Chapter 6 Accumulation and Effects of Persistent Organic Pollutants in Marine Ecosystems and Wildlife

Introduction

The accumulation and effects of persistent organic pollutants (POPs) in marine ecosystems testify to the magnitude and global scale of these pollutants. On the coasts of the continental United States, POPs concentrations in some marine mammals reach very high levels, such as in the bottlenose dolphins familiar off the Atlantic coast. In remote regions in the middle of the North Pacific Ocean, thousands of miles from industrial centers, albatross have been found to have high PCB and dioxin levels that may be interfering with their reproductive success. And POPs are now measured, albeit at much lower levels, in remote reaches of the Southern Hemisphere and Antarctic. Although these oceanic ecosystems often fall under no national jurisdiction, they are valuable economic and esthetic resources for all. The species that inhabit these areas recognize no national boundaries and neither do the contaminants to which they are exposed. As with the Great Lakes, the seemingly limitless potential of these vast waterbodies to absorb chemicals is thwarted by the persistent and bioaccumulative nature of these contaminants, focusing and maintaining their presence in biological food chains amidst the vast oceanic distances.

The very high levels of POPs found in some marine mammals, e.g., dolphins, killer whales, and beluga, inevitably lead to questions about their links to mass strandings and mortality events. The popular scientific literature contains many stories, articles, and opinions suggesting that the concentrations of POPs accumulated by various marine mammal species are sufficient to be causing adverse effects. However, these exposure data are not backed by the rigorous toxicological information necessary to allow a sound estimate of the actual risks posed by these contaminants. Instead, anecdotal studies are reported in which the concentrations of POPs are measured in sick, dying, or stranded marine mammals. The presence of POPs is then suggested to be the cause of the demise, without explaining why apparently healthy animals can be found with similar levels of POPs. Resolution of these questions, necessary because POPs levels are indeed troubling, is difficult because modern society treats these highly intelligent and social creatures with great deference, formalized through the Marine Mammal Protection Act. As with humans, it is very difficult to obtain permission to experimentally dose marine mammals with contaminants. As with human epidemiology, it is difficult to isolate causal agents in the presence of viral and bacterial infective illness, natural toxins (e.g., red tides), food shortages, predation, and the soup of chemical contaminants and exposures.

This chapter examines POPs in the marine environment: their transport and the resulting levels, trends, and distributions. Examples are provided of adverse effects on the species most at risk from POPs—those at the top of the food chain accumulating the greatest concentrations of contaminants. Field observations of ocean birds, inshore and offshore, are summarized, demonstrating adverse reproductive effects both historically and recently. Research data are then provided summarizing the evidence for adverse effects on marine mammals.

Transport of POPs and the Role of Oceans as Sinks

POPs have spread over the entire surface of the Earth, as evidenced by their occurrence in the air, water, and wildlife of the open oceans. The characteristics of POPs that favor their long-range transport are semivolatility, persistence in the atmosphere, and high chemical and biological stability (see Chapters 7, 9). POPs are transported by runoff and rivers. They are deposited and accumulate in marine sediments, particularly in estuaries and other near-shore areas. The POPs present in these sediments represent an ongoing source of contaminants that can be recirculated into the food chain and accumulated by top predators. Atmospheric transport is also of considerable importance to near-shore marine environments where, despite the influence of terrestrial runoff, short-range transport of contaminated particles may represent a significant contribution to the total input. Calculations by the Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) indicate that atmospheric transport dominates the inputs of POPs to the oceans, with 80% to almost 100% coming from the atmosphere (Preston, 1992; GESAMP, 1989) (Table 6-1).

Available information demonstrates that the oceans, including deep ocean waters, also function as a major and ultimate sink for POPs (Ballschmiter, 1992; Preston, 1992; Loganathan and Kannan, 1991; 1994; Tanabe et al., 1994; Wania and Mackay, 1999; Macdonald et al., 2000; Froescheis et al., 2000). In the late 1970s, the U.S. Environmental Studies Board (1979) estimated that about 50% to 80% of the total PCB residues in the U.S. environment were present in North Atlantic waters. It has also been estimated that over 60% of the world's environmental PCB load is present in open-ocean waters (Tanabe, 1988). The GESAMP estimated mean depositional rates of POPs for different oceans of the world (Table 6-2).

Status and Trends of POPs in North American Marine Ecosystems

Analysis of time trends in environmental concentrations is necessary for understanding and managing the causes and effects of POPs contamination in marine ecosystems. On the simplest level, time trend studies indicate the persistence of a substance by simply watching its rate of decrease with time. On a more complex level, historical trend studies can be used to predict future toxic impacts or indicate times when such impacts are no longer significant.

Time-trend monitoring programs for POPs exist in several countries. The International Mussel Watch Program and the National Status and Trends Program of the United States National Oceanic and Atmospheric Administration (NOAA) are examples (O'Connor, 1996). Temporal trend studies of POPs in biota (fish and oysters) from inshore/coastal aquatic ecosystems have shown clear declines in tissue concentrations following bans on the use of POPs. Although the concentrations of POPs in marine biota are generally declining, the rates of decline are slow (Loganathan and Kannan, 1991, 1994). This slowness results from the long residence time of POPs in the marine environment.

To assess the current status and long-term trends of POPs in U.S. coastal marine environments, NOAA's Status and Trend Mussel Watch Program has been monitoring the coastal waters since 1986 (*http://ccma.nos.noaa.gov/NSandT/New_NSandT.html*). Concentrations of POPs have generally declined in mussels from 154 sites along the U.S. coast, although some local trends remain uncertain (O'Connor, 1996, 1998) (Figures 6-1, 6-2). For fish, the National Benthic Surveillance Project, a component of NOAA's Status and Trends Program, has monitored levels from the West Coast of the United States since 1984 and

Table 6-1. Comparison of atmospheric and riverine inputs of some organochlorines to the oceans (tons/year) (GESAMP, 1989)

Compound	Atmospheric	Estimated riverine	% Atmospheric
ΣHCH	4754	40-80	99
HCB	77.1	4	95
Dieldrin	42.9	4	91
ΣDDT	165	4	98
Chlordane	22.1	4	85
ΣPCBs	239	40-80	80

Table 6-2. Estimated total deposition of organochlorines to the oceans (tons/year) (data from GESAMP, 1989)

Compound	Atlantic Ocean	Pacific Ocean
ΣHCH	948	3111
HCB	27	39
Dieldrin	19	18
ΣDDT	30	92
Chlordane	9.7	10
ΣPCBs	114	65

found no consistent trend in POP concentrations (Brown et al., 1998). These studies have also documented that fish, mussels, and sediment collected near urbanized coastal areas continue to contain relatively high concentrations of POPs (Daskalakis and O'Connor, 1995; O'Connor, 1996; Brown et al., 1998).

Although the concentrations of POPs in sediment and oysters from coastal areas have declined following use restrictions, concentrations in marine mammals have declined only slightly (slow response) over the past few decades. Very few studies have examined temporal trends of POPs using marine mammal tissues. No significant differences in concentrations of PCBs and DDT were observed in striped dolphins collected from the western North Pacific Ocean between 1978–79 and 1986 (Loganathan et al., 1990) (Figure 6-3). Similarly, fur seals collected from the northern North Pacific Ocean showed no decline in concentrations of PCBs during the 1980s (Tanabe et al., 1994). Six- to ten-year trends in the concentrations of POPs in the Canadian Arctic have been examined in female ringed seals and male narwhal and beluga whales. Concentrations of DDTs, PCBs, chlordanes, and toxaphene showed no significant decline in these marine mammals (Muir et al., 1999). In fact, concentrations of PCBs have increased in minke whales from the Antarctic since 1984, and the other POPs have remained at a steady state (Aono et al., 1997). Similarly, concentrations of PCBs and chlordanes in minke whales collected from the North Pacific Ocean in 1994 were higher than those collected in 1987 (Aono et al., 1997). The current absence of a reduction in the residue levels of POPs in marine mammals is consistent with an ongoing redistribution of POPs up the food chain

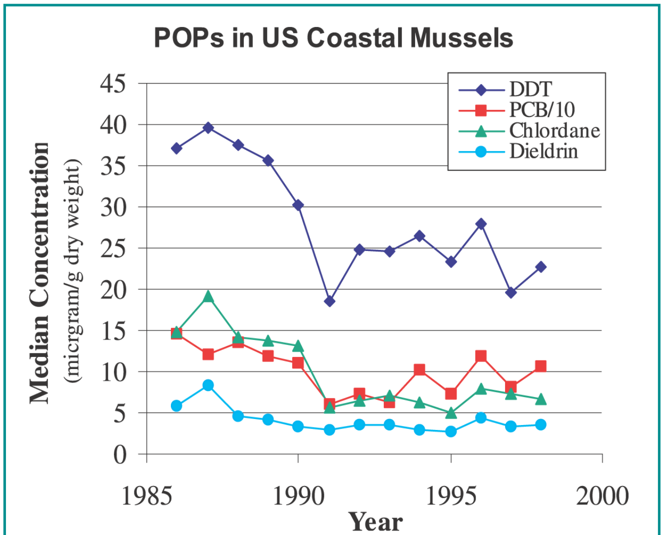

Figure 6-1. Decreasing national median concentrations of contaminants in mussels.
Source: NOAA

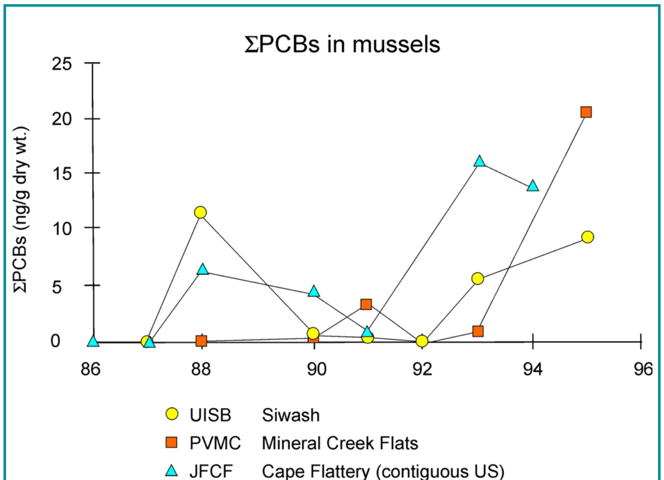

Figure 6-2. Time trends of PCBs of Alaskan mussels.
Source: NOAA

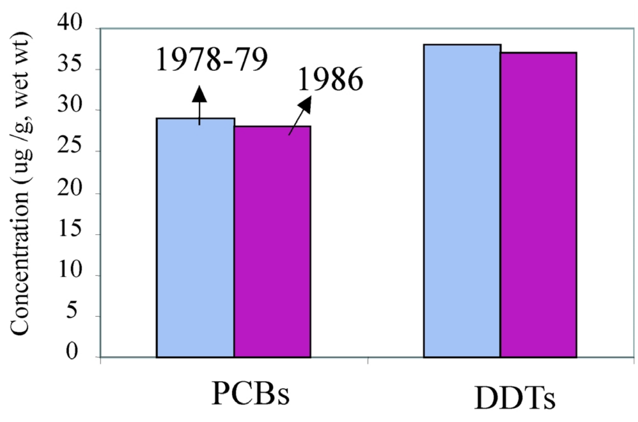

Figure 6-3. POPs concentrations in North Pacific dolphins (Loganathan et al., 1990).

into long-lived animals, and emphasizes the long-term potential toxic impacts of POPs.

Routes of Exposure of Marine Mammals and Seabirds to POPs

POPs reach marine environments from atmosphere deposition or in terrestrial runoff from rivers. Because of their relatively low water solubility, POPs tend to be strongly bound to particulate matter in sediments in aquatic ecosystems. Invertebrate animals living in sediments, such as worms and shellfish, eat the POPs bound to food particles and also receive some uptake of POPs directly from water. Fish and other predators consume these invertebrates and accumulate a variety of organic contaminants. POPs biomagnify at each subsequent step up the food chain (i.e., reach higher concentrations in the predator than in its food). In marine ecosystems, many of these top predators are birds or marine mammals. A study from the western North Pacific Ocean has shown the concentrations of PCBs and DDTs in striped dolphins to be 10 million times greater than in surface waters (Tanabe et al., 1984) (Table 6-3). Although the concentrations in water or prey items were low, marine mammals have large pools of fatty tissues, long lifespan and reduced capacity to metabolize POPs. As a result, they accumulate high concentrations of POPs in their tissues (Tanabe et al., 1994).

Levels of squid in the North Pacific provide further evidence of the link between environmental pollution and subsequent contamination high on the food chain in marine mammals and birds. Hashimoto et al. (1998) measured polychlorinated dioxin and furan levels in predatory squid in a transect of the North Pacific from Japan to near the coasts off Canada and the United States, with additional samples from New Zealand waters (Figure 6-4). Squid taken from waters near industrial centers, such as Japan, showed considerably higher levels of dioxins and furans. The lowest levels were found in the far South Pacific. The levels of dioxins and furans in these relatively short-lived (1–2 years) predatory animals in the remote North Pacific Ocean provide a clear marker of the level, extent, and contemporary nature of POPs contamination. This contamination of the North Pacific offers potential insights when considering the sources of elevated POPs levels in albatross on Midway Atoll (see below).

Further POPs transfers can still occur once contaminants have reached adult predators at the top of food chains. POPs are passed on to the next generation by transfer into the eggs of birds; they are also passed either directly, or via milk, into the progeny of marine mammals. Such transfers of relatively high concentrations of POPs to the developing young raise serious questions about the potential effects of these compounds on wildlife populations.

Adverse Effects on Wildlife

Global distribution of POPs has been well documented (Tanabe et al., 1983; Iwata et al., 1993; Zell and Ballschmiter, 1980). The adverse effects of

Table 6-3. PCBs and DDTs biomagnify in the North Pacific Ocean from surface waters through plankton to marine mammals (Tanabe et al., 1984)

Concentration (pg/g)	PCBs	DDTs
Surface water	0.28	0.14
Zooplankton	1800	1700
Myctophids	48000	43000
Squid	68000	22000
Striped dolphin	3700000	5200000

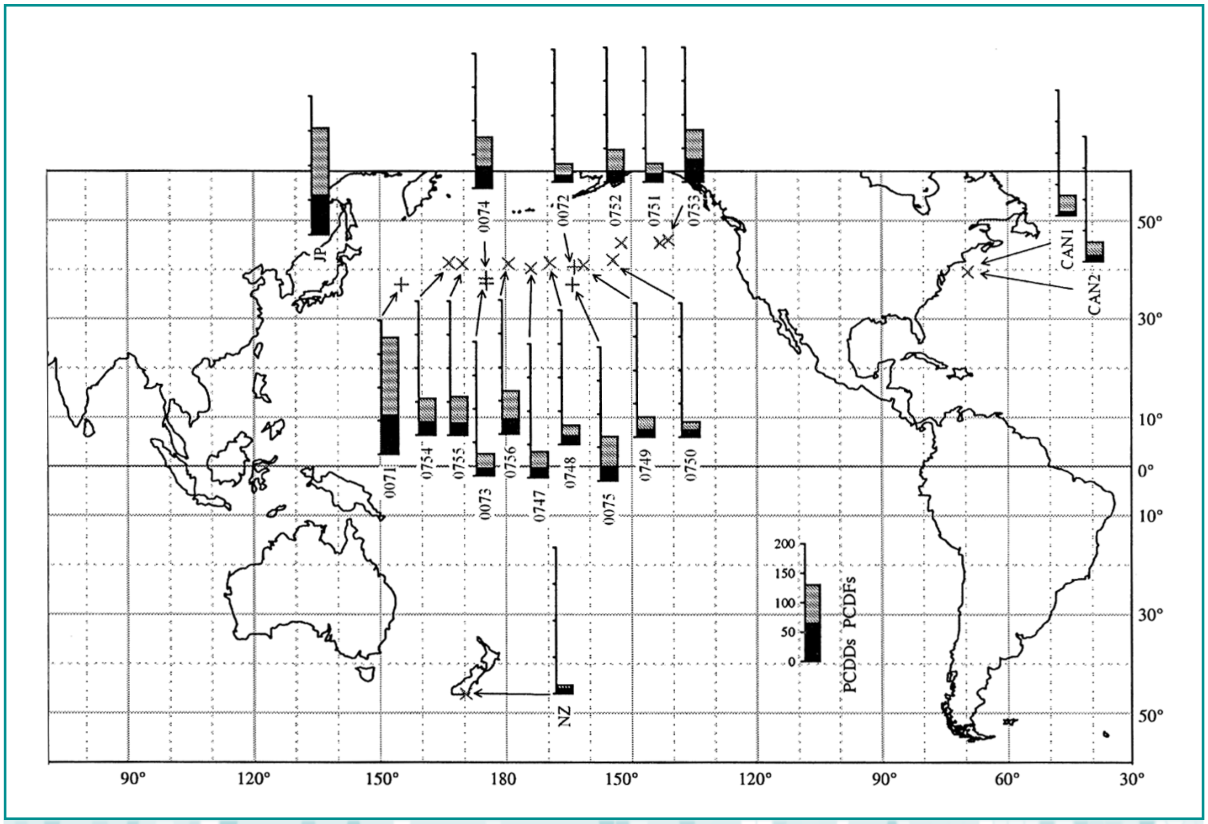

Figure 6-4. Polychlorinated dioxin and furan levels in North Pacific squid (pg/g squid liver) (Hashimoto et al., 1998).

POPs on test animals have also been demonstrated in many studies in laboratories and in the field. It is the challenge of ecotoxicology to identify and/or predict the possibility of adverse effects in wildlife populations based on the known chemical distribution and known toxic effects. In many cases where discrete chemical applications occur and adverse effects are clearly evident, such as acute lethality (e.g., fish kills), it is easy to link the chemical exposure to the adverse effect. However, in the case of POPs, many of the adverse effects are chronic and subtle, and evaluating their relevance to population level effects is difficult. Wildlife are also more commonly exposed to relatively low environmental concentrations of POPs, and these exposures accumulate over long time periods—whole lifetimes for most animals. Finally, although most POPs are generally metabolized slowly, if at all, there are differences in uptake and elimination that lead to differences in the patterns of compounds accumulated in different species. These dissimilarities can result in different contaminant exposures to different animals living in the same environment. In the following sections of this report, we provide information on the exposures and effects of POPs on different classes of marine species.

Inshore Birds

The effects of organochlorines on a number of bird species have been demonstrated in laboratory and field studies (SETAC, 1996; Blus 1996; Wiemeyer, 1996). At high concentrations, many organochlorines can be acutely toxic. This toxicity is particularly pronounced in pesticides, such as dieldrin, which are neurotoxins. However, of more concern at current, relatively low environmental concentrations are the chronic effects of a number of POPs on the reproductive success of birds. PCBs and dioxins have been shown to adversely affect the reproductive success of fish-eating water birds in the Great Lakes (see Chapter 3). A more historically significant effect was the near extinction of some bird species resulting from the adverse effects of DDT residues.

Pelicans (*Pelicanus occidentalis*) in the Gulf of Mexico were particularly affected by DDT and its residues. These compounds accumulated in the adult birds and caused the production of abnormally thin eggshells (Figures 6-5, 6-6). During incubation in the nest, these thin shells were easily broken by the relatively clumsy adults. The result was severe depopulation in this and other species around the Gulf of Mexico and in other parts of the United States. On the Pacific coast, daily discharges into the Los Angeles sewer system contained hundreds of pounds of DDT in the 1960s. This DDT was eventually discharged to the Pacific Ocean. Coastal waters became contaminated, and brown pelicans nesting on Anacapa Island more than 60 miles away suffered near-complete nesting failure. The colony was littered with broken eggs, with eggshells averaging 31% (for intact eggs) to 50% (broken eggs) thinner than normal. In 1969, 1,125 pairs of pelicans were able to fledge only 4 young birds, and in 1970, 727 pairs produced 5 chicks (Anderson et al., 1975).

By the end of 1970, the DDT discharge had been stopped, leading to a remarkable recovery in reproductive success (Anderson et al., 1975). By 1974,

Figure 6-5. DDT has had a severe impact on brown pelicans in Florida.

Photo: NOAA

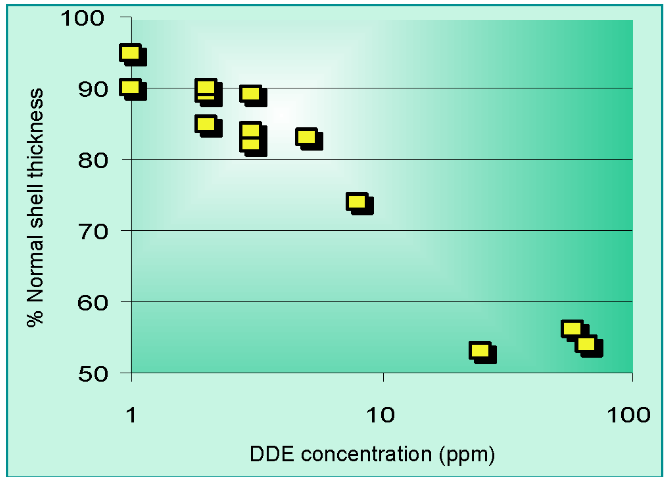

Figure 6-6. Effect of DDE on eggshell thickness in pelicans from North America (data from Blus, 1996).

DDT residues in anchovies, the primary food of the pelicans, had declined by 97%—from 4.3 ppm to 0.15 ppm. Contamination of pelican eggs by DDT residues had declined 89%. Eggshell thickness also improved to 16% (intact eggs) and 34% (broken eggs) thinner than normal, indicating that by 1974 the abnormal shell thinning caused by DDT and DDE had been reduced by nearly half. Eggshell thinning up to about 10% can be tolerated without affecting reproduction. The improvement in reproductive success was spectacular. In 1974, 1,286 pairs of pelicans fledged 1,185 young birds. In terms of the number of young birds raised per nest, the figure rose from a low of 0.004 in 1969 to 0.922 in 1974—an improvement in reproductive success of more than 200-fold. These figures confirm the severity of DDT's effect on the reproduction of carnivorous birds, and exemplify the remarkable improvement possible when the release of DDT into the local environment is stopped.

Although the principal chemical of concern to aquatic birds has been DDT, other organochlorines are usually measured at the same time. In North America, much of the data collected have been compiled by the U.S. Geological Survey and are available electronically (*http://www.pwrc.usgs.gov/bioeco/*). The distribution of POPs in marine ecosystems is such that most bird samples analyzed contain most, if not all, of the "dirty dozen" POPs (Table 6-4). In most cases, the absence of a par-

Table 6-4. POPs detected in North American coastal birds

POPs	Pelican	Cormorant	Gull	Eagle	Albatross
Aldrin[a]					●
Chlordane	●	●		●	●
DDT[b]	●	●	●	●	●
Dieldrin	●	●	●	●	●
Dioxins		●	●	●	●
Endrin	●	●		●	
Heptachlor	●	●	●	●	●
HCB		●		●	●
Mirex	●		●	●	●
PCBs	●	●	●	●	●
Toxaphene	●			●	●

Note: Data from USGS *(http://www.pwrc.usgs.gov/bioeco/)*; Great Lakes data NOT included.
[a]In birds, aldrin is rapidly converted to dieldrin.
[b]"DDT" includes DDT and residues (e.g., DDE).

ticular POP from one of these species is because it has not been looked for, not because the POP is not present.

The degree of concern regarding POPs contaminants in birds has led to the use of DDE, PCBs, and dioxin concentrations in double-crested cormorant eggs as a "National Environmental Indicator" in Canada. Although many of these initial monitoring programs focused on the Great Lakes, their use as indicators has been extended to coastal and maritime regions. Concentrations of POPs have declined in maritime regions since their peaks in the 1970s and 1980s, but the rate of decline has slowed in recent years (Figure 6-7). It is also gratifying to see that, as concentrations in the environment have decreased, so have the adverse effects attributable to these compounds. The decreases in adverse effects are demonstrated by the recovery of many inshore bird populations since their historic lows.

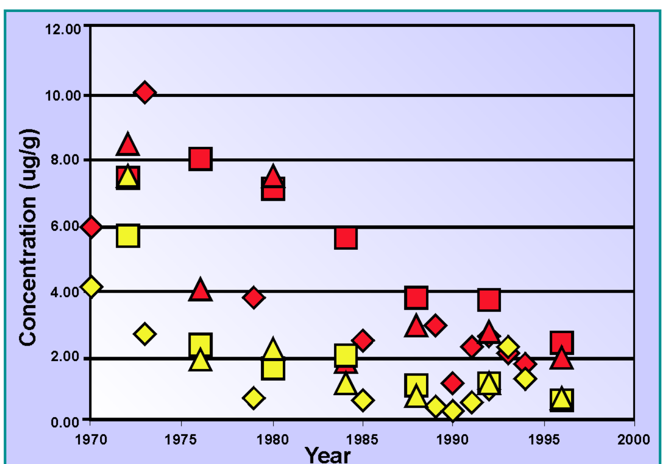

Figure 6-7. PCB (red) and DDT (yellow) concentrations in Canadian coastal birds (double-crested cormorant eggs) have declined since their peaks in the 1960s and 1970s. Monitoring sites in the Straits of Georgia (diamonds) [British Columbia], Bay of Fundy (triangles), and St. Lawrence estuary (squares). Data from Environment Canada.

Much of the contaminant decline in inshore-living bird species can be attributed to the large decrease in inputs to near-shore environments as a result of the banning, deregistration, and emission reductions for all of the POPs in North America. However, now that these chemicals are no longer intentionally produced and released in the United States, we cannot expect to see a continuation of the dramatic decreases of the past. Future declines in near-shore environments will result from the global redistribution of the contaminants currently present. Whether sequestered to the deep ocean and sediments, or transferred to open waters and accumulated in animals, this process can be expected to continue for a long time (Loganathan and Kannan, 1994). Furthermore, although production and use of POPs such as DDT has ceased in North America, it continues in other regions through limited exemptions available under the Stockholm Convention and in nonsignatory countries.

Offshore Birds

Recent studies of offshore-living birds have highlighted the presence of relatively high concentrations of POPs in remote ocean environments (Jones et al., 1996; Auman et al., 1997). Several studies have focused on albatross colonies on Midway Atoll in the North Pacific Ocean (Figure 6-8). The atoll, a former U.S. air base, is located 2,800 miles west of San Francisco and 2,200 miles east of Japan (see Figure 6-9). It is situated close to the northwestern end of the Hawaiian archipelago. These studies demonstrate that, despite living in remote parts of the North Pacific Ocean, albatross accumulate concentrations of POPs similar to those observed in the North American Great Lakes. POPs concentrations in one albatross species were sufficient to suggest a significant risk to the reproductive success of the population (Figure 6-10).

Populations of albatross species in the North Pacific are presently on the increase following the cessation of hunting, which in the early 1900s almost drove the birds to extinction (McDermond and Morgan, 1993). Nonbreeding Laysan albatross (*Diomedea immutabilis*) mainly frequent the western Pacific and Asian coasts, while black-footed albatross (*D. nigripes*) are more common along the northeastern Pacific and North American coasts. Thus, albatross are useful indicators of different sources of marine pollution in the North

Figure 6-8. Birds on remote Midway atoll in the north Pacific are exposed to POPs.
Photo: NASA

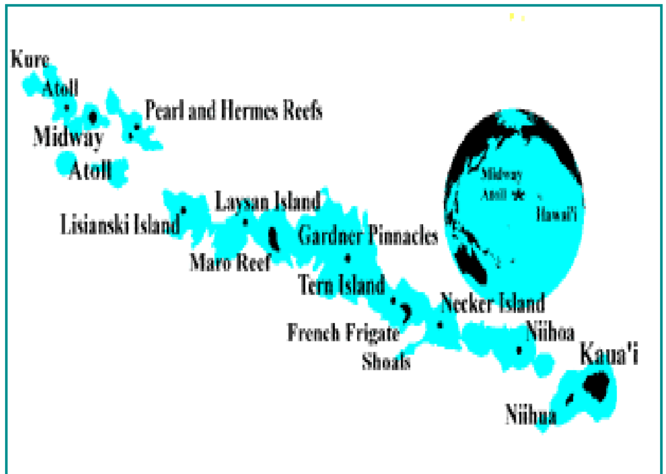

Figure 6-9. Midway Atoll is located in the very center of the North Pacific Ocean at the end of the Hawaiian chain. (U.S. FWS)

Pacific Ocean. Studies on similar species in the South Pacific also provide useful information on the global distribution of organochlorines in these species (Jones, 1999). Relatively high concentrations of chlorinated aromatic compounds, including PCBs and their hydroxylated and methyl sulfone metabolites (Klasson-Wehler et al., 1998), polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCDD/Fs), and non-ortho-substituted PCBs were found in fat samples from the North Pacific populations (Jones et al., 1996). Lower concentrations were measured in the South Pacific species (Jones, 1999). Ortho-substituted PCBs and DDT-related compounds were also reported in plasma samples from these populations, as were a range of other POPs (Auman et al., 1997). The concentrations in the black-footed albatross are reported as sufficient to cause subtle adverse effects on reproduction (Auman et al., 1997; Jones et al., 1996). Although historical data are scarce, they indicate that the egg-crushing rates and hatching rates for black-footed albatross between 1962 and 1964 were comparable to those of Laysan albatross from 1993 to 1994. However, the recent rate of cracked eggs in black-footed albatross, which accumulate greater concentrations of POPs, is greater than 5%, twice the rate observed in the less contaminated Laysan albatross. A statistically significant difference was also found in the hatching rate for the black-footed albatross compared with the Laysan albatross (Auman et al., 1997).

Figure 6-10. Levels of POPs may be sufficient to decrease reproduction in albatross from remote Pacific islands.

Photo: John Giesy

The concentrations of some POPs measured in North Pacific albatross do not differ greatly from concentrations measured in fish-eating birds in the North American Great Lakes. For example, pooled egg samples of black-footed albatross contained 3.8 ppm of total PCBs and 1.8 ppm of DDE, whereas similar samples from Laysan albatross contained 1.0 ppm PCBs and 0.3 ppm DDE (Auman et al., 1997). These concentrations can be compared with concentrations of 9.4 ppm and 6.1 ppm of total PCBs found in the eggs of Great Lakes Caspian terns and doubled-crested cormorants, respectively, in the early 1990s. The finding of elevated POPs concentrations in such a remote location underscores the global transportation of these chemicals and the limitations of the global environment to assimilate or “dilute” these chemicals to safer concentrations (Jones et al., 1996).

Marine Mammals

No situation indicates more the global nature of the POPs problem than the concentrations of POPs detected in marine mammals from around the world. Even though many marine mammals are not directly exposed to POPs sources, particularly in the oceans of the Southern Hemisphere, every marine mammal tissue analyzed contains at least some of the “dirty dozen” POPs (Colborn and Smolen, 1996).

The life history parameters of many marine mammals result in their accumulating high concentrations of POPs. Marine mammals inhabit aquatic environments that are the ultimate sinks for many of these compounds. They have a unique lifestyle that requires thick layers of fatty blubber to provide thermal insulation and energy reserves for fasting periods in their life cycles. These fatty tissues act as a reservoir for the accumulation of POPs, and also act as a continual source “resupplying” the rest of the body with these contaminants when fats are metabolized. The long lifespan and generally predatory feeding habits of marine mammals lead to high levels of POPs in blubber. In addition, marine mammals appear to be limited in the biochemical processes required to metabolize and eliminate these chemicals (Tanabe et al., 1988). Finally, because of the high lipid content of marine mammal milk, POPs are passed via lactation to the developing young.

Of the compounds studied in marine mammals, PCBs appear to accumulate to the greatest concentrations in the widest range of species (Table 6-5) (Tanabe et al., 1983; 1994). Very high POPs levels have been measured on the U.S. East Coast in bottlenosed dolphins, reaching 620 and 200 ppm lipid weight for PCB and DDE, respectively, in mature males (Geraci, 1989). To put these concentrations in perspective, U.S. hazardous waste regulations for PCB liquids commence at 50 ppm.

It has been contended that, since 1968, 16 species of aquatic mammals have experienced population instability, major stranding episodes, reproductive impairment, endocrine and immune system disturbances, or serious infectious diseases (Colborn and Smolen, 1996). The authors also suggest that organochlorine contaminants, particularly PCBs and DDTs, have caused reproductive and immunological disorders in aquatic mammals (Colborn and Smolen, 1996). The presence of high concentrations of PCBs in tissues have also been associated with

- High prevalence of diseases and reduced reproductive capability of the Baltic grey seal

Table 6-5. Global PCB distribution in marine mammal populations

Species	Location	PCBs, μg g^{-1} Wet weight, blubber	Reference
Bottlenose dolphin	East USA	81.4	Kuehl et al., 1991
White-sided dolphin	East USA	50.1	Kuehl et al., 1991
Common dolphin	East USA	36.5	Kuehl et al., 1991
Minke whales	West USA	3.3	Varanasi et al., 1993
Pilot whale	East USA	17	Varanasi et al., 1993
Common dolphin	New Zealand	0.75 –>1.0	Jones et al., 1999
Dusky dolphin	South of New Zealand	1.4	Tanabe et al., 1983
Pilot whale	New Zealand	0.31	Schroder, 1998
Bottlenose dolphin	South Africa	13.8	Cockroft et al., 1989
Dall's porpoise	North Pacific	8.6	Tanabe et al., 1983
White-sided dolphin	Japan	37.6	Tanabe et al., 1983
Harbour porpoise	United Kingdom	55.5	Morris et al., 1989
Baird's beaked whales	Japan	3	Subramanian et al., 1988
Pilot whale	United Kingdom	36.9	Law, 1994

(*Halichoerus grypus*) and ringed seal (*Phoca hispida*) (Olsson et al., 1994)

- Reproductive failure in the Wadden Sea harbor seal (*Phoca vitulina*) (Reijnders, 1986) and St. Lawrence estuary beluga whales (*Delphinapterus leucas*) (Martineau et al., 1987)
- Viral infection and mass mortalities of the U.S. bottlenose dolphin (*Tursiops truncatus*) (Kuehl et al., 1991; Lipscomb et al., 1994), Baikal seal (*Phoca sibirica*) (Grachev et al., 1989), and Mediterranean striped dolphin (*Stenella coeruleoalba*) (Aguilar and Raga, 1993; Kannan et al., 1993)

However, unequivocal evidence of a "cause-effect" linkage between disease development and mass mortalities in marine mammals is lacking, because of confounding factors that limit the ability to extrapolate results from field studies.

Compelling evidence that marine mammals can experience toxic effects comes from data on the feeding of wild-collected fish from different regions to confined seals (Reijnders, 1986, 1994; Ross et al., 1995, 1996, 1997). In the Reijnders study, two matched groups of captive harbor seals were maintained in the same location. One group was fed Baltic Sea herring, the other North Atlantic herring. The group fed Baltic Sea fish suffered near-complete reproductive failure, while the group fed less contaminated Atlantic fish reproduced normally. Similar impacts were evident on immune function. However, although the effects were clear, the specific causal agent(s) were not. As with field studies, several confounding factors prevent a conclusive connection between cause and effect in these studies. These factors include limited sample sizes, the presence of chemicals other than POPs in the food fish, the presence of chemical contaminants in the "control" diet, as well as general concerns about the nutritional quality and similarity of the "control" and "exposed" diets. Additional toxic effects attributed to PCBs and DDT in seals resident in the Baltic Sea include uterine stenosis and occlusions in ringed seals, skull-bone lesions (osteoporosis) in Baltic gray seals and harbor seals, adrenocortical hyperplasia in Baltic ringed and gray seals, lowered levels of vitamin A and thyroid hormones in harbor seals, and lowered immunocompetence in harbor seals (Reijnders, 1994; Hutchinson and Simmonds, 1994).

Much of the controversy over marine mammal levels of POPs centers on the widely publicized mortality episodes among bottlenose dolphins along the Atlantic Coast of North America. Numerous causal agents, or combinations of agents, have been proposed, but none proven. Apart from chemical contaminants, exposure to natural marine toxins has been hypothesized as a possible cause for the bottlenose dolphin mortality episodes (Anderson and White, 1989). However, later studies have indicated that this evidence is circumstantial (Lahvis et al., 1995). Morbillivirus infection appears to have been at least a contributing factor in the dolphin mortality (Belfroid et al., 1996). It has been hypothesized that synthetic chemicals, specifically AhR-active POPs, render marine mammals more susceptible to opportunistic bacterial, viral, and parasitic infection (Lahvis et al., 1995). Debilitating viruses such as morbillivirus may result in further immunosuppression, starvation, and death (Lahvis et al., 1995). Conclusions about causality are further complicated by the fact that marine mammals are exposed simultaneously to a number of synthetic halogenated hydrocarbons, many of which are not quantified or identified. Despite the high accumulation and possible adverse effects of PCBs in marine mammals, tissue concentrations of PCBs that would affect the immune system in marine mammals have not been established. Similarly, factors such as population density, migratory movement, habitat disturbance, and climatological factors have been proposed as playing roles in mass mortalities of marine mammals.

Probably the most convincing case for observable adverse effects of chemical contaminants on marine mammals is in beluga whales (*Delphinapterus leucas*) resident in the Gulf of St. Lawrence on the U.S.-Canada border (*http://www.medvet.umontreal.ca/services/beluga/beluga_homepage.html*). Whales from this population have been shown to have a high incidence of cancers not common in other animals, as well as a variety of other lesions (Figure 6-11) (DeGuise et al., 1994). It is known that these animals accumulate large concentrations of POPs, and it has been suggested that such accumulation may be a contributing factor to the observed effects. However, these animals also accumulate, or are exposed to, high concentrations of other environmental pollutants, including polycyclic aromatic hydrocarbons (PAHs). Because Ah-receptor active POPs are tumor promoters, POPs may be a significant contributing factor to the observed cancer occurrence in these animals. However, because of the limited amount of quantitative toxicological information available about marine mammals (as discussed above), the relative contribution of POPs to these effects may never be known.

Figure 6-11. Intestinal cancer in a beluga whale.
Source: Daniel Martineau, University of Montreal

Even though the above-mentioned field studies do indicate an association between POPs and adverse health effects in marine mammals, the association is not conclusive. Other studies have focused on the in vivo and in vitro effects of POPs on marine mammal immune function (De Guise et al., 1998). These studies have shown some effects, but it is difficult to relate the effects observed to a functional deficit in the immune systems of free-living marine mammals. It has been suggested that immune suppression as the result of POPs exposure may be a contributing factor to marine mammal mass die-offs, such as occurred in the Mediterranean in 1992 (Aguilar and Raga, 1993).

Toxicological data for the effects of PCBs and dioxins on marine mammals were recently compiled, analyzed, and used to derive toxicity reference values (TRVs) (Table 6-6) (Kannan et al., 2000). The TRVs express the best available estimate of the concentrations of chemicals that will result in adverse effects. If the concentration of the chemical in an animal's tissue exceeds the TRV,

Table 6-6. Toxicity reference values for marine mammals (lipid weight basis)

	Total PCBs
Food based	10-150 ng/g
Tissue based (blubber)	17,000 ng/g

adverse effects may result. The TRVs were derived on the basis of the concentration of chemicals in the food that marine mammals were consuming, or on the concentrations of these chemicals in the blubber (fat) of the animals. They were based on studies examining physiological effects such as vitamin A depletion, suppression of natural killer cell activity, and the proliferative response of lymphocytes to mitogens. Details regarding derivation of the TRVs are discussed in Kannan et al. (2000). Because PCBs, dioxins, and furans are considered to act through the same mechanism of action, the authors used a weighted sum of all the exposures to these chemicals, called "toxicity equivalence" or TEQ.

Using these TRVs, it is possible to conduct a risk screening of the possibility of adverse effects from these chemicals in marine mammals. For example, the blubber concentrations of PCBs in pilot whales and bottlenose dolphins from the United States exceed the TRV values (Table 6-7). This suggests that these animals may be subject to adverse effects from these chemicals. In contrast to the North American samples, risks posed by PCBs to marine mammals in the Southern Hemisphere are far lower.

Other Marine Mammals: Pinnipeds, Manatees, and Otters

A wide range of POPs have been measured in the tissues of seals, sea lions, and walruses (collectively called pinnipeds) (Figure 6-12). These include chlordanes, heptachlor epoxide, HCB, dieldrin, toxaphene, PCDDs, and PCDFs (Hutchinson and Simmonds, 1994; Ames and Van Vleet, 1996). Relatively high concentrations of PCBs (85 to 700 ppm) found in harbor seal blubber in the Wadden Sea have been implicated in their mass mortalities and reproductive impairment (Reijnders, 1986). Similarly, elevated exposure of California sea lions to DDT in the 1970s has been linked to reproductive problems (Figure 6-13) (DeLong et al., 1973). California sea lions collected in the early 1970s from coastal California contained a mean DDT concentration of 980 ppm lipid weight in the blubber (DeLong et al., 1973). A recent study has reported DDT concentrations of up to 2,900 ppm lipid weight in the blubber of California sea lions from the California coast (Kajiwara et al., 2001). DDT concentrations as high as 169 ppm lipid weight were also found in the livers of sea otters from coastal California (Nakata et al., 1998). The occurrence of several tens of ppm of PCBs and DDTs has been reported in harbor seals collected in 1990–1992 (approximately 20 years after the ban on the use of DDT) from the northeastern United States (Lake et al., 1995). And the Florida manatee, *Trichechus manatus*, an endangered species, has been reported to contain several ppm levels of sum PCBs and DDT in blubber (Ames and Van Vleet, 1996).

Table 6-7. Risk screening of PCBs in marine mammals

Species	Blubber PCB TRV (ng g^{-1})	Blubber PCB (ng g^{-1})	PCB Exceedance
Bottlenose (USA)	13,600	81,400	6.0
Pilot whale (USA)	13,600	17,000	1.25
Dall's porpoise (Pacific)	13,600	8,600	0.63
Pilot whale (NZ)	13,600	310	0.023
Baleen whales (NZ)	13,600	12.9	0.0001
NZ fur seal (NZ)	13 600	1,069	0.08

Note: TRVs are given on a wet weight basis.

Source: Kannan et al., 2000.

Figure 6-12. Pinniped marine mammals, like these walrus, accumulate POPs.
Photo: NOAA

Summary and Conclusions

Adverse effects from POPs on marine mammals and birds demonstrate the potential for these substances to affect species in regions far from the source of the POPs emission. At the peak of DDT use in the United States, many marine bird species suffered eggshell thinning from DDT, most particularly the brown pelican which became threatened with extinction. Impacts on birds continue in some locations, including elevated exposures in remote reaches of the Pacific Ocean. For marine mammals, controlled studies of high environmental pollutant exposures in their food have demonstrated reproductive impairment and immune changes. But although many marine species are exposed to POPs, there are few studies that "prove" a causal link between specific POPs at more general environmental levels and adverse effects in populations or individuals. Nevertheless, risk evaluations indicate cause for concern. Concentrations of POPs in some species and locations are currently at levels close to those with the potential to cause adverse effects. Therefore, it can no longer be assumed that the world's oceans can diffuse or dispose of these chemicals to "safe" concentrations.

Figure 6-13. DDT is believed to have caused reproductive problems in California sea lions in the 1970-80s.
Photo: NOAA

As with the Great Lakes, however, nature is resilient and there have been remarkable recoveries in wildlife populations with the cessation or reduction of POPs release. POPs levels in U.S. coastal regions are declining in sediments and invertebrates. Less evident are reductions in POPs levels in fish and mammal species, testifying to the peculiar hazard posed by these persistent, bioaccumulative toxins and their ability to remain in the food chain and pass from generation to generation. Decreases in marine POPs levels following production and use controls indicate that regulatory actions can be successful. However, because of the problems of global transportation and deposition of these contaminants, the desired decreases in global POPs will not be achieved without global cooperation.

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