1985) which did not have a bibliographic entry]. Using these concentrations with the previous formula gives total annual ΣDDT flux by the Ganges from India as 8.9–46.7 t. Considered with the previous estimate from Farrukhabad unit area loads, the annual Ganges ΣDDT flux would appear to be in the 10–110 t range.

Table 3.27 POPs insecticide flux from the Ganges River estimated from Farrukhabad data.

Insecticide	Annual flux at Farrukhabad kg	† Unit area load g/ha	Annual flux from 600,000 km^2 tonne
ΣHCH	4,229	1.070	64.2
ΣDDT	6,940	1.731	103.9
aldrin	1,262	0.316	19.0
Σdrin	1,942	0.486	29.2
Σendo	776	0.286	17.2
heptachlor	—	—	17–29

†for 40,000 km^2 catchment area between Haridwar and Farrukhabad.

[6] The 600,000 km^2 figure for land in agriculture is an old estimate from 1975 or earlier.

3.6.4 ∑HCH Flux Estimates

The concentration scenarios are listed in Table 3.28. After discounting near-bank contaminant plumes in the Ganges at Varanasi, suspiciously low data for the River Jamuna at Delhi, and small headwater systems like the Jaipur lakes, viable mean concentrations for the Ganges would seem to fall into the 100-660 ng/L range. The peninsular river scenarios were determined independently from Vellar and Periyar River data, but the results were much the same as for the Ganges. Brahmaputra and Meghna concentrations were based on the data of Pathak et al. (1992) in Table 3.2. The *medium-low* and *medium-high* estimates are an attempt to bracket the most plausible mid range estimates. For these two scenarios, the Ganges delivers about 55%, the peninsular rivers about 40%, the Meghna about 4% and the Brahmaputra sub-basin about 0.4% of total riverine ∑HCH flux.

Table 3.28 ∑HCH concentration (ng/L) scenarios for river flux estimation.

	Low	Medium-low	Medium-high	High
Ganges	100	250	390	1,494
Brahmaputra	22	23.5	23.5	23.5
Meghna	100	165	214	214
Peninsula	100	225	421	421

3.6.5 ∑DDT Flux Estimates

DDT flux is more difficult to estimate than HCH flux. Because DDT data are not widely available for peninsular India, the best choice of a peninsular mean DDT water concentration is not obvious. The very low concentrations seen in the small Vellar River watershed (mean 0.3 ng/L) may be atypical of the peninsula at large. Information suggests that malaria incidence is lower in peninsular India where malaria appears to be concentrated in urban centres (Sharma, 1991). Now that DDT is banned for agriculture, usage is expected to be lower in peninsular India than on the Gangetic plains. However, sources indicate that DDT was used in significant quantities (Jani et al., 1988; Kawano et al., 1992) on the peninsula, and has seen recent use in some areas (Bhatnagar et al., 1992; Jani et al., 1991); so that, a higher mean peninsular DDT concentration than the 0.3 ng/L seen in the Vellar River watershed is expected.

A further problem the Ganges is that *p,p'*-DDD was not measured at any of the most important sites (Farrukhabad, Varanasi, and *24 Parganas*). In appropriate conditions, the conversion of DDT to DDD is rapid, and *p,p'*-DDD concentrations can dominate ∑DDT, and conditions favouring DDD formation are difficult to predict. The synoptic

survey data of Iwata et al. (1994) have a mean *p,p'*-DDD composition of 41% and range 0.4–88%. On average, if only DDT and DDE isomers are measured, ΣDDT ≈ 1.7 x (DDT+DDE), and in the worst case, say 90% *p,p'*-DDD, ΣDDT = 10 x (DDT+DDE). The only independent measurement in the Ganges with complete DDT species is the single sample obtained by Iwata et al. (1994) from the Hooghly for which *p,p'*-DDD contributed 31% of ΣDDT = 1.5 ng/L. Consequently, for Ganges flux calculations, *p,p'*-DDD was assumed to constitute 1/3 ΣDDT; however, one abnormally low sample taken at the onset of low discharge season is inadequate to generalize the average *p,p'*-DDD concentrations expected in the lower Ganges with great confidence.

The net result of these considerations is that the plausible range of mean Ganges River ΣDDT concentration in the lower reaches extends from 1.5 ng/L up to 1.5 µg/L, and accordingly, the estimated annual ΣDDT flux from the Ganges ranges from 800 kg to 800 t. Thanks to alternative Ganges DDT flux estimates developed in section 3.6.3, the plausible range was narrowed to the concentrations given in Table 3.29. The *medium-low* and *medium-high* scenarios bracket the most plausible range giving total India ΣDDT flux as 66–180 t of which the Ganges generates 77–87%. Although the results are termed *plausible*, there is enough uncertainty in available data that, if unfavourable circumstances prevail, the true DDT flux could be as high as 500–1,000 t/a. If a single value had to be selected, a total flux near 100 t/a or 1% of annual DDT applied seems reasonable, but the choice is somewhat subjective.

Table 3.29 ΣDDT concentration (ng/L) scenarios for river flux estimation.

	Low	Medium–low	Medium-high	High
Ganges	10	100	300	1,000
Brahmaputra	21	119	218	218
Meghna	13	40	67	67
Peninsula	2.3	13	21	21

3.6.6 Σdrin Flux Estimates

As reliable aldrin and dieldrin data are limited, the flux scenarios (Table 3.30) are based on data for the Ganga River at Farrukhabad, and data from Mizoram state. The concentrations applied to the peninsular rivers rely on the assumption that aldrin usage in peninsular India is greater than in the northeastern hill states, and therefore, Σdrin should be higher in peninsular waters than in hill state waters, but lower than in the Ganges. The scenarios have 65–80% of drins emanating from the Ganges, 21–38% from the peninsula, and <2% from the Brahmaputra and the Meghna.

Table 3.30 ∑drin concentration (ng/L) scenarios for river flux estimation.

	Low	Medium–low	Medium-high	High
Ganges	28	38	56	—
Brahmaputra	1	1	6	—
Meghna	1	6	10	—
Peninsula	10	28	38	—

3.6.7 Heptachlor Flux Estimates

Heptachlor flux scenarios (Table 3.31) are based on mainly on data for the Ganga River at Farrukhabad. Peninsular rivers were assumed to have 1/2 the concentration assigned to the Ganga, while the Brahmaputra and Meghna were assigned the same concentrations as for the drin scenarios. Like the drin scenarios, the results show 66–72% of heptachlor emanating from the Ganges, 21–33% from the peninsula, and <2% from the Brahmaputra and the Meghna.

Table 3.31 Heptachlor concentration (ng/L) scenarios for river flux estimation.

	Low	Medium–low	Medium-high	High
Ganges	16	37	78	—
Brahmaputra	1	1	6	—
Meghna	1	6	10	—
Peninsula	8	19	39	—

3.6.8 Chlordane Flux Estimates

Chlordane flux scenarios (Table 3.32) are based on the 8 sample, India wide synoptic data of Iwata et al. (1994). Concentrations ranged from nondetectable (<8 pg/L) to 2.75 ng/L, with higher levels in urban areas and no obvious distinction between the Ganges and peninsular river sites. The Brahmaputra and Meghna waters were simply assigned 1/2 the 8 pg/L detection limit. The resulting flux estimates are very low. The *medium* estimates (175–490 kg) bracket the most plausible range of chlordane fluxes.

Table 3.32 ∑chlordane concentration (pg/L) scenarios for river flux estimation.

	Low	Medium–low	Medium-high	High
Ganges	35	190	540	840
Brahmaputra	4	4	4	4
Meghna	4	4	4	4
Peninsula	35	190	540	840

3.6.9 PCB Flux Estimates

PCB flux scenarios (Table 3.33) are based on the 4 lowest samples of Iwata et al.'s (1994) India wide synoptic survey that ranged from 340 pg/L to 2.6 ng/L. As discussed previously, 4 samples from large urban centres (Delhi, Madras, Bombay, Calcutta) were too high to apply generally to Indian river waters. The minimum concentration of 340 pg/L was applied to the Brahmaputra and Meghna as a general background level. The *medium-low* scenario is most plausible given the minimal data available. The relatively low PCB flux may be deceptive as the major cities may contribute significant local PCB discharges that are diluted in large rivers like the Ganges during wet monsoon flows, or escape directly to sea via the urban drainage canals of coastal cities. Further PCB data are required for Indian surface waters. Analytical detection capability down to 100 pg/L or less is essential.

Table 3.33 PCBs concentration (ng/L) scenarios for river flux estimation.

	Low	Medium–low	Medium-high	High
Ganges	0.88	1.22	2.60	3.57
Brahmaputra	0.34	0.34	0.34	0.34
Meghna	0.34	0.34	0.34	0.34
Peninsula	0.88	1.22	2.60	3.57

3.6.10 Riverine Flux Summary

Given highly variable POPs data from sites far upstream of the main river outlets, it is prudent to consider a range of scenarios. The resulting total India [7] riverine contaminant fluxes are in Table 3.34. As expected, HCH and DDT are the leading POPs; however, the annual aldrin+dieldrin and heptachlor fluxes determined from data spanning 1986-1992 were surprisingly high. Exports should have declined significantly since the recent bans; however, perceptible residues of heptachlor epoxide and dieldrin may persist for some time yet. Chlordane discharges are minimal.

The relatively low PCB flux may be deceptive. The low to high scenarios derive from four samples with concentrations in the 0.34–2.6 ng/L range. Concentrations in four water samples from drainage canals of major centres (Bombay, Calcutta, Delhi, Madras) ranged from 8-48 ng/L. These data are too high to apply to Indian river waters generally, but do suggest that major cities may contribute significant local PCB discharges that are diluted in large rivers like the Ganges, or escape directly to sea via the urban drainage canals of coastal cities. Further PCB data are required for Indian surface waters. Analytical detection capability down to 100 pg/L or less is essential.

Table 3.34 Estimated annual POPs exports (t) to the sea from India's rivers.

	Low	Medium–low	Medium-high	High
ΣHCH	98	229	381	960
ΣDDT	8	66	180	548
Σdrin	19	31	45	—
heptachlor	13	27	62	—
ΣCHL	<0.1	0.2	0.5	0.8
PCBs	0.8	1.2	2.4	3.3

ΣHCH = α-HCH + β-HCH + γ-HCH.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD; concentration data used did not include *p,p'*-DDD which was assumed to be about 1/3 ΣDDT as defined here.
Σdrin = aldrin + dieldrin.
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

[7] Total excludes India's POPs discharges from the upper reaches of the Indus River in Jammu and Kashmir, and Punjab states.

Chapter 4 Japan

4.1 Hydrology

Hydrologic data for 11 Japanese rivers are listed in Table 4.1. These represent almost 25% of the 360,300 km^2 area of the four main islands of the Japanese archipelago. Extrapolating runoff and contaminant discharge from the GLORI river areas to the entire area of the four islands is simple and unambiguous because Japan is an island state of fairly uniform unit runoff.

The GLORI rivers are presented in two groups that roughly correspond to relatively undeveloped (Northern) and developed (Southern) areas. Northern Japan corresponds to the main island of Honshu above 36° N and the island of Hokkaido, while Southern Japan comprises Honshu Island south of 36° N, plus the islands of Kyushu and Shikoku. The south comprises the great industrial cities of Japan, and the more intensely agricultural areas to the southwest of the archipelago.

Table 4.1 Rivers of Japan that discharge to the sea; after (Milliman et al., 1995).

	Area 10^3 km^2	%	Annual discharge km^3	%	Runoff
A. Northern Rivers					
Shinano	12.0	15.1	16.0	18.1	1,333
Ishikari	14.0	17.6	15.0	16.9	1,071
Tone	17.0	21.3	15.0	16.9	882
Kitakami	10.0	12.5	9.4	10.6	940
Northern subtotal	53.0	66.5	55.4	62.5	1,045
B. Southern Rivers					
Kiso	9.1	11.4	10.0	11.3	1,099
Yodo	8.2	10.3	9.7	10.9	1,183
Yoshino	3.6	4.5	5.3	6.0	1,472
Chikugo	2.9	3.6	3.7	4.2	1,276
Nagara	2.0	2.5	3.6	4.1	1,800
Hii	0.9	1.2	0.9	1.0	978
Southern subtotal	26.7	33.5	33.2	37.5	1,243
Total Rivers	79.7	100.0	88.6	100.0	1,111
Japan total	360.3		400.3		

Greater riverine pollutant delivery to the seas is anticipated in the South, and should be factored into schemes for extrapolating Japan totals. The notion is supported by PCB, DDT, HCH and chlordane burdens observed in fish tissue samples collected from Japanese coastal waters (Figure 5 of Tanabe et al., 1989). With few exceptions, fish from southern coastal waters are distinctly more contaminated. HCH contamination is evident at a site off northeast Honshu, while a site at the eastern tip of Hokkaido showed very high DDT. Nonetheless, for general purposes, the northern and southern river groups can be assigned concentrations for mass export estimation that reflect the differing degrees of contamination seen in the fish tissue data. A south-north contamination ratio of 2–4 seems appropriate as a first approximation.

The Northern Japan drainage area constitutes about 2/3 of the area of the four main islands, and the Southern Japan area about 1/3. The watershed areas of available rivers are split approximately in the same proportions between the Northern and Southern. Data of Table 4.1 suggest that Southern rivers deliver about 200 mm more unit runoff than Northern rivers. Taking this difference into account, the estimated total annual surface water discharge from Japan is estimated as 400 km^3.

The mean annual surface runoff from Japan may be as high as 500 km^3 or more. Long term hydrologic records (20-45 years each) from WMO archives for 5 rivers in Table 4.1 (Chikugo, Ishikari, Shinano, Tone and Yodo) have mean annual runoff of 1293 mm, in contrast to mean runoff of 1098 mm derived from data reported by Milliman et al. (1995). When WMO data are used for these 5 rivers, the estimated mean annual surface water discharge for Japan rises to 443 km^3, an increase of 10% over the 400 km^3 estimate based on GLORI data. If GLORI data for the other 5 rivers in Table 4.1 are similarly under reported, the annual total Japan estimate rises to about 486 km^3, a difference that can significantly affect contaminant flux estimates. The potential uncertainties could be readily rectified by assembling more extensive and up to date hydrologic records for Japan.

Additional hydrologic data are given by (Sugiura et al., 1986) for river inputs to Tokyo Bay. They report the combined mean annual discharge of the Edo, Arakawa and Tama tributaries as 7.4 km^3 and with a combined mean annual sediment delivery of 1.2 million t. Together, the Tokyo Bay, Nagara and Yodo watersheds, and adjacent small basins on the Osaka–Tokyo axis drain the most heavily urban and industrial lands of Japan that, a priori, are expected to deliver the greatest contaminant loads to the surrounding seas.

4.2 PCB Production and Usage in Japan

Japan was a significant PCB producer generating about 57,330 t between 1954 and 1971, and importing another 590 t (Sugiura et al., 1986). Peak production of 11,110 t occurred in 1970 (Figure 4.1). Domestic production ceased in 1971. By application, Japanese PCBs were distributed as 66% electrical equipment, 17% heat transfer fluids,

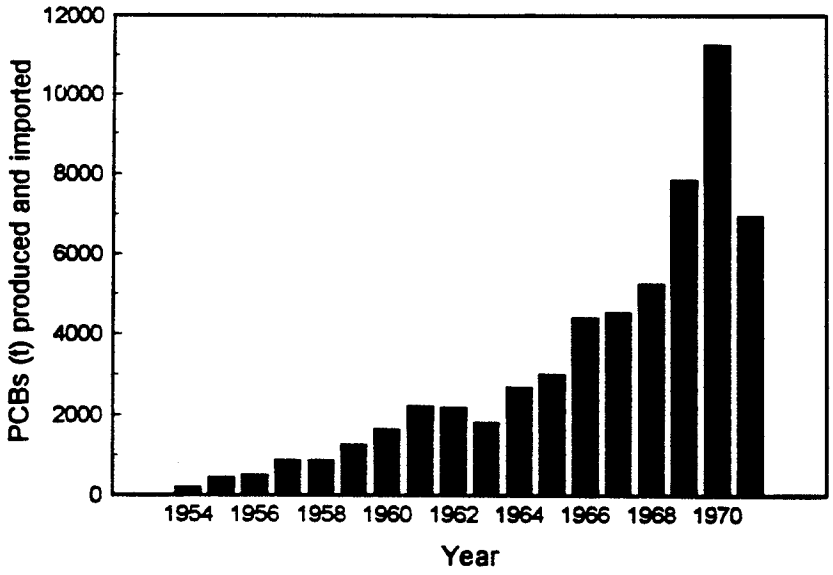


Figure 4.1 Annual production and importation of PCBs to Japan.

12% carbonless paper and 5% in other open systems (Sugiura et al., 1986). PCBs were produced under the trade name Kanechlor in 4 commercial grades known domestically as KC-300 (46%), KC-400 (18%), KC-500 (31%) and KC-600 (5%) where the percentages of product shipped are given in parentheses. The KC series is roughly comparable in composition to the German Clophen series (Kannan et al., 1993b) where increasing numbers indicate increasing chlorine content.

Sugiura et al. (1986) estimated annual release rates to the Japanese environment of certain PCB congeners between 1955 and 1980. Their estimates suggest PCB releases peaked from 1966-74. Assuming that most heat transfer and electrical fluids were recovered and contained, they also estimated that the cumulative environmental release of total PCBs to 1980 was about 7.4 Kt. Tanabe et al. (1989) suggested that, in the late 1980s, 50% or more of total PCB production (ca. 29 Kt) remained in active usage in older electrical equipment.

4.3 POPs Insecticide Usage in Japan

Production and usage of DDT and crude HCH began after WW II and accelerated steadily until 1971-72 when bans were implemented on these compounds. Cumulative production up to the bans has been estimated (Loganathan et al., 1993) at: 30 Kt DDT

and 400 Kt HCH. HCH was used primarily on rice cultivation, while DDT was used mainly on fruits and vegetables. Annual production of DDT and HCH reached about 2.7 Kt and 45 Kt respectively in the late 1960s.

Chlordane importation began in the early 1970s following the ban on DDT and HCHs. Chlordane was used primarily as a termiticide for timber structures. Annual imports approached 2.5 Kt prior to banning in September 1986 at which time cumulative imports were estimated to be 17.5 Kt.

According to statistics cited by (Tanabe et al., 1989), Japanese total pesticide usage was extraordinarily intensive in 1971-72 at 10.8 kg/ha versus 2 kg/ha in Europe and 1.6 kg/ha in the USA. Intensive usage of persistent pesticides during those years may partly account for the continuing presence of certain compounds in the Japanese aquatic systems.

4.4 General Environmental Trends

Loganathan et al. (1993) present evidence of the long term environmental trends of PCBs, DDT, HCH and CHL (chlordane isomers) as manifested in human adipose tissue burdens. For PCBs, concentrations rose appreciably about 1960, and since then have declined marginally at best. The negligible decline since the 1971 ban is at least partly attributable to high Japanese fish consumption that recycles PCBs from the coastal seas into the Japanese terrestrial environment. DDT and HCH concentrations peaked in human adipose tissues through the late 1950s to 1970 and have declined gradually since then. Chlordane levels in human fat have shown marked increase from 1965 that tracks the importation trends.

4.5 Aquatic Ecosystem Trends: River Nagaragawa Fish 1968-1986

In an earlier study, Loganathan et al. (1989) examined organochlorine residues in archived lizard goby [*Rhinogobius flumineus*] fish samples from the River Nagaragawa [River Nagara in Table 4.1] which drains the Nagoya area between Tokyo and Kyoto into Ise Bay. Fish were collected in the mid reaches above the city of Seki and below the city of Mino. River Nagaragawa, which receives discharges from industrial and agricultural sources, has had a history of serious environmental problems (Loganathan et al., 1989).

The 1968–1986 time trends (Figure 4.2) likely reflect broad trends in aquatic ecosystem contamination experienced across the heavily urban-industrial areas of south-central Honshu. Organochlorines that were banned about 1970 after attaining peak usage in the 1960s, show appreciable, and generally parallel, downward trends between 1968 and 1986 with concentrations falling at least 100 fold.

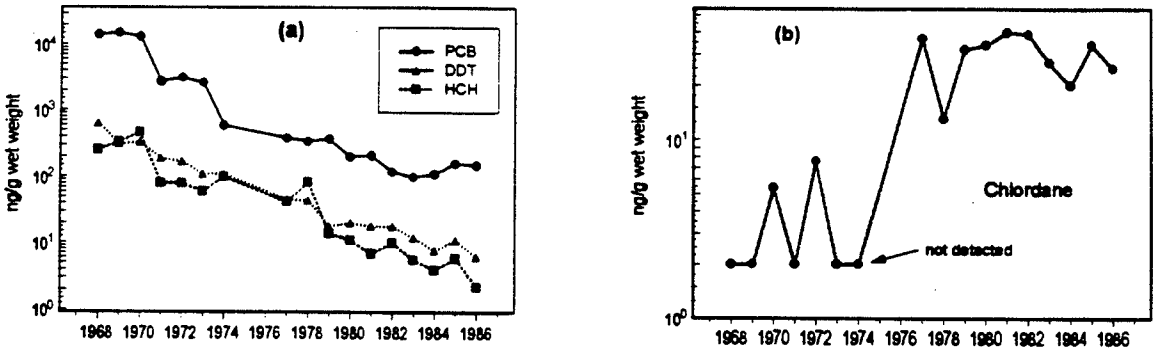


Figure 4.2 POPs trends in lizard goby from River Nagaragawa 1968-86; data from Table 1, Loganathan et al. (1989).

PCBs have been the most significant contaminant. Over 1980–86, the declining trend appears to have slowed or stabilized indicating that PCBs continue to be released to the Nagaragawa at roughly constant rate. In contrast, the insecticides DDT and HCH appear to decline steadily to 1986.

Figure 4.2b shows that chlordane in lizard goby increased from what were likely nil levels in the 1960s, to exceed DDT and HCH concentrations by 1980, but still an order of magnitude below those of PCBs. There have been no obvious trends over 1977-86.

4.6 GEMS POPs Data

The GEMS data base contains records for 15 Japanese monitoring sites that have from 8–305 sample records with a gross total of 853 records. Available POPs data may be summarized as follows:

- Most data were collected from 1979-85, and do not reflect recent conditions.
- PCBs were most frequently available with 722 observations; however, all but 2 are non-detections, and the 2 detections are possible transcription errors. Because PCB detection levels are excessively high [from 50–5,000 ng/L, but mostly 100 ng/L or 500 ng/L], the available PCB records are essentially uninformative.
- Limited HCH or α-HCH data are available at 6 sites having from 3–90 measurements, but useful data are found only at 3 sites with fully quantified results.

- DDT measurements are available at 6 sites with up to 159 isomer specific or total measurements. Most records yield little information. There are only 5 detections that allow some limited calculations at two sites.

4.7 HCH

Due to minimal GEMS data availability, Japanese HCH fluxes are estimated from combined literature and GEMS data that are summarized in Table 4.2.

4.7.1 Seto Inland Sea

The Seto Inland Sea comprises marine waters bounded by the islands of Honshu, Kyushu and Shikoku. Tributary waters and other environmental compartments were surveyed shortly after the ban on HCH in 1971 (Tanabe et al., 1989). The drainage basin that, at the time, had a population of 34 million engaged in intensive agricultural and urban-industrial activities, includes the Yodo River and many other small rivers and streams. The 1973-77 HCH summary data demonstrate the high HCH concentrations prevalent in surface waters associated with intense usage.

A summary of environmental compartment HCH burdens for the Seto Inland Sea drainage area (Table 1, Tanabe et al., 1989) showed that about 71% of the 100 t of HCH residing in the system (air, rainwater, river water, soil, seawater, sediment, fish, human), was retained in soils. This soil reservoir would seem to be the chief source of the lingering HCH burdens seen in the Yodo River as discussed below.

4.7.2 Yodo River

Four data sets, including Lake Biwa, represent the Yodo River basin. The Yodo River originates in Lake Biwa and discharges through the city of Osaka into Osaka Bay on the Seto Inland Sea. The upper watershed around Lake Biwa and the Kizu tributary are agricultural, while the other main upper tributary, the Katsura, drains the city of Kyoto. A GEMS site at Hirakata Bridge represents about 89% the total Yodo River watershed.

Water concentration trends in the Yodo River estuary (Figure 3, Tanabe et al., 1989) trace the general decline in HCH levels since 1974. Tanabe et al. argue that because HCH is more mobile than other POPs, it should have largely cleared the Yodo River system by the mid 1980s. Moreover, they suggest that the 20–25 ng/L concentrations observed in 1988 were sustained by long range atmospheric inputs from China, India and elsewhere in southeast Asia where HCH may have still been in heavy

Table 4.2 HCH in Japanese surface waters.

Site	Date	Variable	n	Mean	Min	Max	Source
Seto inland sea drainage	1973-1977	ΣHCH		250	30	5600	Fig. 1 Tanabe et al. (1989)
Lake Biwa	1979-1984	ΣHCH	16	55	21	120	GEMS site 80012
Yodo R at Hirakata Bridge	1979-1984	ΣHCH	16	69	36	121	GEMS site 80013
Yodo R estuary	1974	ΣHCH		160			Fig. 3 Tanabe et al. (1989)
	1979	ΣHCH		75			
	1983	ΣHCH		33			
	1988	ΣHCH		20			
Yodo R – 15 site survey	Apr-Dec 1990	α-HCH	117	2.5	0.4	10.0	Fig. 2 Yamaguchi et al. (1992)
		β-HCH	117	10.0	0.6	100.0	
		γ-HCH	117	0.8	0.2	3.5	
		δ-HCH	117	0.4	<0.1	3.0	
		ΣHCH	117	13.7	1.2	116.5	
Tone R at Tone-Ozeki	1980-1982	α-HCH	3	13.5	8	24	GEMS site 80003
Lake Mashu, Hokkaido	1982-1984	ΣHCH		26			Fig. 9 (Kawai et al., 1992)
	1985-1987			20			

use. However, recent Yodo River synoptic survey data (Yamaguchi et al., 1992) that show β-HCH to be dominant species (73% versus 18% α, 6% γ, and 3% δ-isomer) cast doubt on the notion that Yodo River, and generally, Japanese surface water HCH levels are supported by significant long range atmospheric inputs. Because β-HCH is less soluble, less volatile, more likely to accumulate in soil and sediments, more persistent, and resistant to microbial degradation, Yamaguchi et al.'s 1990 data suggest that recent HCH exports represent the lingering residues of the intensive applications 20 years before. The α and γ isomers appear to have greatly dissipated, leaving β-HCH to dominate. It is possible that atmospheric deposition somewhat supports the low levels of α, γ and δ isomers observed recently in the Yodo.

Yamaguchi et al.'s results are confirmed by 3 independent Osaka Bay samples (Iwata, et al., 1994) that show HCH composition nearly identical to that observed in the Yodo River. Within Iwata et al.'s synoptic survey data for southeast Asia, Oceania, and Australia, the 3 Osaka Bay samples are amongst only 5 of 50 samples in which β-HCH constitutes more than 50% of total HCH, demonstrating the uniqueness of the Yodo River – Osaka Bay HCH isomer profile.

4.7.3 Other Sites

Other sites with some data are the Tone River at Tone-Ozeki with a mere 3 α-HCH records over 1980-81, and Lake Mashu in northeast Hokkaido that has 7 ΣHCH observations taken at 30 m depth over 1982-87. These data are too sparse to permit reliable generalization across Japan; however, because the remote Lake Mashu measurements of 1987 are similar to the heavily industrialized Yodo River estuary data of 1988, it seems reasonable for first order estimation, to assume that ΣHCH concentrations are uniform across Japan.

4.7.4 ΣHCH Exports to the Sea

The probable range of recent ΣHCH exports from Japanese rivers is shown in Table 4.3 for both the total discharge estimated from GLORI data, and the slightly higher figure obtained by merging WMO and GLORI discharges. Because recent ΣHCH concentrations in the Yodo River are about 13–14 ng/L, and even remote northern waters such as Lake Mashu had concentrations of 20 ng/L as recently as 1987, Japanese riverine HCH exports to surrounding seas were estimated using uniform water concentrations across Japan as in Table 4.2. For regional and global comparisons, *medium* scenario annual exports of 6.33 t/a derived from a mean ΣHCH concentration of 15 ng/L have been selected. This may be too large if water concentrations have continued to decline since 1988.

Table 4.3 Recent annual ΣHCH exports to the sea by Japanese Rivers.

Scenario	Discharge km^3	Annual exports (t) at mean concentration $\overline{c}$ 10 ng/L	15 ng/L	20 ng/L
low	[a] 400.3	**4.00**	6.00	8.01
medium	[b] 421.7	4.00	**6.33**	8.01
high	[c] 443.0	4.43	6.65	**8.86**

[a] GLORI based discharge estimate (see Section 4.1).
[b] average of low and high discharge estimates.
[a] GLORI based discharge estimate modified by WMO discharge data (see Section 4.1).

4.8 DDT

Available data are meagre. Concentrations in river water discharging to the Seto Inland Sea during 1973-77 had a mean of only 7 ng/L and a range of 0.1–25 ng/L. Over 1979-84, Lake Biwa GEMS data showed mean concentrations of 7.7 ng/L *p,p'*-DDT, <1 ng/L *p,p'*-DDD, and 0.6 ng/L *p,p'*-DDE that yield ΣDDT of about 9 ng/L.

For the same period, the GEMS Yodo River – Hirakata Bridge site showed similar raw mean concentrations, but the flow-weighted mean concentrations were about 1.7 ng/L for *p,p'*-DDT. This implied a ΣDDT concentration of 2.3–3.3 ng/L that gives an estimated annual ΣDDT mass flow of 23–33 kg for the entire Yodo Basin.

A recent survey (1987-88) of surficial Lake Biwa sediments (Kawabe et al., 1992) showed continuing presence of DDT isomers with lake wide detection frequencies of 45% *p,p'*-DDT and 60% *p,p'*-DDE where detections showed concentrations of 2–11 ng/g. These data suggest that low level DDT exports will continue for some time from Lake Biwa and generally, Japanese surface waters.

Geographically, fish tissue data (Figure 5, Tanabe et al. 1989) show that fish from southern coastal waters continue to accumulate DDT at levels that are 2–4 times higher than fish in northern areas, with the notable exception of a fish sample from the eastern tip of Hokkaido.

4.8.1 DDT Exports to the Sea

Lacking hard recent data, ΣDDT export calculations are by necessity somewhat arbitrary and speculative. Judging from the limited Lake Biwa and Yodo River data, ΣDDT concentrations have likely declined slowly from the 2.3–3.3 ng/L levels observed

Table 4.4 Annual DDT exports (kg) from Japanese rivers to the sea.

Scenario	Low	Medium	High
Concentration	1.0	1.5	2.0
Discharge	400	422	443
S/N Ratio			
1:1	400	633	886
2:1	275	435	609
3:1	233	369	517
4:1	213	336	471

over 1979-84. This would place current levels in the 1–2 ng/L range that can be used to define ΣDDT concentrations for the southern rivers.

As cited previously, a ratio of southern to northern concentrations of 2–4 seems plausible according to fish tissue data. Prorated total Japanese riverine exports based on these assumptions are presented in Table 4.4 for *low*, *medium* and *high* concentration–discharge scenarios. Recent annual ΣDDT exports in the range 200–600 kg seem plausible. If water concentrations have declined broadly to <1 ng/L, the total Japanese ΣDDT exports may be <200 kg/a.

4.9 Chlordane

Table 4.5 gives the chlordane isomer concentrations observed in the 15 site survey conducted in 1990 by Yamaguchi et al. (1992). The proportional distribution of the isomers is nearly identical to that observed the synoptic survey of east and south Asia, Oceania, and Australia by Iwata et al. (1994) where mean compositions were 38.4% *trans*-chlordane, 31.7% *cis*-chlordane, 20.4% *trans*-nonachlor, and 9.6% *cis*-nonachlor from the 24 sites with complete records.

Table 4.6 summarizes chlordane measurements by Hirai and Tomokuni (1989) in waters and sediments of streams draining through Saga City on northwestern Kyushu over 1987-88. The Kase, a small river adjacent to the Chikugo River (Table 4.1), splits into several channels that drain through Saga into the Ariake Sea. The survey data are from (A) main river channels, and (B) a dense network of small streams in the urban area. Only *trans*-chlordane and *cis*-chlordane were measured. Here, ΣCHL for water and sediment samples was estimated by prorating the sum of *trans*- and *cis*-chlordane to include *trans*- and *cis*-nonachlor isomers by a multiplier of 1.43 based on the typical isomer composition profile observed by Iwata et al. (1994) in both water and sediments.

Table 4.5 Chlordane in the Yodo River; after Yamaguchi et al. (1992).

	n	% detection	Mean	Min.	Max.
trans-chlordane	117	96.6	2.5	<0.1	11.0
cis-chlordane	117	93.1	2.0	<0.1	11.0
trans-nonachlor	117	93.1	1.5	<0.1	10.5
cis-nonachlor	117	93.1	0.5	<0.1	3.0
ΣCHL			6.5	<0.4	35.5

ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

Table 4.6 Chlordane in Saga City streams; after Hirai and Tomokuni (1989).

	n	Mean	G-mean	Min.	Max.
river channels H2O (ng/L)	10	6.6	5.3	1.4	12.9
stream channels H2O (ng/L)	30	5.8	3.3	<0.3	15.7
stream channels sediment (ng/g)	34	110.6	24.4	0.7	572.0

G-mean = geometric mean; ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

Mean ΣCHL water concentrations of Saga City streams and the Yodo River are very similar. In Saga City sediments, the large difference between the mean of 111 ng/g and the geometric mean of 24 ng/g reflects the wide ranging data and the strong influence that some very large measurements from highly contaminated sites have on the arithmetic mean.

4.9.1 Chlordane Exports to the Sea

Riverine chlordane export estimates are based on qualitative assessments of the south/north chlordane levels observed in fish from coastal areas from Figure 5 of Tanabe et al. (1989). Regional differences are more significant and more consistent than for the other POPs considered in this report. If water concentrations follow the fish tissue patterns, the south/north concentration ratio ranges from 5:1 to 10:1. Given the Saga City and Yodo River data, a mean southern river concentration of 7 ng/L is adopted. The loads (t/a) for a range of south/north (S/N) ratios are: 2.90 (1:1); 1.42 (5:1); 1.18 (10:1) where the S/N water concentration ratio is given in parentheses. If average northern Japan water concentrations are 1.4 ng/L (1/5 of the Yodo River / Saga City concentrations), total Japanese chlordane exports would be about 1.4 t/a,

while if average northern Japan water concentrations exceed 1.4 ng/L, total Japanese chlordane exports climb into the 1.4–2.9 t/a range. If mean northern river levels are very low, exports decline towards 1 t/a that comes mainly from southern Japan.

4.10 PCBs

As GEMS data are uninformative, PCB exports from Japan are estimated from limited literature data. Since about 1960, riverine PCB exports from southern rivers have been significant. During 1973-77, rivers draining to the Seto Inland Sea had mean PCB concentration of 250 ng/L. Tanabe et al. (1989) indicate that PCB transport to the sea was relatively rapid as it originates mainly from urban-industrial areas that are concentrated in close to the sea. The fish tissue data shown in Figure 5 of Tanabe et al. (1989) suggest PCB concentrations that are 2–5 times higher in the southern coastal waters than in the north.

4.10.1 PCBs Loading to Tokyo Bay Sediments

The Tokyo Bay watershed is amongst Japan's mostly heavily urbanized and industrial drainage basins. The bay receives mean annual water inflows of about 7.4 km^3 and sediment inflows of about 1.2 million t (Sugiura et al., 1986). With 1,000 km^2 area, mean depth 17 m, water residence time of 1.5 months, and mean annual sedimentation rate of 0.18 g/cm^2, Tokyo Bay also serves a sink for appreciable quantities of sediment bound contaminants.

For 5 sediment cores, Sugiura et al. (1986) gave dated concentration–depth profiles and estimated historical PCB accumulation in Tokyo Bay sediments to 1980 for 4 sets of PCB congeners comprising 2 individuals (IUPAC nos. 73 and 180) and 2 summed pairs (IUPAC nos. 101 plus 136; and 105 plus 153). The concentration profiles show trace PCBs present as early as 1936. Appreciable concentrations began about 1950, and steep increases occurred through the 1960s and early 1970s, followed by decline toward concentrations in the range observed from 1955-65.

From additional composition data for the Kanechlors (KC-300, KC-400, KC-500, KC-600) and the percentage that each contributed to total Japanese PCB production, estimates were derived for the historical time trend of annual total PCB sedimentation rates, and the cumulative tonnage of total PCBs buried in Tokyo Bay to 1980. Three estimates were calculated by prorating total PCBs from the mean percentage of the 6 measured congeners in:

1. A mix of all 4 Kanechlors weighted by their contribution to total Japanese PCB production. This a *high* estimate that gives large weight to light PCBs present in the KC-300 technical mixture. These low chlorinated congeners are much more likely

to have volatilized at the source, in transit to the bay, or from the bay surface; and are more likely to have been transported through the bay in aqueous solution.

2. A mix of 3 Kanechlors [KC-400, KC-500, KC-600] weighted by their contribution to total Japanese PCB production. This is a *low* estimate that assumes that only the 3 heaviest Kanechlors contribute to the PCB sediment burden of Tokyo Bay.

3. The KC-500 technical mixture. This is the *minimal* or *KC-500* estimate.

According to the relative proportions of the 4 constitutent congener measurements given by Sugiura et al. (1986) at five year intervals from 1950 to 1980, the PCB composition of sediments most closely matches that of KC-500. However, the heaviest congener (IUPAC no. 180) is under-represented and the estimated total PCB burden of Tokyo Bay sediments may be low by assuming that PCBs in sediments have KC-500 composition. Thus it seems most likely that the total PCB load in Tokyo Bay sediments is in the range defined by the *low* and *KC-500* estimates. The high estimate may be more indicative of the total PCB emissions initially released into the water courses draining to Tokyo Bay and from direct effluent discharges. As volatilization losses could be appreciable even for higher PCBs, the *high* estimate of PCB sediment load may yet underestimate actual PCB emissions.

The historical trends of the three estimates of total PCB loads into Tokyo Bay sediments are shown in Figure 4.3. The peak sediment loading is estimated to have occurred about 1975, lagging the peak Japanese PCB production that occurred in 1970. During 1975, the estimated total PCB losses into sediments were 120, 146 and 234 kg respectively for the KC-500, low and high estimates, and from 1950 to 1980, the estimated cumulative total PCBs buried in Tokyo Bay sediments are 2.2, 2.8 and 4.5 tonnes respectively.

If PCB release in the Tokyo Bay watershed follows the pattern shown in other heavily industrialized and urban Japanese rivers such as the Yodo and the Nagaragawa, the decline in PCBs transport to Tokyo Bay would have stabilized after 1980 at levels comparable to 1980 or slightly lower. Under these conditions, an additional 1.3 to 2.7 t total PCBs would have been buried in Tokyo Bay sediments between 1981 and 1995.

By normalizing the PCB sedimentation load to the annual water and sediment inflows to Tokyo Bay, the low range of total PCB concentrations in a whole water sample due only to suspended sediments in inflowing Tokyo Bay tributary waters was estimated at 13-15 ng/L during 1980, while the high estimate was about 25 ng/L. Although these estimates ignore PCBs in aqueous phase and losses due to volatilization and transport through the bay, the figures are remarkably close to the whole river water concentration of 30 ng/L observed in the Yodo River since about 1980 (see next section). Taken together, Yodo River water, Nagara River fish and Tokyo Bay sediment data all show essentially the same time trend patterns where

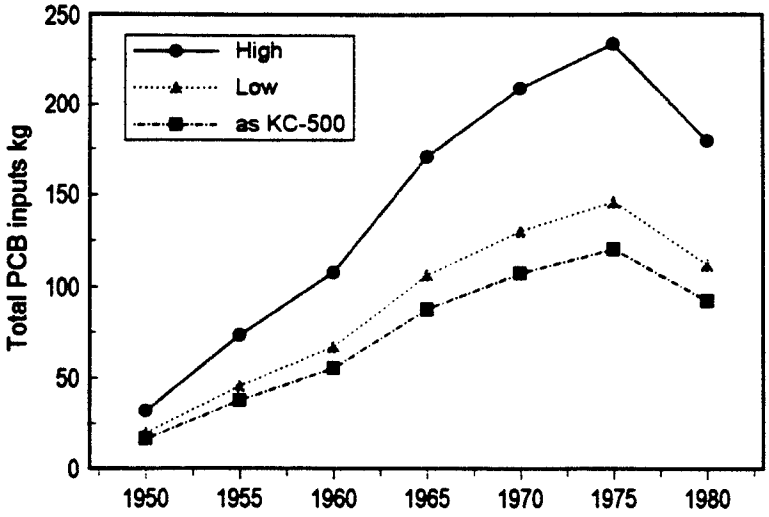


Figure 4.3 Estimated total PCB loads into Tokyo Bay sediments; 1950–1980.

records overlap. This suggests that comparable PCB contamination occurred in freshwater ecosystems across the urban-industrial region of southern Honshu Island.

4.10.2 PCBs in Yodo and Neya Rivers

PCB concentrations in the polluted lower reaches of the Yodo River have declined from an annual mean of about 160 ng/L in 1974 to an annual mean of 30 ng/L in 1988 (Figure 2, Tanabe et al., 1989), and appeared to be declining very slowly in the late 1980s. PCB concentration trend was also shown for a *"less polluted"* upper reach of the Yodo River where levels have fluctuated between 10-20 ng/L over the same period from 1974-88. Tanabe et al. (1989) suggest that 10 ng/L is a *"background level"* for the Yodo watershed that represents leakage from many small sources that will likely persist well into the future.

Within 2 km of the outlet of the Yodo River to Osaka Bay, the River Neya and the River Second Neya intersect and discharge to Osaka Bay through Hokko Basin. The Osaka City municipal incinerator is located on River Second Neya about 1 km upstream of the junction with River Neya, and 8-10 km upstream of Osaka Bay. Sediments collected in the Neya and Second Neya rivers, and the Hokko Basin in August 1985 and June 1986 were highly contaminated with PCBs. The river site nearest the incinerator had PCBs of 5,300 μg/g, and levels declined to an average of about 1,300 μg/g in the Hokko Basin.

Though PCB emissions from the Osaka incinerator are not given, the incinerator is a significant local source of PCBs. Japan relies heavily on incinerators for disposal of municipal wastes, and the many incinerators in operation may contribute appreciable quantities of PCBs to Japanese aquatic systems.

4.10.3 PCBs Exports to the Sea

With concentration data limited to only the Yodo River basin, estimates of total Japan PCB exports are speculative. While the Yodo River typifies the most intensely urbanized and industrialized watersheds of southern Honshu, the 1988 *"polluted reach"* concentration of 30 ng/L may be too high for the rivers of the two southern islands Kyushu and Shikoku. Thus Japanese riverine total PCB exports were calculated according to the following scheme:

1. for southern Japan, a total PCB concentration of 30 ng/L was selected to represent the area of heavy urbanization and industrialization which was determined arbitrarily as two times the approximate area of the Tokyo Bay, Nagaragawa, and Yodo watersheds (area $\approx$ 32,300 km^2),

2. for *non-industrial* southern Japan including Kyushu and Shikoku (area $\approx$ 87,800 km^2), the upper Yodo River *"background"* concentration of 10 ng/L was taken to reflect the greater levels of development relative to northern Japan,

3. for northern Honshu and Hokkaido, a total PCB concentration of 5 ng/L was taken to reflect the lesser developed character of the region,

4. because assumptions 1–3 may be too high, especially outside the heavy industrial areas, a *low* scenario was developed by reducing all concentrations 1/2 to allow for decline since 1988 and overestimation of northern Japan PCBs concentrations,

5. a *medium* scenario was developed taking 3/4 high scenario concentrations.

The resulting *medium* scenario Japanese riverine total PCB export estimates to coastal waters are given below in Table 4.7. Although the heavy urban-industrial Osaka–Tokyo area is a strong source, northern Japan by virtue of its relatively large areal extent and water discharge could be an equally prominent source. The *low* and *high* scenarios give annual flux estimates of 1,780 and 3,940 kg respectively.

Table 4.7 Estimated *medium* scenario annual total PCBs exports by Japanese rivers.

Region	PCBs ng/L	Area 10^3 km^2	Discharge km^3	Exports kg
Southern Japan — heavy industrial	22.50	32.3	43	956
Southern Japan — non-industrial	7.50	87.8	116	866
Northern Japan	3.75	240.2	264	989
Japan Total		360.3	422	2,811

4.11 POPs River Flux Summary

Low, *medium* and *high* estimates of total Japanese riverine exports of PCBs, HCH, DDT and chlordane developed in this chapter are summarized below in Table 4.8.

Table 4.8 Estimated annual POPs exports (t) to coastal seas by Japanese rivers.

	Low	Medium	High
ΣHCH	4.00	6.33	8.86
PCBs	1.78	2.81	3.94
ΣCHL	1.18	1.42	2.90
ΣDDT	0.21	0.37	0.61

ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

Chapter 5 China

5.1 Hydrology

Long term mean annual discharges for major Chinese rivers are summarized in Table 5.1. Rivers are ordered from north to south by latitude of the outlet. The Changjiang (Yangtze) clearly dominates both in drainage area and annual discharge volume. Together the Changjiang and Xijiang (Pearl) account for 87% of Chinese river discharge to the sea.

Table 5.1 Major Chinese rivers discharging to the sea; after Milliman et al. (1995).

	Basin area		Annual discharge		Runoff
	10^3 ha	%	km^3	%	mm
Daliaohe	27	0.8	9	0.6	333
Liaohe	230	6.5	16	1.1	70
Daling	20	0.6	1	0.1	50
Luanhe	44	1.2	3	0.2	68
Huanghe	770	21.8	59	4.2	77
Changjiang	1,900	53.8	930	65.8	489
Menjiang	61	1.7	58	4.1	951
Jiulongjiang	15	0.4	15	1.1	1,000
Dongjiang	25	0.7	23	1.6	920
Xijiang	440	12.5	300	21.2	682
Totals	3,532		1,414		

Table 5.2 Prorating factors [PRFs] for GEMS sites on Chinese Rivers.

River	Annual mean sample discharge	Mean annual discharge at outlet	PRF
	km^3	km^3	
Changjiang	725.1	930	1.28
Xijiang	202.7	300	1.48
Huanghe	28.3	59	2.08

5.1.1 Discharge Data at GEMS Sites

HCH data are available at three GEMS sites on major Chinese Rivers: Changjiang (30° 35' N, 114° 47'E), Xijiang (23° 4' N, 112° 27'E), and Huanghe (36° 44' N, 116° 59'E), and each water quality measurement is accompanied by a discharge record that was interpreted as the mean daily discharge on the day of sampling.

To estimate total China river flux, it is required to prorate fluxes estimated at GEMS sites upwards to the respective entire watersheds; however, drainage areas for the sampling sites were not available. To effect the upward adjustment, approximate areal prorating factors [PRFs] were developed as the ratio of long term GLORI discharges for the entire watersheds, to the mean of the sample discharges reported at each GEMS site. These PRFs are listed in Table 5.2. At each GEMS site, there were over 200 sample flows from 1980-92; hence, the long term mean sample flows give reasonable approximations of the long term mean annual discharges.

The approximate PRFs in Table 5.2 were validated by review of historical WMO/UNESCO river discharge records that contained data for several sites on the three rivers. Near the Changjiang and Xijiang GEMS sites, there were discharge gauging stations with drainage area data suggesting that the estimated PRFs are approximately correct. On the Huanghe River, it possible that the PRF is too high and that Huanghe River flux will be overestimated, but, because Huanghe water discharge is a relatively small portion of total China discharge, the effect on total China river flux will be negligible.

5.2 POPs Usage in China

5.2.1 PCBs

Chu et al. (1995) claim that from 1966-76, China produced 10 Kt of a PCB mixture similar to Aroclor 1242. It is not clearly stated whether these were all the PCBs produced, nor is it clear whether China imported any PCBs. The deployment and fate of the 10 Kt Chinese PCBs are unknown. If these PCBs were deployed in heavy electrical equipment, it is likely that most remain in service. As will be shown, there is some limited evidence of low level PCB contamination.

5.2.2 Insecticides

It is reported that between 1965 and 1983, Chinese pesticide production rose from 534 Kt to 1,224 Kt, and that DDT and crude HCH accounted for 50% of production (Dhaliwal and Pathak, 1993). If so, river discharge of these POPs would have been appreciable up to bans on agricultural use imposed in 1983. According to Voldner and Li (1995), cumulative HCH usage in China was >100 Kt, while cumulative DDT usage was between 10–100 Kt. Kannan et al. (1995) claim that annual HCH usage was 20 Kt

in the late 1970s; however, Liu and He (1987) gave the combined DDT and HCH 1980 usage for Hunan province as 154 Kt. HCH likely dominated the latter figure suggesting that total China HCH usage in 1980 was much higher than the 20 Kt claimed by Kannan et al. (1995). Some DDT use continues, possibly for public health purposes.

Voldner and Li (1995) also indicate that 1-10 Kt of toxaphene was used in China up to 1975 when it appears to have been banned. Usage and status of other POPs insecticides including aldrin, dieldrin, heptachlor and chlordane is not known.

5.3 POPs in the Chinese Environment

5.3.1 Xiamen Harbour — Jiulong River Estuary

Xiamen Harbour is located off the Jiulong River estuary between Xiamen Island and the mainland. In November 1993, Hong et al. (1995) collected 7 sediment samples in the harbour and 2 in the estuary. These were analyzed for HCH, DDT and PCBs. Data are summarized in Table 5.3. The order of contamination is DDT > PCBs > HCH. The *o,p'*- isomers were not measured. These can be significant in sediment; so that, the actual ΣDDT levels may be somewhat higher than the reported values.

The mean DDT concentration is driven upwards by a single very high sample of 311 ng/g from near a densely populated urban district near the mouth of the harbour and the Jiulong River. The mean and maximum of the remaining samples are 9 and 17 ng/g respectively. In 3 of 9 samples unmetabolized DDT exceeds 50% and in 2 others equals 49% of ΣDDT. Generally, these data indicate ongoing DDT usage, possibly for public health purposes. In a broad regional comparison, DDT contamination is modest, except for the singularly high urban sample. HCH contamination is modest. According to Hong et al. (1995), levels are lower than in a previous survey of the mid 1980s.

PCBs levels are low. In a broad regional comparison [Chapter 2], Xiamen sediments are the least contaminated of 19 areas across Asia-Pacific region. But, there is contamination in the vicinity of the same urban site where high DDT was observed, and also near the discharge point of the city sewage lagoons. The isomer composition is similar to Aroclor 1254 (Hong et al., 1995).

Table 5.3 POPs in Xiamen Harbour and Jiulong estuary sediments (ng/g dry); Hong et al. (1995).

	Mean	G-mean	Min.	Max.
ΣHCH	0.47	0.39	0.17	1.12
ΣDDT	42.70	12.40	4.50	311.00
PCBs	1.74	0.83	0.05	7.24

ΣHCH = α-HCH + β-HCH + γ-HCH + δ-HCH; ΣDDT = *p,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.

5.3.2 PCBs in Mussels

In November 1992, Chu et al. (1995) sampled mussels and other shellfish from Da Chen Island (ca. 28° 30' N, 121° 50' E) and a river estuary on the adjacent mainland. The mussel sample from the island had 1.4 ng/g wet weight, while other shellfish had from 2-9 ng/g PCBs. These levels are very low.

PCB concentrations of 32.4 ng/g wet weight observed in the late 1980s in mussels from the vicinity of Dalian City in northern China at the entrance to the Gulf of Chihli (Wu et al., 1989, cited by Chu et al., 1995). While PCBs levels of the Dalian City mussels are 20 fold higher than Da Chen Island mussels, contamination is modest on a global scale. Nonetheless contamination is occurring.

5.3.3 POPs in Chinese Foodstuffs

A 1990 survey of foodstuffs in 12 Chinese provinces sheds light on recent HCH and DDT levels in the Chinese environment (Chen and Gao, 1993). The estimated per capita average daily intakes [ADIs] of DDT and HCH were 20.5 and 5.0 μg/person/day respectively. From comparative data in (Kannan et al., 1992a), Chinese ADIs rank second with Vietnam well behind India, but notably ahead of other countries in the region. Over 1973-78, respective mean ADIs of DDT and HCH were 41.4 and 150.0 μg/person/day, close to recent ADIs in India. The older ADI data seem to confirm that China used DDT and HCH at very high rates in the 1970s.

Also given are concentration data for *aquatic foods* including freshwater fishes and marine seafoods that comprise only 1.3% of typical Chinese diets. DDT wet weight concentrations (mean 141 ng/g) rank with certain River Ganges, India and Rivers Diyala and Shatt al-Arab, Iraq fishes as the most contaminated in the region. For HCH, China mean wet weight concentrations (mean 9 ng/g) rank after most Indian and Iraqi sites, but ahead of most other countries in the region. If the decline in respective ADIs between 1973-78 and 1990 is a valid indicator of conditions in aquatic systems, DDT will have declined by half, while HCH will have declined by 30 fold.

5.4 POPs in the Chinese Environment: GEMS Data

The GEMS water data base contains 1980-92 HCH and DDT data for 3 Chinese river sites — Changjiang (Yangtze), Xijiang (Pearl) and Huanghe (Yellow) rivers — and one lake (Tai). At river sites, there were 24 samples taken annually to 1984 and 12 thereafter giving 205–215 observations at each site. These time series were sufficient to estimate HCH fluxes for the entire period, and to show clearly the long term time trends from the pre-ban 1983 years through the post-ban decade.

5.4.1 GEMS HCH Data

Chinese HCH records occur in GEMS files as α-HCH (1980-86), and a generic HCH variable (1980-92) that represents either: (1) ΣHCH = α-HCH + γ-HCH, or (2) ΣHCH = α-HCH + β-HCH + γ-HCH. If the former interpretation is correct, the true ΣHCH concentration could be higher by roughly 30%[8] if the β-HCH isomer had been measured. If the latter interpretation is correct, data need no adjustment.

5.4.2 ΣHCH River Load Calculation Method

The availability of long term HCH time concentration series with contemporaneous river discharge data permitted more advanced flux estimation methods than for any other data set considered in this report.

Within a one year period, the mass flux is estimated as $L = \overline{Q} \cdot \overline{c}$ where $\overline{Q}$ is the mean annual discharge and $\overline{c}$ is an estimate of mean chemical concentration. For Chinese river data, the flow-weighted mean concentration $\overline{c} = \sum c_i q_i / \sum q_i$ is used where c_i and q_i are respectively sample concentration and river flow on the day of sampling. Annual discharge data were not available; so that, mean annual discharges $\overline{Q}$ were estimated as the mean of sample flows, i.e., $\overline{Q} \approx \overline{q} = \frac{1}{n}\sum q_i$ observed in any 1 year period. This estimate roughly approximates year-to-year variations in mean annual discharge rates, but requires area prorating by the factors given in Table 4.2 to adjust the flux upwards from the sampling site to the entire watershed.

Using the preceding methods, raw fluxes were estimated as *annualized quarterly moving averages*, i.e., at each quarter, the load was estimated from data within ± 6 months of the current time. Then the quarterly raw flux series was smoothed by further moving averaging to approximate a continuous curve.

5.4.3 Estimating Total Chinese River Flux

To estimate all China flux, the 10 rivers of Table 5.1 were regrouped as:

1. Northern rivers (Daliaohe, Liaohe, Daling, Luanhe, Huanghe),
2. Changjiang,
3. Southern rivers (Menjiang, Jiulongjiang, Dongjiang),
4. Xijiang.

POPs flux from Northern rivers was estimated using Huanghe data. Northern rivers export only 6% of total Chinese water discharge. Flux from Southern rivers was

[8] An average estimate derived from Asia-Pacific survey data in Iwata et al. (1994).

estimated by prorating averaged properties (flow-weighted-mean concentrations and unit area loads) of the two surrounding basins (Changjiang and Xijiang). As the three rivers with GEMS data account for about 91% of total China water discharge, the errors involved in the upward extrapolation to all China are relatively small.

5.4.4 ΣHCH River Loads

Table 5.4 summarizes contaminant flux data for the three rivers having the actual measurements and the total China estimates. Figure 5.1 presents the final smoothed time series of ΣHCH fluxes from Chinese rivers stacked to show the total flux by Chinese rivers and emphasize the overwhelming dominance of the Changjiang River. The decline following the 1983 ban is striking.

The peak flux observed about mid 1984 is believed to be an artefact of corrupted ΣHCH records in the GEMS data file between 1984-86 for the Changjiang River only [9]. The true peak is most likely the earlier one occurring about the beginning of 1983.

During the pre-ban period, total China ΣHCH riverine exports rose from 555 t (721 t) in 1980 to a peak of 780 t (1020 t) in the 12 month period from April 1982 to March 1983, and by 1992, fell to 31 t (40 t) where the figures in parentheses are prorated to include β-HCH isomer. As the plot suggests, gradual decline of HCH concentrations was continuing at the time available records ended in December 1992.

The China HCH case illustrates the significant impacts that result and the time scales involved when a large country commits to severely restricting the usage of a persistent organochlorine pollutant. The Changjiang (Yangtze) was and remains the dominant contributor to total Chinese ΣHCH riverine exports (ca. 70% over 1981-83 falling to ca. 54% over 1991-92). Recently, the Xijiang River has assumed a greater proportion of total China flux because concentrations have declined more slowly and now exceed those in the Changjiang by almost 2 fold.

5.4.5 Lake Tai ΣHCH Trends

HCH data for Lake Tai (31° 19' N, 120° 6'E) located to the south of the Changjiang estuary and inland from Shanghai, show generally similar temporal patterns as the river HCH data. However, Lake Tai data show some features not common to the river data,

[9] The α-HCH / ΣHCH ratio oscillates wildly between 0 and 1 from 1984–86. This would imply that ΣHCH suddenly comprised all α-HCH, then abruptly all γ-HCH, and so on. The scenario is physically implausible given the great hydrochemical inertia of a river as large as the Changjiang. Similar behaviour is not observed at other Chinese sites. It seems likely that the oscillations are the artefacts of transcription or data entry errors in the Changjiang record of 1984-86.

Table 5.4 ΣHCH in Chinese rivers; pre-ban (1981-83) and recent (1991-92) periods; ΣHCH = α-HCH + γ-HCH — adjust upwards by 30% to approximately include β-HCH.

		Changjiang	Xijiang	Huanghe	China total
Annual river load (t)	81-83 maximum	560	142	28.6	783.0
	81-83 mean	457	130	20.5	661.0
	91-92 mean	23	13	0.2	39.3
Annual flow-weighted-mean concentration (ng/L)	81-83 maximum	546	498	390	—
	81-83 mean	467	407	310	—
	91-92 mean	24	44	8	—
Annual unit area load (g / km^2)	81-83 maximum	295	322	37.1	—
	81-83 mean	240	295	26.6	—
	91-92 mean	12	29	< 0.3	—

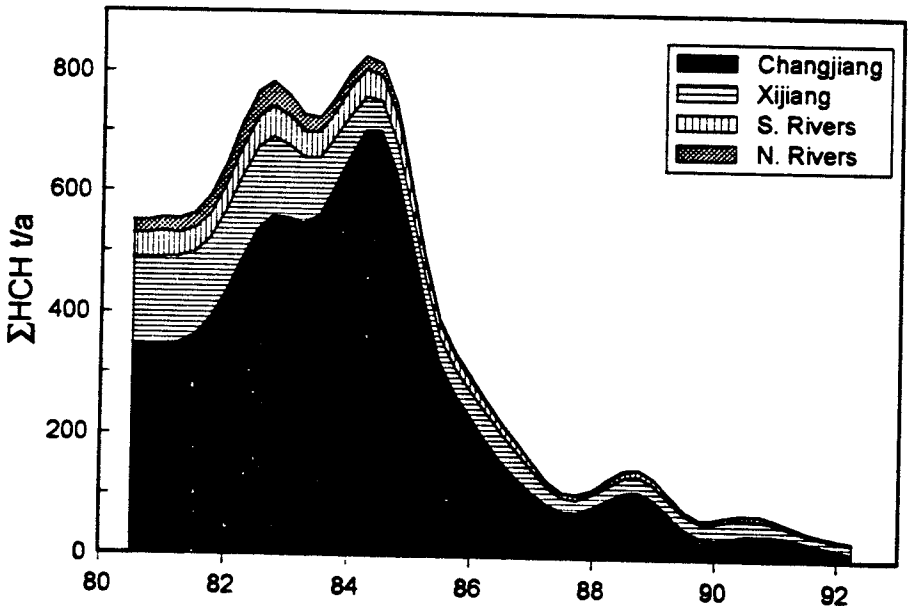


Figure 5.1 Annual ΣHCH flux from Chinese rivers 1980–1992.

namely: (1) water concentrations during the high usage years (1980-83) were more variable ranging up to 4 μg/L during the peak summer usage periods, (2) unlike river data that suggest widespread usage of crude HCH, in Lake Tai watershed, it appears that both lindane (γ-HCH) and crude HCH were used, with lindane dominating, and (3)

Lake Tai HCH concentrations have declined somewhat more rapidly than in the rivers that have much larger drainage areas.

5.4.6 DDT Flux

While DDT time series at Chinese river sites have 205–215 observations for 1980-92, almost all are non-detections after 1980. The details of the 1980 ∑DDT estimates are shown in Table 5.5. Total DDT exports were estimated for 1980 on the Xijiang and Huanghe rivers which had enough fully quantified measurements, and then 1980 estimates for the Changjiang River which had no detections were derived by averaging concentrations and unit area loads of the other two rivers. Variations on the basic statistical assumptions yield the *high* and *low* estimates given in Table 5.5.

The remarkably high ∑DDT concentrations and relatively high flux seen in the Huanghe River during 1980 are due to extraordinarily high sediment concentrations. The Huanghe drains an area known for the most erodable soils in the world, and accordingly, the river waters have sediment concentrations of the order 20 g/L. The strong partitioning tendency of DDT to fine sediments would assure that more DDT is retained by eroding soils. Other POPs chemicals with strong particle affinity are expected to behave similarly in similar circumstances.

Table 5.5 ∑DDT † exports ca. 1980. Huanghe and Xijiang River exports calculated from monitoring data; Changjiang and unmonitored rivers estimated using (A) Huanghe UAL, and (B) Xijiang UAL.

	Flow	Concentration		Load		Unit area load	
	km^3	ng/L		t/a		g / km^2	
		Low	High	Low	High	Low	High
Huanghe R	45	135.5	182.3	6.1	8.2	7.9	10.7
Xijiang R	275	3.6	7.6	1.0	2.1	2.3	4.8
A. Changjiang and unmonitored rivers at Huanghe 1980 UAL							
Changjiang	1,003	15.0	20.2	15.1	20.3	7.9	10.7
all China	1,452	17.6	24.2	25.5	35.1	7.2	9.9
B. Changjiang and unmonitored rivers at Xijiang 1980 UAL							
Changjiang	1,003	4.3	9.0	4.3	9.1	2.3	4.8
all China	1,452	8.5	14.7	12.4	21.4	3.5	6.1

† ∑DDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.

Because of the unusual circumstance of the Huanghe watershed, the Changjiang DDT data were used to extrapolate Xijiang ΣDDT flux. This gives total Chinese riverine ΣDDT flux of 12-21 t for 1980.

After 1980, only speculative estimates are possible due to inadequate analytical detection capability. If DDT water concentration trends paralleled HCH water concentration trends, in the post-ban era, ΣDDT river concentrations would have fallen at least one order of magnitude by 1990-92, in which case annual ΣDDT exports from Chinese rivers would be about 1-2 t. If DDT water concentration trends were similar to DDT trends in food ADIs, ΣDDT river concentrations would have fallen by only 1/2, in which case 1990 ΣDDT exports from Chinese rivers would be about 6-11 t/a. For present purposes, 1990 DDT exports are estimated as 5 t, the mean of these estimates, with a range of 1–11 t/a. To accurately determine the concentration of the DDT metabolites, detection capability down to 100 pg/L or less is currently required.

5.4.7 PCBs Flux

Given that at least 10 Kt of PCBs were produced in China, and that there has been minor contamination of coastal sediments and biota by PCBs, the rivers of China likely carry some minimal PCB load, particularly from those watersheds with large urban-industrial complexes. Approaching the problem in reverse, given the total China mean annual river discharge of 1,414 km^3, a China wide river concentration of about 750 pg/L would be necessary to deliver 1 t/a PCBs to the sea, and a river concentration of 75 pg/L would deliver about 100 kg PCBs. Considering the regional surface water PCB data of Iwata et al. (1994), it is likely that the net China wide PCBs concentration in water is <100 pg/L. Local surface waters with somewhat higher concentrations might be found in urban-industrial areas where PCBs were deployed.

Chapter 6 Iraq — Shatt al-Arab River

6.1 Hydrology

The Shatt al-Arab River is a short channel between the confluence of the Tigris and Euphrates rivers and the Persian Gulf. Milliman et al. (1995) give the Shatt al-Arab drainage area as 3,800,000 km^2, and annual discharge as 46 km^3.

Figure 6.1 shows a flow schematic of the Shatt al-Arab River that is variably cited as being about 110–190 km in length. The city of Basra lies about 20 km below the Tigris–Euphrates junction, and about 20–30 km further downstream, the Shatt al-Arab splits into two distributary channels that debouch to the Persian Gulf another 50 km or more downstream. The southern branch appears to be the main channel. At the head of the two distributaries, the Shatt al-Arab is joined by a sizable tributary from Iran, the Karun River which has a mean annual discharge at of 18 km^3 at Avhaz, Iran about 100 km upstream of the junction with the Shatt al-Arab. Below Avhaz, a significant tributary, the Jarrahi River, joins the Karun before it joins the Shatt al-Arab.

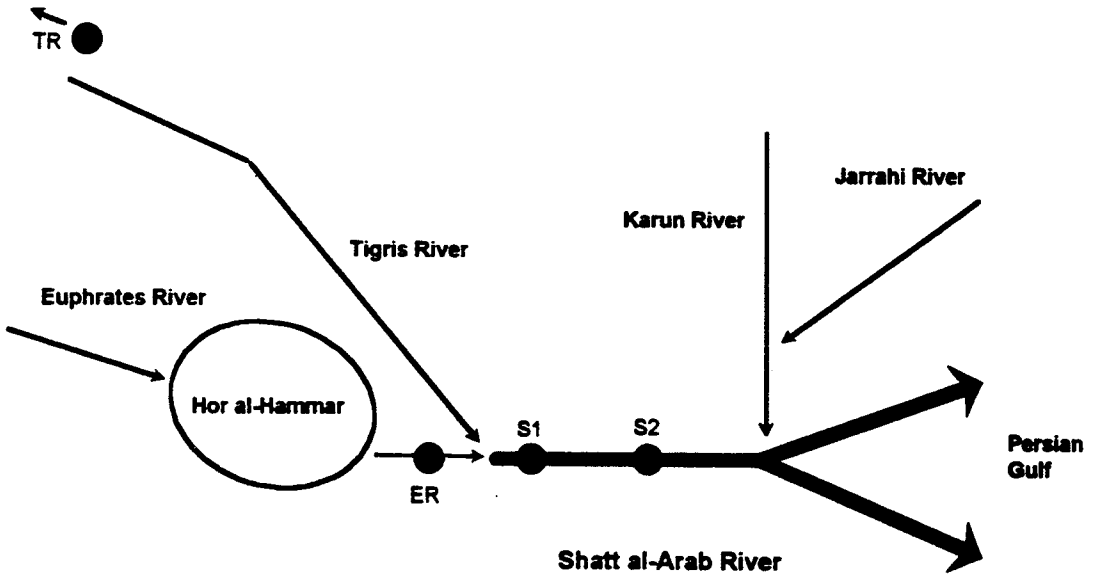


Figure 6.1 Shatt al-Arab River schematic with water sampling sites.

Water sample data to be used in POPs river flux estimation were obtained at sampling sites S1 and S2, above and below Basra respectively. DouAbul et al. (1988, citing Mohammad, 1982) give the mean discharge at the S1 and S2 sampling sites as 919 m^3/s or 29 km^3 on an annual basis. This approximately equals the annual discharge volume estimate of 46 km^3 of Milliman et al. (1995) less the contribution of the Karun River (18 km^3 + Jarrahi River discharge – evaporation / consumption losses). Accordingly, the discharge of 29 km^3 should be used to estimate POPs fluxes at sites S1 and S2. Separate calculations should be done for the Karun River.

6.2 POPs Usage in the Shatt al-Arab Basin

The Shatt al-Arab watershed extends over Syria, Turkey, Iran and Iraq. Generally, the history of POPs usage in these countries is poorly documented. Iraq appeared to be the main source of contaminants borne by the Shatt al-Arab River in the late 1980s. According to Al-Omar et al. (1986), Iraq used organochlorine insecticides extensively against malaria and in agriculture for 25 years [beginning about 1950] before banning in 1976. However, data to be presented suggest that usage continued into the late 1980s. PCBs have likely been used in all the Shatt al-Arab basin countries, but no information regarding their environmental occurrence has been reported.

Some POPs residues in the Shatt al-Arab may have originated in Syria, Turkey and Iran. POPs usage history in Syria is unknown. Turkey began implementing restrictions on organochlorine pesticides in 1979, and in 1985, prohibited all except toxaphene and endosulfan (Burgaz et al., 1994). It is expected that the organochlorine insecticide burden entering the rivers in Iraq would likely overwhelm those originating upstream in Syria and Turkey.

The Karun River tributary may carry significant lindane (γ-HCH) residues from Iran. It is reported that Iran banned organochlorine insecticides except lindane in 1985 (Burgaz et al., 1995).

There is no readily available information on PCB usage or environmental occurrence data in the Shatt al-Arab watershed.

6.3 POPs Insecticides in Iraqi Aquatic Systems

6.3.1 POPs in Aquatic Biota

Several studies have been conducted of POPs accumulation in aquatic biota in Iraqi surface waters (Al-Omar et al., 1986; DouAbul et al., 1987; DouAbul and Al-Saad, 1987; DouAbul et al., 1988). Generally, these findings confirm the significant presence of the same organochlorine insecticides observed in surface waters and sediments [HCH, aldrin, dieldrin, endrin, heptachlor, chlordane, and DDT] that are discussed in

the following sections. These studies provide additional perspective for Hor al-Hammar lake. Evidently spraying has long been conducted near or even over lake waters. Fish (*Barbus xanthropterus*) from Hor al-Hammar have high levels of endrin, chlordane, and DDT (DouAbul et al., 1987). Endrin levels standout as being especially high. In the Shatt al-Arab River, another species (*Tenualosa ilisha*) showed high levels of DDT, heptachlor, chlordane, aldrin, dieldrin, and endrin. Sporadic samples showed significant presence of β-HCH indicating that crude HCH was being used to some extent in upstream areas.

6.3.2 POPs in Baghdad Sewage Effluents and the Diyala River

During 1984, Al-Omar et al. (1989) studied the presence of organochlorine insecticides in water and sediments of the Diyala River downstream of the Baghdad (Rustamiya) sewage treatment plant [STP] outfall that is located not far above the confluence with the Tigris River. The plant was expanded in 1983 to serve a population of 1.5 million and has an estimated dry weather peak discharge of 155,000 m^3/d. POPs concentrations and mass flux estimates are presented in Table 6.1. The estimated annual mass flows of insecticides in STP effluents are remarkably high for a country that claimed to have banned the chemicals in 1976. The dry weather peak discharge is a design flow that may overestimate actual discharges, but even at a more conservative 1/2 or 1/3 design flow, discharges of insecticides would still be very high.

Table 6.1 POPs insecticides in Baghdad sewage treatment plant effluents 1984, and POPs fluxes in effluents at 155,000 m^3/d peak dry weather flow (Al-Omar et al., 1989).

Insecticide	Concentration mg/L	Daily load kg	Annual load t
lindane (γ-HCH)	0.206	31.93	11.66
aldrin	0.092	14.26	5.21
dieldrin	0.061	9.46	3.45
Σdrin	0.153	23.72	8.66
endrin	0.194	30.07	10.98
ΣDDT	1.135	175.93	64.26

Σdrin = aldrin + dieldrin.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD.

Table 6.2 POPs insecticides (ng/L) in Diyala River March–October 1984.

	Mean	† G-mean	Min.	Max.
γ-HCH	377	269	<84	1,589
Σdrin	596	419	61	2,141
Σhep	546	343	<44	3,667
Σchlordane	1,752	1,205	168	6,100
ΣCHL	3,047	2,195	393	12,373
ΣDDT	1,320	997	259	5,577

† geometric mean.
Σdrin = aldrin + dieldrin.
Σhep = heptachlor + heptachlor epoxide.
Σchlordane = *trans*-chlordane + *cis*-chlordane.
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor + heptachlor + heptachlor epoxide; nonachlors estimated as mean proportions in survey data of Iwata et al. (1994).
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD.

Water data collected at 5 sites in the Diyala River upstream and downstream of the STP effluent outfall from March to October 1984 are summarized in Table 6.2. Differences between upstream and downstream sites were marginal. Upstream data show that the river was already quite contaminated with POPs insecticides; hence, the STP effluents do not greatly increase insecticide concentrations in the river. River water data also show that collectively heptachlor and chlordane isomers are the dominant contaminants followed by DDT. According to DouAbul et al. (1988) heptachlor has never been used in Iraq which would imply that heptachlor seen in the Diyala River originated from heavy chlordane usage.

6.3.3 POPs in Shatt al-Arab River Waters

DouAbul et al. (1988) systematically investigated the occurrence of several organochlorine insecticides (DDT, endrin, aldrin, dieldrin, heptachlor and chlordane) in water (filtered), suspended sediments (retained on filters), bottom sediments and river mussels (*Corbicula fluminea*) at two sites on the Shatt al-Arab, and one site each on the Euphrates and Tigris rivers. HCH isomers were analyzed but were below the detection limits [0.1 ng/L for water and 1 ng/g dry weight for other media] on all samples. Sampling was conducted during July – December 1986. Generally, the low flow months extend from August through November. Higher, but below annual average, discharges occur in July and December. Despite not representing peak

discharges that occur from late March through May [roughly 50% of annual discharge volume], these data are the best available for determining POPs fluxes by the Shatt al-Arab River.

DouAbul et al.'s (1988) survey data are summarized in Table 6.3. The Shatt al-Arab sites (S1 and S2) were located as on Figure 6.1, separated by the city of Basra. The Tigris River site was just below the confluence with the Diyala River near Baghdad, while the Euphrates River site was in the channel draining Hor al-Hammar lake. Six or more samples were obtained at each site. Whole water concentrations reconstructed from the separate aqueous and suspended solids phase data given by DouAbul et al. (1988) are summarized in Table 6.3.

Table 6.3 POPs insecticides (ng/L) in Shatt al-Arab waters, 1986.

	S1	S2	Tigris R (TR)	Euphrates R (ER)
SS	† 9.8 ‡ 9.6–9.9	9.8 9.7–9.9	20.9 19.8–21.8	9.9 9.7–10
ΣDDT	151 74–213	95 32–152	96 48–152	414 219–542
Σdrin	96 64–122	42 21–66	41 21-64	29 13–49
endrin	1.6 1.3–1.8	0.9 0.7–1.1	4.6 3.6–6.3	24 14–36
Σchlordane	72 48–88	39 18–59	52 35–72	62 40–86
heptachlor	11 2.5–20	81 52–93	33 22–56	4 <0.1–7
ΣCHL	114 71–146	137 77–177	108 71–158	92 57–130

† mean; ‡ range.
SS = suspended solids concentration (mg/L).
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD.
Σdrin = aldrin + dieldrin.
Σchlordane = *trans*-chlordane + *cis*-chlordane.
ΣCHL = Σchlordane + *trans*-nonachlor + *cis*-nonachlor + heptachlor; nonachlors estimated from chlordanes as mean proportions in survey data of Iwata et al. (1994).

Data show that Shatt al-Arab waters were highly contaminated with POPs insecticides in 1986; although, not as heavily as the Diyala River in 1984. ΣDDT and Σdrin levels rival those seen recently in India's Ganges River where there has been heavy active usage of DDT and aldrin. The chlordane levels are by far the highest reported for Asian waters. The heptachlor levels are similar to those reported recently for Ganges River waters.

6.3.4 POPs in Shatt al-Arab River Bottom Sediments

Comparing sites S1 and S2, whole water concentrations are, except for heptachlor, lower at site S2 downstream of Basra. Surficial sediment (upper 2 cm) concentrations, summarized in Table 6.4, that the concentration decline between the two sites is results from sedimentation, particularly for DDT, endrin, aldrin and dieldrin.

Heptachlor and chlordane data are somewhat anomalous. In water, chlordane isomers decline in concentration between sites S1 and S2; however, in surficial sediments, they are not detectable at either site, thus suggesting that sedimentation is not occurring. In contrast, heptachlor concentrations increase significantly from site S1 to S2 in both water and sediments. This could only occur if heptachlor were being introduced between sites S1 and S2; however, investigators have consistently claimed that heptachlor has not been used in Iraq. These inconsistencies cannot be resolved without further data collection.

Table 6.4 POPs insecticides (ng/g dry weight) in Shatt al-Arab River surficial (2 cm) sediments.

site	ΣDDT	Σdrin	endrin	heptachlor
S1	24	16	3	<0.1
S2	85	25	16	24

ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD.
Σdrin = aldrin + dieldrin.

6.3.5 POPs in Shatt al-Arab River Mussels

Freshwater mussel data given in Table 6.5 offer added perspective and raise further questions. The data represent approximately 100 mussels from each site, analyzed in 6 or more subsets. Most importantly, note the extraordinarily high levels of DDT,

Table 6.5 POPs insecticides (ng/g wet) in freshwater mussels from the Shatt al-Arab River, 1986.

	S1	S2
ΣDDT	70 48–95	769 664–935
aldrin	26 19–41	72 51–82
dieldrin	24 18–34	493 407–667
aldrin + dieldrin	50 37–75	565 458–749
endrin	166 140–181	540 520–583
trans-chlordane + *cis*-chlordane	11 6–19	46 31–58
fat	0.78 0.75–0.80	0.78 0.76–0.79

† mean; ‡ range.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD.
N.B. Heptachlor was not detected, and heptachlor epoxide was not analyzed due to interference problems with the particular analytical setup used in this study (DouAbul et al., 1988).

dieldrin and endrin observed at site S2 below Basra. The mussels are filter feeders and the data should represent POPs accumulated from aqueous and suspended solids phases. Thus we would expect consistency with both water and surficial sediment samples, rather than the inconsistencies observed with both data sets. There are no immediately obvious explanations for the observed inconsistencies among water, surficial sediment and mussel data, and the issue can only be resolved by further sampling. The endrin flux from the Shatt al-Arab may be somewhat higher than indicated by the water concentration data for sites S1 and S2 in Table 6.4.

6.4 Annual POPs Insecticide Exports from the Shatt al-Arab River

Shatt al-Arab River flux estimates were estimated using the maximum of the S1 and S2 mean concentrations for each contaminant, and a river discharge of 29 km^3. The highest of the two concentrations was used for flux estimation. These are dry season concentrations and it is likely that contaminants lost to surficial sediments between

sites S1 and S2 during the dry season are scoured and transported to the estuary during the wet months. Analysis of Tigris and Euphrates discharge data[10] indicates that 50% of the annual discharge occurs during March–May. While the marshes ("hor") of the lower Tigris and Euphrates would attenuate discharges from upstream, wet season flows of the Shatt al-Arab are likely appreciably higher than dry season flows. Furthermore, the low suspended solids concentrations (9.8 mg/L) of the dry season seem unrealistic for wet season discharges. Milliman et al. (1995) give a suspended sediment flux estimate for the Shatt al-Arab that corresponds to a mean annual suspended sediment concentration of 2,170 mg/L. For this figure to be reliable, there must be significant mass mobilization of sediments during the peak flow months.

No attempt was made to estimate POPs flux contributed from Iran by the Karun River because no data are available. If claims are correct that Iran prohibited use of POPs insecticides except lindane in 1985, the Karun River may contribute lindane and possibly low levels of lingering residues of prohibited insecticides. With no data of any kind, it is impossible to estimate fluxes even crudely.

POPs flux estimates for the Shatt al-Arab River circa 1986 are listed in Table 6.6. DDT and chlordanes appear to have been the leading contaminants transported to the Persian Gulf. From 4 to 5 t total chlordanes (including heptachlor) were discharged annually depending on how the calculation is performed. The endrin flux is remarkably low considering that the endrin load in the Diyala River exceeds that of aldrin + dieldrin in 1984, and the endrin concentration in certain Shatt al-Arab fish (DouAbul et al., 1987) rivaled that of chlordane and DDT in 1984.

In the absence of better data, flux estimates for other contaminants, specifically HCH and PCBs are not attempted. DouAbul et al. (1988) failed to detect α, β and γ-HCH down to measurement limits of 0.1 ng/L. If this condition prevailed generally, the maximum potential load at the detection limits could only be about 30 kg/a; however, the 11 t/a γ-HCH discharged by the Baghdad sewage treatment plant in 1984 (Table 6.1) certainly raises the possibility that there may have been significant, but irregular HCH loadings to the Persian Gulf by the Shatt al-Arab River. Moreover, if Iran is still using γ-HCH as reported by (Burgaz et al., 1995), the Karun River may be delivering a significant γ-HCH load.

Considering PCBs, there is a possibility that the Shatt al-Arab may be delivering a modest PCB load that would originate mainly from urban–industrial areas including Baghdad and Basra in Iraq, and Ahvaz and Abadan in Iran. For the Iraqi Shatt al-Arab (discharge = 29 km^3) or the entire Shatt al-Arab including the Karun River (discharge = 46 km^3) to deliver 1 t/a PCBs would require water concentrations of 35 and 22 ng/L respectively. Neither prospect seems likely; however, the PCB concentrations in the Shatt al-Arab should be determined if possible.

[10] WMO records for Tigris River at Baghdad and Euphrates River at Hindiya; 1965–1972.

Table 6.6 Annual POPs insecticide exports by Shatt al-Arab River (in t except endrin in kg).

	Low	Mean	High
ΣDDT	2.1	4.4	6.2
Σchlordane I	1.4	2.1	2.6
Σchlordane II	2.0	3.0	3.7
heptachlor	1.5	2.3	2.7
ΣCHL	2.2	4.0	5.1
Σdrin	1.9	2.8	3.5
endrin (kg)	39	45	52

ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD.
Σchlordane I = *trans*-chlordane + *cis*-chlordane.
Σchlordane II = Σchlordane I + *trans*-nonachlor + *cis*-nonachlor; nonachlors estimated from chlordanes as mean proportions in survey data of Iwata et al. (1994).
ΣCHL = Σchlordane II + heptachlor.
Σdrin = aldrin + dieldrin.

6.5 DDT and PCBs in Kuwaiti Coastal Waters

As a footnote to discussion of Shatt al-Arab River contamination, Villeneuve et al. (1987) examined 24 archived samples of crab, shrimp, squid and 12 fish species taken in February–March 1979 from Kuwaiti coastal waters. Superficially, Kuwaiti waters would appear to be at high risk from contaminants discharged by the adjacent Shatt al-Arab River. Because data were given in dry weight and no lipid concentrations were given, the results are compared approximately to wet weight concentrations by assuming that the wet to dry weight ratio was 0.3.

On approximate wet weight basis in ng/g, the Kuwaiti sea life concentrations are:

- ΣDDT mean: 1.7, range: <0.1 – 9.8,
- PCBs mean: 3.3, range: <0.4 – 11.4.

The low PCB concentrations are possible if the area has had little or no usage history of PCBs; however, the ΣDDT concentrations are so low that they raise questions regarding the fate of DDT and other POPs insecticides in the Persian Gulf originating from the Shatt al-Arab River. Further investigation is necessary.

Chapter 7 Vietnam

7.1 Hydrology

Long term mean annual discharges for Vietnamese rivers are summarized in Table 7.1. Vietnamese river discharges are clearly dominated by the Mekong River with 62% of total, and the Hong He and Song-Koi at 16% of total each. Together the 3 account for 94% of total discharges by Vietnamese rivers to the sea. As only 65,000 km^2 or 8% of Mekong drainage area lies in Vietnam, Mekong contaminant flux through Vietnam could be largely determined by inputs from upstream countries. Using a net runoff of 867 mm from neighbouring Cai and Dong-Nai watersheds, the annual Mekong discharge volume generated within Vietnam is estimated at 56.3 km^3. Thus, Mekong runoff originating upstream of Vietnam is about 414 km^3 or 55% of the total river discharge of 759 km^3 to the sea from Vietnam.

7.2 POPs in the Vietnamese Environment

Synoptic POPs data for air, surface waters, sediments, soils, and foodstuffs are available for examination. Three 3 air samples show that DDT is used in urban areas across the country [Hanoi, Hue, Ho Chi Minh], most likely for public health purposes. The Hue air sample had one of the highest HCH levels in the survey of Iwata et al. (1994) indicating that spraying was likely in progress nearby at the time of sampling. Chlordane and PCB levels were moderate. Other compartments are examined below.

7.3 POPs in Vietnamese Soils

Thao et al. (1993) surveyed soils in Vietnam, Thailand and Taiwan for DDT, HCH and PCBs. While the survey was opportunistic and not representative in any rigorous statistical sense, the results suggest that Vietnam had the highest HCH and DDT levels in surficial soils, and ranked between Thailand (very clean) and Taiwan (moderately contaminated) for PCBs contamination. The soil samples came from 4 areas: Hanoi (4), Bin Tri Thien (8), Ho Chi Minh (8), and Tay Ninh (2) where the number of samples is given in parentheses. Bin Tri Thien is near Hue about midway between Hanoi and Ho Chi Minh. Tay Ninh is just north of Ho Chi Minh near the Cambodian frontier.

Data are summarized in Table 7.2. The geometric means are the most appropriate statistics for comparisons. The general order of contamination is HCH > PCBs > DDT. Hanoi paddy fields were the most contaminated of the three areas for all three POPs. Barring a few high values, there is little practical difference in contamination levels between Binh Tri Thien and Ho Chi Minh area soils. The two samples from Tay Ninh

Table 7.1 Hydrologic data, Vietnam; after Milliman et al. (1995).

River	Area 10^2 km^2	Discharge km^3	Runoff mm
Mekong	790.0	470.0	595
Hong He	160.0	120.0	750
Song-Koi	158.9 E	120.0	755 E
Dong-Nai	22.0	16.0	727
Da-Rang	12.0	9.1 E	755 E
Cai	2.0	4.8	2400
An-Nong	1.5	3.7	2467
Thach-Han	1.4	2.5	1786
Thu-Bon	3.3	2.5 E	755 E
Tra-Khuc	2.8	2.1 E	755 E
Kone	1.4	1.8	1286
Tam-Giang	1.8	1.1 E	633 E
Total	1157.1	753.6	651

E — denotes estimates derived by extrapolating.

paddy fields show one with very low levels of all three contaminants, while the other had high HCH and the highest level of PCBs observed in the survey. PCBs were observed in Hanoi samples and sporadically about the rest of the country. Thao et al. (1993) theorized that PCBs were released by munitions, weaponry and defoliants deployed in the Indochina war of 1961-71. The site of the highest sample (Tay Ninh) is in an area that endured fierce battles and heavy spraying with defoliants, but is remote from typical sources of PCB contamination. Otherwise, PCBs likely originate from older electrical equipment imported from France, USSR, and the USA.

7.4 POPs in Vietnamese Sediments

The synoptic survey of Iwata et al. (1994) included freshwater sediment data at 18 sites, mainly in the vicinity of Ho Chi Minh in the south. The skewed spatial distribution makes the data difficult to organize. The highest POPs concentrations were observed in sewage canals of Ho Chi Minh where the order of contamination was PCBs (446 ng/g) > DDT (411 ng/g) > chlordanes (17.3 ng/g) > HCH (8.4 ng/g) where mean concentrations standardized to mean DOC concentration of 6.75% are given in parentheses. In contrast to soils, ΣHCH [comprising α, β, and γ isomers] has the lowest concentration, even lower than chlordanes.

Table 7.2 POPs in Vietnamese soils (ng/g dry weight); after Thao et al. (1993).

POP		Mean	† G-mean	Min.	Max.
ΣHCH	Hanoi	139	97.6	31.0	330
	Binh Tri Thien	182	22.2	1.6	1300
	Ho Chi Minh	56	19.3	1.9	280
	Tay Ninh	178	43.8	5.5	350
	Vietnam	128	29.4	1.6	1300
ΣDDT	Hanoi	19.60	8.47	1.30	55.0
	Binh Tri Thien	2.56	1.14	0.09	5.7
	Ho Chi Minh	1.84	1.43	0.23	4.0
	Tay Ninh	1.78	0.71	0.15	3.4
	Vietnam	5.32	1.71	0.09	55.0
PCBs	Hanoi	22.4	17.9	5.50	39.0
	Binh Tri Thien	4.6	3.8	0.61	12.0
	Ho Chi Minh	20.3	5.7	1.50	130.0
	Tay Ninh	160.8	22.6	1.60	320.0
	Vietnam	27.8	6.4	0.61	320.0

† geometric mean.

Iwata et al. (1994) arranged Ho Chi Minh area samples in hydrologic order from upstream of the city through the estuarine system below. This showed that the high concentrations seen in municipal canal sediments decreased downstream through the estuarine system. ΣHCH and Σchlordane returned to upstream levels, but ΣDDT and PCBs remained perceptibly higher than in upstream sediments. A few remote samples from south and central Vietnam had perceptibly lower concentrations of all pollutants, e.g., PCBs (0.35 ng/g), ΣDDT (3 ng/g), Σchlordane (0.2 ng/g), and ΣHCH (0.6 ng/g).

7.5 POPs in Vietnamese Market Fish

Vietnamese foodstuffs surveyed in Tay Ninh and Ho Chi Minh provinces in the south, and Thua Thien province in central Vietnam over 1990-91 (Kannan et al., 1992a) were contaminated by organochlorine pesticides, and PCBs levels were near or higher than in western countries that have discontinued using these chemicals. The survey included 16 pooled samples of various fish species obtained fresh at markets,

Table 7.3 POPs (ng/g lipid weight) in Vietnamese fresh market fish; after Kannan et al. (1992).

	[a] Fish		[b] Prawn	[c] Shellfish	[d] Crab
	Mean	Range			
ΣDDT	2000	300 – 5,900	220	660	980
PCBs	760	220 – 1,500	830	1,400	740
ΣHCH	140	21 – 310	190	250	18
Σdrin	26	<1.16 – 112	35	44	6
Σhept	11	1.6 – 64	19	11	4
HCB	3.7	0.6 – 19	3.8	3.6	2.1

[a] 11 of 16 pooled samples for which fat content was determined.
[b] pooled sample of 4.
[c] pooled sample of 37.
[d] pooled sample of 3.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.
ΣHCH = α-HCH + β-HCH + γ-HCH + δ-HCH.
Σdrin = aldrin + dieldrin.
Σhept = heptachlor + heptachlor epoxide.

plus single pooled samples of prawn, shellfish and crab from the Duyen Hai district in the estuarine zone downstream of Ho Chi Minh city. These market fish samples indicate general freshwater and coastal ecosystem contamination in central and south Vietnam. About 1/2 the Vietnamese diet is supplied by fish; hence, contamination affects humans also.

POPs concentrations in market fish are summarized in Table 7.3. The general order of contamination in Vietnamese fish was DDT > PCB > HCH > aldrin + dieldrin > heptachlor + heptachlor epoxide > HCB. The Vietnamese fish and seafood samples had POPs levels below those in comparable market fish samples from India (Kannan et al., 1992b), but above those from Bangkok, Thailand (Tanabe et al., 1991a).

PCB levels are remarkably high in broad conformance with high levels seen in other environmental compartments. In comparison with several Asian and developed countries, Vietnamese estimated daily dietary intake of PCBs ranked close behind Japan and the former West Germany; and about twice that in Thailand and the USA.

Mean DDT levels were actually higher than in comparable market fish baskets from India (mean 1,100 ng/g fat), but lower than a pooled sample of Ganges River fish [4,700 ng/g fat, (Kannan et al., 1994a)]. Thus, there may be relatively high DDT usage in Vietnam that is not fully evident in currently available environmental data. Estimated DDT daily food intake ranked second only to India in a comparison of several Asian and developed countries (Kannan et al., 1992a). Other data show DDT is the main contaminant in 12 human milk samples from south Vietnam (Schecter et al., 1989).

Otherwise the market fish data show moderate HCH presence, and that heptachlor, aldrin, and dieldrin have been used on limited scales. HCB levels are very low.

7.6 POPs in Vietnamese Surface Waters

POPs data for Vietnamese surface waters are limited to the synoptic survey data of Iwata et al. (1994) for HCH, DDT, PCB and chlordane. This includes 7 water samples (Table 7.4) from paddy field, mangroves and urban canals that show the presence HCHs, DDT, chlordane (ΣCHL), and PCBs. Levels vary from minimal to moderate except for the Ho Chi Minh canal which receives sewage effluents. The canal measurement skews the mean of 7 measurements. The mean and median of the remaining 6 samples are more generally representative of Vietnamese surface waters.

7.7 Annual POPs Exports from Vietnamese rivers

River fluxes were calculated by assigning separate POPs concentrations to (A) Mekong discharge originating in Vietnam plus other rivers discharging from Vietnam, and (B) Mekong river discharge upstream of Vietnam. *Low, medium* and *high* scenarios were calculated using the three concentration summary statistics given in Table 7.4 for Vietnamese surface water discharges.

The chief difficulty was assigning *low, medium* and *high* concentrations to the 414 km^3 of Mekong River discharge originating upstream of Vietnam. Mekong Basin POPs usage figures are unavailable but, one upstream country (China) has severely restricted DDT and HCH usage, while three remaining upstream countries (Cambodia, Laos, Myanmar) would likely have very low pesticide usage. Thailand likely uses DDT for malaria control in its 182,000 km^2 Mekong drainage area, but usage figures were unavailable. PCB presence should be low to nil in all upstream countries.

Concentration scenarios for upstream Mekong River drainage were ultimately selected by aggregating surface water data from Iwata et al. (1994) for several modestly contaminated countries of the region (Thailand, Vietnam, Solomon Islands, Indonesia) and choosing the 10th and 33rd percentiles respectively for the low and high scenarios, and averaging these to define the medium scenario. These concentrations are listed in Table 7.5, and the Vietnamese mass fluxes in Table 7.6. The order of mass export is ΣHCH > PCBs > ΣDDT > ΣCHL. This agrees with the water and soils data; however, in market fish, ΣHCH ranks after ΣDDT and PCBs. DDT exports may be higher due to influx from upstream Mekong River countries that is higher than estimated here. Monitoring data are needed for the lower Mekong River.

Table 7.4 POPs (ng/L) in Vietnamese surface waters; after Iwata et al. (1994).

Location		ΣHCH	ΣDDT	ΣCHL	PCBs
Hanoi	urban area	3.2	0.7	0.04	0.8
Thua Bien	paddy field	18.4	0.3	0.21	1.2
Hue	sub-urban area	1.9	1.1	0.07	1.6
My Hung Village, Cu Chi	paddy field	9.5	0.6	0.16	1.9
Duyen Hai, Ho Chi Minh	paddy field & mangroves	5.2	0.5	0.13	0.6
Ferry pier, Ho Chi Minh	paddy field & mangroves	30.5	4.6	0.55	2.7
Ho Chi Minh	municipal sewage canal	18.9	25.2	1.02	8.0
mean		12.5	4.7	0.31	2.4
mean ‡		11.4	1.3	0.19	1.5
geometric mean ‡		7.4	0.8	0.14	1.3

ΣHCH = α-HCH + β-HCH + γ-HCH.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.
‡ excluding sewage canal measurement.

Table 7.5 POPs concentrations (pg/L) assigned to Mekong River inflows to Vietnam.

scenario	ΣHCH	ΣDDT	ΣCHL	PCBs
low	184	223	14.3	50
medium	906	258	42.8	250
high	1627	292	71.3	450

Table 7.6 Annual POPs exports by Vietnamese rivers (kg/a except HCH in t/a).

Scenario	ΣHCH	ΣDDT	ΣCHL	PCBs
low	2.64	381	54	468
medium	3.30	562	82	621
high	4.99	1,754	137	787

Chapter 8 Malaysia

Geographically, Malaysia (329,750 km^2) includes peninsular Malaysia (128,000 km^2) and Malaysian Borneo (201,700 km^2) which contains the provinces of Sarawak (125,200 km^2) and Sabah (76,500 km^2). Currently neither hydrologic nor POPs data are readily available for Malaysian Borneo; hence, this chapter focuses on POPs occurrence in peninsular Malaysia.

8.1 Hydrology

Mean annual rainfall over peninsular Malaysia ranges from 1.7–4.2 m with an overall mean of about 2.6 m. About 2/3 of 64 rain gauge sites had mean annual rainfall of 2–3 m, and 11 had rainfalls exceeding 3 m.

Available runoff data are listed in Table 8.1. The Pahang, Perak and Kelantan are the three largest watersheds of peninsular Malaysia. The remainder of the peninsula comprises 45 small watersheds [see Fig. 1 in Tan and Chong (1993)]. The prorata estimate of total annual surface runoff from peninsular Malaysia is 122 km^3. From 65-75% of surface waters flow eastward to the South China Sea.

8.2 Peninsular Malaysian POPs Usage

There are few readily available published reports on POPs usage in Malaysia. A 1981 survey of organochlorine pesticide residues in water, sediments and fish from paddies used jointly for rice and food fish cultivation (Meier et al., 1983) found modest contamination by DDT, HCH, aldrin, dieldrin and chlordane. Dieldrin and chlordane had the highest residues in paddy fish. It is clear from recent reports (Tan et al., 1991; Tan and Vijayaletchumy, 1994) that the major organochlorine insecticides (DDT, HCH, heptachlor, dieldrin) have been used recently in Malaysia. As of 1991, use of lindane and dieldrin was still permitted (Tan and Vijayaletchumy, 1994), but DDT usage may have been restricted. No information is available concerning Malaysian PCB stocks.

8.3 POPs in Peninsular Malaysian Coastal Waters

Hungspreugs (1988) gave some fish and shellfish POPs data from coastal waters circa 1987 that showed PCBs at 20–44 ng/g (wet weight), DDT at 3–50 ng/g, lindane at 1–12 ng/g and dieldrin at <1–5 ng/g. Lindane, PCB and DDT levels indicate modest contamination, while dieldrin levels seem low.

Table 8.1 Hydrologic data: peninsular Malaysia.

	Area 10^3 km^2	Discharge km^3	Runoff mm
Pahang †	26.00	21	808
Perak	13.00	12	923
Kelantan	12.00	15	1,250
Selangor ‡	1.45	2	1,380
Subtotal	52.45	50	953
total peninsular Malaysia	128.02	122	953

† Pahang, Perak, and Kelantan from Milliman et al. (1995).
‡ from McMahon et al. (1992).

8.4 POPs in Peninsular Malaysian Surface Waters

Results of synoptic stream surveys in 1989 (Table 8.2) and 1990-91 (Table 8.3) give sufficient data to estimate peninsular river fluxes of the main POPs insecticides. The 1989 survey gave data for 32 samples collected in streams flowing westward to the Straits of Malacca, and 2 samples in streams flowing eastward to the South China Sea; while the 1990-91 survey gave data for 16 westward flowing and 9 eastward flowing streams. Comparative analysis between the two data sets is complicated by inconsistent lists of measured contaminants and sites between the two surveys.

Together, the surveys indicate that DDT, HCH, heptachlor, aldrin, dieldrin, and endrin were all used in peninsular Malaysia during the late 1980s and early 1990s. Contaminant levels observed in the 1990-91 survey are perceptibly higher than observed in the 1989 survey, and insecticide levels in westward flowing streams are perceptibly more contaminated than eastward flowing streams. Because these are spatially extensive synoptic surveys, differences between surveys and between east-west streams may be artefacts of temporally sparse sampling that fails to adequately represent the annual usage cycles.

Because only *p,p'*-DDT and *p,p'*-DDE were analyzed, ΣDDT given in Tables 8.2–8.3 will be too low. The mean ratio of *p,p'*-DDT to ΣDDT = *p,p'*-DDT + *p,p'*-DDE for the 1989 samples is about 0.91, suggesting that virtually all samples represent fresh DDT inputs. Using Iwata et al.'s (1994) survey data as a guide, *o,p'*-DDT and *p,p'*-DDD, had they been measured, would have approximately doubled ΣDDT concentrations.

Table 8.2 POPs insecticides in peninsular rivers, 1989; after Tan et al. (1991).

	unit	Mean	† G-mean	Min.	Max.
ΣDDT [a]	ng/L	13.93	9.86	1.44	69.39
heptachlor	ng/L	1.39	1.19	<0.29	3.38
β-HCH	pg/L	125	3	<10	770
γ-HCH	pg/L	139	4	<40	1,940
β-HCH + γ-HCH	pg/L	265	40	<10	1,940
ΣHCH [b]	pg/L	458	70	<17	3,360
aldrin	pg/L	34	2	<20	520
dieldrin	pg/L	62	35	<60	250
Σdrin [c]	pg/L	96	53	<60	630
endrin	pg/L	511	136	<140	3,320

[a] ΣDDT = *p,p'*-DDT + *p,p'*-DDE; [b] approximate sum of α, β and γ isomers; [c] Σdrin = aldrin + dieldrin.
† geometric mean.

Table 8.3 POPs insecticide in peninsular rivers, 1990-91; after Tan and Vijayaletchumy (1994).

	unit	Mean	† G-mean	Min.	Max.
A. West Coast					
ΣDDT [a]	ng/L	65.1	49.0	7.8	190
ΣHCH [b]	ng/L	56.9	7.3	<0.2	320
heptachlor	ng/L	26.0	9.8	<0.1	120
dieldrin	ng/L	12.0	4.6	<1.3	47
A. East Coast					
ΣDDT [a]	ng/L	16.0	1.0	<0.1	88
ΣHCH [b]	ng/L	2.6	1.3	<0.2	7.7
heptachlor	ng/L	3.9	2.0	0.3	13
dieldrin	ng/L	1.6	0.8	<1.3	8.8

[a] ΣDDT = *p,p'*-DDT + *p,p'*-DDE; [b] approximate sum of α, β and γ isomers.
† geometric mean.

The survey data of Iwata et al. (1994) included single December 1991 water and sediment samples from the Selangor River that offer a small added perspective. The water sample yielded a ΣHCH concentration of 1.9 μg/L that appeared to represent fresh lindane inputs. The ΣHCH concentration was higher than any seen in the two surveys of Tables 8.2– 8.3; and shows that much higher levels of HCH might be observed by more temporally representative sampling. The Selangor sample had a modest ΣDDT level of 1.7 ng/L and a Σchlordane level of 2.1 ng/L comprising over 50% *cis*-chlordane and *cis*-nonachlor which suggests that technical chlordane has been in use as well as heptachlor. Lastly, the Selangor sample had a PCBs concentration of 450 pg/L which is a reasonable concentration to use as representative of general PCBs background levels in peninsular Malaysian surface waters.

8.5 Annual POPs Exports from Peninsular Malaysian rivers

Estimated annual POPs riverine exports from peninsular Malaysia to the sea are given in Table 8.4. Three scenarios are considered: (A) *very low* using 1989 concentrations, (B) *low* using 1990-91 geometric mean concentrations, and (B) *high* using 1990-91 mean concentrations. Scenarios B and C also took into account observed concentration differences between west- and eastward flowing streams by splitting the total discharge from peninsular streams into approximately 35% west coast and 65% east coast drainage. For PCBs and chlordane, flux estimates must be regarded cautiously as only the single Selangor River sample of Iwata et al. (1994) was available.

Generally, ΣDDT shows the greatest mass flux. The true figure may be somewhat higher depending on the significance of the unmeasured *o,p'*-DDT and *p,p'*-DDD isomers. If the full complement of isomers had been observed in the average proportions suggested by Iwata et al.'s (1994) survey data, true annual ΣDDT flux would range from 3.4–7.8 t/a.

After DDT, the next most prevalent POPs insecticide may be either HCH or heptachlor. Had the full complement of heptachlor and chlordane isomers been measured, the total may well have exceeded that of ΣHCH. The total of aldrin and dieldrin appears to be lower, but of a similar order to HCH and heptachlor/chlordane. Both endrin and PCBs are present at low levels, and both require further monitoring to clarify their significance.

Currently, no data are available for Sarawak and Sabah on Malaysian Borneo. If DDT and other POPs insecticides are being used there, the total exports generated by Malaysia could be much larger.

Table 8.4 Annual POPs exports (kg) by peninsular Malaysian rivers.

	1989	1990-91 Low	1990-91 High
ΣDDT [a]	1,700	2,088	3,903
ΣDDT [b]	3,499	4,177	7,807
ΣHCH	56	443	4,839
heptachlor	170	576	1,422
ΣCHL	—	† 256	—
aldrin	4	—	—
dieldrin	8	256	640
Σdrin	12	‡ 397	‡ 991
endrin	52	—	—
PCBs	—	† 55	—

† based on single Selangor River sample in Iwata et al. (1994).
‡ Σdrin prorated as the aldrin / dieldrin composition observed in 1989 samples.
[a] ΣDDT = *p,p'*-DDT + *p,p'*-DDE; in survey data of Iwata et al. (1994).
[b] ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD; prorated as 2 x ΣDDT [a] according to average composition observed in survey data of Iwata et al. (1994).
ΣHCH = α-HCH + β-HCH + γ-HCH.
Σdrin = aldrin + dieldrin.
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

Chapter 9 Thailand

9.1 Hydrology

Surface water discharges from Thai watersheds draining directly to the sea are given in Table 9.1. The Chao Phraya covers about one third of the country. Two drainage areas are commonly given for the Chao Phraya: 160,000 km^2 (Takeuchi, 1993), and 177,000 km^2 (Hungspreugs et al., 1990). As it was unclear which area was correct, Table 9.1 gives both with discharge estimate (A) from Milliman et al. (1995), and estimate (B) determined by prorating 1956-86 discharge data from the Nakhon Sawan gauge (Takeuchi, 1993) to 177,000 km^2. The discrepancy may result from separate accounting of the Ta Chin River distributary that leads off to the west below Nakhon Sawan. The difference between the two Chao Phraya discharge estimates does not greatly affect riverine POPs flux estimates, and the larger value was adapted herein. Considering that annual rainfall over the mainland ranges from 1–1.5 m, unit runoff of ca. 200 mm for the Chao Phraya seems low. Takeuchi (1993) attributes long term declining runoff to major reservoir construction and dry season irrigation that enhance evapotranspiration losses.

The combined annual discharges of the Chao Phraya, Mae Klong and Bang Pakong are 53–60 km^3, but there remains significant ungauged direct drainage from Thailand to the sea. Of Thailand's total land area of approximately 512,000 km^2, about 182,000 km^2 drains to the Mekong River that debouches to the sea through Vietnam (Piper et al., 1991). After taking account of Mekong, Chao Phraya, Mae Klong and Bang Pakong drainage, there remains roughly 105,000 km^2 land area of which it was very crudely estimated that 15% drained to Burma and about 89.4 km^2 drained directly to the sea. If the unit runoff of 481 mm observed for the Mae Klong watershed applies, then this ungauged area generates 43 km^3 discharge annually. This is a very rough estimate that should be used with caution. Over the mainland, the runoff of 481 mm used to extrapolate discharges from ungauged areas represents a reasonable 35% of incoming rainfall. However, over peninsular Thailand, annual rainfall can range from 1–4 m with a mean of about 2 m; hence, the specific runoff of 481 mm may be too low. Mean unit runoff over peninsular Malaysia immediately to the south is about 940 mm suggesting that ungauged drainage from peninsular Thailand may contribute similarly. Thus total ungauged areas draining directly to the sea may generate as much as 50-60 km^3 discharge annually rather than the 43 km^3 estimate used in this study.

9.2 POPs insecticide use in Thailand

Historically, Thailand has used the principal POPs insecticides, most of which are now banned or restricted. DDT was banned for agricultural purposes in 1983, but public health usage continues. Table 9.2 gives 1987-90 antimalarial usage in the Thai provinces draining to the Gulf of Thailand. Average usage was about 90 t/a of which

Table 9.1 Hydrologic data, Thailand; after Milliman et al. (1995).

	Area 10^2 km^2	Discharge km^3	Runoff mm
Chao Phraya (A)	166	30.0	188
Chao Phraya (B)	177	36.6	207
Mae Klong	27	13.0	481
Bang Pakong †	‡ 21	9.9	481
ungauged ††	89	43.0	481
Total (A)	297	95.9	323
Total (B)	314	102.4	326

† from (Hungspreugs et al., 1990). ‡ estimated area using runoff of 481 mm. †† estimate see text.

Table 9.2 Antimalarial DDT usage (t) in Thai provinces draining to Gulf of Thailand, 1987-90.

Province	† No.	1987	1988	1989	1990	Sum	Rank	‡ Trend
Mainland								
Trad	I	4.0	16.7	5.1	6.1	31.9	6	0
Chachaengsao	II	5.4	16.3	7.7	10.1	39.4	5	0
Samutsakorn	III	0	0	0	0	0	7	0
Samutsongkram	IV	0	0	0	0	0	7	0
Peninsula								
Petchburi and	V	11.3	14.4	16.7	15.1	57.6	4	+
Prachaubkirihan	VI							
Chumporn	VII	18.0	19.6	20.6	33.4	91.6	1	+
Nakornsrithammarat	VIII	8.7	11.8	28.8	18.7	68.0	3	+
Pattani	IX	22.0	7.8	12.3	28.4	70.6	2	0
sums		69.4	86.8	91.2	111.7	359.0		+

† Numerals correspond to mussel sampling sites (see Section 9.7); ‡ "+" indicates net increasing trend from 1987-90; data from Ruangwises et al. (1994).

about 80% was used in peninsular Thailand. Figures for Thai Mekong River drainage were unavailable, but if the usage rates are similar, there could be up to 50 t/a or more DDT used there for public health. Crude HCH was banned for agricultural use in 1980, and for public health use in 1987. Lindane use is permitted in small quantities for selected purposes (Siriwong et al. 1991). Heptachlor was banned in 1988. Aldrin and

dieldrin were used until 1988 (Tabucanon et al. 1992) for termite control (Siriwong et al. 1991). Over 1976-87, 250 and 270 t were imported respectively (Tabucanon et al. 1992). In 1987, usage was 43,264 kg and 2,028 kg respectively (Tanabe et al. 1991a). Endrin usage was prohibited in 1981 (Siriwong et al. 1991).

9.3 PCBs in Thailand

In Thailand, the major stocks of PCBs are precisely known in terms of the numbers of transformers and capacitors imported by the country's three leading electrical authorities (Watanabe et al., 1996). One third of this equipment has been taken out of service pending disposal. The precise mass of PCBs was not given, but should be obtainable from authorities.

From August 1991 to December 1992, Watanabe et al. (1996) monitored air concentrations of PCBs near a storage facility for PCB containing equipment located in suburban Bangkok. Results show that PCBs are being lost by volatilization from this nominally "closed system" electrical equipment, and that losses likely increase with air temperature. No attempt was made to estimate loss rates; however, such figures would be useful. The authors suggest that dumping of PCBs in tropical environments will lead to large and rapid loss of PCBs to the atmosphere, and general contribution to global contamination.

9.4 POPs in Thai Soils

Thao et al. (1993) surveyed soils in Vietnam, Thailand and Taiwan for DDT, HCH and PCBs. Thai soils (Table 9.3) were the least contaminated of the three countries for all three variables. The 15 surficial soil samples came mainly from 3 areas in the north (Cheng Mai, Chiengrai, Lampang), and two areas of the south (Bangkok, Ayutthaya). Two of the 15 samples, both from the north, had high ΣHCH [34 and 98 ng/g] indicating recent usage, but otherwise levels were very low. Low levels were evident also for ΣDDT and PCBs, for which the single Bangkok sample had the highest value.

Table 9.3 POPs in Thai soils (ng/g dry weight); after Thao et al. (1993).

	Mean	† G-mean	Min.	Max.
ΣHCH	10.71	2.92	0.5	98.0
ΣHCH ‡	2.20	1.85	0.5	4.9
ΣDDT	0.43	0.31	0.07	1.6
PCBs	2.99	2.76	1.4	6.2

† geometric mean; ‡ 2 high values (34, 98) removed.

9.5 POPs in Thai Sediments

Four sediment samples collected from Bangkok area canals January 1990 are reported by Iwata et al. (1994). The same data are reported by Tabucanon et al. (1992) with minor differences. In addition to the four usual DDT isomers, the latter report includes *o,p'*-DDE and *o,p'*-DDD that increase ΣDDT by about 10%, and 2 of 4 PCB results are higher than given by Iwata et al. (1994).

The Bangkok area sediment data (Table 9.4) do show that DDT was relatively high, driven up by what appeared to be fresh inputs at two sites along one canal, and high metabolized DDT levels at another. Mean PCB concentrations were high due to 2 very high sample results. Likewise, mean total chlordanes are high thanks to one very high sample. Taken together, these data show that Bangkok canals can be very strong localized sources for several leading POPs.

Table 9.4 POPs in Bangkok area sediments (ng/g dry weight); after Tabucanon et al. (1992).

	Mean	† G-mean	Min.	Max.
ΣHCH	1.8	1.5	0.5	3.1
ΣDDT	106	59	5.5	205
ΣCHL	57	12	1.4	202
PCBs	282	85	11	700

† geometric mean; ΣHCH = α-HCH + β-HCH + γ-HCH;
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *o,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD;
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

9.6 POPs in Thai Market Fish

Tanabe et al. (1991a) conducted a survey of Bangkok foodstuffs over December 1989 – January 1990 that included 3 specimens each of 5 food fish species [finbarb, catfish, snake-head, Nile tilapia, climbing perch] obtained fresh from fish markets. As in Vietnam (Kannan et al., 1992a), fish likely constitute a major fraction (ca. 50% in Vietnam) of the Thai diet. The aggregated results (Table 9.5)[11] indicate general POPs presence in Bangkok area aquatic ecosystems. The broad order of contamination is ΣDDT > aldrin + dieldrin > PCBs > ΣHCH > Σheptachlor > HCB. The relatively high

[11] Data are derived from Table II Tanabe et al. (1991) giving fat weight concentrations by species; however, several table entries contained obvious discrepancies, in which case, fat weight concentrations were approximated by wet concentrations normalized to mean species lipid content given by Tanabe et al. in Table I.

Table 9.5 POPs in Bangkok market fish (ng/g fat weight); after Tanabe et al. (1991).

	mean	† G-mean	min.	max.
PCBs	50	32	7	240
ΣHCH	29	15	6	220
ΣDDT	120	109	28	195
aldrin	26	5	0.5	275
dieldrin	86	55	9	285
Σdrin	112	92	—	—
heptachlor	1.3	0.4	<0.05	14
heptachlor epox.	7.1	3.9	0.3	21
Σhep	8.4	5.7	—	—
HCB	3.7	1.9	0.3	26

† geometric mean.
ΣHCH = α-HCH + β-HCH + γ-HCH; ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD;
Σdrin = aldrin + dieldrin; Σhep = heptachlor + heptachlor epoxide.

drin levels should have begun to decline in the wake of the 1988 ban. Heptachlor levels were very low reflecting limited usage. Regionally, Bangkok area fish generally contain lower levels of POPs contamination than fish from elsewhere. Kannan et al. (1992a) compared average daily intakes (ADIs) for Thailand, Vietnam, India, and several developed countries, showing that except for aldrin + dieldrin, ADIs for most POPs are comparable to or marginally greater than levels in developed countries.

9.7 POPs in Gulf of Thailand Mussels

Surveys of organochlorine insecticides in Gulf of Thailand mussels (Mensaveta and Cheevaparanapiwat, 1981; Ruangwises et al., 1994; Siriwong et al., 1991) have shown relatively low contamination. The most recent surveys from 1989 and 1991 are contrasted in Table 9.6 for 9 sites along the Thai coast from the frontiers with peninsular Malaysia to Cambodia. HCH and heptachlor, which were detected only sporadically, are not shown. The order of contamination in both surveys was DDT > aldrin + dieldrin > HCB > HCH ≥ heptachlor.

The Thai mussel data show that contamination by the leading POPs (DDT, drins) is widely distributed along the coast. Off the peninsular and the easternmost mainland coasts, levels are generally higher than in the estuaries of the main rivers (Mae Klong, Ta Chin, Bang Pakong), and for both surveys, contamination was consistently higher off peninsular Thailand than off the mainland. Moreover, DDT contamination has increased at 4 of 5 peninsular sites in accordance with the DDT usage trends evident in Table 9.2. Thus, POPs mass exports estimated solely from Chao Phraya water

Table 9.6 POPs in Gulf of Thailand mussels (ng/g fat).

Site	Province	ΣDDT			Σdrin			HCB		
		1989	1991	† Δ	1989	1991	Δ	1989	1991	Δ
A. Peninsula										
Laem Tachi	‡ IX	255	196	-59	98	35	-63	10.0	4.8	-5.3
Paknakom	VIII	90	335	245	66	63	-3	5.8	1.9	-3.9
Tungka	VII	135	297	162	76	46	-30	5.6	3.1	-2.6
Ko Luk	VI	183	195	12	119	72	-47	4.7	4.1	-0.6
Ban Laem	V	67	155	88	83	36	-48	7.4	1.8	-5.6
Peninsula mean		146	236	90	88	50	-38	6.7	3.1	3.6
B. Mainland										
Mae Klong estuary	IV	97	128	31	52	34	-17	1.5	13.3	11.8
Ta Chin estuary	II	99	76	-24	50	29	-21	1.9	1.7	-0.2
Bang Pakong	II	145	112	-33	65	37	-28	3.9	2.1	-1.8
Ban Laem Hin	I	172	156	-16	55	53	-2	6.2	2.8	-3.4
Mainland mean		128	118	-10	55	38	-17	3.4	5.0	1.6
Grand Mean		138	183	45	74	45	-29	5.2	3.9	-1.3

— 1989 data from Siriwong et al. (1991); 1991 data from Ruangwises et al. (1994).
† Δ = 1991 mean - 1989 mean; ‡ numerals correspond to Provinces listed in Table 9.2;
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *o,p'*-DDE + *p,p'*-DDD + *o,p'*-DDD; Σdrin = aldrin + dieldrin.

concentration data, may be somewhat conservative. The maximum DDT concentrations approach levels seen at the least contaminated sites in a south India coastal survey (Ramesh et al., 1990b).

The mussel data also show definite declining trends in aldrin + dieldrin concentrations following the 1988 ban. The relatively proportions of aldrin and dieldrin also changed — from 80% aldrin / 20 % dieldrin in 1989, to 73% aldrin / 27 % dieldrin in 1991 — indicating the ongoing metabolism of lingering aldrin residues to dieldrin. HCB levels in Gulf of Thailand mussels are thought to be low, but there is little data with which to compare them. Except for the anomalous 1991 data observed in the Mae Klong estuary, HCB levels have declined at all other sites.

9.8 POPs in Thai Surface Waters

Organochlorine data for Thai rivers are available in reports by Onodera and Tabucanon (1986) and Tabucanon et al. (1992). The former paper covering 1982-84 was unavailable for present review. The latter paper summarizes synoptic data for 70 samples at 30 sites along the lower 376 km of the Chao Phraya River gathered on six dates from November 1988 to April 1991. Samples were analyzed for HCH, DDT, aldrin and dieldrin. In addition, there are synoptic data (Iwata et al., 1994) for 5 Bangkok area sites for January, 1990.

The 1988-91 summary data from Tabucanon et al. (1992) are difficult to use in the manner they were presented as the original data contained numerous non-detections. Table 9.7 summarizes the detection frequencies observed on each of the 6 sampling dates. The detection frequencies for the 3 familial POPs classes (HCHs, DDTs, and drins = aldrin + dieldrin) are the average of contributions by component isomers.

Table 9.7 data suggest that Chao Phraya levels of HCHs and DDTs have declined significantly after restrictions imposed in 1988. Data further suggest that sporadically high levels may continue to be observed for some time. The detection frequency of the drins does not support a decline; however, the median concentrations reported by Tabucanon et al. (1992) do suggest that a decline is in progress. More systematic data collection over the next 5 years is needed to clarify the situation.

Table 9.8 presents Bangkok area water sample data from the survey of Iwata et al. (1994) that includes chlordane (ΣCHL) and PCBs. The upstream sample shows high ΣHCH that likely indicates recent application in the vicinity, and probably waters that were not a well-mixed, representative sample of the main Chao Phraya discharge. The next two samples indicate that Bangkok area canals are more contaminated.

Table 9.7 POPs insecticide detection frequencies in the lower Chao Phraya River, 1988-91; after Tabucanon et al. (1992).

Date	Nov-88	May-89	May-90	Oct-90	Jan-91	Apr-91
n	10	10	17	9	12	12
HCHs	87	83	27	7	31	31
DDTs	90	77	57	11	0	14
drins	85	90	100	67	92	80

Table 9.8 POPs in lower Chao Phraya River and Bangkok canal waters (ng/L); from Iwata et al. (1994).

Water body	Locale	ΣHCH	ΣDDT	ΣCHL	PCBs
Chao Phraya R (above Bangkok)	paddy field	75.40	0.35	0.21	<0.24
Klong Sam-rong Canal	industrial canal	1.11	2.55	1.34	3.20
Krung Kasem Pumping Station	municipal sewage canal	1.04	0.23	0.43	4.40
Chao Phraya R	urban area	—	0.31	0.44	0.58
Chao Phraya R estuary	urban area	0.18	0.23	0.18	0.45

9.9 Annual POPs Exports from Thai rivers

Direct Thai riverine exports are shown in Table 9.9 for 1988-89, and 1990-91. The Chao Phraya load was calculated with flow estimate (b) of Table 9.1. Loads are low compared to certain other countries of the region. The order of exports is aldrin + dieldrin > PCBs > chlordanes > DDT > HCH. Exports of HCH, DDT and aldrin+dieldrin seem to have declined since the restrictions imposed in 1988. If the mussel data (Section 9.7) are a reliable indicator, DDT loads from the peninsula may have actually increased. Further monitoring data from the lower Chao Phraya and other Thai streams entering the Gulf of Thailand, particularly from peninsular Thailand, would help confirm the apparent trends in the estimates of Table 9.8.

Table 9.9 Annual POPs exports by Thai rivers to coastal seas; (A) Chao Phraya, (B) Thailand.

	1988-89†			1990-91‡		
	Concentration ng/L	Flux (A)	(B)	Concentration ng/L	Flux (A)	(B)
ΣHCH	1.1	40	113	0.30	11	31
ΣDDT	1.3	48	133	0.31	11	32
ΣCHL	—	—	—	0.44	16	45
PCBs	—	—	—	0.58	21	59
aldrin	7.2	263	738	1.50	55	154
dieldrin	6.8	249	697	0.74	27	76
Σdrin	13.0	512	1,435	2.24	82	230

†1988–1989 based on median concentrations from Tabucanon et al. (1992).
‡1990–1991 based on median concentrations from Tabucanon et al. (1992) for aldrin and dieldrin; and mean of samples 4 and 5 from Table 9.7 for others; PCBs and total chlordane (ΣCHL) based on Chao Phraya data from Iwata et al. (1994) [see Table 9.8].

Chapter 10 Indonesia

The Indonesian archipelago of more than 13,000 islands (6,000 inhabited) sprawls over a wide swath of the equatorial eastern Indian and western Pacific oceans. With high rainfall and runoff, a burgeoning population of about 200 million, and accelerating development, the potential for pollution of aquatic ecosystems is significant.

10.1 Hydrology

The Indonesian climate is generally humid. Mean annual rainfall averages about 2.4 m across the archipelago, and most areas receive annual rainfall between 1.5 and 4 m. Annual rainfall can vary from 800 mm in a few relatively dry areas (Timor) to 6 m or more on uplands of high relief islands (Sumatra, Java, Sulawesi, Irian Jaya).

10.1.1 Dubious Discharge Data

The poor quality of available Indonesian hydrologic data is a serious obstacle to estimating contaminant mass fluxes. The LOICZ GLORI data file (Milliman et al., 1995) identifies 32 Indonesian rivers; however, mean annual discharges are supplied for only 11 mostly Javanese rivers. Calculation of specific unit runoff by normalizing the annual discharge volume to the reported drainage areas reveals that 8 rivers have unit runoff of 4.4–22.5 m. On Java, mean annual rainfall averages about 2.3 m, ranges from about 1.4–4.1 m over most of the island, and may exceed 6 m at some high elevations. Thus, according to available data, most Javanese rivers have runoff well in excess of rainfall. Whether the errors lie with the reported discharges, the drainage areas, or both is unknown. The severity of these errors renders it infeasible to prorate available discharge data to ungauged basins.

As a provisional work-around for developing river flux estimates, mean annual surface water discharges from the large Indonesian islands were estimated as 50% of mean annual rainfall. Hydrologic studies have found that in the humid tropics, mean runoff averaged about 50% of rainfall (McMahon et al., 1992). Using the 50% runoff assumption, surface runoff from Indonesia would be about 2300 km^3 of which about 80% would originate from Kalimantan, Irian Jaya, Sumatra and Sulawesi. For Java, the only island with any POPs aquatic data, the mean annual water discharge at 50% runoff is about 146 km^3. Allowing for potential variability in the runoff coefficient, say 35%–70%, mean annual water discharge from Java is likely in the range 102–205 km^3.

10.2 POPs Usage in Indonesia

10.2.1 POPs Insecticides

With >70% of Indonesia's population, currently about 200 million, engaged in agriculture, Indonesia has been a major consumer of pesticides. After severe rice crop failures in the mid 1980s due to pest resistance, in 1986 the Indonesian government prohibited the use of 57 pesticides in rice cultivation and promoted a national *integrated pest management* [IPM] policy aimed at reducing pesticide use. A recent report (Stackhouse, 1996) claims that there are now 10,000 IPM field schools in the country, and that chemical pesticide use in rice agriculture has declined. However, other informal reports (PANNA, 1995a) offer contradictory claims that Indonesia's pesticide consumption jumped from 28 Kt in 1991 to 41 Kt in 1993, and that organochlorine use has increased in rice cultivation. The current situation remains unclear.

The extent of public health usage of POPs insecticides in Indonesia is not known.

10.2.1 PCBs

No data are available concerning PCB stocks held by Indonesia. Environmental data suggest that PCB stocks may be significant in more densely populated and developed areas.

10.3 POPs in the Indonesian Environment

10.3.1 Ciliwung River

The Ciliwung River descends from near Bogor in west Java, down through Jakarta to Jakarta Bay. No drainage area or discharge data are available. Mean annual rainfall increases from about 1.8 m in Jakarta to about 4.2 m inland near Bogor. The synoptic surveys of Iwata et al. (1994) included 3 water and 4 sediment samples from the Ciliwung River in the cities of Jakarta and Bogor, and 5 fish samples from Bogor were taken by Kannan et al. (1995). Data are summarized in Table 10.1.

In regional comparisons [Chapter 2], Ciliwung River samples show moderate contamination. HCH and PCBs have high enough water concentrations to be of marginal concern. The isomer profile indicates that crude HCH was being used in late 1991. DDT levels in water were low and metabolized DDT dominated the isomer profile. Jakarta sediments had perceptibly higher PCBs, DDT and chlordane than Bogor sediments. In regional comparisons, Bogor fish ranked low except for DDT content that ranked below Shatt al-Arab River fish and ahead of Indian market fish suggesting that DDT occurrence may be more prevalent than indicated by the limited water and sediment data available.

Table 10.1 POPs in Ciliwung River water (W, pg/L), sediments (S, ng/g) and fish (F, ng/g wet weight).

		Mean	Min.	Max.
PCBs	W	1,260	380	2,100
	S	111	6	220
	F †	2.6	2.0	3.8
ΣHCH	W	10,050	3,070	21,900
	S	0.067	0.035	0.100
	F	0.73	0.06	1.4
ΣDDT	W	227	194	265
	S	21	3.4	42.1
	F	28	0.66	76
ΣCHL	W	191	71	262
	S	11.6	4.2	37.6
	F ‡	0.45	0.24	0.69
Σdrin	F	1.2	<0.1	2.3
HCB	F	0.05	0.01	0.08

ΣHCH = α-HCH + β-HCH + γ-HCH; † fish analysis includes δ-HCH.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor; ‡ fish analysis includes oxychlordane; Σdrin = aldrin + dieldrin.

10.3.2 Citarum River

Some additional organochlorine pesticide data are given for the Citarum River by Djuangsih and Salim (1994). The Citarum (also Tjitarum) River rises about 40 km south of the city of Bandung in West Java province, and runs north about 225 km to enter the Java Sea east of Jakarta. Mean annual rainfall varies from about 2 m near Bandung to about 1.5 m near the Java Sea.

Discharge and drainage area data for the Citarum River (Table 10.2) demonstrate the poor quality of hydrologic data available for Indonesia. The discharge reported by Milliman et al. is far too high, while the basin area given by Djuangsih and Salim seems far too low. In all cases, derived annual runoff is 5-10 fold greater than annual rainfall.

Djuangsih and Salim state that DDT ranged from "undetectable to 14 µg/L". There is no indication when or where samples were collected, what DDT species were measured, nor what concentrations correspond to "undetectable". Very high DDT usage would be required to explain water concentrations of 14 µg/L. The only country in the region reporting such levels is India where DDT is used at high rates.

Djuangsih and Salim also state that HCB ranged from "37.2 to 56.2 µg/L". These are implausibly high levels that would require either extraordinarily high usage of HCB

Table 10.2 Hydrologic data for Citarum River.

	Area km^2	Annual discharge km^3	Runoff mm
Milliman et al. (1995)	6,000	76.0	12,667
Djuangsih and Salim (1994)	452	5.0	11,054
		† 6.9	15,349

† In addition to giving mean annual discharge as 5 km^3, Djuangsih and Salim also state that the mean flow rate is 220 m^3/s. This corresponds to a mean annual discharge volume of 6.9 km^3.

as a fungicide, or great industrial discharges. At an annual water discharge of 5 km^3, these concentrations correspond to annual river HCB flux of 186–281 t, while at 6.9 km^3, the flux increases to 258–390 t. The implausible Citarum discharge of 76 km^3 reported by Milliman et al. (1995) would raise these estimates another order of magnitude. HCB concentrations in Solo and Brantas River waters from East Java were only 6–12 pg/L in 1984, almost 7 orders of magnitude lower than is claimed for the Citarum River. At these concentrations, HCB exports from the Citarum River would be insignificant. The HCB data reported by Djuangsih and Salim should be regarded with skepticism, and at the very least, confirmed by further sampling.

Djuangsih also claims to have observed DDT, dieldrin, endrin and lindane in samples of fish taken from the Citarum at Bandung.

10.3.3 Solo and Brantas River Waters

In 1984, measurements of PCBs and several POPs insecticides in water, suspended sediments and biota were made in the straits between the islands of Java and Madura, the surrounding coastal seas, and the estuaries of the two major rivers of the area: the Solo (Bengawan Solo) and the Brantas that drain intensely agricultural watersheds of northeastern Java (Hillebrand et al., 1989). About 30-40 km inland, the Brantas splits into two distributaries: the Surabaya River that exits to the Strait of Surabaya at the city of Surabaya, and the Kali Porong that carries about 80% of the Brantas flow (Hoekstra et al., 1989) during wet season and debouches to the Strait of Madura 50-60 km south of Surabaya.

Solo and Brantas River POPs data for dissolved phase samples are summarized in Table 10.3. Measurements on suspended particulate matter [SPM], showed negligible concentrations; hence, the dissolved concentrations represent the total POPs concentrations in the water column. The low levels of PCBs and DDT on SPM, and the apparent high solubility in the dissolved phased is unusual. Aldrin, dieldrin, endrin, heptachlor epoxide and *o,p'*-DDD were not detected.

Table 10.3 POPs concentration in dissolved phase (pg/L); Solo and Brantas Rivers 1984.

River		n	PCBs	ΣDDT	ΣHCH	HCB
Solo	Dry season	3	884	627	489	6
	Wet season	6	1,740	584	455	10
	mean		1,312	606	472	8
	PW-mean †		1,483	597	465	9
Kali Porong	Dry season	2	818	547	577	6
	Wet season	2	3,518	847	431	12
	mean		2,168	697	504	9
	PW-mean		2,708	757	475	10

ΣDDT = *p,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD; *o,p'*-DDD not detected at 38 pg/L.
ΣHCH = α-HCH + β-HCH + γ-HCH, β-HCH not detected at 8 pg/L.
† PW-mean = precipitation weighted mean [70% wet season, 30% dry season].

The PCBs levels are remarkable given that both watersheds are primarily agricultural, and are not home to Java's major urban centres. PCBs concentrations were also appreciably higher during the wet monsoon than in the dry season. DDT levels were low and comprised mainly *p,p'*-DDD indicating the release of old inputs from bottom sediments. HCH comprised mainly γ-HCH indicating lindane rather than crude HCH usage. HCB levels were very low.

10.3.4 POPs in Aquatic Biota from Solo and Brantas Rivers

Aquatic biota were surveyed in the estuaries and nearshore seas of the Solo and Brantas Rivers, and in offshore coastal seas (Boon et al., 1989). DDT and PCBs were the most frequently observed contaminants in 96 samples of teleost fish, crustaceans, and other species. Traces of γ-HCH and HCB were found sporadically, but endrin, dieldrin and heptachlor epoxide were not detected.

Generally, DDT, which occurred primarily as *p,p'*-DDE, was highest at the estuary sites, lower at nearshore sites and lowest at offshore sites. In teleost fish, mean ΣDDT concentrations were respectively 4,500; 1,100 and 327 ng/g fat at estuary, nearshore and offshore sites. Boon et al. (1989) suggest that the DDE concentrations in the estuarine biota were at the low end of the range that might induce harmful effects to high end predator birds.

Boon and colleagues measured only 8 specific PCB congeners (IUPAC nos. 26, 44, 49, 52, 99, 118, 180, and 187) whose sum, Σ_8PCB, showed that PCBs were present in estuarine and nearshore waters. Regrettably, the partial sum of 8 congeners is not directly comparable to most available total PCB data. The highest Σ_8PCB measurement was 1 µg/g fat, but most samples contained <500 ng/g fat. The most

contaminated samples were from estuarine and nearshore waters of Surabaya city. Boon et al. (1989) remarked that PCB content of the most contaminated samples was about 1 order of magnitude lower than observed in biota of the Dutch Wadden Sea and nearshore areas of the North Sea.

10.3.5 POPs Flux from Solo and Brantas Rivers

Attempts to estimate POPs discharges from the Solo and Brantas Rivers must contend with questionable discharge data. Table 10.4 lists the Solo and Brantas annual discharge and drainage area data given by Milliman et al. (1995). Of two discharge figures given for the Solo River, the first, 14.6 km^3, yields a mean annual unit runoff estimate that is remarkably close to 1/2 the mean annual rainfall over the Solo and Brantas basins. This area of Java is relatively dry with a mean rainfall of about 1.75 m, somewhat less than the island average of about 2.3 m.

The Solo-b and Brantas River discharges, which yield runoff estimates in excess of the mean annual rainfall, are discounted. Assuming runoff to be 1/2 mean annual precipitation, the mean annual water discharge from the Brantas is estimated as 10.5 km^3. Using average wet and dry season discharge rate estimates for 1984 from Hoekstra et al. (1989) suggests that 1984 Brantas discharge was 12–16 km^3. The latter are very rough estimates, but they suggest that 10.5 km^3 is a reasonable estimate of mean annual Brantas discharge.

Annual flux estimates using these water discharge estimates for the Solo and Brantas Rivers are given in Table 10.5. Despite the apparently high PCB concentrations, the river loads total only 50 kg for the two rivers. DDT, HCH and HCB loads of 17 kg, 12 kg, and 236 g respectively were very low.

Table 10.4 Hydrologic data for Solo and Brantas Rivers; as in (Milliman et al., 1995).

	Area km^2	Annual discharge km^3	Runoff mm	‡ % Rainfall
Solo - a	16,000	14.6	913	52
Solo - b	16,000	239.0	14,938	852
Brantas	12,000	25.0	2,083	119

‡ at mean annual rainfall of 1,754 mm for Solo and Brantas watersheds.

Table 10.5 Annual POPs fluxes from Solo and Brantas Rivers; kg except HCB in g.

River	PCBs	ΣDDT	ΣHCH	HCB
Solo	22	9	7	128
Brantas	28	8	5	107
sum	50	17	12	236

10.3.6 DDT in Fish Ponds near Yogyakarta

In 1989, Noegrohati et al. (1992) investigated DDT residues in soil, water, fish, poultry and human milk from villages of Central Java near Yogyakarta that had been treated in anti-malarial operations 2, 8 and 25 years previously. The soil data (Table 10.6) have slightly higher ΣDDT concentrations than seen in sediments of the Ciliwung River. The longevity of *p,p'*-DDT and *o,p'*-DDT in soils from areas treated years ago is remarkable. The same pattern is evident in the DDT isomer composition of fish pond waters from the three villages (Table 10.7). Fish pond water ΣDDT concentrations are about 200 fold higher than reported in the Ciliwung River, and about 65-75 fold higher than reported in the Solo and Brantas Rivers. As the anti-malarial operations were described as standard indoor residual spraying, the high DDT concentrations seen in outdoor soil and pond water raise the question: ***"How does spraying DDT on interior walls and ceilings lead to such high contamination of exterior soils and surface waters?"*** If unreported spraying has not occurred, the apparent longevity of DDT in the soils of central Java suggests that DDT residues from anti-malarial treatments may persist for a remarkably long time afterwards in the Javanese environment. Other data (Noegrohati et al., 1992) show that humans take up DDT form contaminated poultry and fish, and that intake may be high enough to put infants at risk.

Table 10.6 DDT in soils from villages near Yogyakarta (ng/g dry weight).

Village	Years since treatment	*p,p'*-DDT + *o,p'*-DDT	%	ΣDDT
Slanden	2	72.8	83	87.7
Kalijeruk	8	6.8	60	11.3
Sonowetan	25	22.3	74	30.2

ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.

Table 10.7 Table 10.7 DDT in fish pond waters from villages near Yogyakarta (ng/L).

Village	Years since treatment	*p,p'*-DDT + *o,p'*-DDT	%	ΣDDT
Slanden 1	2	34.2	53.9	63.5
Slanden 2		39.1	54.1	72.3
Kalijeruk 1	8	36.8	72.1	51.0
Kalijeruk 2		14.7	63.6	23.1
Sonowetan	25	14.4	62.9	22.9

ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.

10.3.7 Miscellaneous POPs data

Some reports of limited POPs data in the Indonesian environment include:

- Hungspreugs (1988) cites DDT and PCBs concentrations in Jakarta Bay sediments taken August 1982 as: DDT [1-13] and PCBs [4-9] μg/g wet weight.
- Eng (1989) cites maximum DDT and PCBs concentrations in Jakarta Bay waters as 13 μg/L DDT and 9 μg/L PCBs according to a 1985 Indonesian report.
- Oshawa et al. (1985) give HCH and DDT in Bali stream water as 72 ng/L and 3 ng/L respectively.
- Kuwatsuka et al., (1986) give 1982 HCH and DDT soil concentrations (ng/g dry) as:
  - paddy field: HCH [mean 8.9, range 0.5-20], and DDT [mean 1.2, range 0-20]
  - upland soil: HCH [mean 3.9, range 0.8-11], and DDT [mean 10, range 0.6-24].

The reported concentrations of DDT and PCBs in Jakarta in the early mid 1980s are disturbingly high, and should be confirmed by further investigations. The HCH levels reported for Bali streams are high enough to be of concern if HCH usage has continued since the 1986 pesticide restrictions were imposed. The Bali DDT levels seem to be of the same order as reported recently for the Ciliwung River. The 1982 HCH soils' levels were higher than those reported for Thailand and lower than those for Taiwan in the survey conducted recently by (Thao et al., 1993). The 1982 DDT data for paddy fields are comparable to levels seen recently in Taiwan paddies, but lower than in paddy soils of Vietnam where DDT remains in use (Thao et al., 1993).

10.4 Annual POPs Exports from Javanese rivers

Annual river flux estimates from Java are given in Table 10.8. *Low*, *medium* and *high* scenarios were based on combined POPs water concentration data from Iwata et al. (1994) for the Ciliwung River and Hillebrand, et al. (1989) for the Solo and Brantas Rivers. The *medium* scenario was based on mean annual total Java river water discharge of 146 km^3, and plausible variations (102–205 km^3) were considered in estimating low and high scenarios. The *very high* scenario used the maximum concentrations with the maximum flow, except for DDT for which the minimum Yogyakarta area fish pond water concentrations (Noegrohati et al., 1992) were used.

The figures reveal that the potential annual POPs mass fluxes from Java by surface water discharges are relatively low, i.e., < 1 tonne. Under very high, but seemingly implausible scenarios given the available information, HCH and DDT loads could approach 3-5 t/a each. Better water discharge and POPs water concentration data are necessary to confirm the estimates in Table 10.8, as the available data are very limited.

Table 10.8 Annual POPs exports (kg) by Javanese rivers.

	Low	Medium	High	Very High
PCBs	103	178	307	720
ΣHCH	103	242	756	4,500
ΣDDT	39	60	92	3,000
ΣCHL	10	38	50	—

ΣHCH = α-HCH + β-HCH + γ-HCH.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

Chapter 11 Taiwan

11.1 Hydrology

Taiwan has humid climate with a mean annual rainfall of about 2.5 m for the island. The island is covered with numerous small rivers. Milliman et al. (1995) give area and discharge data for 19 streams representing a total drainage area of 25,520 km^2, an annual mean discharge of 52 km^3, and a mean runoff of about 2,039 mm. The area of Taiwan is variably given as 32,260 – 35,980 km^2 which suggests that the total mean annual discharge from the island is 66 – 73 km^3.

In addition to water discharges, the 19 streams in the GLORI data base also have annual sediment discharges. Because POPs water concentration data for Taiwan are minimal, the sediment flux data provide an alternative means to estimate POPs fluxes from more abundant POPs soil and sediment concentration data. The total mean annual sediment flux from Taiwan is about 292 million tonnes.

11.2 Taiwan POPs Usage

The history of POPs insecticide usage in Taiwan is not well documented. According to a secondary source (Chu and Lin, 1983; cited in Thao, et al., 1993), organochlorine insecticides were prohibited circa 1975. Evidence to be presented suggests that there has been some relatively recent usage of HCH, while DDT environmental concentrations have declined suggesting that prohibition has been effective.

Taiwan was the site of the *Yucheng* PCB poisoning incident in 1979 (Chen and Hsu, 1986); hence, the territory should be well aware of the potential hazards posed by PCBs. Because of its association with the West and its relatively high level of development, PCBs may be present in significant quantity.

11.2.1 Er-Jen River PCB contamination

PCBs have also been released by waste disposal activities in the Er-Jen River watershed of southwest Taiwan. Since the mid 1960s, the town of Wan-Li at the mouth of the Er-Jen River has been the site of metal recovery by open air incineration of scrap electrical wire and magnetic circuit cards, as well as the disposal of waste motors and electrical capacitors and transformers (Huang et al., 1992; Ling et al., 1995). Incineration releases PCBs used in plastics in wire coverings and circuit cards. The area is considered to be amongst the most polluted in Taiwan.

11.3 POPs In Taiwanese Soils

Thao et al. (1993) surveyed soils in Vietnam, Thailand and Taiwan for DDT, HCH and PCBs. Fourteen Taiwanese samples were collected in August 1990 in two districts only, the capital city of Taipei, and the nearby district of Taoyuan that includes both rural and urban areas. Data are summarized in Table 11.1. Results showed that this limited sampling of Taiwanese surficial soils were less contaminated with HCH and DDT than Vietnamese soils, but were perceptibly more contaminated than Thai soils. Some 1980 era Taiwanese soil data cited by Kawano et al. (1992) suggest that HCH levels have increased while DDT levels have declined.

Table 11.1 POPs in Taiwanese soils (ng/g dry weight); after Thao et al. (1993).

		mean	† G-mean	min.	max.
ΣHCH	Taipei	28.3	20.1	4.2	78
	Taoyuan	12.1	8.1	2.4	42
	Taiwan total	20.2	12.8	2.4	78
	upland soils — 1981 [a]	3.5	—	1.1	6
	paddy soils — 1980 [b]	1.9	—	‡ nd	36
ΣDDT	Taipei	1.7	1.3	0.45	4.8
	Taoyuan	1.2	0.9	0.33	3.2
	Taiwan total	1.4	1.1	0.33	4.8
	upland soils — 1981 [a]	15.0	—	5.4	25
	paddy soils — 1980 [b]	12.0	—	nd	120
PCBs	Taipei	163	40	12	960
	Taoyuan	27	16	1.6	53
	Taiwan	95	25	1.6	960
	Wan Li (Er-Jen River) [c]	27,890	9,160	450	77,000

ΣHCH = α-HCH + β-HCH + γ-HCH + δ-HCH.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.
[a] (Wang et al., 1981) cited by (Kawano et al., 1992).
[b] (Li and Lee, 1983) cited by (Kawano et al., 1992).
[c] soils from 6 incineration sites in Wan Li (Huang et al., 1992).
† geometric mean; ‡ nd = not detected.

Soils of Taiwan exhibited the highest level of PCB contamination of the three survey countries. The mean PCB concentrations in Table 11.1 are inflated by 3 samples with PCBs content exceeding 50 ng/g. If these three samples are deleted the mean PCB concentration falls to about 21 ng/g for the remaining 11 samples. That is comparable to PCBs observed in 4 samples from Hanoi, Vietnam, but higher than most other samples from Vietnam and Thailand. Soils of the Er-Jen River incineration sites show high PCB contamination that is about 300 fold greater than in typical Taiwanese soils.

11.4 POPs In Taiwanese Sediments

Iwata et al. (1994) collected 3 sediment samples from Taipei canals that are summarized in Table 11.2. Sediments have very low HCH contamination, but are modestly contaminated with DDT, chlordane, and PCBs. HCH and DDT data for sediments from agricultural streams taken in 1980 (Wu et al., 1982; cited in Ramesh et al., 1991) suggest current HCH and DDT levels are lower than 1980 levels in rural sediments. As expected, the Er-Jen River sediments show very high levels of PCB contamination.

Table 11.2 POPs in Taipei sediments (ng/g dry); after Iwata et al. (1994).

		Mean	† G-mean	Min.	Max.
ΣHCH	Taipei — 1990	0.46	0.40	0.29	0.79
	rural — 1980 [a]	8.20	—	—	—
ΣDDT	Taipei — 1990	7.0	3.5	0.4	10.6
	rural — 1980 [a]	18.0			
ΣCHL	Taipei — 1990	2.32	0.99	0.14	5.60
PCBs	Taipei — 1990	104	34.8	2.3	230
	Er-Jen R — 199? [b]	4,480	3,090	870	13,600

† geometric mean.
[a] (Wu et al., 1982) cited in (Ramesh et al., 1991).
[b] 6 samples from sites downstream of incineration sites (Ling et al., 1995), date not given.
ΣHCH = α-HCH + β-HCH + γ-HCH.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

11.5 POPs In Taipei Surface Waters

The only available surface water data are for 3 samples obtained by Iwata et al. (1994) from Taipei canals. These are summarized in Table 11.3. Levels of the measured contaminants are relatively low. The extent to which these minimal data represent general Taiwanese surface water conditions is unknown.

Table 11.3 POPs in Taipei water samples (pg/L); after Iwata et al. (1994).

	mean	† G-mean	Min.	Max.
ΣHCH	192	171	82	253
ΣDDT	86	48	10	192
ΣCHL	47	27	11	114
PCBs	978	512	85	2,100

† geometric mean.
ΣHCH = α-HCH + β-HCH + γ-HCH.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.
ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor.

11.6 Annual POPs Exports from Taiwanese rivers

Because surface water data are limited to three Taipei urban drainage samples, estimation of POPs fluxes entails great uncertainty. Consequently, additional flux scenarios were constructed by applying soil and sediment POPs concentration data to river sediment flux data. The resulting estimated annual POPs riverine exports from Taiwan to the sea are given in Table 11.4 where the PCB flux estimates ignore potential Er-Jen River loads.

Generally, river load estimates derived from the limited water concentrations are very low, while estimates derived from sediment concentration and flux data can be very high. The bulk sediment flux from Taiwan of circa 300 million t/a is very high, and most likely dominated by relatively clean material washed down from the mountains and scoured from river banks and channel bottoms. Contaminated surficial soils eroded from agricultural, urban and industrial areas may constitute only a small fraction of the bulk sediment load, so that, sediment derived contaminant fluxes are likely too high. In the absence of better data, these sediment based flux estimates do indicate that a potential exists for high POPs river discharges, particularly for HCH and PCBs that have relatively high concentrations in Taiwanese surficial soils.

Table 11.4 Annual POPs exports (kg) by Taiwanese rivers to the sea; flux scenarios based on water, sediment and soil concentrations given in Tables 11.1–11.3.

Basis	Concentration Statistic	ΣCHL	ΣDDT	ΣHCH	PCBs
Water	G-mean	1.1	3.5	12	37
	Mean	3.4	6.3	14	71
Sediment	Min.	41	113	85	586
	G-mean	289	1,010	118	10,194
	Mean	681	2,047	133	30,466
Soil	Min.	—	97	703	469
	G-mean	—	322	3,746	7,284
	Mean	—	419	5,911	27,807

11.6.1 Potential PCB Flux from the Er-Jen River

Attempts were also made to assess potential PCB loads from the Er-Jen River. Applying the maximum Taipei water concentration of 2.1 ng/L to annual Er-Jen water discharge of 0.5 km^3 yields an annual PCB flux of only 1 kg that seems too low.

As reported by Milliman et al. (1995), bulk sediment exports from the Er-Jen watershed are 13 million t/a. The specific annual unit area delivery is 371 t/ha, and the flow-weighted mean suspended sediment concentration is 26 g/L. Both figures are the highest reported for Taiwan which has overall mean unit area sediment delivery of 91 t/ha, and flow-weighted mean suspended sediment concentration of 4.45 g/L. If the Er-Jen bottom sediment PCB concentrations of 3–4.5 μg/g are applied to bulk Er-Jen sediment delivery, the estimated annual fluxes are an implausible 40–60 t/a.

Er-Jen PCB flux is generated by only a small area, perhaps a few ha, located near the outlet of the watershed. For a 1 ha incineration site generating 370 t/a of sediment, the estimated PCB flux is only 3–10 kg/a. Erosion losses from the Wan Li town area are probably less than the basin average of 371 t/ha, but the area receiving PCB laden fall-out from open air incineration may be much larger, e.g., 1 km^2 or more. Without more detailed data, it is impossible to derive a reasonably certain estimate of Er-Jen PCB flux; however, it does seem very likely that the annual river load is less than 1 tonne, and likely less than 500 kg. Also, it should be expected that supplies of PCB laden scrap wire and circuit cards should diminish as old products are taken out of service. If Wan Li also serves as a disposal site for old Taiwanese electrical equipment containing PCBs, the possibility exists that improper disposal will yield significant environmental releases of PCBs.

Chapter 12 Australia

12.1 Hydrology

The nation of Australia comprises mainland or continental Australia (7,611,800 km^2) and the island of Tasmania (68,200 km^2).

Much of mainland Australia is desert with no well-defined perennial drainage, and large tracts of ephemeral *closed* drainage systems. While mean rainfall over Australia is 450 mm (R. James, Bureau of Meteorology, Melbourne, pers. comm.), the map of median annual rainfall (e.g., Kinimonth, 1983) shows that only 25–30% of the continent receives >400 mm rainfall. That area extends in a 300–600 km band that arcs across the northern, eastern, and southeastern margins of the mainland. In the north, much of Arnhem Land and the Cape York Peninsula receive more than 1 m rainfall annually. Along the eastern seaboard there is a very narrow coastal strip extending to 35° S that receives 1 m or more annual rainfall. On the southwestern coast from Geraldton to Esperance, a narrow band of terrain enjoys a maritime climate with more than 400 mm mean annual rainfall.

The major surface drainage divisions and discharges reported in a comprehensive 1985 water resources study (AWRC, 1987) are summarized in Table 12.1. Practically, the Northeast and Southeast Coast drainage divisions are likely the most important sources of POPs delivery to coastal waters. The *Northeast Coast* division comprises waters flowing eastward of the Great Dividing Range in the state of Queensland from Cape York to south of Brisbane. In places the watershed extends inland up to 500 km from the Pacific coast. The *Southeast Coast* division is defined mainly by waters flowing eastward or southward of the Great Dividing Range in the states of New South Wales and Victoria, with a small extension into eastern South Australia. At most, the watershed extends inland no more than 200 km, but this narrow strip is home to the greatest concentration of population and industry, as well as intense agriculture.

Two northern drainage divisions, *Gulf of Carpentaria* and *Timor Sea*, have large water discharge that should be relatively contaminant free given the sparse population of northern Australia. Though *Tasmania* has a large runoff yield, the Derwent River that flows through the state capital Hobart, may be the only contaminant source of significance. Waters of the *Murray-Darling* system, a vast tract of semi-arid drainage, may be contaminated with pesticides, but the runoff yield is rather low. Otherwise, the remaining drainage divisions — *South Australian Gulf*, *Southwest Coast*, and *Indian Ocean* — drain mostly arid terrain. *South Australian Gulf* is a relatively small division draining the Adelaide area. *Southwest Coast* is the small corner of maritime climate in the vicinity of Perth, Western Australia, and *Indian Ocean* designates ephemeral desert drainage to the Indian Ocean between Southwest Coast and Timor Sea divisions.

Table 12.1 Australian annual surface water discharge by drainage divisions; from (AWRC, 1987).

Drainage division	No.	Drainage area 10^3 km^2	Discharge km^3	Runoff mm
Northeast Coast	I	451.0	83.900	186
Southeast Coast	II	274.0	41.900	153
Tasmania	III	68.2	52.900	776
Murray-Darling	IV	1,060.0	12.200	12
South Australian Gulf	V	82.3	0.757	9
Southwest Coast	VI	315.0	6.600	21
Indian Ocean	VII	519.0	3.840	7
Timor Sea	VIII	547.0	80.700	148
Gulf of Carpentaria	IX	641.0	92.500	144
Total		3,957.5	375.297	95

12.2 POPs Usage in Australia

12.2.1 Insecticides

No readily available quantitative data on historical Australian production and consumption of POPs insecticides were found. Historically, Australia has used most common POPs insecticides including DDT, aldrin, dieldrin, heptachlor, chlordane and HCH. Originally, DDT appears to have been widely used up to the early 1960s when dieldrin usage became popular. In the 1970s, aldrin, heptachlor, chlordane, HCH, and lindane (γ-HCH) usage became more prevalent. HCB was used as a fungicidal seed dressing, but HCB originating as an impurity in other POPs insecticides and as an industrial by product may have been the major source in recent years.

The historical chronology of POPs insecticide status in Australia is vague. Some claim DDT was banned in 1981; others suggest it was still registered for use in some areas as of 1993. Registration requirements vary across the states and territories. Heavy usage of most POPs insecticides ended about 1985 except for termiticides. About 280 t heptachlor and 10 t chlordane were imported over 1987-89 (Kannan et al., 1995). Circa 1994, DDT, endrin, HCB, and crude HCH had no permitted uses, while selected uses of lindane, aldrin, dieldrin, heptachlor, and chlordane continued in some states (Richardson, 1995). In mid 1995, heptachlor and chlordane were prohibited except in the Northern Territory which has to June 30, 1997 to phase out stocks (ANRA, 1995). In mid 1996, aldrin and dieldrin had no registered uses, lindane was registered in Queensland for use on pineapples, and mirex was registered for use in Western Australia and Northern Territory (D. Owens, ANRA[12], pers. comm.).

[12] For Australian National Registration Authority contacts see http://www.dpie.gov.au/nra/welcome.html.

12.2.2 PCBs

Australia never manufactured PCBs. Imports before 1971 are unknown. Bulk import data for 1971-80 were given by Richardson (1986). From 1971-75, about 290 t/a were imported, excluding PCBs contained within imported manufactured goods. Restrictions introduced after 1975, reduced bulk imports to 0 by 1980.

The fate of Australian PCB stocks is only partially known. It was estimated that electrical authorities accounted for 75% of PCBs stocks much of which may still be in use. Before 1970, PCBs were disposed in landfills without any record of location (Thompson et al., 1992). Some imported PCBs were exported to neighbouring countries (New Zealand, Vietnam) in manufactured electrical equipment. About 300 t were shipped to the U.K. for high temperature incineration, and about 150 t were destroyed on the incinerator ship *Vulcanus* in the early 1980s.

12.2.3 Organochlorine Management Plans

Since 1994, Australia has been in the process of developing management plans aimed at the ultimate destruction of existing stocks certain hazardous materials and wastes. Recent bulletins [13], have announced specific plans are under development for (1) organochlorine pesticides, (2) PCBs, and (3) HCB. Recently, Australia has commissioned a plant near Perth to destroy stockpiles of organochlorine pesticides (predominantly DDT) and PCBs [14].

12.3 POPs in the Australian Environment

12.3.1 POPs In Australian Market Fish

A survey of POPs contaminants in Australian foodstuffs conducted in 1990-92 (Kannan et al., 1995; Kannan et al., 1994b) included baskets of freshly caught food fish from the local freshwater and nearshore fisheries of Sydney, Perth, and Hobart [sampled August – September 1990]; and Brisbane, Townsville and Atherton [sampled February – March 1992]. Townsville is in northern Queensland on the Pacific coast near the mouth of the Burdekin River where sugar cane is cultivated. Atherton is a town of 3,000 further north near the Tineroo dam on the upper Barron River that exits to the sea at Cairns. Atherton fish were taken from the Tineroo reservoir. There may have been POPs insecticide usage, likely DDT and dieldrin, associated with past tobacco cropping in the Atherton area. Together, the fish data give a broad geographic view of POPs contamination of Australian aquatic systems unavailable from other data sets. The survey data are summarized in Table 12.2.

[13] Summer 1996, Australian Environmental Protection Agency; see http://www.erin.gov.au/portfolio/epa/publications.html.
[14] See http://www.mfe.govt.nz/organo.htm.

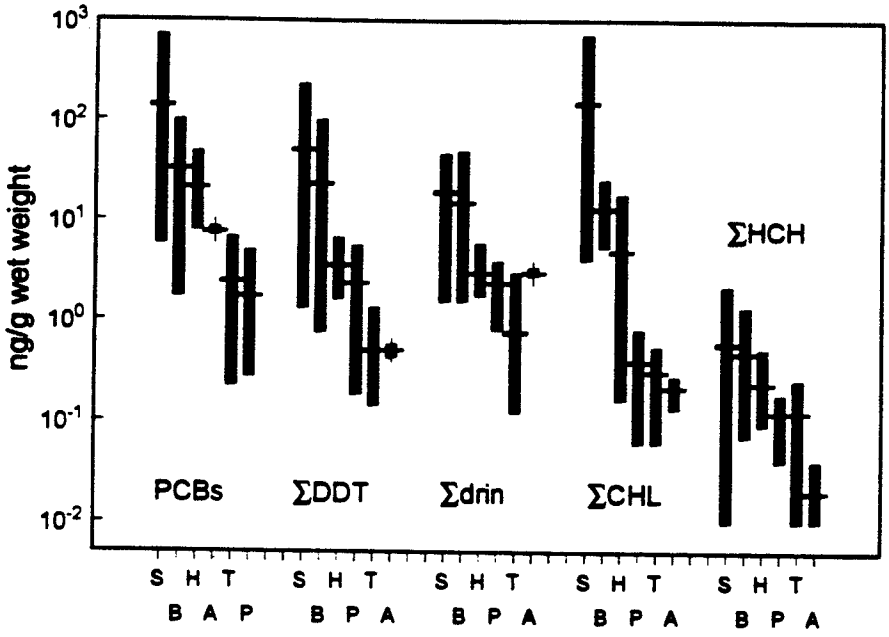


Figure 12.1 POPs in locally-caught market fish from 6 Australian cities: Sydney (S), Brisbane (B), Hobart (H), Perth (P), Townsville (T), Atherton (A); vertical bars span concentration range and cross-bar shows the mean; data from (Kannan et al., 1995; Kannan et al., 1994b).

The general order of contamination is PCBs > chlordanes > DDT > drins > HCB > heptachlor > HCH. Geographically, wet weight concentration data (Figure 12.1) show a broadly similar pattern for the 5 POPs contaminants: Sydney > Brisbane > Hobart > Perth > Townsville > Atherton. The pattern is broken for chlordanes as Hobart > Brisbane, and for PCBs and drins, Atherton > Townsville > Perth.

Ideally, comparisons should have relied on lipid weight concentrations because multiple local species with a broad range of lipid content (0.14 – 20%) were employed. The precise fat weight concentrations were not given by location and can only be roughly approximated from information given by Kannan et al. (1994) [15]. Some rankings could change if the analysis were performed on lipid weight concentrations.

Sydney fish have especially high levels of PCBs and chlordanes. Earlier studies of fish from the Sydney coast (Thompson et al., 1992) have shown similar contamination. Concentrations are so high that they skew the national averages (Table 12.2) toward

[15] Mean of individual lipid weight concentrations is **not equal** to mean wet weight concentration divided by mean lipid concentration. See Section 2.4.1.

Table 12.2 POPs in Australian market fish.

	ng/g wet weight			ng/g lipid weight		
	Mean	Min.	Max.	Mean	Min.	Max.
fat %	3.40	0.14	20.00	—	—	—
PCBs	55.0	0.22	720.0	6,900	18.0	130,000
ΣCHL	51.0	0.06	720.0	6,000	5.4	36,000
ΣDDT	22.0	0.14	230.0	2,300	12.0	43,000
dieldrin	9.5	0.12	47.0	940	10.0	8,800
aldrin	0.8	<0.01	7.5	65	<0.2	500
Σdrin	10.3	0.12	54.5	1,005	10.1	9,300
HCB	4.2	<0.01	60.0	470	<0.5	9,000
heptachlor epox.	1.0	0.03	4.5	170	1.2	1,100
heptachlor	0.1	<0.01	0.8	11	<0.4	140
Σhep	1.1	0.03	5.3	181	1.2	1,240
ΣHCH	0.3	<0.01	2.1	46	<.5	460

ΣCHL = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor + oxychlordane.
ΣDDT = *p,p'*-DDT + *o,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.
Σdrin = aldrin + dieldrin.
Σhep = heptachlor epoxide + heptachlor.
ΣHCH = α-HCH + β-HCH + γ-HCH.

Sydney. Though HCB and heptachlor data were not broken out by location, the high mean HCB concentrations may likewise be the product of high contamination in Sydney area fish. Brisbane and Hobart area fish are moderately contaminated, while fish from the remaining sites have low POPs levels. Thus, surface waters of the Northeast Coast (excepting Brisbane area) and Southwest Coast drainage divisions appear lightly contaminated with PCBs, DDT, chlordane and HCH.

12.3.3 POPs In Australian Coastal Waters

Contamination in the coastal waters in the vicinity of Melbourne, Victoria may rival that seen in Sydney area waters. Various studies (Phillips et al., 1992; Richardson, 1985b) have shown significant PCB contamination of sediments and biota from Port Phillips Bay. Melbourne is located on Hobsons Bay at the head of greater Port Phillips Bay. Late 1980s' data showed declining PCBs in greater Port Phillips Bay, but unchanged levels in Hobsons Bay. DDT had also been found in Hobsons Bay and was attributed to agricultural use before 1975 in the Yarra River tributary.

Shaw and Connell (1980) analyzed PCBs in water, sediments, fish and other intertidal organisms of the Brisbane River, Queensland about 1980. PCBs were quantified as both as Aroclor 1254 and Aroclor 1260. The data are not directly comparable with modern PCB data, but do suggest that water concentrations ranged from nondetectable (detection limit not given) to 50 ng/L. Mean concentrations in fish muscle samples were roughly about 10 and 50 μg/g fat weight for Aroclor 1254 and 1260 respectively. Relative to the recent market fish survey, PCB levels in Brisbane area waters seem to have declined between 1980 and 1992.

Recent investigations (von Westernhagen and Klumpp, 1995) of marine and estuarine fish along the Queensland coast show minimal POPs contamination of livers and ovaries. Fish were taken between October 1992 and March 1993 at 11 sites along the Queensland coast between 12° 50′ S [Port Stewart] and 21° 07′ S [Mackay] including 3 remote offshore sites in the Great Barrier Reef. Of 142 samples analyzed for *p,p′*-DDE and dieldrin, only 11 (7.7%) showed detectable presence of these two POPs which were found off the sugar cane growing districts [Ayr, Mackay] and urban centres [Cairns, Cooktown] at what appear to be low levels. Evidently, both DDT and dieldrin remained registered for use in Queensland as of 1993; although, the fish data would suggest that they are no longer being used in perceptible quantities, if at all. PCBs were apparently analyzed on only 7 additional samples. Low levels were found only off Cairns and the Burdekin River estuary at Ayr.

A recent study of Perth area coastal waters (Burt and Ebell, 1995) showed minimal POPs contamination. In 1991, sediments were sampled at 175 sites, and mussels at 35 sites in a 500 km^2 area in the vicinity of Perth. Low levels of PCBs and DDT were detected at only 2% of sites. There were a few sporadic detections of chlordane, heptachlor and dieldrin. Generally, these results concur with the results of the market fish survey showing that Perth area waters have only light POPs contamination.

12.3.3 POPs Insecticides in Groundwaters of the Burdekin River Delta

The Burdekin River exits to the Pacific Ocean near the town of Ayr located south of Townsville, Queensland. In 1976, the 620 km^2 delta supported about 309 km^2 sugar cane cultivation. A 1976-77 survey of delta groundwaters (Brodie et al., 1984) is dated, but gives some historical context on POPs insecticide usage in tropical sugar cane agriculture. Initially, DDT was widely used to about 1964, when it was superseded by heptachlor which dominated to 1973 when lindane (γ-HCH) became the preferred pesticide. For 1975-76, annual γ-HCH and heptachlor usage was 5.5 t/a and 0.55 t/a respectively. The figures correspond to respective annual usage rates of 210 and 18 g/ha. The groundwater data of 1976/77 showed that the heavy monsoonal rains of January – February seemed to carry pesticide residues into the aquifer, but that by the end of the April – November dry season, γ-HCH and heptachlor had dissipated to non-detectable levels. Average wet season groundwater concentrations of 0.6–0.8 ng/L

γ-HCH and heptachlor were quite low. Investigators mysteriously neglected dieldrin which was widely used against sugar cane pests elsewhere in Australia (McDougall et al., 1995; McDougall et al., 1987). Residues were found recently in fish from the Burdekin estuary (von Westernhagen and Klumpp, 1995).

12.3.4 POPs Insecticides in New South Wales

POPs insecticide presence has been documented in soils, fish, birds, drinking water and grain storage areas of New South Wales (Allender, 1989; Ang et al., 1989; McDougall et al., 1989; McDougall et al., 1995; McDougall et al., 1987; Wan et al., 1989). Relevant studies are discussed below.

12.3.4.1 Fish

The most immediately relevant survey reports dieldrin and ΣDDT observed in fish from three major rivers of northern New South Wales: the Clarence, Richmond and Tweed (McDougall et al., 1989). The Clarence and Richmond watersheds have been dedicated mainly to livestock production with sugar cane cultivation on the flood plains of the lower reaches. The Tweed watershed has had intensive cultivation of sugar cane, bananas, and other tropical fruits and vegetables.

A limited survey of fish from all three rivers was originally conducted from October 1983 to February 1984. Table 12.3 gives the mean dieldrin and ΣDDT observed in mullet and bream, the two species common to all three rivers. For reference, dieldrin and ΣDDT levels in the recent market fish survey are included. The order of contamination among rivers is obvious. Dieldrin in Tweed River fish rivaled that observed recently in Sydney market fish; however, DDT levels were generally lower in all fish from all three rivers.

Following the initial survey, more intensive monitoring the mullet species was initiated in the Tweed and Clarence Rivers. Data are summarized in Table 12.4 with the mullet concentration data from the earlier survey and recent market fish data. In the Tweed River, dieldrin residues in mullet were higher than in recently surveyed

Table 12.3 Dieldrin and DDT (ng/g fat) in New South Wales river fish, 1983-84.

	† Market fish	Tweed R.	Richmond R.	Clarence R.
dieldrin	940	894	224	20
ΣDDT	2,300	510	219	39

† lipid weight concentrations from (Kannan et al., 1994b).

Table 12.4 Dieldrin and DDT (ng/g fat) in Tweed and Clarence River mullet.

	date	dieldrin	ΣDDT
Market fish †	Aug–Sep 1990	940	2,300
Tweed R.	Oct 83 – Feb 84	1,104	625
	Jul 85 – Aug 86	1,135	562
Clarence R.	Oct 83 – Feb 84	17	34
	Jun 84 – May 85	98	224

† lipid weight concentrations from (Kannan et al., 1994b).

market fish, while DDT levels were significantly lower. For reasons unknown, in the Clarence River, dieldrin and DDT levels were 5-7 times higher in the 1984/85 survey than in the earlier 1983/84 survey.

12.3.4.2 POPs New South Wales Soils

In 1987, DDT was still prevalent in soils of fruit and vegetable growing areas (Wan et al., 1989), while heptachlor, dieldrin and some HCH residues were found in pasture, sugar cane and banana plantation soils (McDougall et al., 1989). Heptachlor was the preferred insecticide used on lands with mixed dairy pasture and crop rotations, while dieldrin was the dominant residue in sugar cane and banana soils. Recent results (McDougall et al., 1995) of a long term study tracking the decay of heptachlor, *trans*-chlordane [16] and dieldrin, suggest that soil half-lives are about 1 year for heptachlor, 3 years for heptachlor epoxide and *trans*-chlordane, and from 4–7 years for dieldrin.

Together, these results suggest that persistent residues of past POPs insecticide usage in New South Wales will continue to contaminate surface waters via residues in eroded soil for some years to come as levels slowly decline.

12.3.5 POPs Insecticides in Ovens and King Rivers of Northeast Victoria

The Ovens and King Rivers of northeastern Victoria state are tributaries of the Murray River. DDT and dieldrin were used intensively through the 1960s and 1970s on tobacco crops grown on about 4,000 ha of narrow flood plains of these rivers and their tributaries. Recently, McKenzie-Smith et al. (1994) reported water and sediment data collected in 1990. The survey showed that DDT and dieldrin residues remain, mostly associated with sediments. Storm flow sampling showed that remarkably high DDT

[16] Introduced as a component of technical heptachlor.

levels (up to 340 ng/L) could be observed in whole water samples where the DDT was associated with suspended particulates. Dieldrin was present in storm water samples only to maximum whole water concentrations of 2 ng/L.

Curiously, most total DDT in high concentration stormwater samples appeared to exist in unmetabolized forms (*p,p'*-DDT, *o,p'*-DDT). The authors' speculative explanations notwithstanding, area soils may be similar to certain U.S. soils in which negligible microbial degradation of applied DDT has been observed for reasons that are not yet understood (Hitch and Day, 1992).

While this study reveals that significant DDT residues remain in the soils and sediments of headwater catchments of the Murray River, and that residues are mobilized during storm runoff events, it remains unclear how far the mobilized sediments are transported downstream.

12.3.6 Termiticides

As a mostly sub-tropical / tropical nation, termites ("white ants") are ubiquitous across Australia, and organochlorine termiticidal treatments have been widely applied to houses, sheds, fences, utility poles and other wood structures. Dieldrin appears to have been the original preferred treatment, giving way to aldrin, heptachlor and latterly, chlordane. Studies in the vicinity of Perth, Western Australia (Stacey et al., 1985; Stacey and Tatum, 1985) have shown how rapidly these compounds are absorbed by lactating mothers after domicile treatments. Interestingly, HCB was also absorbed after treatments. While it is widely stated that HCB occurs as an impurity in organochlorine pesticides, Stacey and Tatum's (1985) study is one of the few that clearly document its release as an unintentional contaminant in other organochlorine compounds.

Termiticidal usage of chlordane was thought by Kannan et al. (1994) to explain the unusually high presence of chlordane compounds in market fish caught near Sydney. Sewage discharge and urban storm runoff were posited as the likely means by which chlordanes enter aquatic systems; however, the precise mechanisms remain to be elaborated. In rural areas, technical heptachlor usage as a soil insecticide would contribute *trans*-chlordane and *trans*-nonachlor to the total of these isomers that also originate from rural termiticidal treatments.

Usage of heptachlor and chlordane as termiticides became illegal across most of Australia in August 1995 except Northern Territory where it is to be phased out by June 1997 (ANRA, 1995).

12.3.5 Regional Comparison of POPs in Surface Waters and Sediments

The synoptic survey conducted by Iwata et al. (1994) included sufficient samples (20 water, 19 sediment) to permit broad comparison between New South Wales (Sydney area), Tasmania (Hobart area) and Western Australia (Perth area) waters and sediments. Samples were collected in September 1990. The results are shown in Figure 12.2. An overall ranking scheme gave the general order of contamination in water as New South Wales > Western Australia > Tasmania, and in sediments as New South Wales > Tasmania > Western Australia. By contaminant, in water, the net order of contamination was PCBs > ΣHCH > Σchlordane > ΣDDT, while in sediments ΣDDT > PCBs > Σchlordane > ΣHCH. With a few exceptions, surface waters of the Sydney, New South Wales area stand out as the most contaminated.

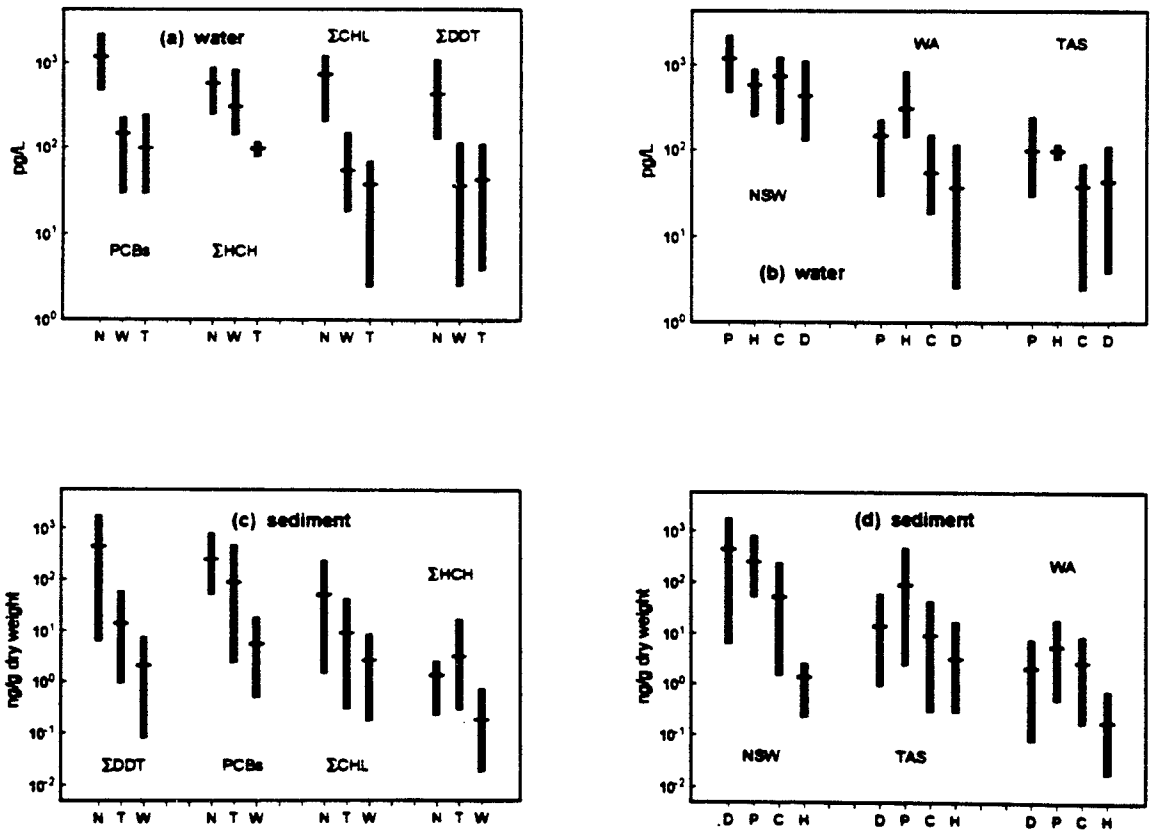


Figure 12.2. Regional comparison of POPs in water and sediments; P ≡ PCBs, D ≡ DDT, C ≡ chlordane, H ≡ HCH, N ≡ New South Wales (Sydney area), T ≡ Tasmania (River Derwent, Hobart area), W ≡ Western Australia (Perth area); vertical bars indicate range, cross-bars indicate means.

While waters, sediments and fish of Western Australia are relatively uncontaminated, for 5 of 9 water samples, the fraction of unmetabolized DDT (*p,p'*-DDT + *o,p'*-DDT) exceeded 50% ΣDDT, and for 6 of 9 exceeded 49%, suggesting that there had been recent DDT inputs. For similar reasons, in their global survey of POPs insecticides in tree bark, Simonich and Hites (1995) observed that there had been possible recent use of DDT on the west coast of Australia. Evidence in sediments is weaker as only 3 of 7 samples suggest the possibility. Western Australian fish have modest DDT levels, well below those seen in Sydney and Brisbane fish.

12.4 POPs Riverine Flux from Australia

To estimate POPs riverine flux from Australia, the 19 concentration data from Iwata et al.'s (1994) synoptic survey are used in the same regional grouping as in the previous section, i.e., New South Wales, Tasmania and Western Australia. These are harmonized with grouped hydrologic drainage divisions in Table 12.5.

The New South Wales, Tasmania, and Western Australia water concentrations are natural choices for representing *SE Coast*, *Tasmania* and *SW Coast* drainage divisions respectively. The New South Wales water concentrations represent Sydney (and likely Melbourne) area waters, but may overestimate POPs flux from the rest of *SE Coast* drainage division.

Table 12.5 POPs water concentrations assigned to drainage divisions for river flux estimation.

Group	Drainage Divisions †	Area 10^3 km^2	Discharge km^3	% total discharge	Water concentrations ‡
A	*SE Coast*	274.0	41.9	11	New South Wales
B	*Tasmania*	68.2	52.9	14	Tasmania
C	*NE coast*	451.0	83.9		
	Murray-Darling	1,060.0	12.2		
	SW coast	315.0	6.6		
	S Australian Gulf	82.3	0.8		
	subtotal	1,908.3	103.5	28	Western Australia
D	*Gulf of Carpentaria*	641.0	92.5		
	Timor Sea	547.0	80.7		
	Indian Ocean	519.0	3.8		
	subtotal	1,707.0	177.0	47	Low decile

† from Table 12.1. ‡ from Iwata et al. (1994).

Table 12.6 Annual POPs exports by Australian rivers to coastal seas.

POP	Drainage Group	Low		High	
		Concentration pg/L	Load kg	Concentration pg/L	Load kg
PCBs	A	1,000	42	1,200	50
	B	65	3	100	5
	C	130	13	145	15
	D	29	5	50	9
	Australia		**64**		**79**
ΣDDT	A	320	13	430	18
	B	40	2	45	2
	C	20	2	35	4
	D	5	1	10	2
	Australia		**18**		**26**
ΣCHL	A	600	25	725	30
	B	25	1	40	2
	C	45	5	55	6
	D	5	1	25	4
	Australia		**32**		**43**
ΣHCH	A	540	23	570	24
	B	95	5	100	5
	C	240	25	300	31
	D	80	14	100	18
	Australia		**67**		**78**

Waters of the remote, relatively pristine Gulf of Carpentaria, Timor Sea, and Indian Ocean drainage divisions are expected to have low background POPs levels, and thus are taken as a group. For mass flux estimation, a low percentile, e.g., the first decile, from the ranked set of 20 Australian water samples for New South Wales, Tasmania and Western Australia defines suitably low concentrations.

This leaves *NE Coast* (Queensland), *Murray-Darling* and *South Australia Gulf* drainage divisions. Market fish (Section 12.3.1) from Atherton and Townsville areas of northern Queensland have contaminant levels similar to Perth area fish; hence, *NE Coast* drainage division is assigned Western Australian water concentrations. While higher concentrations are expected in Brisbane area waters, underestimation of Brisbane contributions should be more than compensated, by overestimation of *SE Coast* POPs flux.

In the absence of information, *Murray-Darling* and *South Australia Gulf* runoff was arbitrarily assigned the Western Australia water concentrations. Pesticide concentrations may be higher in *Murray-Darling* discharge, but discharge volume is quite low which limits the effects of errors.

POPs riverine flux estimates are presented in Table 12.6 by drainage division group with the water concentrations used in the calculations. *Low* and *high* scenarios correspond to using the geometric and arithmetic mean concentrations respectively for New South Wales, Tasmania and Western Australia water concentration data from Iwata et al. (1994). The flux estimates are very low. *These estimates should be regarded as* ***base flow flux estimates***. Higher contaminant delivery during high flow season and urban storm runoff could raise total annual POPs by a factor of 2 or more.

Chapter 13 New Zealand

13.1 Hydrology

The islands of New Zealand have a combined area of about 266,800 km^2. The two main islands, North Island and South Island, have areas of about 114,000 and 151,000 km^2 respectively. Located off the southern end of South Island, the 1,750 km^2 Stewart Island has also been included in the present analysis.

The New Zealand climate is humid. Over most of the country, mean annual precipitation ranges from 0.6 to 1.5 m. Dominant westerly air flows deposit heavy precipitation on western slopes, particularly those of the Southern Alps mountains on the South Island where mean annual precipitation can exceed 10 m; however, areas in the lee of the mountains can receive as little as 300 mm mean annual rainfall. Specific runoff maps (Duncan, 1987) show that roughly the western half of South Island has high mean annual unit runoff of 1.2–2.5 m, while the eastern half has unit runoff in the 0.3–1.2 m range with a few large areas having <0.3 m.

In contrast, on North Island, high specific runoff areas are scattered and constitute only a relatively small proportion, likely ≤10%, of land area. North Island unit runoff is mostly in the 0.3–1.2 m range with scattered areas having <0.3 m.

Available river discharge data have been compiled in Table 13.1. The selection of New Zealand rivers cited in the GLORI data file (Milliman et al., 1995) was heavily biased to high discharge, westward flowing sites of South Island. These were supplemented with additional rivers cited in Duncan (1987) and McMahon et al. (1992) Which gave better coverage of low discharge areas? As represented by available data, net unit runoff for North and South Island are 900 mm and 1.5 m respectively.

The lower part of Table 13.1 gives the total New Zealand discharges extrapolated from the gauged sites. The combined gauged drainage area and discharge represent only about 1/3 of the respective total area and discharge from New Zealand. The extrapolated discharge estimates are still thought to be mildly biased to high discharge areas. Duncan's report (1987) indicates that there is more extensive hydrometric data available that could be obtained to improve the estimates.

For contaminant flux estimation, South Island discharges are broadly distinguished by westward flowing waters that should be relatively uncontaminated, and eastward flowing waters that are expected to become contaminated by the agriculture and other human activity prevalent on the plains of the eastern half of the island. Very roughly, about 90 km^3 is discharged from the east side of the island, and about 140 km^3 from the west side.

Table 13.1 New Zealand hydrologic data.

	Total area $10^3 km^2$	Discharge km^3	Runoff mm
A. New Zealand discharge data at gauged sites.			
North Island			
Waikato	19.00	13.000	684
Wanganui	6.64	7.069	1,064
Mohaka	2.37	2.746	1,158
Hutt	0.64	2.600	4,063
Tarawera	0.91	1.041	1,149
Awanui	0.22	0.189	853
Porirua	0.04	0.031	760
Waitangi	0.02	0.001	48
subtotal	29.84	26.677	894
South Island			
Clutha	20.31	17.14	844
Waiau	8.20	17.00	2,073
Buller	6.35	13.51	2,127
Grey	3.90	9.30	2,385
Waitaki	9.70	7.40	763
Haast	1.02	6.41	6,281
Rakaia	2.60	6.20	2,385
Waimakariri	3.20	3.80	1,188
Karamea	1.20	3.50	2,917
Hokitika	1.10	3.10	2,818
Hooker	0.10	0.54	5,209
subtotal	57.68	87.89	1,524
NZ Total Gauged	87.52	114.56	1,309
B. New Zealand mean annual water discharge prorated to total areas.			
North Is.	114.05	101.96	894
South Is.	151.01	230.09	1,524
Stewart Is.	1.75	2.29	1,309
NZ total	266.80	334.34	1,253

13.2 POPs Usage in New Zealand

PCBs were imported into New Zealand up to about 1980. Richardson et al. (1986) give partial data for bulk imports to 1980. There were 143 t PCBs in electrical transformers and capacitors, and another 23 t PCBs imported for use as plasticizers. New Zealand has exported some PCBs to France for destruction by high temperature incineration. Under the New Zealand Ministry for the Environment's *Organochlorines Programme* [17], new destruction technologies are currently being evaluated for disposing hazardous wastes and PCBs.

Most POPs insecticides have likely seen use in New Zealand. The *Organochlorines Programme* appears to be most concerned with DDT, dieldrin and chlordane. DDT and dieldrin were used in horticulture and agriculture, while dieldrin and chlordane were used as timber treatments and termiticides. DDT was used extensively to control grass grub damage to pastures from 1947 to 1970 when the practice was prohibited (Boul et al., 1994). Quantities consumed have not been given. Regarding chlordane, Fox et al. (1988) noted that "*large* quantities of this pesticide have been added to plywood glues to prevent insect attack since the early 1960s." The term "*large*" is vague. New Zealand has about 400 timber treatment yards, but the number using chlordane is unknown (Simpson et al., 1995). A study of an Auckland timber yard where chlordane had been used showed that there had been contamination of adjacent aquatic systems via surface runoff of contaminated treatment yard soils.

13.3 POPs in New Zealand Aquatic Systems

13.3.1 POPs Runoff from Remote South Island Catchments

PCBs, HCB, α-HCH and γ-HCH were amongst contaminants studied in rainfall and runoff of South Island upland watersheds during 1985 (Herrmann, 1987). The study catchments were generally undisturbed; so that, the only POPs inputs were by atmospheric deposition. Rainfall originates from easterly moving air masses with long residence time in the atmosphere. Thus, POPs contaminants in rainfall would ostensibly indicate mid latitude southern hemispheric background levels about 1985.

The most detailed runoff data were collected in a small (2.64 ha) forest catchment designated M15 in the upper Grey River on the windward side of South Island where about 20 samples were collected over 4 rainfall events. Previous work had shown that stream runoff response to rainfall events was from a well-mixed groundwater reservoir with about 4 months mean residence time, and that runoff was mostly "old" water mixed with <25% "new" rain water. Water samples were also collected in Lake Tekapo in the headwaters of the Waitaki River that drains eastward on the lee side of the island.

[17] Organochlorines Programme, Ministry for the Environment, PO Box 10-362, Wellington, Phone (04) 4734 090, Fax (04) 4710 195; see also: http://www.mfe.govt.nz/organo.htm.

Table 13.2 POPs (ng/L) in remote South Island catchments; after Herrmann (1987).

	PCBs	α-HCH	γ-HCH	ΣHCH	HCB
M15 streamflow	1.11	0.08	0.59	0.67	13.5
L. Tekapo	<0.04	0.47	0.67	1.13	0.6

The water concentration data summarized in Table 13.2 reveal odd inconsistencies. The only variable exhibiting similar concentrations between the two sites is γ-HCH. PCBs and HCB had significantly concentrations in catchment M15 runoff than in Lake Tekapo waters, while α-HCH was much lower in M15 runoff than in Lake Tekapo. Also, PCBs of 1 ng/L and HCB of 14 ng/L seem remarkably high for southern hemispheric background levels. Though these may be valid data, they are very difficult to explain.

13.3.2 Manukau Harbour

POPs occurrence in Manukau Harbour, a large inlet (344 km^2) adjacent to Auckland city on the North Island has been studied by Fox et al. (1988) and Holland et al. (1993). The watershed includes urban, industrial, rural and native forest lands. In both studies, sediment samples were collected at mid-tide range. The first study showed higher levels of contamination than the second, partly because finer sediments were sampled. In the first study, the sampled sediments may still have been comparatively coarse material yielding deceptively low POPs concentrations. Water, suspended matter and biotic samples would have better indicated the status of these aquatic systems. Both articles refer to biotic surveys that suggest contamination may be more significant than indicated by these sediment samples.

13.3.2.1 First Survey

The earlier study (Fox et al., 1988) analyzed sediments at the inlets of 5 of Manukau tributary streams for 51 PCB congeners, DDT, HCH, drins, heptachlor, chlordane and HCB. Sampling dates were not given, but appear to be in 1987 or earlier. Contamination was generally low, and appeared to originate mainly from a site known as Mangere Inlet that is surrounded by industry and receives treated sewage effluents from domestic and industrial sources. Fox et al. reviewed other biological data for Manukau that suggests that at the relatively low contamination levels seen in sediments, POPs may be exerting effects on biota at 3 of the 5 sites sampled.

Summed concentrations of 51 PCBs ranged from 14.2 ng/g (dry weight) at Mangere Inlet down to 0.5 ng/g at the least contaminated site. Examination of congener profiles shows that the PCBs comprise certain low chlorinated PCBs (IUPAC nos. 18, 31, 40,

44, 49, 52) typical of mixtures such as Aroclors 1016 and 1242, or Clophens A30 and A40; plus higher chlorinated PCBs (IUPAC nos. 118, 138, 153, 159, 170, 180) more prevalent in Aroclors 1254 and 1260, or Clophens A50 and A60. Assuming an equally proportioned mixture of 4 PCB grades, total PCBs in the sediments are estimated as 2.35 times the sum of the 51 congeners. Thus total PCBs in Mangere Inlet sediments are about 33 ng/g dry weight, and about 12 ng/g at the next most contaminated site. This modest level of contamination exceeds some sediment quality guidelines and meets others.

After PCBs, chlordane and heptachlor were the most prominent POPs. Total chlordane [18] was estimated to range from 0.9–6.1 ng/g dry weight. These levels indicate modest contamination comparable to sediments of PNG, Solomon Islands, Taiwan and Perth, Western Australia. The most contaminated site was Mangere Inlet.

ΣDDT [19] was low, ranging from 1.2–2.3 ng/g dry weight. Though *o,p'*-DDT was not quantified, the isomer profile indicates old DDT mainly in the form of DDE and DDD isomers. HCH [20] levels were negligible at 3 sites, but reached 1.5–2.0 ng/g dry weight at Mangere Inlet and another site. Unlike other POPs, HCH quantities in sediment are relatively high, and values of 1.5–2.0 ng/g are similar to lower concentrations seen in India where HCH is heavily used. Thus there are indications of recent HCH, possibly lindane, use. Water concentrations are more reliable indicators of HCH presence. Of drins, only dieldrin was found at 0.3–0.5 ng/g dry weight indicating very low, and likely very old contamination. HCB levels were <0.1 ng/g, the detection limit.

13.3.2.2 Second Survey

Manukau Harbour was surveyed again in December (summer) 1989 by Holland et al. (1993). Investigators collected triplicate sediment samples at 5 sites on the north side of Manukau Harbour including 2 in Mangere Inlet. The sediments sampled were coarser than in the previous survey, and except for DDT at one site, contaminant concentrations were lower. Data showed a gradient that declined away from Mangere Inlet. The most contaminated site was in a bay where Mangere Inlet opens into Manukau harbour. DDT levels are 15-50 fold higher, and PCBs are 5–7 fold higher than at other sites. ΣDDT levels were 6–22 ng/g dry weight mostly in the form of DDD isomers. Under the strongest assumption that PCBs originated from an Aroclor 1260 mixture, total PCBs would range 4–12 ng/g dry weight. Other contaminants, chlordane, dieldrin and lindane, were all present at <1 ng/g dry weight.

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[18] Σchlordane = *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor + heptachlor + heptachlor epoxide; *cis*-nonachlor estimated as mean proportions in sediment data of Iwata et al. (1994).
[19] ΣDDT = *p,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD + *o,p'*-DDE + *o,p'*-DDD.
[20] ΣHCH = α-HCH + β-HCH + γ-HCH; β-HCH not detected at any site.

13.3.2.3 Chlordane Pollution by Timber Treatment Facility

Concern about the levels of chlordane observed in Manukau Harbour in previous studies, prompted an investigation that identified the primary source as a timber treatment facility near Mangere Inlet (Simpson et al., 1995). Chlordane was used in pressure treatments and in plywood adhesives. Surface runoff from the yard drains through a marsh into Mangere Inlet. In 1988, sediment samples from the drain exiting the yard showed chlordane levels of about 16 µg/g [21] ; however, in stream sediments at the outlet to Mangere Inlet, levels had fallen 3 orders to about 17 ng/g. About 1990, city officials excavated soil from the area in an attempt to decontaminate the site.

13.3.3 Waikareao Estuary, Tauranga Harbour

POPs contamination in shellfish and sediments has been studied in Waikareao Estuary, an inlet of Tauranga Harbour on the north coast of North Island (Burggraaf et al., 1996). The watershed is not described, but appears to be non-urban. Samples were collected at 10 sites. Chemical analyses included: DDT (*p,p'*-DDT, *p,p'*-DDE, *p,p'*-DDD); PCBs (9 congeners: IUPAC nos. 101, 110, 138, 149, 151, 153, 170, 180, 187); *cis*- and *trans*-chlordane; and dieldrin.

Sediments show modest contamination by PCBs and DDT, and virtually non-detectable (<100 pg/g dry weight) chlordanes and dieldrin. Congener specific PCB data were given for only one sediment sample. It had a congener profile similar to Clophen A60 or Aroclor 1260 for which total PCBs would be about 2.0-2.2 times the sum of concentrations of the 9 congeners. Thus, the mean and maximum total PCBs in Waikareao sediments would be about 9 and 36 ng/g dry weight respectively.

Amongst the shellfish, the 5 samples of filter feeding rock oyster *S. glomerata* do show moderate PCB contamination. Assuming that total PCBs are about 2 times the sum of the 9 congener concentrations, and converting reported dry weights to wet weights by a conversion factor of 0.15 [22], the estimated mean and range of total PCBs in Waikareao oyster are about 74 and 38–87 ng/g wet weight respectively. This is much lower than the mean total PCBs of 170 ng/g wet weight reported for U.S. Gulf of Mexico oysters for 1980, but higher than other U.S. sites, e.g., Chesapeake Bay 10 ng/g and South Carolina 50 ng/g [23]. PCB contamination appears to be more significant than indicated by sediments alone.

Mean and maximum ΣDDT in sediments are respectively about 1.2 and 5.4 ng/g dry weight that represent very low contamination. In oyster *S. glomerata*, the dry weight mean and range of ΣDDT = *p,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD were 104 and 62–176

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[21] About 10–12 µg/g *trans*-chlordane + *cis*-chlordane + *trans*-nonachlor + *cis*-nonachlor + heptachlor, plus 4.6 µg/g other chlordane components.
[22] Bivalves average about 15% dry matter.
[23] See Table 2 Ramesh et al. (1990) for a global comparison of PCBs, DDT and HCH in bivalves.

ng/g respectively. For the same range of dry weight concentrations, in American oyster *C. Virginica*, the *o,p'*- isomers of DDT and metabolites accounted for 50% [24]. Assuming similar relations hold for *S. glomerata*, the full dry weight ΣDDT including *o,p'*- isomers would be about twice the values given above. Converting to wet weight, the resulting wet weight means and ranges of the complete ΣDDT measurements for Waikareao *S. glomerata* are estimated as 33 and 20–56 ng/g. These concentrations are lower than reported for *C. Virginica* in U.S. Gulf of Mexico coastal waters, but higher than other areas such as Chesapeake Bay and South Carolina. DDT contamination in Waikareao oyster is moderate, but more significant than indicated by sediments alone.

While chlordane and dieldrin were not found in sediments, they were found in oyster *S. glomerata* at low, nearly constant concentrations. The wet weight means and ranges were 2.5 and 1.8–5.4 ng/g respectively for dieldrin, and 2.5 and 2.2–2.8 ng/g respectively for *cis*-chlordane + *trans*-chlordane. Had the full complement of chlordane isomers including *cis*- and *trans*-nonachlor, oxychlordane, heptachlor, heptachlor epoxide been determined, the level of chlordane contamination observed could have been 2-3 fold higher. Chlordane contamination of Waikareao estuary seems to be low.

13.4 DDT in New Zealand's Pasture Soils

Much of New Zealand's agricultural land is devoted to pasture. From 1947 to 1970, DDT was applied extensively to pastures to control grass grub damage. Long term field data from trials conducted at an agricultural experimental station on South Island from 1950 to 1970, show that significant DDT remained in pasture soils (Boul et al., 1994). Levels in 1990 ranged from about 1-2 μg/g, down from 2.6–6.4 μg/g in 1965. Though erosion losses from untilled pasture land will be relatively low, some losses to streams will likely continue for the indefinite future given the apparent longevity of DDT in New Zealand soils.

13.5 Potential POPs Exports from New Zealand's Rivers

The available data are inadequate to estimate riverine POPs flux from New Zealand. Though reliable load estimation is not possible, Table 13.3 based on the water concentrations from remote catchments gives a sense of POPs flux magnitudes that are possible from New Zealand. Given the estimated mean annual water discharge from New Zealand of 334 km^3, a mean New Zealand water concentration of 1 ng/L translates to flux of 334 kg. Even at the seemingly high concentrations of just above 1 ng/L observed for PCBs and ΣHCH in remote catchments by Herrmann (1987), the New Zealand flux would be only 375 kg. Applied across the country, the HCB concentration of 13.5 ng/L observed at catchment M15 yields an annual load of 4.5 t

[24] based on analysis of U.S. NOAA National Status and Trends Program (i.e., U.S. "Mussel Watch") data for *C. Virginica* obtained at http://seaserver.nos.noaa.gov/projects/nsandt/nsandt.html.

Table 13.3 Potential POPs exports by New Zealand's rivers.

	Concentration (ng/L)	Flux (kg)
PCBs	0.04–1.11	13–371
ΣHCH	0.67–1.13	224–378
HCB	0.60–13.5	201–4,514

that seems implausibly high for background conditions. The lower HCB concentration observed in Lake Tekapo may be more generally appropriate.

13.6 The New Zealand Organochlorines Programme

Under the *Organochlorines Programme*, comprehensive surveys of air, soil, river water, river biota, estuarine sediment and estuarine shellfish are under way [25]. The river surveys involved the collection of 3 samples from 16 sites representing about 13% of the country's surface drainage. Sampling was conducted from January to March 1996. The precise contaminant list was not given, but the information bulletin suggests that PCBs, DDT, dieldrin and chlordane will be amongst the variables measured. When these data become available, New Zealand river flux can be readily estimated.

[25] Organochlorines Programme, Ministry for the Environment, PO Box 10-362, Wellington, Phone (04) 4734 090, Fax (04) 4710 195; see also: http://www.mfe.govt.nz/organo.htm

Chapter 14 Other Countries

14.1 Iran

Iran generates about 40% (18 km^3) of the total Shatt al-Arab River mean annual water discharge via the Karun River that discharges to the head of the Shatt al-Arab delta. In the past, Iran may have contributed several *dirty dozen* POPs insecticides to Shatt al-Arab discharges. It is reported that Iran banned organochlorine insecticides except lindane in 1985 (Burgaz et al., 1995).

14.2 Pakistan

River discharge from Pakistan is virtually limited to the Indus River (Table 14.1). The Indus rises in Tibet and flows through the territory of Jammu and Kashmir before entering Pakistan. Several significant tributaries rise within Indian borders including the Jhelum, Chenab, Ravi and Sutlej. Under terms of the Indo-Pakistani water treaty of 1962, India retains certain rights to consumptive use of the latter tributaries, but ceded rights on the upper Indus River itself.

Within Pakistan, Indus River waters support one of the world's largest irrigation systems with 170,000 km^2 under irrigation in 1992 (CIA, 1996).

Table 14.1 Hydrologic data for Indus River.

Area km^2	Annual discharge km^3	Runoff mm
970,000	240	247

14.2.1 Pesticide Use

Pakistan began using pesticides in 1954. Historical POPs insecticide usage is not well documented. Endrin was used on sugar cane in the 1960s and 1970s, and "DDT was used abundantly" in the past (Jabbar et al., 1993). According to Soerjani (1988), about the mid 1980s, Pakistan annually produced about 2.6 Kt ai DDT and HCH in approximately equal amounts. Endrin has been banned since 1993 (Jabbar et al., 1993) or earlier. An informal report claims that as of 1995 or earlier, DDT, endrin and toxaphene were "banned" in Pakistan, while the status of HCH, aldrin, dieldrin,

heptachlor and chlordane was not known (PANNA, 1995a). The status of POPs insecticide usage for public health purposes in Pakistan is unknown.

14.2.2 PCBs

There are no data concerning PCB stocks or environmental occurrence of PCBs in Pakistan. It is likely that PCB laden electrical equipment was imported to the major cities.

14.2.3 POPs in the Pakistani Environment

Environmental POPs data for Pakistan were found only in three sources discussed below. Some POPs insecticide data are also available for waters of the Indus and tributaries upstream in India.

High DDT and HCH levels were found in human blood and fat samples from the city of Quetta, Baluchistan province (Krawinkel et al., 1989). Sampling dates are not given. Though Quetta lies outside the Indus watershed, the samples showed high exposure to HCH and DDT that was likely typical of much of Pakistan at the time [most likely between 1985 and 1989]. In humans, DDT and HCH levels were comparable to those observed in India (Jani et al., 1988). No HCB was detected in either blood or fat.

Groundwaters and soils were sampled in aldrin, dieldrin, endrin and DDT in the vicinity of Faisalabad (Jabbar et al., 1993). Sampling dates are not given, but are likely early 1990s. About 100-200 ng/L endrin was found in 3 of 10 groundwater samples. While deeper soils (2-3 feet) showed low levels (<1 – 10 ng/g, weight basis not given) of drins and DDT species, these organochlorines were not found in the surficial layer.

A general water quality survey of sites along the Degh Nala, a stream near Lahore, included sediment samples analyzed for DDT, HCH and HCB (Tehseen et al., 1994). DDT and HCB were observed at 8 µg/g and 95 ng/g wet weight in a sample from the downstream near field of the effluent outfall of a former pesticide manufacturing facility that now produces other chemicals. It is unclear whether the DDT and HCB originated from former activities or recent discharges. The DDT and HCB concentrations in sediments dissipate progressively downstream.

A survey in India over 1986-89 (Pathak et al., 1992) obtained 3 samples from the Indus River and its Tawi tributary in Jammu and Kashmir, and the Beas River in Himachal Pradesh. The Indus sample had 13 ng/L HCH, while the Beas River sample had 6 ng/L HCH and 25 ng/L DDT. Aldrin was not detected. The Beas River passes into India's Punjab state where intensive agricultural activity and high malaria incidence rates likely induce heavy DDT and HCH usage; such that, drainage waters entering Pakistan could be more contaminated than suggested by the Beas River sample.

14.2.4 Indus River POPs Exports

Available data are inadequate to estimate Indus River mass fluxes to the Arabian Sea. The only water concentration data available are a few samples for the Indus and tributaries upstream in India. To estimate Indus River contaminant loads, data are needed from the lower reach downstream of Hyderabad.

14.3 Bangladesh

Though the vast Ganges– Brahmaputra–Meghna river system debouches to the Bay of Bengal through Bangladesh, Bangladesh itself remains a small consumer of pesticides and a negligible contributor to POPs transported by these rivers. Total pesticide consumption for a 1988-89 accounting year was about 5.2 Kt of which about 395 t (7.6%) were unspecified organochlorines (Rahman et al., 1995)[26]. Total usage had climbed steadily from about 2.1 Kt in 1981-82. If increases continued at the same rate into the 1990s, 1995-96 consumption of organochlorines would exceed 600 t. As pesticide use rose, so did agricultural productivity. Through the 1980s, 8.5–9.2 million ha of potentially more than 14 million ha crop lands were cultivated (Rahman et al., 1990; Rahman et al., 1995). Demand for pesticides can only be expected to increase.

Precisely which organochlorines are currently in legal use is not known. DDT was banned over 20 years ago, while endrin and HCH (including lindane) have also been banned since at least 1985 (Soerjani, 1988). In 1985, aldrin, dieldrin, heptachlor, chlordane, and endosulfan had "restricted" status, and toxaphene was neither restricted nor banned (Soerjani, 1988). Some of these may still be used.

DDT and other banned pesticides are evidently available on the open market. Recently, authorities discovered that dried fish were being dusted or dipped in DDT to kill insect larva that infest the sun dried fish (IAEA, 1996). The extent of black market organochlorine insecticide usage is not known, and in the absence of hard information, is assumed to be negligibly small for present purposes.

14.4 Sri Lanka

Sri Lanka is a relatively small pesticide user consuming about 4 Kt (gross formulation weight) annually in the mid 1980s (Soerjani, 1988). Historically, Sri Lanka has had a high malaria incidence rate. In the 1950s, Sri Lanka engaged in the WHO

[26] Rahman et al. may have given gross formulation mass not active ingredient mass, in which case active ingredient mass will likely be only 25-50% of the cited figures. Also "unspecified organochlorines" may comprise mostly endosulfan and other low priority POPs.

sponsored campaign of anti-malarial DDT spraying, and virtually eradicated the disease by 1963. Spraying stopped in 1964, apparently due to lack of funds, and by 1970, malaria cases had returned to pre-treatment levels (Spindler, 1983). Despite the high malaria rate, Sri Lanka banned DDT some years ago. It is not known if other POPs insecticides are used for public health purposes. An informal report (PANNA, 1995a) suggests that most POPs insecticides are banned or restricted.

Some PCBs may have been imported to Sri Lanka, most likely to Columbo. Other than localized contamination in the Columbo area, PCBs exports by riverine discharges from Sri Lanka are expected to be minimal.

No environmental data for POPs occurrence in Sri Lanka were found.

14.5 Myanmar (Burma)

The Union of Myanmar, or Burma as it is still commonly known, remains politically isolated and relatively undeveloped. FAO data suggest that Burma for the present is a minimal user of agricultural pesticides (FAO, 1996). PCB holdings are likely negligible.

14.6 Philippines

The Philippine archipelago extends over 299,000 km^2. The two main islands Luzon and Mindanao account for almost exactly 2/3 of the total land area. Mean annual rainfall averages about 2.2 m, and ranges from about 1 m to 3.6 m. Luzon averages about 2.6 m rainfall, while Mindanao averages 1.7 m annually.

The GLORI river discharge file (Milliman et al., 1995) gives discharges and drainage areas for 18 rivers representing about 1/3 of the Philippine land area. The prorated discharge for the entire archipelago is 528 km^3. The corresponding mean annual runoff of 1.7 m constitutes a relatively high 78% of average annual rainfall. Discharge data or drainage areas may be in error. At 1/2 mean annual rainfall, mean annual Philippine water discharge would be 331 km^3.

14.6.1 POPs in the Philippines

Most POPs insecticides have seen use in the Philippines. Informal reports (PANNA, 1995a) suggest that in 1995, DDT, crude HCH, aldrin, dieldrin, endrin, heptachlor and toxaphene were banned, while restricted usage of chlordane and lindane was permitted. It is not known if POPs Insecticides are used for public health purposes.

Evidence of POPs presence in the environment is meagre. Hungspreugs (1988) cited the following water concentrations: aldrin (trace – 180 ng/L), dieldrin (10–88 ng/L), endrin (12–24 ng/L), *p,p'*-DDT (12 –112 ng/L), but indicates neither what waters (paddy, river, coastal) nor what time periods the data represent. Also given are some data for marine biota that suggest POPs contamination was relatively low, but again it is unclear what waters the data represent. Eng (1989) claimed that endrin in the Pampanga River that drains to Manila Bay exceeded 200 ng/L during the May to September planting season of 1988.

There are no data on Philippine PCB stocks, nor on the occurrence of PCBs in the local environment. Given the association with the west, and the USA in particular, it is probable that PCB laden electrical equipment was imported to the Philippines, likely to the large cities, chiefly Manila, and the former large U.S. military bases.

Currently, data are inadequate to estimate even crudely POPs fluxes by Philippine rivers. Given the high runoff, a potential for perceptibly large mass flux of contaminants exists. Assuming that the GLORI discharge data are correct, mean Philippine-wide water concentrations of 2 ng/L are sufficient to generate about 1 t of annual exports. The 200 ng/L endrin concentration for the Pampanga River would generate about 100 t endrin riverine exports if similar concentrations prevailed across the archipelago.

14.7 Papua New Guinea

With a large area [about 462,000 km^2], a very wet climate [2-4 m annual rainfall, >10 m in the highlands], Papua New Guinea [PNG] has a high annual water discharge of about 745 km^3. PNG remains sparsely populated and relatively undeveloped, but the country has the potential to become a significant consumer and emitter of persistent pollutants as development accelerates. POPs insecticides, notably DDT and dieldrin, have seen use since the 1950s (Mowbray, 1986; Mowbray, 1988). Mowbray reported that in 1982-83, more than 40 t DDT were used in PNG, and that there have been incidents of fish kills attributed to DDT and lindane usage. The fragmentary environmental data available suggest current contamination levels are low. Informal reports (PANNA, 1995a) suggest that the drins (aldrin, dieldrin, endrin) were banned in 1995 or earlier.

Port Moresby area POPs data exist for 3 sediment samples from local mangroves (Iwata et al., 1994), and fresh market fish, crab and oyster collected in a foodstuff survey (Kannan et al., 1994). Regional analyses [Chapter 2] suggest that PCBs, DDT, HCH and chlordane are present at comparatively low levels. DDT species were the most prominent contaminant seen in the limited sampling of sediments. The mean level of 50 ng/g is about 1/2 as much as in Bangkok canals, but higher than sediments of Djakarta area and Vietnam (exclusive of Ho chi Minh sewage canals) aquatic

systems. While PCBs, HCH and chlordane content are relatively low, there are clear signs that these chemicals have been used in the Port Moresby area.

In addition to PCBs, DDT, HCH and chlordane, the fish and shellfish data include measurements for aldrin, dieldrin, heptachlor, heptachlor epoxide, and HCB. Though, POPs in PNG fish rank medium to low against other countries of the region, there is clear evidence that these chemicals have seen some use in PNG. The leading contaminants in fish appear to be drins (virtually all as dieldrin in fish, but mostly aldrin in mud crab and oyster), and PCBs. The drin levels rank behind India, a significant user of aldrin at the time of the survey, but ahead of Thailand, previously a significant user. PCB levels rank behind Solomon Islands, Vietnam, and India's Ganges River; but ahead of India's Vellar River and Indian market fish. PCB contamination is low, but definitely present. DDT and chlordane levels are the lowest of any locality having data.

Outside Port Moresby, DDT was found in 1990 human breast milk samples from remote, and until recently unknown, highland villages near the western PNG frontier with Irian Jaya (Spicer and Kereu, 1993). DDT was seen in all samples from 41 females. Mean ΣDDT was about 890 ng/g fat that is relatively low an a global scale, but greater than anticipated. The villages have never been sprayed with DDT; however, vector control spraying was conducted during 1983-87 in the vicinity of a mining town about 2 1/2 days walk from the villages. For about 25% of samples, DDT > DDE + DDD, suggesting relatively recent exposure.

14.8 Solomon Islands

With an area of 27,600 km^2, the Solomon Islands located just east of PNG, are the largest of the small Pacific island nations. With a wet climate similar to PNG, the Solomons could generate annual surface discharge of about 44 km^3 (as much as the Shatt al-Arab River). The main island is Guadalcanal with an area of 6,500 km^2. The population is low (400,000) and the islands are mostly forested. Foodstuffs and pesticides have been imported from Australia in the past. Pesticide usage prior to about 1985 has been discussed by Mowbray (1986,1988).

Synoptic survey POPs data are available for water (6 samples), sediment (2 samples) and fish (10 specimens of 3 species) from the vicinity of Honiara, Guadalcanal (Iwata et al., 1994; Kannan et al., 1994). Regional comparisons reveal that water and sediment samples had low HCH, chlordane and PCBs; however, ***DDT levels in water samples were roughly tied with Vietnam behind India***. At the time of sampling in September 1990, DDT was very high at 4 of 6 Guadalcanal water sampling sites, and 1 of 2 sediment sampling sites. Moreover, the isomer profile showed mainly unmetabolized forms indicative of fresh inputs. Two rural sites showed signs of lindane (γ-HCH) usage, while the urban Honiara site showed perceptible chlordane and PCB concentrations.

Contrary to the waters and sediments, ***Honiara fish had PCB levels that ranked second only to Australia*** (representing mainly fish from the Sydney area), about equal to Vietnam, and ahead of India, PNG, and Thailand. PCBs levels were about 1 order lower than in Australian (Sydney) fish, and 1 order greater than Thai (Bangkok) fish. DDT levels were not as high as expected considering the water and sediment data. Concentrations are in the range seen in the Vellar River south India where DDT usage has been relatively low. However, Honiara market fish have unequivocally higher DDT levels than fish from PNG and Thailand. Though POPs levels are medium to low for HCH, DDT, chlordane, heptachlor, drins and HCB, evidence confirms that all have been used to some extent.

14.9 South Korea

There are no readily available POPs usage or environmental occurrence data for South Korea (Republic of Korea). According to Soerjani (1988), DDT, HCH, aldrin, dieldrin, endrin, heptachlor and chlordane were prohibited in South Korea by the mid 1980s. An informal report (PANNA, 1995a) claims that toxaphene was also banned in 1995 or earlier.

The chief concern posed by South Korea is potential emissions from improperly disposed PCB stocks. No data are available concerning South Korean PCB holdings; however, given the level of industrialization and the long association with the west, stocks could be significant. No readily accessible data exist on the occurrence of PCBs in the South Korean environment. Mean annual river discharge to the sea from South Korea (Republic of Korea) is about 55 km^3. If PCBs concentrations in surface waters were similar to Japan's Yodo River (20 ng/L), the annual PCB riverine flux from South Korea would be about 1 t to coastal waters.

14.10 North Korea

There are no POPs usage or environmental data available for North Korea (Democratic Republic of Korea). POPs emissions from North Korea are thought to be negligible, but it is possible that past political association with the former USSR may have lead to the importation of some POPs insecticides and PCBs.

14.11 Hong Kong

Hong Kong has a land area of 990 km^2, and mean annual rainfall of 2.12 m. At 50% runoff, the annual surface water discharge from the territory is about 1.1 km^3, within a likely range of about 0.77–1.64 km^3.

Surveys of human breast milk in 1976 and 1985 showed Hong Kong females to have amongst the highest levels of DDT and HCH isomers reported anywhere (Ip and Phillips, 1989). Prodigious local consumption of insecticides may have been the main source. Over a 25 month period in the late 1980s, net imports were 736 t DDT, 25 t crude HCH, 12 t lindane (γ-HCH), and 1 t dieldrin (Ip, 1990). Inspection of foodstuffs imported from China has shown few detections of organochlorines (Phillips, 1989); however, these results seemingly contradict the results of recent Chinese foodstuff surveys (Chen and Gao, 1993; see Chapter 5). DDT and HCH may also have originated from seafood contaminated by insecticides in the discharges of the Xijiang (Pearl) River that exits to the sea adjacent to Hong Kong. Since China banned DDT and HCH in 1983, this DDT source should have decreased significantly. Over 1987-88, Hong Kong appears to have banned aldrin, dieldrin, DDT and HCH.

Victoria Harbour sediment data from 1993 (Table 14.2) show moderate levels of DDT and HCH contamination. The DDT occurred mostly as metabolized forms, except for the highest sample (97 ng/g dry) which suggested recent inputs. Except for a few high samples, HCH levels were generally very low.

For PCBs, sediment data show modest contamination in Victoria and Tolo Harbours, but very high concentrations in Junk Bay where industry and a large landfill of domestic and industrial waste are thought to be sources. In 1988, Hong Kong was expected to have introduced a *Code of Practice for Handling, Transport and Disposal of PCBs* (Ip, 1990). The status of the proposed code is unclear. Hong Kong may have significant PCB holdings.

Table 14.2 POPs in Hong Kong sediments (ng/g dry weight).

	POP	Mean	G-mean	Min.	Max.
Victoria Harbour	ΣHCH	1.81	0.46	0.06	9.43
	ΣDDT	22.72	12.39	1.38	97.00
	PCBs	18.6	11.7	3.2	81.0
Tolo Harbour	PCBs	18.5	14.6	6.6	45.0
Junk Bay	PCBs	492.0	223.0	31.0	2,200.0

Sources: Victoria Harbour (Hong et al., 1995); Junk Bay and Tolo Harbour (Kannan et al., 1989).
ΣHCH = α-HCH + β-HCH + γ-HCH + δ-HCH.
ΣDDT = *p,p'*-DDT + *p,p'*-DDE + *p,p'*-DDD.

References

Agarwal, H.C., Kaushik, P.K., and Pillai, M.K.K. 1987. Organochlorine insecticide residues in the rain water in Delhi, India. Water Air Soil Pollut. 32:293-302.

Agarwal, H.C., Mittal, P.K., Menon, K.B., and Pillai, M.K.K. 1986. DDT residues in the River Jamuna in Delhi, India. Water Air Soil Pollut. 28:89-104.

Agnihotri, N.P., Gajbhiye, V.T., Kumar, M., and Mohapatra, S.P. 1994. Organochlorine insecticide residues in Ganga River water near Farrukhabad, India. Environ. Monit. Assessment. 30:105-112.

Ahlborg, U.G., Becking, G.C., Birnbaum, L.S., Brouwer, A., Derks, H.J.G.M., Feeley, M., Golor, G., Hanberg, A., Larsen, J.C., Liem, A.K.D., Safe, S.H., Schlatter, C., Wærn, F., Younes, M., and Yrjänheikki, E. 1994. Toxic equivalency factors for dioxin-like PCBs. Chemosphere. 28(6):1049-1067.

Ahmad, N., Harsas, W., Maroit, R.S., Morton, M., and Pollack, J.K. 1988. Total DDT and dieldrin content of human adipose tissue. Bull. Environ. Contam. Toxicol. 41:802-808.

Allender, W.J. 1989. Organochlorine pesticide residues on grain storages of New South Wales. Bull. Environ. Contam. Toxicol. 42:603-608.

Al-Omar, M.A., Al-Ogaily, N.H., and Shebil, D.A. 1986. Residues of organochlorine insecticides in fish from polluted water. Bull. Environ. Contam. Toxicol. 36:109-113.

Al-Omar, M.A., Al-Ogaily, N.H., and Shebil, D.A. 1989. Effect of a sewage plant in the distribution of organochlorine residues in the Diyala River, Iraq. Water Air Soil Pollut. 44(1-2):1-7.

Ang, C., Meleady, K., and Wallace, L. 1989. Pesticide residues in drinking water in the north coast region of New South Wales, Australia, 1986-87. Bull. Environ. Contam. Toxicol. 42:595-602.

Anon. 1992. *Report Prepared by the Industrial Toxicology Research Centre, Lucknow, for the Ganga Action Plan of the Ministry of Environment and Forests, Government of India.* Industrial Toxicology Research Centre, Lucknow.

ANRA. 1995. Building industry adjusts to the phase out of organochlorines NRA News, vol. 2(2). Australian National Registration Authority, http://www.dpie.gov.au/nra/nra2_082.html.

Arnold, S.F., Klotz, D.M., Collins, B.M., Vonier, P.M., Guillette, L.J.J., and McLachlan, J.A. 1996. Synergistic activation of estrogen receptor with combinations of environmental chemicals. Science. 272:1489-1492.

AWRC. 1987. *1985 Review of Australia's Water Resources and Water Use. Vol. 1 Water Resources Data Set.* Australian Water Resources Council, Australian Government Publishing Services, Canberra, ACT.

Bakre, P.P., Misra, V., and Bhatnagar, P. 1990a. Organochlorine residues in water from the Mahala water reservoir, Jaipur, India. Environ. Pollut. 63:275-281.

Bakre, P.P., Misra, V., and Bhatnagar, P. 1990b. Residues of organochlorine insecticides in fish from the Mahala water reservoir, Jaipur, India. Bull. Environ. Contam. Toxicol. 45:394-398.

Battu, R.S., Singh, P.P., Joia, B.S., and Kalra, R.L. 1989. Contamination of bovine (buffalo, *bubalus bubalis* (L.)) milk from indoor use of DDT and HCH in malaria control programmes. Sci. Tot. Environ. 86:281-287.

Béland, P., deGuise, S., Girard., C., Lagacé, A., Martineau, D., Michaud, R., Muir, D.G., Norstrom, R.J., Pelletier, É., Ray, S., and Shugart, R. 1993. Toxic compounds and reproductive effects in St. Lawrence beluga whales. J. Great Lakes Res. 19(4):766-775.

Bhatnagar, V.K., Patel, J.S., Variya, M.R., Venkaiah, K., Shah, M.P., and Kashyp, S.K. 1992. Levels of organochlorine insecticides in human blood from Ahmedabad (Rural) India. Bull. Environ. Contam. Toxicol. 48:302-307.

Bolton, H.S., Breteler, R.J., Vigon, B.W., Scanlon, J.A., and Clark, S.L. 1985. *National perspective on sediment quality*. Battelle, Washington Environmental Program Office EPA Contract No. 68-01-6986, Washington, DC.

Boon, J.P., Everaarts, J.M., Kastoro, W.W., Razak, H., Sumanta, I., Sumarno, Nelissen, P.H., Stefels, J., and Hillebrand, M.T.J. 1989. Cyclic organochlorines in epibenthic organisms from coastal waters around East Java. Neth. J. Sea Res. 23(4):427-439.

Boul, H.L., Garnham, M.L., Hucker, D., Baird, D., and Aislabie, J. 1994. Influence on agricultural practices on the levels of DDT and its residues in soil. Environ. Sci. Technol. 28(8):1397-1402.

Bradley, D.J. 1991. Malaria - whence and whither? *In:* G. A. T. Targett (ed.) *Malaria, Waiting for the Vaccine*. Wiley, Chichester. pp. 11-29.

Brodie, J.E., Hicks, W.S., Richards, G.N., and Thomas, F.G. 1984. Residues related to agricultural chemicals in the groundwater of the Burdekin River delta, North Queensland. Environ. Pollut. B 8(3):187-215.

Burgaz, S., Afkham, B.L., and Karakaya, A.E. 1995. Organochlorine pesticide contaminants in human adipose tissue collected in Tebriz (Iran). Bull. Environ. Contam. Toxicol. 54:546-553.

Burgaz, S., Afkham, B.L., and Karakaya, A.E. 1994. Organochlorine pesticide contaminants in human adipose tissue collected in Ankara (Turkey) 1991-1992. Bull. Environ. Contam. Toxicol. 53:501-508.

Burggraaf, S., Wilkins, A.L., Langdon, A.G., and Wilcock, R.J. 1996. Organochlorine compounds in three species of shellfish from Waikareao estuary, Tauranga Harbour, New Zealand. Bull. Environ. Contam. Toxicol. 57:111-117.

Burt, J.S., and Ebell, G.F. 1995. Organic pollutants in mussels and sediments of the coastal waters off Perth, Western Australia. Mar. Pollut. Bull. 30(11):723-732.

Buser, H.-R., and Müller, M.D. 1993. Enantioselective determination of chlordane components, metabolites, and photoconversion products in environmental samples using chiral high-resolution gas chromatography and mass spectrometry. Environ. Sci. Technol. 27(6):1211-1220.

Chapin, G., and Wasserstrom, R. 1981. Agricultural production and malaria resurgence in Central America and India. Nature. 293:181-185.

Chen, J., and Gao, J. 1993. The Chinese total diet study in 1990. Part I. Chemical contaminants. J. Assoc. Off. Anal. Chem. 76(6):1193-1205.

Chen, P.H., and Hsu, S.-T. 1986. PCB poisoning from toxic rice-bran oil in Taiwan. *In:* J. S. Waid (ed.) *PCBs and the Environment, Volume III*. CRC Press, Inc., Boca Raton, FL. pp. 28-37.

Chowdhury, K.R., and Bhuiya, A.H. 1990. Environmental processes: flooding, river erosion, siltation, and accretion - physical impacts. *In:* A. A. Rahman, S. Huq and G. R. Conway (eds.) *Environmental Aspects of Surface Water Systems of Bangladesh*. University Press, Dhaka. pp. 93-99.

Chu, S.-G., Xi, Z.-Q., and Xu, X.-B. 1995. Polychlorinated biphenyl (PCB) congeners in mussel and other mollusc from Da Chen Island, East China Sea. Bull. Environ. Contam. Toxicol. 55:682-688.

Chu, Y.I., and Lin, C.Y. 1983. Pesticide use in Taiwan - present and future Symposium on Tropical Agricultural Research. Tropical Agriculture Research Center, Ministry of Agriculture, Forestry and Fisheries, Japan, Kyoto.

CIA. 1996. The World Factbook 1995. U.S. Central Intelligence Agency, http://www.odci.gov/cia /publications/pubs.html.

Colborn, T., vom Saal, F.S., and Soto, A.M. 1993. Developmental effects of endocrine-disrupting chemicals in wildlife and humans. Environ. Health Persp. 101(5):378-384.

Cummins, J.E. 1988. Extinction: The PCB threat to marine mammals. Ecologist. 18:193-195.

David, B. 1992. *Pest Management and Pesticides: Indian Scenario*. Namrutha Publications, Madras.

David, B.V., Tamilselvan, C., and Manickavasagam, S. 1993. Bioaccumulation of pesticides in the environment. *In:* G. S. Dhaliwal and B. Singh (eds.) *Pesticides: their ecological impact in developing countries*. Commonwealth Publishers, New Delhi. pp. 81-111.

Dearth, M.A., and Hites, R.A. 1991. Complete analysis of technical chlordane using negative ionization mass spectrometry. Environ. Sci. Technol. 25(2):245-254.

Dhaliwal, G.S., and Pathak, M.D. 1993. Pesticides in the developing world: a boon or a bane. *In:* G. S. Dhaliwal and B. Singh (eds.) *Pesticides: their ecological impact in developing countries*. Commonwealth Publishers, New Delhi. pp. 1-29.

Dikshith, T.S.S., Raizada, R.B., Kumar, S.N., Srivastava, M.K., Kulshrestha, S.K., and Adholia, U.N. 1990. Residues of DDT and HCH in major sources of drinking water in Bhopal, India. Bull. Environ. Contam. Toxicol. 45:389-393.

Djuangsih, N. 1993. Understanding the state of river basin management from an environmental toxicology perspective: an example from water pollution at Citarum River Basin, West Java, Indonesia. Sci. Tot. Environ. Suppl. 1993:283-292.

Djuangsih, N., and Salim, H. 1994. Preliminary study on the environmental pollution in the Citarum River basin, West Java, Indonesia. *In:* B. Widianarko, K. Vink and N. M. van Straalen (eds.) *Environmental Toxicology in South East Asia*. VU University Press, Amsterdam. pp. 277-288.

DouAbul, A.A.Z., Al-Omar, M.A., Al-Obaidy, S., and Al-Ogaily, N. 1987. Organochlorine pesticide residues in fish from the Shatt-al-Arab River, Iraq. Bull. Environ. Contam. Toxicol. 38:674-680.

DouAbul, A.A.Z., and Al-Saad, H.T. 1987. Residues of organochlorine pesticides in environmental samples from the Shatt-al-Arab River, Iraq. Environ. Pollut. 43(3):175-187.

DouAbul, A.A.Z., Al-Saad, H.T., Al-Timari, A.A., and Al-Rekabi, H.N. 1988. Tigris-Euphrates delta: a major source of pesticides to the Shatt al-Arab River (Iraq). Arch. Environ. Contam. Toxicol. 17(3):405-418.

Dua, V.K., Pant, C.S., and Sharma, V.P. 1994. HCH residues in rain water from Hardwar, India. Bull. Environ. Contam. Toxicol. 52:797-801.

Duncan, M.J. 1987. River hydrology and sediment transport, Chapter 4. *In:* A. B. Viner (ed.) *Inland Waters of New Zealand*, vol. DSIR Bulletin 21. DSIR Science Information Publishing Centre, Wellington, New Zealand. pp. 113-137.

Dureja, P., Nair, A., and Pillai, M.K.K. 1991. Aldrin and dieldrin in maternal serum, cord serum and breast milk in human samples from Delhi. Intern. J. Environ. Anal. Chem. 44:253-256.

Eichelberger, J.W., and Lichtenberg, J.J. 1971. Persistence of pesticides in water. Environ. Sci. Technol. 5(6):541-544.

Eng, C.T., Paw, J.N., and Guarin, F.Y. 1989. The environmental impact of aquaculture and the effects of pollution on coastal aquaculture development in southeast Asia. Mar. Pollut. Bull. 20:335-343.

Environment Canada. 1992. *Interim criteria for quality assessment of St. Lawrence River sediment*. St. Lawrence Action Plan, St. Lawrence Centre and Ministere de l'Environnement du Quebec. ISBN 0-662-19849-2, ,

Erickson, M.D. 1986. *Analytical Chemistry of PCBs*. Butterworth, Boston.

FAO. 1996. Statistics on pesticide usage (around 1990-93). Food and Agriculture Organization of the United Nations, Statistical Analysis Service (ESSA), Statistics Division, Rome, Italy, http://www.fao.org/WAICENT/faoinfo/economic/pesticid.htm.

FASE. 1996. *Exporting Risk: Pesticide Exports from US Ports, 1992-1994*. Foundation for Advancements in Science and Education, Research Report, , Los Angeles.

Fischer, R.C., Krämer, W., and Ballschmiter, K. 1991. Hexachlorocyclohexane isomers as markers in the water flow of the Atlantic Ocean. Chemosphere. 23(7):889-900.

Fox, M.E., Roper, D.S., and Thrush, S.F. 1988. Organochlorine contaminants in surficial sediments of Manukau Harbour, New Zealand. Mar. Pollut. Bull. 19:333-336.

Frank, R., Holdrinet, M., Braun, H.E., Thomas, R.L., Kemp, L.W., and Jaquet, J.-M. 1977. Organochlorine insecticides and PCBs in sediments of Lake St. Clair (1970 and 1974) and Lake Erie (1971). Sci. Tot. Environ. 8:205-277.

Frank, R., Thomas, R.L., Braun, H.E., Gross, D.L., and Davies, T.T. 1981. Organochlorine insecticides and PCB in surficial sediments of Lake Michigan (1975). Sci. Tot. Environ. 7(1):42-50.

Frank, R., Thomas, R.L., Holdrinet, M., Kemp, L.W., Braun, H.E., and Dawson, R. 1979. Organochlorine insecticides and PCBs in sediments of Lake Huron (1969) and Georgian Bay and North Channel (1973). Sci. Tot. Environ. 13:101-117.

Gajbhiye, V.T., Naveeen, K., Mohapatra, S.P., and Agnihotri, N.P. 1995. Non-point pollution of Ganga River with pesticides: a quantitative approach. J. Environ. Systems. 24(1):95-110.

Giesy, J.P., Ludwig, J.P., and Tillitt, D.E. 1994. Deformities in birds of the Great Lakes region: assigning causality. Environ. Sci. Technol. 28(3):128A-135A.

Goldberg, E.D. 1975. Synthetic organohalides in the sea. Proc. R. Soc. Lond. B. 189:277-289.

Gopal, B., and Sah, M. 1993. Conservation and management of rivers in India: case-study of the River Yamuna. Environ. Conserv. 20(3):243-254.

Gupta, P.K. 1986. *Pesticides in the Indian Environment*. Interprint, New Delhi, India, pp. 206.

Haldar, P., Kole, R.K., Bhattacharya, P., Chowdhary, A., and Adityachaudhury, N.C. 1990. Studies on residues of BHC isomers (α, β, γ, and δ) occurring in Ganga water. Pollution Research. 9:51-56.

Haldar, P., Raha., P., Bhattacharya, P., Choudhury, A., and Adityachaudhury, N.C. 1989. Studies on residues of DDT and endosulfan occurring Ganga water. Indian J. Environ. Health. 31(2):156-161.

Herrmann, R. 1987. Temporal variations of some trace organics (HCB, HCH, PCB and PAH) in rainfall, runoff and in lake sediments in remote sites of New Zealand. Catena. 14:233-248.

Hillebrand, M.T.J., Everarts, J.M., Razak, H., Moeylo, D.M., Stolwij, K., and Boon, J.P. 1989. Input of selected chlorinated hydrocarbons into the coastal area of east Java and adjacent waters: Distribution patterns in the dissolved and suspended phase. Neth. J. Sea Res. 23:369-377.

Hirai, Y., and Tomokuni, K. 1989. Levels of chlordane in water and sediment of rivers around Saga City, Japan. Bull. Environ. Contam. Toxicol. 42(4):589-594.

Hitch, R.K., and Day, H.R. 1992. Unusual persistence of DDT in some western USA soils. Bull. Environ. Contam. Toxicol. 48:259-264.

Hoekstra, P., Nolting, R.F., and van der Sloot, H.A. 1989. Supply and dispersion of water and suspended matter of the rivers Solo and Brantas into the coastal waters of East Java, Indonesia. Neth. J. Sea Res. 23(4):501-515.

Holland, P.T., Hickey, C.W., Roper, D.S., and Trower, T.M. 1993. Variability of organic contaminants in inter-tidal sandflat sediments from Manukau Harbour, New Zealand. Arch. Environ. Contam. Toxicol. 25:456-463.

Hong, H., Xu., L., Zhang, L., Chen, J.C., Wong, Y.S., and Wan, T.S.M. 1995. Environmental fate and chemistry of organic pollutants in the sediment of Xiamen and Victoria Harbours. Mar. Pollut. Bull. 31(4-12):229-236.

Huang, C.-W., Miyata, H., Lu, J.-R., Ohta, S., Chang, T., and Kashimoto, R. 1992. Levels of PCBs, PCDDs and PCDFs in soil samples from incinerator sites for metal reclamation in Taiwan. Chemosphere. 24(11):1669-1676.

Hungspreugs, M. 1988. Heavy metals and other non-oil pollutants in southeast Asia. Ambio. 23(3):178-182.

Hungspreugs, M., Dharmvanij, S., Utoomprukporn, W., and Windom, H.L. 1990. A comparative study of the trace metal fluxes of the Bang Pakong and the Mae Klong Rivers, Thailand. Sci. Tot. Environ. 97/98:89-102.

Hungspreugs, M., Utoomprukporn, W., Dharmvanij, S., and Sompongchaiyakul, P. 1989. The present status of the aquatic environment of Thailand. Mar. Pollut. Bull. 20(7):327-332.

Hutchinson, J.D., and Simmonds, M.P. 1994. Organochlorine contamination in pinnipeds. Rev. Environ. Contam. Toxicol. 136:123-167.

IAEA. 1996. Bangladesh: More dried fish, not DDT. INSIDE Technical Cooperation, 1(1). International Atomic Energy Agency, Vienna, http://www.iaea.or.at/programs/tc/tcflyer/itc2bng.htm.

IARC. 1979. *Some Halogenated Hydrocarbons*, vol. 20. International Agency for Research on Cancer, Lyons.

IARC. 1985. *Hexachlorobenzene: Proceedings of an International Symposium Held at IARC, Lyon, France 24-28 June 1985*. International Agency for Research on Cancer, Oxford University Press [distributor], Oxford.

Ip, H.M.H. 1990. Chlorinated pesticides in foodstuffs in Hong Kong. Arch. Environ. Contam. Toxicol. 19:291-296.

Ip, H.M.H., and Phillips, D.H. 1989. Organochlorine chemicals in human breast milk in Hong Kong. Arch. Environ. Contam. Toxicol. 18:490-494.

Iwata, H., Tanabe, S., Sakai, N., Nishamura, A., and Tatsukawa, R. 1994. Geographical distribution of persistent organochlorines in air, water and sediments from Asia and Oceania, and their implications for global redistribution from lower latitudes. Environ. Pollut. 85:15-33.

Iwata, H., Tanabe, S., Sakai, N., and Tatsukawa, R. 1993. Distribution of persistent organochlorines in the oceanic air and surface seawater and the role of ocean on their global transport and fate. Environ. Sci. Technol. 127:1080-1098.

Jabbar, A., Masud, S.Z., Parveen, Z., and Ali, M. 1993. Pesticide residues in cropland soils and shallow groundwater in Punjab Pakistan. Bull. Environ. Contam. Toxicol. 51:268-273.

Jani, J.P., Patel, J.S., Shah, M.P., Variya, M.V., Shah, Y.H., Gupta, S., and Kashyap, S.K. 1988. Levels of dichlorodiphenyltrichloroethane and hexachlorocyclohexane in human adipose tissue of the Indian population. Scand. J. Work. Environ. Health. 14:201-204.

Jani, J.P., Raiyani, C.V., Mistry, J.S., Patel, J.S., Desai, N., M,, and Kashyap, S.K. 1991. Residues of organochlorine pesticides and polycyclic aromatic hydrocarbons in drinking water of Ahmedabad City, India. Bull. Environ. Contam. Toxicol. 47:381-385.

Kannan, K., Sinha, R.K., Tanabe, S., Ichihashi, H., and Tatsukawa, R. 1993a. Heavy metals and organochlorine residues in Ganges River dolphins from India. Mar. Pollut. Bull. 26(3):159-162.

Kannan, K., Tanabe, S., Quynh, H.T., Hue, N.D., and R., T. 1992a. Residue pattern of dietary intake of persistent organochlorine compounds in foodstuffs of Vietnam. Arch. Environ. Contam. Toxicol. 22:367-374.

Kannan, K., Tanabe, S., Ramesh, A., Subramanian, A.N., and Tatsukawa, R. 1992b. Persistent organochlorine residues in foodstuffs from India and their implications on human dietary exposure. J. Agric. Food Chem. 40:518-524.

Kannan, K., Tanabe, S., and Tatsukawa, R. 1995. Geographical distribution and accumulation features of organochlorine residues in fish in tropical Asia and Oceania. Environ. Sci. Technol. 29(10):2673-2683.

Kannan, K., Tanabe, S., Tatsukawa., R., and Sinha, R.K. 1994a. Biodegradation capacity and residue pattern of organochlorines in Ganges River dolphins from India. Toxicol. Environ. Chem. 42:249-261.

Kannan, K., Tanabe, S., Williams, R.J., and Tatsukawa, R. 1994b. Persistent organochlorine residues in foodstuffs from Australia, Papua New Guinea, and the Solomon Islands: contamination levels and human dietary exposure. Sci. Tot. Environ. 153:29-49.

Kannan, N., Schulz-Bull, D.E., Petrick, G., and Duinker, J.C. 1993b. High resolution PCB analysis of Kanechlor, Phenoclor and Sovol mixtures using multidimensional gas chromatography. *In:* J. Albaigés (ed.) *Environmental Analytical Chemistry of PCBs*. Gordon and Breach Science Publishers, Yverdon, Switzerland. pp. 289-303.

Kannan, N., Tanabe, S., Okamoto, T., Tatsukawa, R., and Phillips, D.J.H. 1989. Polychorinated biphenyls (PCBs) in sediments in Hong Kong: a congener-specific approach to the study of coplanar PCBs in aquatic systems. Environ. Pollut. 62:223-235.

Kaushik, C.P. 1989. Loss of HCH from surface soil layers under subtropical conditions. Environ. Pollut. 59:253-264.

Kaushik, C.P., Pillai, M.K.K., Raman, A., and Agarwal, H.C. 1987. Organochlorine insecticide residues in air in Delhi, India. Water Air Soil Pollut. 32:63-76.

Kawabe, K., Maegawa, A., Yamanaka, S., and Nomura, K. 1992. Distribution of DDTs in the sediment of Lake Biwa. Water Sci. Technol. 25(11):93-98.

Kawai, T., H., S., Nojiri, Y., Tanaka, A., Furuta, N., Horiuchi, S., Ambe, Y., Soma, M., and Otsuki, A. 1992. Studies on methods for long-term monitoring of environmental pollutants in a remote region - Lake Mashu, Japan. Water Sci. Technol. 25(11):213-230.

Kawano, M., Inoue, T., Hidaka, H., and Tatsukawa, R. 1984. Chlordane compound residues in Weddell seals (*Leptonychotes weddelli*) from the Antarctic. Chemosphere. 13(1):95-100.

Kawano, M., Ramesh, A., Thao, V.D., Tatsukawa, R., and Subramanian, A.N. 1992. Persistent organochlorine insecticide residues in some paddy, upland and urban soils of India. Intern. J. Environ. Anal. Chem. 48:163-174.

Kinimonth, W.R. 1983. Variability of rainfall over northern Australia. *In:* A. Street-Perrott (ed.) *Variations in the Global Water Budget*. Reidel. pp. 265-272.

Krawinkel, M.B., Piehn, G., Kruse, H., and Kasi, A.M. 1989. Organochlorine residues in Baluchistan / Pakistan: blood and fat concentrations in humans. Bull. Environ. Contam. Toxicol. 43:821-826.

Kulshrestha, S.K. 1991. Pesticide residues in the Indian aquatic environment. *In:* K. V. Sastry, U. H. Mane, S. Sahai, A. K. Awasthi, P. B. Deshmukh and A. K. Gupta (eds.) *Environmental Pollution and Resources of Land and Water*. The Academy of Environmental Biology, Muzzaffarnagar. pp. 43-53.

Kumar, S., Lal, R., and Bhatnagar, P. 1988. Residues of organochlorine insecticides in two lakes of Jaipur, India. Water Air Soil Pollut. 42:57-65.

Kuwatsuka, S., Moon, Y.-H., Oshawa, K., Yamamoto, I., Hartadi, S., Sastrohamidjoyo, H., Untung, K., Arya, N., and Sumiartha, K. 1986. Comparative studies of on pesticide residues in soil, water and vegetables in Indonesia and Japan *Proceedings of the International Seminar on Comparative Agricultural Studies in Southeast Asia, Denpasar, Jan 20-24*. pp. 121-133.

Li, G.C., and Lee, T.L. 1983. The residual level of chlorinated hydrocarbon insecticides in different localities of Taiwan. Plant Prot. Bull. (Taiwan, R.O.C.). 25:271-279.

Ling, Y.-C., Soong, D.-K., and Lee, M.-K. 1995. PCDD/DFs and coplanar PCBs in sediment and fish samples from the Er-Jen River in Taiwan. Chemosphere. 31(3):2863-2872.

Liu, J.K., and He, X. 1987. Quantitative and quantitative aspects of fish crop in relation to environmental quality. Ecotoxicol. Environ. Safety. 13:61-75.

Loganathan, B.G., and Kannan, K. 1994. Global organochlorine contamination trends: an overview. Ambio. 23(3):187-191.

Loganathan, B.G., Tanabe, S., Goto, M., and Tatsukawa, R. 1989. Temporal trends of organochlorine residues in lizard goby *Rhinogobius flumineus* from River Nagaragawa, Japan. Environ. Pollut. 62:237-251.

Loganathan, B.G., Tanabe, S., Hidaka, Y., Kawano, M., Hidaka, H., and Tatsukawa, R. 1993. Temporal trends in persistent organochlorine residues in human adipose tissue from Japan, 1928-1985. Environ. Pollut. 81:31-39.

MacDonald, D.D. 1994. *A Review of Environmental Quality Criteria and Guidelines for Priority Substances in the Fraser River Basin.* Environment Canada, Environmental Conservation Branch, Pacific and Yukon Region DOE FRAP 1994-30, North Vancouver, British Columbia.

McDougall, K.W., Ahmad, N., Harris, C.R., and Higginson, F.R. 1989. Organochlorine insecticide residues in fish and birds from three river systems on the north coast region of New South Wales. Bull. Environ. Contam. Toxicol. 42(6):884-889.

McDougall, K.W., Harris, C.R., Fenton, I.G., and Dowman, A. 1995. Persistence and effect of management practices on organochlorine residues in soils of sub-tropical New South Wales. Bull. Environ. Contam. Toxicol. 54:177-184.

McDougall, K.W., Singh, K.W., Harris, C.R., and Higginson, F.R. 1987. Organochlorine insecticide residues in some agricultural soils on the north coast region of New South Wales. Bull. Environ. Contam. Toxicol. 39:286-293.

McKenzie-Smith, F., Tiller, D., and Allen, D. 1994. Organochlorine pesticide residues in water and sediments from the Ovens and King rivers, North-East Victoria, Australia. Arch. Environ. Contam. Toxicol. 26:483-490.

McMahon, T.A., Finlayson, B.L., Haines, A.T., and Srikanthan, R. 1992. *Global Runoff — Continental Comparisons of Annual Flows and Peak Discharges.* Catena-Verlag, Cremlingen, Germany.

MDNR. 1989. *Rule 57(2): Guideline Levels.* Surface Water Quality Division, Michigan Department of Natural Resources, Lansing, MI.

Mehrota, K.N. 1993. Pesticide resistance in India. Chemistry & Industry, 7 June, pp 399-402.

Meier, P.G., Fook, D.C., and Lagler, K.F. 1983. Organochlorine pesticide residues in rice paddies in Malaysia, 1981. Bull. Environ. Contam. Toxicol. 30:351-357.

Mensaveta, P., and V., C. 1981. Heavy metals, organochlorine pesticides and PCBs in green mussels, mullets and sediments of river mouths in Thailand. Mar. Pollut. Bull. 16(2):78-81.

Milliman, J.D., Rutkowski, C., and Meybeck, M. 1995. *River Discharge to the Sea: A Global River Index (GLORI).* LOICZ Core Project Office, Netherland Institute for Sea Research, P.O. Box 59, 1790-AB Den Burg, Texel, The Netherlands.

Misra, V., and Bakre, P.P. 1994. Organochlorine contaminants and avifauna of Mahala water reservoir, Jaipur, India. Sci. Tot. Environ. 144:145-151.

Miyata, H., Takayama, K., Ogaki, J., Mimura, M., and Kashimoto, T. 1988. Study on polychlorinated dibenzo-p-dioxins and dibenzofurans in rivers and estuaries in Osaka Bay in Japan. Toxicol. Environ. Chem. 17:91-101.

Mohammad, L.A., 1982 A desk study of the Shatt al-Arab estuary. M.Sc. Thesis. U. Strathclyde, Glasgow.

Mowbray, D.L. 1986. Pesticide control in the south Pacific. Ambio. 15:22-29.

Mowbray, D.L. 1988. *Pesticide use in the south Pacific.* UNEP Regional Seas Reports and Studies No. 89 and South Pacific Regional Environment Programme, Topic Review No. 26, Nairobi.

Mudroch, A., and MacKnight, S.D. 1991. *Handbook of Techniques for Aquatic Sediments Sampling.* CRC Press, Boca Raton, FL.

Muir, D.C.G., Norstrom, R.J., and Simon, M. 1988. Organochlorine contaminants in Arctic marine food chains: accumulation of specific polychlorinated biphenyls and chlordane-related compounds. Environ. Sci. Technol. 22(9):1071-1079.

Mussalo-Rauhamaa, H., Häsänen, E., Pyysalo, H., Antervo, K., Kauppila, R., and Pantzar, P. 1990. Occurrence of beta-hexachlorocyclohexane in breast cancer patients. Cancer. 66:2124-2128.

Nair, A., Dureja, P., and Pillai, M.K.K. 1991. Levels of aldrin and dieldrin in environmental samples from Delhi, India. Sci. Tot. Environ. 108:255-259.

Nair, A., Mandapati, R., Dureja, P., and Pillai, M.K.K. 1996. DDT and HCH load in mothers and their infants in Delhi, India. Bull. Environ. Contam. Toxicol. 56:58-64.

Nair, A., and Pillai, M.K.K. 1989. Monitoring of hexachlorobenzene residues in Delhi and Faridabad, India. Bull. Environ. Contam. Toxicol. 42:682-686.

Nair, A., and Pillai, M.K.K. 1992. Trends in ambient levels of DDT and HCH residues in humans and the environment of Delhi, India. Sci. Tot. Environ. 121:145-157.

Nakagawa, R., Hirakawa, H., and Hori, T. 1995. Estimation of 1992-1993 dietary intake of organochlorine and organophosphorus pesticides in Fukuoka, Japan. J. AOAC Intern. 78(4):921-929.

Nayak, A.K., Raha, R., and Das, A.K. 1995. Organochlorine pesticide residues in middle stream of the Ganga River, India. Bull. Environ. Contam. Toxicol. 54:68-75.

Noegrohati, S., Sardjoko, Kasoembogo, U., and Hammers, W.E. 1992. Impact of DDT spraying in the residue levels in soil, chicken, fish-pond water, carp, and human milk samples from Malaria infested villages in central Java. Toxicol. Environ. Chem. 34:237-251.

Norstrom, R.J., Simon, M., Muir, D.C.G., and Schweinsburg, R.E. 1988. Organochlorine contaminants in Arctic marine food chains: identification, geographical distribution, and temporal trends in polar bears. Environ. Sci. Technol. 22(9):1063-1071.

NTP. 1996. List of Chemicals Anticipated to be Carcinogens 7th Annual Report on Carcinogens. http://ntp-server.niehs.nih.gov/Main_Pages/NTP_ARC_PG.html, U.S. National Toxicology Program, National Institute of Environmental Health Sciences.

OMEE. 1994. *Provincial Water Quality Objectives and Guidelines.* Ontario Ministry of Environment and Energy, Toronto, Ontario.

Onodera, S., and Tabucanon, M.S. 1986. Organochlorine pesticide residues in the lower Chao Phraya river and klongs along the river at Bangkok metropolitan area, 1982-1984. J. Sci. Soc. Thailand. 12:225-238.

Oshawa, K., Hartadi. S., Noegrohati, S., Sastrohamidjoyo, H., Untung, K., Arya, N., Sumiartha, K., and Kuwatsuka, S. 1985. Residue analysis of organochlorine and organophosphorus pesticides in soils, waters and vegetables from central Java and Bali Island. *In:* I. Yamamoto and S. Soromarsono (eds.) *Ecological Impact of Pest Management in Indonesia - Crop Protection Studies in the Framework of the Agroecosytem*. Tokyo University of Agriculture, Tokyo. pp. 59-70.

PANNA. 1990. India to Ban Agricultural Use of DDT. PANNA Outlook, Pesticide Action Network North America Regional Chapter, San Francisco, January.

PANNA. 1991a. Demise of the Drins. Global Pesticide Campaigner, Pesticide Action Network North America Regional Chapter, San Francisco, January.

PANNA. 1991b. From pesticide trade to production: new reform strategies. Global Pesticide Campaigner, Pesticide Action Network North America Regional Chapter, San Francisco, June.

PANNA. 1995a. Demise of the Dirty Dozen Chart. Pesticide Action Network North America Regional Chapter, San Francisco.

PANNA. 1995b. Used chlor-alkali plant shipped from Norway to India for pesticide production. Pesticide Action Network North America Regional Chapter, San Francisco, December.

Pathak, S.P., Kumar, S., Ramteke, P.W., Murthy, R.C., Singh, K.P., Bhattacherjee, J.W., and Ray, P.K. 1992. Riverine pollution in some northern and northeastern states of India. Environ. Monit. Assessment. 22:227-236.

Persaud, D., Jaagumagi, R., and Hayton, A. 1992. *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario*. Water Resources Branch, Ontario Ministry of the Environment, Toronto, Ontario.

Phillips, D.J. 1989. Trace metals and organochlorines in the Coastal waters of Hong Kong. Mar. Pollut. Bull. 20(7):319-327.

Phillips, D.J.H., Richardson, B.J., Murray, A.P., and Fabris, J.G. 1992. Trace metals organochlorines and hydrocarbons in Port Phillip Bay, Victoria: a historical review. Mar. Pollut. Bull. 25:200-217.

Pillai, M.K.K. 1986. Pesticide pollution of soil, water and air in Delhi area, India. Sci. Tot. Environ. 55:321-327.

Piper, B.S., Gustard, A., Green, C.S., and Sridurongkatum, P. 1991. Water resource developments and flow regimes on the Mekong River *Hydrology for the Water Management of Large River Basins*, vol. Publ. no. 201. IAHS. pp. 45-56.

Puri, R.K., Orazio, C.E., Kapila, S., Clevenger, T.E., Yanders, A.F., McGrath, K.E., Buchanan, A.C., Czarnezki, J., and Bush, J. 1990. Studies on the transport and fate of chlordane in the environment. *In:* D. A. Kurtz (ed.) *Long Range Transport of Pesticides in the Environment*. Lewis, Chelsea, MI. pp. 271-289.

Rahman, A.A., Huq, S., and Conway, G.R. 1990. Environmental aspects of surface water systems of Bangladesh: an introduction. *In:* A. A. Rahman, S. Huq and G. R. Conway (eds.) *Environmental Aspects of Surface Water Systems of Bangladesh*. University Press, Dhaka. pp. 1-10.

Rahman, M.S., Malek, M.A., and Matin, M.A. 1995. Trend of pesticide usage in Bangladesh. Sci. Tot. Environ. 159:33-39.

Ramesh, A., Tanabe, S., Iwata, H., Tatsukawa, R., Subramanian, A.N., Mohan, D., and Venugopalan, V.K. 1990a. Seasonal variations of persistent organochlorines insecticide residues in Vellar River waters in Tamil Nadu, south India. Environ. Pollut. 67:289-304.

Ramesh, A., Tanabe, S., Kannan, K., Subramanian, A.N., Kumaran, P., and Tatsukawa, R. 1992. Characteristic trend of persistent organochlorine contamination in wildlife from a tropical agricultural watershed. Arch. Environ. Contam. Toxicol. 23:26-36.

Ramesh, A., Tanabe, S., Murase, H., Subramanian, A.N., and Tatsukawa, R. 1991. Distribution and behaviour of persistent organochlorine insecticides in paddy soil and sediments in tropical environment: a case study in south India. Environ. Pollut. 74:293-307.

Ramesh, A., Tanabe, S., Subramanian, A.N., Mohan, D., Venugopalan, V.K., and Tatsukawa, R. 1990b. Persistent organochlorine residues in green mussels from coastal waters of south India. Mar. Pollut. Bull. 21(12):587-590.

Ramesh, A., Tanabe, S., Tatsukawa, R., Subramanian, A.N., Palanichamy, S., Mohan, D., and Venugopalan, V.K. 1989. Seasonal variations of organochlorine insecticide residues in air from Porto Novo, south India. Environ. Pollut. 62:213-222.

Ramesh, R., and Subramanian, V. 1988. Nature of the dissolved load of the Krishna River basin, India. J. Hydrol. 103:139-155.

Ramesh, R., and Subramanian, V. 1993. Geochemical characteristics of the major tropical rivers of India *Hydrology of Warm Humid Regions*, vol. Publ. no. 216. IAHS. pp. 157-164.

Rao, C.V., and Banerji, S. 1988. Polychlorinated biphenyls in the human adipose tissue and liver samples of Bombay. Toxicol. Environ. Chem. 17:313-317.

Rao, C.V., and Banerji, S. 1989. Polychlorinated biphenyls in human blood samples of Bombay. Bull. Environ. Contam. Toxicol. 43:656-659.

Rao, K.L. 1975. *India's Water Wealth - Its Assessment, Uses, and Projections*. Orient Longman Ltd, New Delhi.

Ray, P.K. 1989. *Measurements on Ganga River water quality - heavy metals and pesticides*. Industrial Toxicology Research Centre, Lucknow.

Ray, P.K. 1992. *Measurements on Ganga River water quality — heavy metals and pesticides*. Industrial Toxicology Research Centre, Lucknow.

Richardson, B.J. 1985a. Disposal of polychlorinated biphenyl (PCB) waste in Australia. *In:* M. W. H. Chan, R. W. M. Hoare, P. R. Holmes, R. J. S. Law and S. B. Reed (eds.) *Pollution in the Urban Environment: Polmet 85*. Elsevier, London. pp. 500-506.

Richardson, B.J. 1985b. Polychlorinated biphenyls (PCBs) in Australian waters. *In:* M. W. H. Chan, R. W. M. Hoare, P. R. Holmes, R. J. S. Law and S. B. Reed (eds.) *Pollution in the Urban Environment: Polmet 85*. Elsevier, London. pp. 652-658.

Richardson, B.J. 1995. The problem of chlorinated compounds in Australia's marine environment *State of the Marine Environment Report for Australia, Technical Annex: 2*, Canberra, ACT http://kaos.erin.gov.au/portfolio/esd/coast_marine/somer/somer_con.htm.

Richardson, B.J., Smillie, R.H., and Waid, J.S. 1986. Case study: the Australian ecosystem. *In:* J. S. Waid (ed.) *PCBs and the Environment, Volume III.* CRC Press, Inc., Boca Raton, FL. pp. 241-263.

Ritter, L., Solomon, K.R., Forget , J., Stemeroff, M., and O'Leary , C. 1995. *Persistent Organic Pollutants, An Assessment Report on: DDT, Aldrin, Dieldrin, Endrin, Chlordane, Heptachlor, Hexachlorobenzene, Mirex, Toxaphene, Polychlorinated biphenyls, Dioxins and Furans.* International Programme on Chemical Safety (IPCS), Inter-Organization Programme for the Sound Management of Chemicals (IOMC) PCS/95.38, http://irptc.unep.ch/pops.

Ruangwises, S., Ruangwises, N., and Tabucanon, M.S. 1994. Persistent organochlorine pesticide residues in Green Mussels (*Perna viridis*) from the Gulf of Thailand. Mar. Pollut. Bull. 28(6):351-355.

Samuel, T., and Pillai, M.K.K. 1989. The effect of temperature and solar radiation on volatilization, mineralization and degradation of [^{14}C] - DDT in soil. Environ. Pollut. 57:63-77.

Samuel, T., and Pillai, M.K.K. 1990. Effect of temperature and sunlight exposure on the fate of soil-applied [^{14}C] - gamma - hexachlorocyclohexane. Arch. Environ. Contam. Toxicol. 19:214-220.

Samuel, T., and Pillai, M.K.K. 1991. Impact of repeated application on the binding and persistence of [^{14}C] - DDT and [^{14}C] - HCH in a tropical soil. Environ. Pollut. 74:20-216.

Sarkar, A., and Sen Gupta, R. 1987. Chlorinated pesticide residues in sediments from the Arabian Sea along the central west coast of India. Bull. Environ. Contam. Toxicol. 39:1049-1034.

Sarkar, A., and Sen Gupta, R. 1988a. Chlorinated pesticide residues in marine sediments. Mar. Pollut. Bull. 19(1):35-37.

Sarkar, A., and Sen Gupta, R. 1988b. DDT residues in sediments from the Bay of Bengal. Bull. Environ. Contam. Toxicol. 41(5):664-669.

Sarkar, A., and Sen Gupta, R. 1989. Determination of organochlorine pesticides in Indian coastal water using a moored in-situ sampler. Water Res. 23(8):975-978.

Sarkar, A., and Sen Gupta, R. 1991. Pesticide residues in sediments from the west coast of India. Mar. Pollut. Bull. 22(1):42-45.

Saxena, S.P., Khare, C., Farooq, A., Murugesan, K., and Chandra, J. 1987a. DDT residues in blood of residents of areas surrounding a DDT manufacturing factory in Delhi. Bull. Environ. Contam. Toxicol. 38:392-395.

Saxena, S.P., Khare, C., Murugesan, K., Farooq, A., Dogra, T.D., and Chandra, J. 1987b. DDT residues in soil of areas surrounding a DDT manufacturing factory in Delhi, India. Bull. Environ. Contam. Toxicol. 38:388-391.

Schecter, A., Fürst, A., Fürst, C., Groebel, W., Constable, J.D., Kolesnikov, S., Beim, A., Boldonov, A., Trubitsun, E., Vlasov, B., Cau, H.D., Dai, L.C., and Quynh, N.T. 1990. Levels of chlorinated dioxins, dibenzofurans and other chlorinated xenobiotics in food from the Soviet Union and the south of Vietnam. Chemosphere. 20:799-806.

Schecter, A., Fürst, A., Krüger, C., Meemken, H.A., Groebel, W., and Constable, J.D. 1989. Levels of polychlorinated dibenzofurans, dibenzodioxins, PCBs, DDT, DDE, hexachlorobenzene, dieldrin, hexachlorocyclohexanes and oxychlordane in human breast milk from the United States, Thailand, Vietnam and Germany. Chemosphere. 18:290-297.

Sharma, V.P. 1991. Environmental management in malaria control in India. *In:* G. A. T. Targett (ed.) *Malaria, Waiting for the Vaccine*. Wiley, Chichester. pp. 49-63.

Sharpe, R.M. 1995. Another DDT connection. Nature. 375:538-539.

Shaw, G.R., and Connell, D.W. 1980. Polychlorinated biphenyls in the Brisbane River Estuary, Australia. Mar. Pollut. Bull. 11(12):356-358.

Simonich, S.L., and Hites, R.A. 1995. Global distribution of persistent organochlorine compounds. Science. 269:1851-1854.

Simpson, C.D., Smith, T.J., Burggraaf, S., Wilkins, A.L., Langdon, A.G., and Wilcock, R.J. 1995. Identification of a point source of chlordane contamination from a timber treatment facility. Bull. Environ. Contam. Toxicol. 55:289-295.

Singh, D.K., and Agarwal, H.C. 1995. Persistence of DDT and nature of bound residues in soil at higher altitude. Environ. Sci. Technol. 29(9):2301-2304.

Singh, G., Kathpal, T.S., Spencer, W.F., and Dhankar, J.S. 1991. Dissipation of some organochlorine insecticides in cropped and uncropped soil. Environ. Pollut. 70:219-239.

Singh, G., Kathpal, T.S., Spencer, W.F., Yadav, G.S., and Kushwaha, K.S. 1989. Dissipation of hexachlorocyclohexane isomers in flooded rice soil. J. Environ. Sci. Health B. 24(4):335-348.

Singh, P.P., Battu, R.S., and Kalra, R.L. 1988. Insecticide residues in wheat grains and straw arising from their storage in premises treated with BHC and DDT under malaria control program. Bull. Environ. Contam. Toxicol. 40:696-702.

Sinha, A.K. 1991. River Ganga in the stretch between Kalakankar (Pratapgarh) and Phaphamau (Allahabad): a comprehensive study. *In:* C. R. Krishna Murti, K. S. Bilgrami, T. M. Das and R. P. Mathur (eds.) *The Ganga, A Scientific Study*. Northern Book Centre, New Delhi. pp. 125-140.

Siriwong, C., Hironaka, H., Onodera, S., and Tabucanon, M.S. 1991. Organochlorine pesticide residues in Green Mussel (Perna viridis) from the Gulf of Thailand. Mar. Pollut. Bull. 22(10):510-516.

Soerjani, M. 1988. Current trends in pesticide usage in some Asian countries *Pesticides: Food and Environmental Implications*. International Atomic Energy Agency, Vienna. pp. 219-234.

Sovocool, G.W., Lewis, R.G., Harless, R.L., Wilson, N.K., and Zehr, R.D. 1977. Analysis of technical chlordane by gas chromatography / mass spectrometry. Anal. Chem. 49(6):734-740.

Spicer, P.E., and Kereu, R.K. 1993. Organochlorine insecticide residues in human breast milk: a survey of lactating mothers from a remote area in Papua New Guinea. Bull. Environ. Contam. Toxicol. 50:540-546.

Spindler, M. 1983. DDT: health aspects in relation to man and risk/benefit assessment based thereupon. Residue Rev. 90:1-34.

Srivastava, U.K., and Patel, N.T. 1990. *Pesticides Industry In India: Issues and constraints in its Growth*. Oxford and IBH Publishing, New Delhi, India.

Stacey, C.I., Perriman, W.S., and Whitney, S. 1985. Organochlorine pesticide residue levels in human milk: Western Australia, 1979-1980. Arch. Environ. Health. 40(2):102-108.

Stacey, C.I., and Tatum, T. 1985. House treatment with organochlorine pesticides and their levels in human milk - Perth, Western Australia. Bull. Environ. Contam. Toxicol. 35:202-208.

Stackhouse, J. 1996. Farmers, bugs live in harmony. The Globe and Mail, Toronto, August 12.

Sugavanam, B., and Kim, Y.-H. 1996. *UNIDO activities related to a global plan of action for reducing or eliminating risks associated with persistent organic pollutants (POPs).* presented at IFCS meetings on POPs, Manila, Philippines, 17- 22 June 1996, International Register of Potentially Toxic Chemicals, http://irptc.unep.ch/pops/manila/manexp10.html.

Sugiura, K., Kitamura, M., Matsumoto, E., and Goto, M. 1986. Polychlorobiphenyls (PCBs) in sediments of Tokyo Bay. Arch. Environ. Contam. Toxicol. 15:69-76.

Sujatha, C.H., Nair, S.M., Kumar, N.C., and Chacko, J. 1993. Distribution of organochlorine pesticides in a tropical waterway: HCH isomers. Toxicol. Environ. Chem. 39:103-111.

Tabucanon, M.S., Watanabe, S., Siriwong, C., Boonyatumanond, R., and Tanabe, S. 1992. Current status of contamination by persistent organochlorines in the lower Chao Phraya River, Thailand. Water Sci. Technol. 25(11):17-24.

Takeoka, H., Ramesh, A., Iwata, H., Tanabe, S., Subramanian, A.N., Mohan, D., Magedran, D., and Tatsukawa, R. 1991. Fate of HCH in the tropical coastal area, south India. Mar. Pollut. Bull. 22:290-297.

Takeuchi, K. 1993. Analyses of the flow of the Chao Phraya River. *In:* N. E. Peters, R. J. Allan and V. V. Tsirkunov (eds.) *Hydrology of Warm Humid Regions*, vol. IAHS Publ. no. 216. IAHS. pp. 181-193.

Tan, G.H., and Chong, C.L. 1993. Trace monitoring of water-borne phenolics in the Klang River basin. Environ. Monit. Assessment. 24:267-277.

Tan, G.H., Goh, S.H., and Vijayaletchumy, K. 1991. Analysis of pesticide residues in peninsular Malaysian waterways. Environ. Monit. Assessment. 19:469-479.

Tan, G.H., and Vijayaletchumy, K. 1994. Organochlorine pesticide residue levels in peninsular Malaysia rivers. Bull. Environ. Contam. Toxicol. 53:351-356.

Tanabe, S. 1988. PCB problems in the future: foresight from current knowledge. Environ. Pollut. 50:5-28.

Tanabe, S. 1989. A need for reevaluation of PCB toxicity. Mar. Pollut. Bull. 20(6):247-248.

Tanabe, S., Gondaira, F., Subramanian, A.N., Ramesh, A., Mohan, D., Kumaran, P.L., Venugopalan, V.K., and Tatsukawa, R. 1990. Specific pattern of persistent organochlorine residues in human breast milk from south India. J. Agric. Food Chem. 38:899-903.

Tanabe, S., Kannan, N., Fukushimima, M., Okamoto, T., and Wakimoto, T. 1989. Persistent organochlorines in Japanese coastal waters: an introspective summary from a far east developed nation. Mar. Pollut. Bull. 20(7):344-352.

Tanabe, S., Kannan, N., Subramanian, A.N., Watanabe, S., and Tatsukawa, R. 1987. Highly toxic coplanar PCBs: occurrence, source, persistency and toxic implications to wildlife and humans. Environ. Pollut. 47:147-163.

Tanabe, S., Kannan, N., Tabucanon, M.S., Siriwong, C., Ambe, Y., and Tatsukawa, R. 1991a. Organochlorine pesticide and polychlorinated biphenyl residues in foodstuffs from Bangkok. Environ. Pollut. 72:191-203.

Tanabe, S., Ramesh, A., Sakashita, D., Iwata, H., Tatsukawa, R., Mohan, D., and Subramanian, A.N. 1991b. Fate of HCH (BHC) in tropical paddy field: application test in south India. Intern. J. Environ. Anal. Chem. 45:45-53.

Tanabe, S., Subramanian, A.N., Ramesh, A., Kumaran, P.L., Miyazaki, N., and Tatsukawa, R. 1993. Persistent organochlorine residues in dolphins from the Bay of Bengal, south India. Mar. Pollut. Bull. 26(6):311-316.

Tanabe, S., and Tatsukawa, R. 1986. Distribution behaviour and load of PCBs in the oceans. *In:* J. S. Waid (ed.) *PCBs and the Environment, Volume I.* CRC Press, Boca Raton, FL. pp. 143-161.

Tatsukawa, R., Yamaguchi, Y., Kawano, M., Kannan, N., and Tanabe, S. 1990. Global monitoring of organochlorine insecticides - an 11 year case study (1975-1985) of HCHs and DDTs in the open ocean atmosphere and hydrosphere. *In:* D. A. Kurtz (ed.) *Long Range Transport of Pesticides.* Lewis, Chelsea MI. pp. 127-141.

Tehseen, W.M., Hansen, L.G., Wood, S.G., and Hanif, M. 1994. Assessment of chemical contaminants in water and sediment samples from Degh Nala in the province of Punjab, Pakistan. Arch. Environ. Contam. Toxicol. 26:79-89.

Thakkar, N. 1986. Pesticide pollution in some urban water sources. IAWPC Tech. Annual. 13:103-109.

Thao, V.D., Kawano, M., Tanabe, S., and Tatsukawa, R. 1993. Persistent organochlorine residues in soils from tropical and sub-tropical Asian countries. Environ. Pollut. 81:61-71.

Thompson, G.B., Chapman, J.C., and Richardson, B.J. 1992. Disposal of hazardous wastes in Australia: implications for marine pollution. Mar. Pollut. Bull. 25:155-162.

USEPA. 1990. *Quality criteria for water — 1989.* U.S. Environmental Protection Agency, Criteria and Standards Division, Office of Water, Washington, DC.

Vaithiyanathan, P., Ramanathan, A.L., and Subramanian, V. 1992. Sediment transport in Cauvery River basin - sediment characteristics and controlling factors. J. Hydrol. 139:197-210.

Verghese, B.G. 1990. *Waters of Hope: Integrated Water Resource Development and Regional Cooperation within the Himalayan-Ganga-Brahmaputra-Barak Basin.* Oxford & IBH Publishing, New Delhi.

Villeneuve, J.P., Fowler, S.W., and Anderlini, V.C. 1987. Organochlorine levels in edible marine organisms from Kuwaiti coastal waters. Bull. Environ. Contam. Toxicol. 38:266-270.

Voldner, E.C., and Li, Y.-F. 1995. Global usage of selected persistent organochlorines. Sci. Tot. Environ. 160/161:201-210.

von Westernhagen, H., and Klumpp, D.W. 1995. Xenobiotics in fish from Australian tropical coastal waters, including the Great Barrier Reef. Mar. Pollut. Bull. 30(2):166-169.

Wade, T.L., Atlas, E.L., Brooks, J.M., Kennicutt II, M.C., Fox, R.G., Sericano, J.L., Garcia-Romero, B., and deFreitas, D. 1988. NOAA Gulf of Mexico status and trends program: trace organic contaminant distribution in sediments and oysters. Estuaries. 11:171-179.

Wan, H., Higginson, F.R., Harris, C.R., and McDougall, K.W. 1989. Organochlorine insecticide residues in soils used for vegetable and tropical fruit production in the Cudgeon-Duranbah area of New South Wales. Bull. Environ. Contam. Toxicol. 42:177-180.

Wang, Y.S., Tsai, H.S., Lin, H.C., and Chen, Y.L. 1981. Proc. Natl. Sci. Counc. ROC (Taiwan). 5:212-224.

Watanabe, S., Loavakul, W., Boonyathumanondh, R., Tabucanon, M.S., and Ohgaki, S. 1996. Concentrations and composition of PCB congeners in the air around stored used capacitors containing PCB insulator oil in a suburb of Bangkok, Thailand. Environ. Pollut. 92(3):289-197.

WHO. 1979. *DDT and its Derivatives. Environmental Health Criteria 9.* World Health Organization, Geneva.

WHO. 1984a. *Camphechlor. Environmental Health Criteria 45.* World Health Organization, Geneva.

WHO. 1984b. *Chlordane. Environmental Health Criteria 34.* World Health Organization, Geneva.

WHO. 1984c. *Heptachlor. Environmental Health Criteria 38.* World Health Organization, Geneva.

WHO. 1984d. *Mirex. Environmental Health Criteria 44.* World Health Organization, Geneva.

WHO. 1989a. *Aldrin and Dieldrin. Environmental Health Criteria 91.* World Health Organization, Geneva.

WHO. 1989b. *DDT and its Derivatives - Environmental Aspects. Environmental Health Criteria 83.* World Health Organization, Geneva.

WHO. 1991a. *Alpha- and beta- hexachlorocyclohexanes. Environmental Health Criteria 123.* World Health Organization, Geneva.

WHO. 1991b. *Lindane. Environmental Health Criteria 124.* World Health Organization, Geneva.

WHO. 1992. *Endrin. Environmental Health Criteria 130.* World Health Organization, Geneva.

WHO. 1993. *Polychlorinated Biphenyls and Terphenyls, 2nd ed. Environmental Health Criteria 140.* World Health Organization, Geneva.

Wu, S., Liao, H., and Ma, Y. 1989. Study on the method of determining the residue of organochlorine pesticide and PCB in mussels by gas-chromatography. Mar. Environ. Sci. (China). 8(3):83-92.

Wu, T.C., Lee, J.D., Wang, Y.S., and Chen, Y.L. 1982. Residues of chlorinated hydrocarbon insecticides in sediments of rivers in Taiwan. J. Chinese Agr. Chem. Soc. 20:1-8.

Yadav, D.V., Mittal, P.K., Agarwal, H.C., and Pillai, M.K.K. 1981. Persistence of DDT and BHC in earthworm and soil samples from Delhi (India). Pest. Monit. J. 15(2):80-85.

Yamaguchi, Y., Fukushima, M., Fujita, T., Yamamoto, T., and Yamada, A. 1992. Distribution and seasonal variation of pesticide residues in Yodo River basin, Japan. Water Sci. Technol. 25(11):61-68.

Zaidi, S.S.A., Bhatnagar, V.K., Banerjee, B.D., Balakrishnan, G., and Shah, M.P. 1989. DDT residues in human milk samples from Delhi, India. Bull. Environ. Contam. Toxicol. 42:427-430.

Appendix A

POPs Background Information

A.1 Introduction

The study POPs list is given in Table A.1. Generally properties of the UNEP *dirty dozen* POPs were recently reviewed in (Ritter et al., 1995). Basic chemical, toxicological and environmental information for Table A.1 POPs are also given in the WHO/ IPCS [International Programme on Chemical Safety] *red book* series (WHO, 1979; WHO, 1984a; WHO, 1984b; WHO, 1984c; WHO, 1984d; WHO, 1989a; WHO, 1989b; WHO, 1991a; WHO, 1991b; WHO, 1992; WHO, 1993), and for HCB, the proceedings volume (IARC, 1985).

Toxaphene and mirex have been excluded from consideration in the present report because no environmental occurrence data could be found for either. Mirex may never have been used to any extent in the region. Toxaphene has been used in the region, but has been banned in several important countries.

Table A.1 Current study POPs list.

PCBs	heptachlor and chlordane
DDT	HCB (hexachlorobenzene)
HCH (including lindane)	toxaphene (camphechlor) †
drins (aldrin, dieldrin, endrin)	mirex †

† no GEMS or literature environmental data available for Asia-Pacific region.

A.1.1 Carcinogen Status

Although no POPs listed in Table A.1 are firmly proven human carcinogens, all are considered potential carcinogens. DDT, HCB, HCH, mirex, toxaphene and PCBs are under investigation by the U.S. National Institute of Environmental Health as "anticipated carcinogens", while the U.S. EPA has listed heptachlor and chlordane as "probable human carcinogens".

A.1.2 Emerging Toxicological Issues: POPs as Endocrine-Disrupting Chemicals

Isomers or metabolites of all eight familial classes of POPs listed in Table A.1 have been identified as potential sex hormone disrupting chemicals that may be inducing deleterious effects on the reproduction of humans and wildlife (Colborn et al., 1993). A recent report (Arnold et al., 1996) shows that combinations of certain POPs such as dieldrin and toxaphene at levels that are individually negligible, induce synergisms that increase estrogenic potency 1,000 fold. The environmental implications remain to be explored, but evidence is accumulating that POPs exhibit important hormonal effects in addition established modes of toxicity.

A.2 PCBs (Polychlorinated Biphenyls)

The PCB family comprises 209 *congeners* (compounds) that are now usually identified by the IUPAC [International Union of Pure and Applied Chemistry] numbering scheme (Erickson, 1986). Many countries produced PCBs, usually in series of technical mixtures with increasing weight percentages of chlorine atoms. These commercial mixtures were known by national proprietary trade names such Aroclor, Chlofen, Kanechlor, and numerous others. PCBs were initially produced about 1929. Production rose dramatically after WW II as the inertness, stability and dielectric properties of PCBs made them highly useful in many industrial applications. Global production peaked about 1970 when major producing countries began implementing restrictions, initially to "closed system" uses, and ultimately on all new production. By the late 1970s, production had ceased in most developed countries, but may have persisted to 1985 or later in certain European nations and the former USSR.

Total cumulative global PCB production has been estimated at 1.2–2 million t (Hutchinson and Simmonds, 1994; Tanabe and Tatsukawa, 1986). Tanabe and Tatsukawa estimated that to the mid 1980s, about 30% of the global production had been released into the environment, about 4% had been destroyed and 65% were still in use, in storage, or buried in landfills. Those PCBs still in use were mainly in heavy electrical equipment that was expected to end service life in the latter 1990s. Developing countries have been estimated to hold 15%[27] of global PCB stocks (Cummins, 1988).

Advances in PCB toxicology through the 1980s have made it clear that a small number [13] of *coplanar* PCBs with chlorine substitution patterns and toxicity similar to the most potent dioxins and furans [2,3,7,8 tetrachlorodibenzo-*p*-dioxin and 2,3,4,7,8 pentachlorodibenzofuran] pose grave threats to certain marine mammals, piscivorous birds and other species (Giesy et al., 1994; Tanabe, 1988; Tanabe, 1989; Tanabe et al., 1987). Several competing systems of *toxic equivalency factors* [TEFs] for expressing the dioxin-like toxic of the "dirty 13" PCB congeners (Ahlborg et al., 1994)

[27] Neither Cummins nor others citing this oft quoted 15% figure indicate the source of the estimate.

assign the most toxic PCB [IUPAC no. 126] a toxicity 1/10 that of the most potent dioxin. Total *toxic equivalents* [TEQs] of the 13 coplanar PCBs in environmental samples often exceed the total TEQs of dioxins and furans because PCBs are more prevalent in the environment. Small cetaceans (dolphins, porpoises) appear to be most at risk as they evidently lack capacity to degrade coplanar PCBs which are highly bioaccumulative in aquatic organisms.

These findings prompted Cummins (1988) to assert that *if the 65% of global PCB stocks still in use, storage or land-filled were released to the environment, the extinction of marine mammals would be inevitable*, and furthermore to suggest that *the release of the 15% of PCBs believed to be held by developing nations might be sufficient to cause extinction*. Developing nations are generally the least capable of securely containing and destroying existing PCB stocks. On the basis of the available information it is suggested that PCBs be assigned the highest priority among POPs contaminants.

The high toxicity of coplanar PCBs has profound ramifications for future environmental monitoring and survey programs. Leading environmental agencies have begun developing congener-specific laboratory and field monitoring protocols.

A.3 DDT and Metabolites

DDT is the common name for the broad spectrum insecticide *p,p'*-DDT (dichloro-diphenyl-trichloroethane), and technical formulations comprising mainly *p,p'*-DDT and lesser amounts of related compounds. First synthesized in 1874, DDT's insecticidal properties were only recognized in 1939. Successful applications during World War II lead to escalating use against agricultural insect pests and disease vectors such as mosquitoes and tsetse flies. DDT's low acute toxicity to humans and high persistence when applied to surfaces such as the indoor walls of human dwellings made it the primary tool of malaria eradication programs.

DDT's high persistence and high fat solubility promote a strong tendency to bioaccumulate and biomagnify in ecological food webs. Consequently, severe ecological damage, particularly to raptorial birds at the top of food webs, became evident after intensive usage during the 1950s (WHO, 1989b). Concurrently, heavy usage promoted rapid development of resistance amongst target pests. Ultimately DDT was banned about 1970 across much of the developed world, but retained over much of the developing world for public health usage, mainly for anti-malarial operations, but also against insect vectors transmitting encephalitis, filaria, dengue, leishmaniasis (Kala Azar), trypanosomiasis (sleeping sickness), and others.

Presently, DDT continues to be used against disease vectors in tropical Asia, Oceania, Africa, and the Americas. India appears to be the largest producer and user, having averaged about 8.5 Kt annually over the past 40 years with about 15% used in agriculture. In October 1989, India banned agricultural usage and exports, while

restricting annual public health usage to a limit of 10 Kt. Other producing countries have included Pakistan (at least until recently), Mexico, and possibly Indonesia. In 1992, Brazil and Mexico were reported to have each used about 1,000 t for malaria control (Sharpe, 1995).

Technical DDT comprises isomers *p,p'*-DDT (65–80%), *o,p'*-DDT (5%–21%), *p,p'*-DDD (up to 4%), and traces of related compounds. Plants and animals metabolize the DDT isomers to *p,p'*-DDE (dichloro-diphenyl-dichloroethylene) which is more persistent than the parent compound. Degradation product *p,p'*-DDD (dichloro-diphenyl-dichloroethane) is produced by reductive dechlorination and can dominate in anaerobic environments (e.g., anoxic lake bottoms, pond waters, and sediments). DDD, also known as TDE, has been marketed independently as an insecticide (WHO, 1979).

A.4 Hexachlorocyclohexanes (HCH)

HCH is another widely used broad spectrum insecticide. Crude technical HCH comprises five isomers in approximate proportion 60–70% α-HCH, 5–12% β-HCH, 10–15% γ-HCH, 6–10% δ-HCH, 3–4% ε-HCH and 1–2% other chlorinated compounds (Fischer et al., 1991). Crude HCH has also been widely known by the misnomer benzene hexachloride [BHC]. The γ-HCH isomer is the active insecticidal ingredient, and the pesticide *lindane* is >99% γ-HCH. Between crude HCH and lindane, are so-called *fortified* HCH products that contain from 16–99% γ-HCH and reduced percentages of the non-insecticidal isomers.

HCH is of concern because (1) it continues to be used heavily particularly in south Asia against rice pests and disease vectors, and (2) crude HCH and individual isomers have been identified as potential carcinogens (NTP, 1996). The β-HCH isomer may be the most concern as a potential human carcinogen as it is the most bioaccumulative and the most resistant to degradation. A recent Finnish study has linked β-HCH to breast cancer (Mussalo-Rauhamaa et al., 1990).

It is distinctly possible that cumulative global HCH production exceeds that of all other POPs in Table A.1. Cumulative production to 1993 was estimated at 720 Kt lindane and 550 Kt crude HCH (Voldner and Li, 1995); however, crude HCH production for India was underestimated and Japanese production was not included. Japan produced and consumed about 400 Kt before banning HCH in 1971 (Loganathan et al., 1989). India has produced HCH since 1952 and continues producing and using HCH in large quantities. Usage through the 1970s may have averaged about 15 Kt/a and increased to 35 Kt through the 1980s (Gupta, 1986; Srivastava and Patel, 1990); so that, cumulative Indian production from 1952–1993 likely exceeds 500 Kt. In the early 1980s, technical HCH usage was concentrated in the USSR, Poland, Romania, India, China and Brazil with a total annual production approaching 200 Kt (Fischer et al., 1991). Together these figures suggest the cumulative global crude HCH production to

1993 approached 1.5 million t; however, pesticide production figures cited in (Dhaliwal and Pathak, 1993), if correct, suggest that Chinese HCH production between 1965 and 1983 when HCH was banned could easily rival or exceed that figure.

A.5 Drins: Aldrin, Dieldrin and Endrin

The *drins* are a particular family of cyclodiene insecticides that are the most acutely toxic organochlorine pesticides to humans and animals. The group comprises two pairs of stereoisomers aldrin-dieldrin and isodrin-endrin. Dieldrin and endrin are the biological oxidation products (epoxides) of aldrin and isodrin respectively. Only aldrin, dieldrin and endrin have been produced commercially in large quantity. Endrin is the most acutely toxic of the drins.

In the field, dieldrin and endrin are more persistent than aldrin because they have lower vapor pressures. In humans, endrin is metabolized rapidly whereas dieldrin degrades very slowly. Aldrin has been used mainly as a soil insecticide, while less volatile dieldrin has been used as a termiticidal timber treatment, an animal dip, and a moth-proofing agent. The three have also been used on a variety of crops in tropical and temperate climates.

The last major international producer phased out endrin production in 1982; however, Mexico was still producing it as of 1992. Aldrin and dieldrin have now been banned or severely restricted in so many countries that the last major multinational producer claimed it was phasing out production by 1991 (PANNA, 1991a).

A.6 Heptachlor and Chlordane

Heptachlor and chlordane are considered together here because technical heptachlor was originally derived from chlordane and the two share common components. The parent technical chlordane is a complex mixture of more than 200 compounds (Dearth and Hites, 1991) having 6–10 chlorine atoms, while technical heptachlor is a relatively simple mix of 3 components. The main constituents of technical chlordane are heptachlors, chlordanes (octochlors) and nonachlors with 7, 8 and 9 chlorine atoms respectively. The composition of the technical mixtures is given in Table A.2. *Trans*- and *cis*-chlordane are also known respectively as γ-chlordane and α-chlordane. In addition to parent isomers, biological oxidation yields two metabolites, heptachlor epoxide and oxychlordane, which are persistent, bioaccumulative and toxic.

While both heptachlor and chlordane are broad spectrum insecticides that have been used for many purposes, they have been used mostly as soil insecticides, and termiticidal timber and timber structure treatments, a usage pattern similar to that of aldrin and dieldrin.

Usage and production data are limited. A U.S. chemical company (Velsicol) has been the only documented global producer. Production of chlordane between 1960–1988 was reported as about 70,000 tons [63.5 Kt] for an average of 2.19 Kt/a (Dearth and Hites, 1991). The 1974 domestic U.S. consumption was cited as 9.48 Kt (IARC, 1979). Japan imported 17.4 Kt between 1970 and 1986 when further imports were banned.

Table A.2 Primary constituents (weight %) of technical chlordane and heptachlor.

	Mean	Min.	Max.
A. Technical Chlordane †			
heptachlor	11	7.0	14.5
trans-chlordane	20	14.5	24.0
cis-chlordane	16	13.0	20.5
trans-nonachlor	11	6.0	17.0
cis-nonachlor	3	1.6	4.4
B. Technical Heptachlor ‡			
heptachlor		72	74
trans-chlordane		20	22
trans-nonachlor		4	8

† mean of compositions reported by (Buser and Müller, 1993; Puri et al., 1990; Sovocool et al., 1977)
‡ (WHO, 1984c).

Historical heptachlor production and usage data are elusive. Production in 1971 was 2.7 Kt, and from July 1975 to December 1976, 4.5 Kt, for an estimated annual average 2.88 Kt produced in the early to mid 1970s. Domestic U.S. consumption for 1974 was given as 907 t (IARC, 1979).

Although heptachlor and chlordane uses have been virtually banned in the U.S. since 1988, exports continue. Minimum annual U.S. exports for 1990-94 are listed in Table A.3. Data were obtained from U.S. Customs shipping records that fail to identify 75% of outgoing compounds (FASE, 1996); thus, figures are lower bounds on the actual amounts that may have been exported. As there was no domestic U.S. use, Table A.3 entries represent recent global production and consumption.

Table A.3 Minimum U.S. chlordane and heptachlor exports (t) 1990-94.

Year	Chlordane	Heptachlor
1990	590	† na
1991	518	156
1992	570	718
1993	316	1,150
1994	579	561
mean	524	646
1990-95 total	2,618	‡ 3,230

† not available; ‡ estimated using 1991-94 mean.

Both heptachlor and the chlordane exhibit extraordinarily complex environmental behaviour (Buser and Müller, 1993). Isomeric heptachlor usually degrades rapidly in water (Eichelberger and Lichtenberg, 1971), but rapid microbial metabolism to heptachlor epoxide can yield perceptible concentrations of this more persistent compound. Isomers and metabolites of heptachlor and chlordane have attained significant presence in the Arctic and Antarctic where they rank with PCBs and DDT as the dominant contaminants observed in seals and polar bears (Kawano et al., 1984; Muir et al., 1988; Norstrom et al., 1988). This is remarkable because the cumulative production of chlordane and heptachlor is likely an order of magnitude or more lower than either DDT or PCBs. Due to similar usage patterns and common constituents, it can be difficult to infer which parent chemical is the source of contamination from environmental data. Thus, it is practically necessary to monitor the main isomers plus the two metabolites. Other isomers and metabolites may yet prove to environmentally significant, e.g., the nonachlor III isomer that bioaccumulates perceptibly in polar bears and seals of the Canadian Arctic.