p. 595-610. In: The Ecology of Marine Fishes: California and Adjacent Waters. 2006.
L.G. Allen, D.J. Pondella, and M. H. Horn (eds.). University of California Press, Berkeley, 670 pp.

CHAPTER 23

Pollution

M. JAMES ALLEN

Introduction

The California coast has been a focal point of human activity since the arrival of native Americans more than 10,000 years ago. While these early inhabitants may have fished some local populations heavily, they contributed little pollution and habitat alteration that might affect marine fishes. Although alteration of terrestrial habitats increased from the late 1700s onward with increasing non-native populations, significant marine habitat alteration did not begin until the mid-1800s. During the twentieth century, the production of fishing piers, alteration of coastal lagoons, and construction of artificial harbors and marinas along the coast began in earnest while rivers and streams were chanelized to prevent flooding (Dailey et al., 1993). As the population centers grew, pollution problems arose as waste contaminants were discharged directly into the sea or into the waterways leading to the ocean. However, it was not until the late 1960s that the public became aware that increasing levels of pollution and increasingly altered coastal habitats might be having a detrimental effect on fish populations. With this awareness came scientific interest and political necessity for assessing and understanding the effects of pollution and habitat alteration on marine fish populations (Mearns et al. 1988, 1991; Kennish, 1998).

At present, the coastal population of the Californias is distributed unevenly with high population centers in southern California and in the San Francisco Bay area, and low populations along much of the rest of the coast. In 2000, more than 20 million people were living in the coastal areas of southern California and Tijuana, Baja California, Mexico and about 7 million in the San Francisco Bay area (USCB, 2001). Because of the high human population densities in these areas, these are the primary areas of pollution and habitat alteration that has affected fish populations.

This chapter describes the nature and effects of pollution and habitat alteration effects on California marine fishes. Effects of commercial and recreational fishing are discussed in chapter 22.

Marine Pollution in California

Contaminants of Concern

Human activities along the California coast and on the ocean itself contribute chemical contaminants, oil, and anthropogenic debris to the marine environment. These pollutants may cause detrimental effects to individual fish, their populations, or to fish predators. Chemical contaminants that are of concern to fish biology or to the health of human or wildlife consumers include trace metals (e.g., arsenic, mercury, selenium, cadmium), chlorinated pesticides, such as DDT (dichlorodiphenyltrichloroethane) and its isomers and metabolites, chlordane, dieldrin; PCBs (polychorinated biphenyls); and PAHs (polycyclic aromatic hydrocarbons). These contaminants are of concern due to their toxicity (acute and chronic) to fish at different life stages, potential effects on their populations, and their bioaccumulation in fish tissue and the resultant health risks to wildlife or human consumers. Oil spills sometimes foul the ocean over large areas, subjecting fish at different life stages to altered ocean conditions, tar, and PAHs. Plastic bags and pellets are hazards to larger fish when they are mistakenly consumed as prey, and fish sometimes are entangled or encumbered with rubber bands, plastic packaging, and abandoned or lost fishing gear (e.g., nets, traps, set lines).

Sources of Contaminants to the Marine Environment

Contaminants enter the marine environment from a variety of sources, including municipal wastewater discharge; industrial discharges, storm drains, streams, and rivers; aerial fallout; marine vessel activity; ocean dumping; oil exploration and extraction; and oil spills (Mearns et al., 1991, Anderson et al., 1993, Kennish, 1998, Schiff et al., 2000). The types of contaminants discharged vary by source. Chemical contaminants (e.g., trace metals, chlorinated pesticides, PCBs, PAHs) enter the ocean from most of the sources mentioned, although oil drilling activities and oil spills introduce a more restricted list.

From the late 1940s to the late 1980s, municipal wastewater discharge was a major source of contaminants to the Southern California Bight (SCB). Of greatest concern was the discharge of DDT in southern California from the primary manufacturer of this pesticide in the United States. Extremely high levels of DDT produced by Montrose Chemical Company were discharged off the Palos Verdes Peninsula in municipal wastewater effluent from 1947 to 1970. Similarly, high levels of PCBs and metals were discharged at this site during this period. However, high levels of PCBs were also discharged from other wastewater dischargers in southern California during this time period. With the ban of discharge of DDT and PCBs in the early 1970s and with improved effluent quality (the net result of source control and changes to treatment process), the combined mass emissions of these and other chemical contaminants from the four major wastewater dischargers decreased greatly, such that all in southern California are now meeting California State Ocean Plan requirements. Contaminant inputs from all sources have decreased by 70% (Schiff et al., 2000). Although wastewater discharge is no longer a major source of contaminants in southern California, some environmentally persistent contaminants from historical discharges are still high in sediments on the Palos Verdes Shelf and in Santa Monica Bay (Anderson et al., 1993, Schiff et al., 2000). These historically deposited sediments are now a primary source of DDT and an important source of PCBs to the Southern California Bight. Although the quality of the effluent discharged to the ocean has improved, the coastal human population in southern California is increasing at a rate of about 3.5 million people per decade and contamination from urban runoff (which, with few exceptions, is untreated) has become a major concern. Oil intermittently enters coastal waters from a number of sources. Oil spills result from tanker leaks and occasionally from drilling operations. Some is also discharged in storm water or from marine vessel activity (Anderson et al., 1993). Natural oil seeps (e.g., in the Santa Barbara Channel, and Santa Monica Bay) are also important chronic sources. In addition to chemical contaminants, a wide variety of anthropogenic debris (primarily plastics) enters the ocean in storm water, from fishing and marine vessel activity, and from public use of beaches (Moore and M. J. Allen, 2000, Moore et al., 2001).

Similar sources of contaminants exist in the San Francisco Bay area. In addition, the Bay is the receiving water for contaminants from the extensive agricultural regions of the Delta and Central Valley for which it serves as a sink (Kennish, 1998). Chemical contaminants enter San Francisco Bay from wastewater discharge, urban and non-urban runoff, dredging, and rivers.

Major areas of marine pollution (and hence of contaminated fish) in coastal United States occur along the West Coast in southern California, San Francisco Bay, and Puget Sound, Washington, with much less in coastal areas between these regions (Mearns et al., 1988). On the East Coast, important areas of marine pollution occur in the New York Apex and Boston Bay, although fish contaminated at lower levels occur in estuarine and coastal areas from New England to Georgia. On the Gulf Coast, fish with low-level contamination are found from southern Florida to southern Texas. By the early 1970s, DDT in Palos Verdes Shelf sediments were the largest known area of DDT contamination in the world. Thus coastal fish near the Palos Verdes area in southern California typically had the highest levels of DDT from the late 1960s to early 1980s (fig. 23-1). Nevertheless, relatively high DDT concentrations have been found in southern Texas (Laguna Madre), San Francisco Bay, Puget Sound, Boston Bay, and various other locations along all coasts. In the 1970s, fish muscle tissue with concentrations above 0.500 ppm extended from Gray's Harbor, Washington to the U.S.-Mexico border. In contrast to many other areas where highest DDT concentrations were in estuarine fish, highest DDT concentrations off southern California were in coastal fishes (Mearns et al., 1988).

Fates of Contaminants

The repositories of contaminants in the ocean off California are the water column, sediments, or marine organisms (Anderson et al., 1993, Schiff et al., 2000). Contaminants that enter the ocean from the various sources are distributed by ocean currents and eddies. While some remain dissolved, most are physically or chemically bound to particulate matter and settle to the bottom, resulting in contamination of the sea floor sediments. Contaminants flow to organism repositories through food-web pathways within the water column and benthic habitats resulting in bioaccumulation in fish and other organisms. The degree to which bioaccumulation occurs depends on the solubility, particle affinity, oxidation state, volatility, degradability of the chemicals. These differences influence how contaminants are distributed along the California coast and within biological communities.

WATER COLUMN

In the water column of the SCB, near surface waters (0–200 m) are density stratified with strong vertical gradients of contaminants (Eganhouse and Venkatesan, 1993, Schiff et al., 2000). Natural (e.g., kelp) and anthropogenic debris (e.g., plastics) are most visible at the sea surface. At the surface, the sea-surface microlayer (<1 mm thick) greatly concentrates bacterioneuston, phytoneuston, and zooneuston, as well as anthropogenic contaminants (e.g., DDTs, PCBs, PAHs, chromium, copper, iron, lead, manganese, nickel, silver, and zinc) (Cross et al., 1987, Schiff et al., 2000). For example, concentrations of some trace metals, chlorinated hydrocarbons, and PAHs in the microlayer were much higher in Los Angeles–Long Beach (LA/LB) Harbors than in other nearshore locations, suggesting sources within the harbors. The distribution of high concentrations in sediments away from wastewater outfalls suggests transport of sewage particulates in the water column by currents (Zeng and Venkatesan, 1999). Dissolved contaminants (e.g., DDT, PCB, and PAHs) are generally found in very low concentrations in the water column but are generally higher above contaminated sediments, indicating fluxes between sediments and the water column.

SEDIMENTS

Sediments are major repositories of contaminants in the marine environment. In southern California, high levels of sediment contamination occur on the Palos Verdes Shelf, Santa Monica Bay, and San Diego Bay (Schiff et al., 2000). Sediments on the Palos Verdes Shelf and Santa Monica Bay are high in DDTs, PCBs, and many trace metals whereas San Diego Bay sediments are high in PCBs and PAHs (Mearns et al., 1991). Contaminants in sediments are generally highest near their sources. The discharge of 1,800 metric tons of DDT onto the Palos Verdes shelf between 1953 and 1970 created what may be the largest known area of DDT contamination in the world. Sediment concentrations of > 200 ppm [parts per million = mg/kg] DDT and >10 ppm PCBs were found in sediments there in the early 1970s (MacGregor, 1976). Although surface sediment concentrations of DDT have decreased since the 1970s, the highest sediment concentrations are still on the Palos Verdes Shelf (Schiff et al., 2000; Connolly and Glaser, 2002). As of 1992, about 67 metric tons of *p, p'*–DDE occurred in the sediments of the Palos Verdes Shelf, with 76% of this occurring at water depths of 30 to 100 m with the highest concentrations (300 ppm) 30 cm below the sediment surface (Stull, 1995). Due to upcoast flowing currents

FIGURE 23-1 Total DDT in a) liver and b) muscle of coastal and estuarine fish from 19 sites sampled in 1976 and 1977. Bar represents mean of all individual or composite values for all species collected at a site (after Mearns et al. 1988).

at the discharge depth (61 m), DDT concentrations in sediments at the Palos Verdes shelf decrease upcoast of the discharge site, with much lower levels downcoast of the site. Overall, highest concentrations of PAHs are found in San Francisco Bay and San Diego Bay sediments (Kennish, 1998).

On the seafloor, natural debris (marine algae and terrestrial vegetation debris) is more common on the southern California shelf than anthropogenic debris, and is more common on the inner shelf (<30 m depth) in 1994 and 1998 (Moore and M. J. Allen, 2000; M. J. Allen et al., 2002a). Some of this contributes to a drift algae zone, which is an important habitat on the soft-bottom, providing cover for juvenile white seabass (*Atractoscion nobilis*), barcheek pipefish (*Syngnathus exilis*), and other species (L. G. Allen and Franklin, 1992, M. J. Allen and Herbinson, 1991, Chapter 6). Anthropogenic debris (debris consisting of plastic and metal materials, fishing gear, bottles, and cans) on the shelf of southern California was more common in bays and harbors and on the middle (31–100 m) and outer shelf (101–200 m) zones (Moore and M. J. Allen, 2000). Bay and harbor debris consisted largely of plastics and cans (suggesting land-based and boating sources) (M. J. Allen et al., 2002a) whereas the deepwater debris consisted largely of fishing gear, cans, and bottles (suggesting fishing sources) (Moore and M. J. Allen, 2000).

ORGANISMS

Marine organisms that accumulate contaminants are an important repository and means of redistributing contaminants in the marine environment (Schiff et al., 2000). In a review of contaminant trends in the SCB (Mearns et al., 1991), trace metals, chlorinated pesticides (DDTs, chlordane, dieldrin), and PCBs were found in invertebrates, fishes, seabirds, and marine mammals. Trace metals such as silver, cadmium, chromium, and copper were accumulated in macroinvertebrates but not fish. Mercury did not accumulate in invertebrates but did in fish. Among invertebrates, elevated levels of DDTs and PCBs have been found in southern California in scallops, clams, squid, mysids, shrimp, lobsters, sand crabs, and crabs (Mearns et al., 1991). DDTs, PCBs, chlordane, dieldrin, and PAHs have been found in fishes.

Bioaccumulation of Contaminants in Fish

Bioaccumulation in fish is the accumulation of contaminants in tissue as a result of dietary consumption and bioconcentration (Cardwell, 1991). Bioconcentration occurs when contaminants are absorbed from the water by the gills or epidermis. However, most bioaccumulation in fish results from dietary consumption of food organisms with relatively low levels of contaminants. Most contaminants found in marine organisms are hydrophobic, and accumulate in lipid reservoirs in the organism. As accumulation of hydrophobic contaminants from the environment occurs at a higher rate than the catabolic transformation of these compounds, concentrations increase in the fish above that of the environment or their prey. When predators consume contaminated fish and predators at higher trophic levels in turn consume these, tissue concentrations are biomagnified up the food chain. This results in species at higher trophic levels having higher levels of contaminants than those at lower levels.

As noted above, marine fishes collected from contaminated bay and coastal waters of California often have elevated tissue concentrations of certain contaminants. Although assessments of chemical contaminants in some areas (e.g., parts of southern California and San Francisco Bay) find elevated concentrations of many organic compounds and trace metals in sediments, only some are elevated in fish tissue. Contaminants likely to be high in fish tissue include chlorinated hydrocarbons (e.g., PCBs and pesticides DDT, chlordane, and dieldrin in liver and muscle tissue; PAHs in stomach contents and PAH metabolites in bile) and trace metals (e.g., arsenic, mercury, selenium, and tin) (Mearns et al., 1991). The chlorinated hydrocarbons are hydrophobic and associate with lipids in fish tissue. Similarly, mercury, arsenic, selenium, and tin are complexed with organic compounds, resulting in their accumulation in lipids in fish tissues. The low rate of catabolic destruction of hydrophobic contaminants relative to a higher rate of accumulation from diet or water generally results in higher concentrations in fish tissue relative to the environment (Muir et al., 1990, Pastor et al., 1996). Bioaccumulation of trace metals occurs among those that are organically complexed (e.g., methyl mercury, arsenobetaine, and organotins). However, not all contaminants consumed by fish are bioaccumulated. Trace metals (e.g., cadmium, copper, and zinc) are detoxified in the liver and safely assimilated by California scorpionfish (*Scorpaena guttata*), white croaker (*Genyonemus lineatus*), and Dover sole (*Microstomus pacificus*) in southern California (Brown et al., 1985).

Distribution of Contaminants in Fishes

The highest concentrations of contaminants in fish tissue are generally found in fish living near the source of the contaminants (e.g., discharge site, contaminated sediment) (Mearns et al., 1991) but pelagic fish and wide-ranging demersal fish that become contaminated in these areas may carry contaminants away from the source. In California, the highest levels of contaminants in marine fishes usually occur in coastal southern California and San Francisco Bay (Mearns et al., 1988). In a comparison of fish contamination in coastal areas around the United States (Mearns et al., 1988), DDTs in muscle and liver tissue were much higher in the SCB at the Palos Verdes Shelf than in the rest of California (fig. 23-1). The highest values in muscle tissue were 200 ppm (concentrations are reported as wet weight in this chapter) in spiny dogfish (*Squalus acanthias*), 176 ppm in white croaker, and 123 ppm in Dover sole (all from the Palos Verdes Shelf in the 1970s and early 1980s) (Mearns et al., 1991). Striped bass (*Morone saxatilis*) from the Sacramento River mouth in San Francisco Bay had 7 ppm DDT in the 1970s (Mearns et al., 1988). DDT concentrations in liver tissue are typically about 13 times higher than in muscle tissue from the same fish. The highest concentrations of DDT in fish livers were 1,589 ppm in Dover sole from the Palos Verdes Shelf in 1977 and 1,026 ppm for rockfish (*Sebastes* sp.) (Mearns et al., 1988, 1991). Rockfish from Santa Monica Bay had median liver values of 349 ppm in 1970 (Mearns et al., 1988). The average liver value in San Francisco Bay fish in the 1970s was 1 ppm whereas average concentrations from the Palos Verdes Shelf ranged from 0.9 to 419 ppm (Mearns et al., 1988, 1991). At depths of 500–1000 m off the Farallon Islands in central California in 1985, DDT concentrations were 9 ppm in a sablefish (*Anoplopoma fimbria*) and 2 ppm in a Dover sole (Melzian et al., 1987). Although the maximum Dover sole values were much lower than in the SCB, the sablefish concentration was higher (Melzian et al., 1987, Mearns et al., 1991).

PCBs were also higher for muscle and liver tissue in southern California than in San Francisco Bay (fig. 23-2) (Mearns

FIGURE 23-2 Total PCBs (Aroclor) in a) liver and b) muscle of coastal and marine fish from 19 sites sampled in 1976 and 1977. Bar represents mean of all individuals from all species collected at a site (after Mearns et al. 1988).

et al., 1988). Highest muscle values were 14.8, 10.4, and 10.0 ppm in spiny dogfish, Dover sole, and white croaker, respectively, from the Palos Verdes Shelf in 1975-1981 (Mearns et al., 1991). In contrast, the highest concentration in San Francisco Bay was 4.0 ppm in striped bass from the Sacramento River estuary (Gadbois and Maney, 1982, Mearns et al., 1988). Liver concentrations were highest in Dover sole (162 ppm), followed by Pacific sanddab (*Citharichthys sordidus*) (39 ppm), and English sole (*Parophrys vetulus*) (22 ppm) from the Palos Verdes Shelf in the 1977 (Mearns et al., 1991). Mean concentrations in San Francisco Bay fish were up to 2 ppm (Mearns et al., 1988). PCB concentrations at depths of 500–1000 m off the Farallon Islands were 4 ppm for sablefish and 2 ppm for Dover sole in 1985 (Melzian et al., 1987). Liver concentrations for sablefish and Dover sole were higher (28 and 162 ppm) in the SCB (Melzian et al., 1987, Mearns et al., 1991).

From the late 1960s to early 1980s, PCBs in fish were high on the East Coast from New England to Georgia, along the Gulf Coast, and on the west coast in Puget Sound, San Francisco Bay, and the SCB (fig. 23-2; Mearns et al., *19*88). The highest reported values were 22.0 ppm in muscle tissue of white perch (*Morone americana*) from the Hudson River area in 1980. The highest whole fish concentration was 6.9 ppm in Atlantic menhaden (*Brevoortia tyrannus*) from New Jersey in 1972 (Mearns et al., 1988). Highest liver concentrations were 162 ppm for Dover sole on the Palos Verdes Shelf (Mearns et al., 1991) and 160.0 ppm in starry flounder (*Platichthys stellatus*) in Puget Sound, both in 1977 (Sherwood, 1982). With the exception of southern California, PCBs were highest in urban embayments on Pacific and East Coasts, and near Pensacola, FL, but were low in fish tissue from southeastern and Gulf of Mexico estuaries (Mearns et al., 1988).

In the 1970s, fish contaminated with other pesticides (chlordane, dieldrin, toxaphene) were widespread in coastal estuaries of the East and Gulf Coasts and less so on the west coast (Mearns et al., 1988). Chlordane was highest in the New Jersey area and southern Texas; dieldrin in the New Jersey area, southwest Florida, and southern Texas; and toxaphene in southern Texas, and near Louisiana and Alabama.

In contrast to DDT and PCB, dieldrin was found only in estuaries, and was much higher in fish livers from San Francisco Bay than in the SCB (Mearns et al., 1988). Highest values (0.09 ppm) in San Francisco Bay were found in livers of starry flounder and white croaker. Whole tissue values for dieldrin, endrin, and chlordane in juvenile estuarine fish were low and were similar between San Francisco Bay and southern California (Mearns et al., 1988).

In a regional survey of the southern California shelf (depths of 10-200 m) in 1994, flatfish (Dover sole; Pacific sanddab; longfin sanddab, *Citharichthys xanthostigma*) tissue was analyzed for 14 chlorinated hydrocarbons, including PCBs and 13 pesticides (including DDTs), and of these, only DDTs and PCBs were found (Schiff and Allen, 2000). Flatfishes, which live in these sediments, accumulate DDT and PCB in proportion to the concentration in the sediments (fig. 23-3) (Schiff and Allen, 2000; M. J. Allen et al., 2002a). Hence, the highest levels of DDT in southern California fish have been found near the Palos Verdes area (Mearns et al., 1988, 1991). Within the SCB, DDT concentrations are highest on the Palos Verdes Shelf near the wastewater outfall, followed by Santa Monica Bay, and from there decreasing to the northwest, although low levels of DDT are found in fish from the Palos Verdes Shelf to San Diego (Mearns et al., 1991; M. J. Allen et al et al., 2002a). Fish with highest PCB concentrations in fish tissue are also found

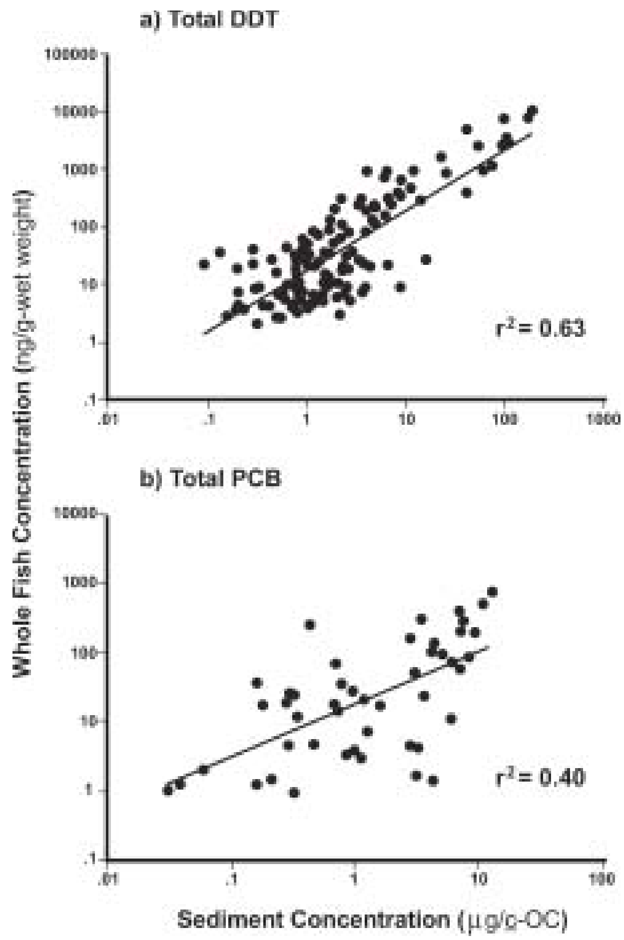

FIGURE 23-3 Relationships between a) total DDT and b) total PCB concentrations in whole fish composites of sanddab guild species and in sediments on the southern California shelf at depths of 3–187 m, July–September 1998. Dashed lines are best fit to the data from linear regression (after M. J. Allen et al. 2002a).

on the Palos Verdes Shelf, but high concentrations are also found in bays and harbors.

Chlordane concentrations in fish tissue have not been widely studied in the SCB. High muscle tissue values were 0.020 ppm for California corbina (*Menticirrhus undulatus*) in 2001 (M. J. Allen et al., 2004) and 0.019 ppm in kelp bass (*Paralabrax clathratus*) from the Palos Verdes Shelf in 1985 (Mearns et al., 1991). The highest liver concentration (162 ppm) occurred in Dover sole from the Palos Verdes Shelf (Mearns et al., 1991). In whole fish tissue, the highest concentration of 0.022 ppm was recorded in arrow goby (*Clevelandia ios*) from Newport Bay in 2002 (M. J. Allen et al., 2004).

The Palos Verdes Shelf did not have the same importance as a locus of high fish tissue concentrations of trace metals as it did for DDTs and PCBs. High arsenic concentrations (0.012 and 0.009 ppm) were found in Pacific sanddab at Santa Catalina Island in 1973 and spotted turbot (*Pleuronichthys ritteri*) in Newport Bay in winter 2001, respectively (Mearns et al., 1991; M. J. Allen et al., 2004). In whole fish tissue, an arsenic concentration of 0.001 ppm was found in diamond turbot (*Pleuronichthys guttulatus*) and California halibut from Newport Bay (M. J. Allen et al., 2004). Highest values of selenium in fish muscle tissue (0.004 and 0.002 ppm) and have been found in Pacific bonito (*Sarda chiliensis*) at Huntington Beach and kelp bass on the Palos Verdes Shelf, respectively (Mearns et al., 1991). Finally, selenium concentrations as high as 2 ppm were found in whole fish tissue from California killifish (*Fundulus parvipinnis*) in Newport Bay in 2002 (M. J. Allen et al., 2004).

High concentrations of mercury (10.6 ppm) have been found in muscle tissue of white shark (*Carcharodon carcharias*) from Santa Catalina Island as might be expected from its high trophic position (Mearns et al., 1991). Among ray-finned fishes, concentrations as high as 5.5 ppm have been found in California scorpionfish from Dana Point and 1.9 ppm in shortfin mako (*Isurus oxyrinchus*) from the waters over the San Pedro Basin (Mearns et al., 1991).

As inputs of contaminants from all sources have decreased (Mearns, 1993, Schiff et al., 2000), concentrations of contaminants in water, sediments, and marine organisms have also decreased. Contaminants in southern California marine fishes have decreased by an order of magnitude or more in wastewater discharge and reference areas from the 1970s to the 1990s (Mearns, 1993; Schiff and Allen, 2000). However, present and historical inputs continue to contaminate California marine fishes.

Bioaccumulation of Contaminants by Trophic Types of Fish

Fishes of different trophic levels and feeding habits sometimes show differences in bioaccumulation of contaminants. Within an ecosystem, the food web typically consists of primary producers (e.g., phytoplankton), primary consumers (e.g., zooplankton), secondary consumers (e.g., planktivorous fishes), tertiary consumers (e.g., carnivorous bony fishes, seabirds), and higher-level consumers (e.g., large carnivorous sharks, marine mammals). Although determination of the trophic level of a species is typically determined from examination of a species diet, it has also been determined chemically, for instance using ratios of cesium and potassium concentrations in fishes (Young et al., 1980). Potassium, an essential electrolyte, is maintained at constant levels in tissues but cesium (a trace element) is not. Cesium, however, has a half-life 2 to 3 times that of potassium and accumulates up the food chain. In a structured food web (as described above), the cesium/potassium (Cs/K) ratio increases with increasing trophic level, whereas in an unstructured food web (where species feed at more than one trophic level), an increase in CS/K ratio is not apparent with presumed trophic position (Isaacs, 1972, Young et al., 1980). The increase of Cs/K with trophic level in structured food webs parallels increases in contaminants that biomagnify in these food webs. An examination of the food web (including fish species) in the Salton Sea, Newport Bay, Palos Verdes Shelf, and San Pedro Channel of southern California from 1975 to 1978 showed that the simple food webs of the Salton Sea (all introduced species) and San Pedro Channel (pelagic) were structured and those of Newport Bay and Palos Verdes Shelf (benthic and pelagic) were unstructured. Mercury and chlorinated hydrocarbons (DDT and PCB) generally increase up structured food webs but other trace metals do not (Young et al., 1980).

Contaminant concentrations from fish in a given area are often similar among fish species that feed in the same way but differ among species with different feeding habits. M. J. Allen et al. (2002b) compared bioaccumulation of DDT in sanddab-guild species of southern California (speckled sanddab, *Citharichthys stigmaeus*; longfin sanddab; Pacific sanddab; and slender sole, *Lyopsetta exilis*). Species of this guild (benthic pelagobenthivore guild of M. J. Allen, 1982; also see chapter 7) are small flatfishes with medium-sized mouths that feed on nektonic and benthic prey and represent the most widespread guild of fishes on the soft bottom shelf of southern California (M. J. Allen et al., 1998, 2002a). DDT concentrations in species pairs from the same sites increased with increasing concentrations (fig. 23-4). Log-transformed DDT concentrations in whole fish samples of different species of this guild of the same age and from the same site were highly correlated among all species of the guild. Relationships between species were linear and did not differ from unity. The variability of DDT levels in species pairs between sites was 60 times that of replicates within a species. In contrast, the variability among species and among ages was 4 and 2 times, respectively, that of replicates. The similar response of species of this guild to the same environmental exposure of DDT suggests that this widespread guild could be used as a super species to assess the extent of fish contamination in bathymetrically diverse shelf areas.

Effects of Contamination

The previous sections have focused on the distribution of contaminants in the environment and specifically in fish on the California coast. In addition to accumulating in marine organisms, these contaminants can have detrimental effects on individual fish, populations and assemblages, fish predators, and human consumers of fish. The primary focus of these studies has been on human health risks to humans and to bird and mammal predators that consume them.

HUMAN HEALTH RISKS

Public concern regarding potential health risks from eating contaminated fish and invertebrates came to light as persons consuming mercury contaminated marine organisms from Minamata Bay became ill or died in the 1950s and 1960s (Irukama, 1966, Ui, 1969). Concerns also developed for other contaminants, and in 1970–1971, white croaker, canned jack mackerel (*Trachurus symmetricus*), and Pacific bonito with DDT concentrations above 5 ppm were seized by the U.S. Food and Drug Administration (MacGregor, 1974, Stull et al., 1987). Further concerns regarding high levels of DDT and PCB contamination in fish caught on the Palos Verdes Shelf and Santa Monica Bay prompted posting of fish consumption guidelines for these areas by California Department of Health Services (Stull et al., 1987). This health advisory recommended that all white croaker not be consumed from these areas and no consumption of any fish caught in the outfall area of Palos Verdes Shelf and in parts of Los Angeles/Long Beach (LA/LB) Harbors. It also recommended restricted consumption by all women of childbearing age and young children; no consumption of fish livers; and healthy cooking methods. Since 1991, health advisories have been issued for about 50 species in southern California that had DDT and PCB concentrations in muscle tissue above 0.1 ppm (OEHHA, 2001). Health advisories were issued for nine locations from Point Dume to Dana Point. Advisories restricting fish consumption were posted for eight types of fish (rockfishes; California scorpionfish; kelp bass; black croaker, *Cheilotrema saturnum*; white croaker; California corbina; queenfish, *Seriphus politus*; and surfperches, Embiotocidae) at one or more locations, with white croaker having the most. Similar advisories have also been posted in San Francisco Bay for fishes with high levels of mercury, PCBs,

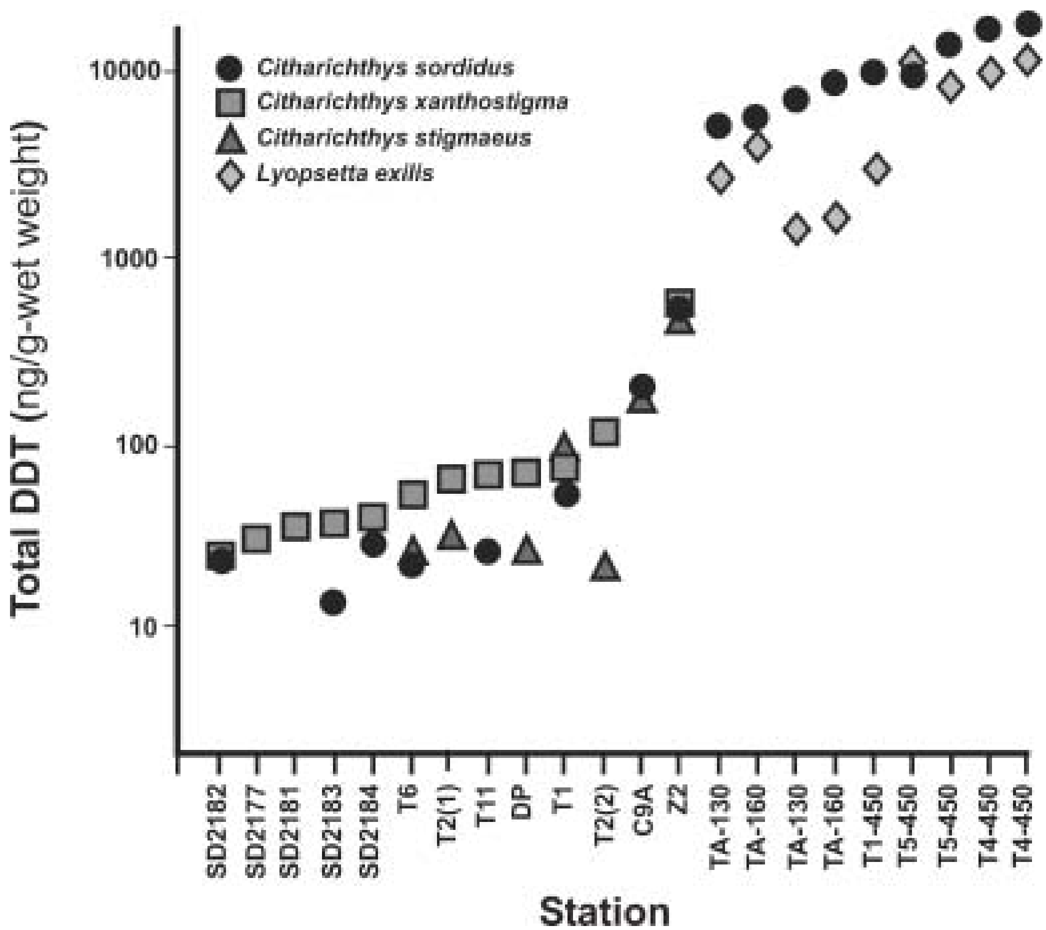

FIGURE 23-4 Total DDT concentrations in whole fish composites of sanddab guild species: Pacific sandab (*Citharichthys sordidus*); longfin sanddab (*Citharichthys xanthostigma*); speckled sanddab (*Citharichthys stigmaeus*); and slender sole (*Lyopsetta exilis*). Sediment contamination gradient increases to right. SD stations are off San Diego, T6 to T2 on San Pedro Shelf, C9A Santa Monica Bay, Z2 outfall station Santa Monica Bay, and stations to right of Z2 are on the Palos Verdes Shelf, California. Samples collected in 1997 (after M. J. Allen et al., 2002b).

and other contaminants, including sturgeon (*Ascipenser* spp.), striped bass, sharks, croakers, surfperches, and gobies (OEHHA, 2001).

As noted above, advisories are posted when contaminants exceed State of California (OEHHA) screening values, which are lower (and hence more conservative) than United States Environmental Protection Agency values (M. J. Allen et al., 2004). For contaminants found in California marine fishes consumed by anglers, current OEHHA screening values are 0.100 ppm DDT, 0.020 ppm for PCBs, 0.030 ppm chlordane, 0.300 ppm mercury, and 2 ppm selenium) (M. J. Allen et al., 2004). Health concerns are based on toxic or carcinogenic effects of the contaminant. Contaminants of toxic concern include trace metals (e.g., inorganic arsenic, mercury, selenium, and PCBs) and those of carcinogenic concern are DDTs, PCBs, and chlordane (USEPA, 1995). Risk-based screening values for carcinogens are based on the effective dose of the contaminant (over a 70-year period), its concentration in the tissue, the mean daily consumption rate, relative absorption coefficient, and mean body weight. For toxic contaminants, the effective dose is the effective ingested dose with a specified level of risk from dose-response studies, with the remaining variables the same as for carcinogens (USEPA, 1995). Fish consumption rates are now based on field studies that obtain species-specific consumption rates by California anglers in the Los Angeles area (Puffer et al., 1982), Santa Monica Bay (M. J. Allen et al., 1996), and San Francisco Bay (SFEI, 2000).

WILDLIFE HEALTH RISKS

In addition to posing health-risks to humans, contaminated fish can pose health-risks to wildlife predators. Brown pelican (*Pelecanus occidentalis*) and double-crested cormorant (*Phalocrocorax auritus*) (both piscivorous) and bald eagle (*Haliaeetus leucocephalus*) (partly piscivorous) nesting on southern California islands had poor reproductive success during the late 1950s to 1970s due to eggshell thinning (Anderson and Hickey, 1970; Risebrough et al., 1971). Birds with high tissue DDT concentrations had thinner eggs and, over time, egg thickness increased as tissue DDT concentrations decreased (Gress, 1994). California sea lions with high levels of DDT gave birth to more premature births than those with lower levels (Connolly and Glaser, 2002).

Contaminant levels in California marine fish have generally not been assessed relative to wildlife (predator) health-risk guidelines, as there are currently no accepted Federal or California State guidelines. However, Canada has developed guidelines for wildlife health-risk to marine and aquatic organisms from DDT and PCBs (Environment Canada, 1997, 1998).

These guidelines are for DDT and PCB concentrations in whole fish tissue. The DDT guideline is 0.014 ppm (Environment Canada, 1997). The PCB guideline is 0.79 ppt (parts per trillion) toxicity equivalency quotient (TEQ) (Environment Canada, 1998). Whereas human health-risk guidelines are based on total PCB, the wildlife PCB guideline is based on concentrations of 12 toxic PCB congeners, and the relative toxicity of these to tetrachloro-dibenzo-p-dioxin (TCDD). The 12 congeners vary in toxicity to birds and mammals.

In 1998, the sanddab-guild species were targeted (all of which had similar uptake of DDT with the same exposure; M. J. Allen et al., 2002b) to obtain comparable samples across the entire shelf from depths of 5–200m and assess fish contamination throughout the southern California shelf (M. J. Allen et al., 2002a). In this study, sanddab-guild fish with DDT concentrations higher than the wildlife health-risk guideline (Environment Canada, 1997) occurred in 71% of the southern California shelf area (M. J. Allen et al., 2002a). Whereas fish with DDT levels above the guideline were widespread, including on the Channel Islands, those with high PCB levels were not. Fish with PCB toxicity equivalent quotient (TEQ) concentrations higher than the mammal and bird wildlife health-risk guideline (Environment Canada, 1998) occurred in 8% and 5% of the area, respectively. These were areas near large wastewater discharges with historically contaminated sediments, and within harbors (for marine mammal risk) and around the southeast Channel Islands (for bird risk). DDT levels above these guidelines were also widespread in Newport Bay in 2002, occurring in samples of all nine species of forage fishes examined (M. J. Allen et al., 2004). Although a large area of concern for DDT was identified using this guideline in these studies, more research is needed to determine to what degree there is a real risk to specific bird and mammal predators of marine fish in these areas.

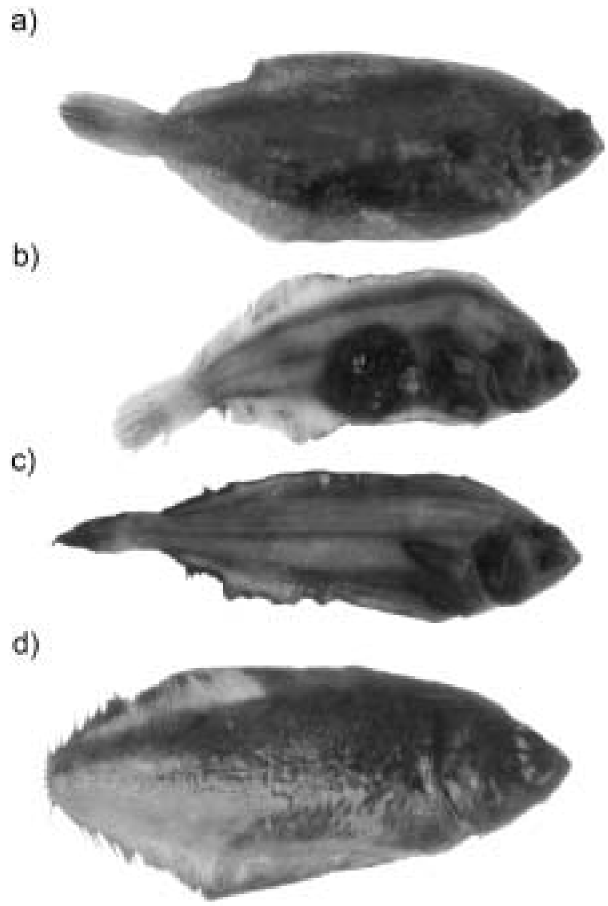

FIGURE 23-5 Dover sole *(Microstomus pacificus)* collected on the Palos Verdes Shelf, California, in 1981: a) normal; b) epidermal tumor and fin erosion; c) fin erosion; and d) skeletal abnormality (no caudal fin). (Photos by Dario Diehl, Southern California Coastal Water Research Project).

EXTERNAL DISEASES, ANOMALIES, AND PARASITES

Beginning in the late 1950s fisheries biologists, anglers, and divers began to note observations of morbid and abnormal fish from Santa Monica Bay, the Palos Verdes Shelf, and LA/LB Harbor area and became aware of many fish with external diseases and anomalies (Young, 1964). These were interpreted as resulting from wastewater discharge in Santa Monica Bay, the Palos Verdes Shelf, and from sites in LA/LB Harbor. Anomalies included abnormally soft California halibut, spotted turbot of low body weight, exophthalmia (abnormal protrusion of the eye) in spotfin croaker (*Roncador stearnsii*) and white seabass, lesions in white seabass and juvenile Dover sole, lip papillomas in white croaker, and papillomas in California tonguefish (*Symphurus atricaudus*), basketweave cusk-eel (*Ophidion scrippsae*), and Pacific sanddab (Young, 1964). As more comprehensive studies were conducted during the 1970s, several important diseases or anomalies were identified in coastal fishes of southern California: fin erosion, skin tumors in flatfishes, oral papillomas in croakers, color anomalies, exophthalmoses, and skeletal anomalies (Mearns and Sherwood, 1977).

Fin erosion levels were very high in the early 1970s on the Palos Verdes Shelf near the POTW outfall, affecting 33 of 151 species examined (Mearns and Sherwood, 1977). In fin erosion disease, the fins that erode are those in contact with the sediments (i.e., dorsal, anal, and caudal fins of flatfishes; pelvic, anal, and ventral part of caudal fins in roundfishes that live or rest on the bottom; personal observations by author). Dover sole was the species with the highest prevalence (30%) of fin erosion in the 1970s, almost all of which occurred on the Palos Verdes Shelf (fig. 23-5 and fig. 23-6). By the middle 1980s, fin erosion had nearly disappeared in Dover sole from that area (Stull, 1995) and has been virtually absent there and throughout the Southern California Bight since the early 1990s (fig. 23-6)(M. J. Allen et al., 1998, 2002a). Although intensely studied, the cause of this disease was not determined. It is assumed to be related to sediment contamination, based upon its greatest prevalence in benthic fishes found near the POTW outfall and its disappearance as wastewater effluent quality improved. Tail erosion in white croaker (in which the entire rather than the lower part caudal fin was eroded) appeared to be a different disease and was found from Santa Monica Bay to San Pedro Bay in the 1970s (Mearns and Sherwood, 1977); this disease is rarely reported at present. [As a historical side note, it is because white croaker sometimes had no caudal fin in the 1970s that all non-fisheries demersal fish surveys in southern California measure standard length rather than total length.]

Tumor-like diseases have been found in Dover sole, white croaker, and occasionally in other species. These are variously called lesions, tumors, papillomas, pseudotumors, and neoplasms. Epidermal tumors were found consistently in

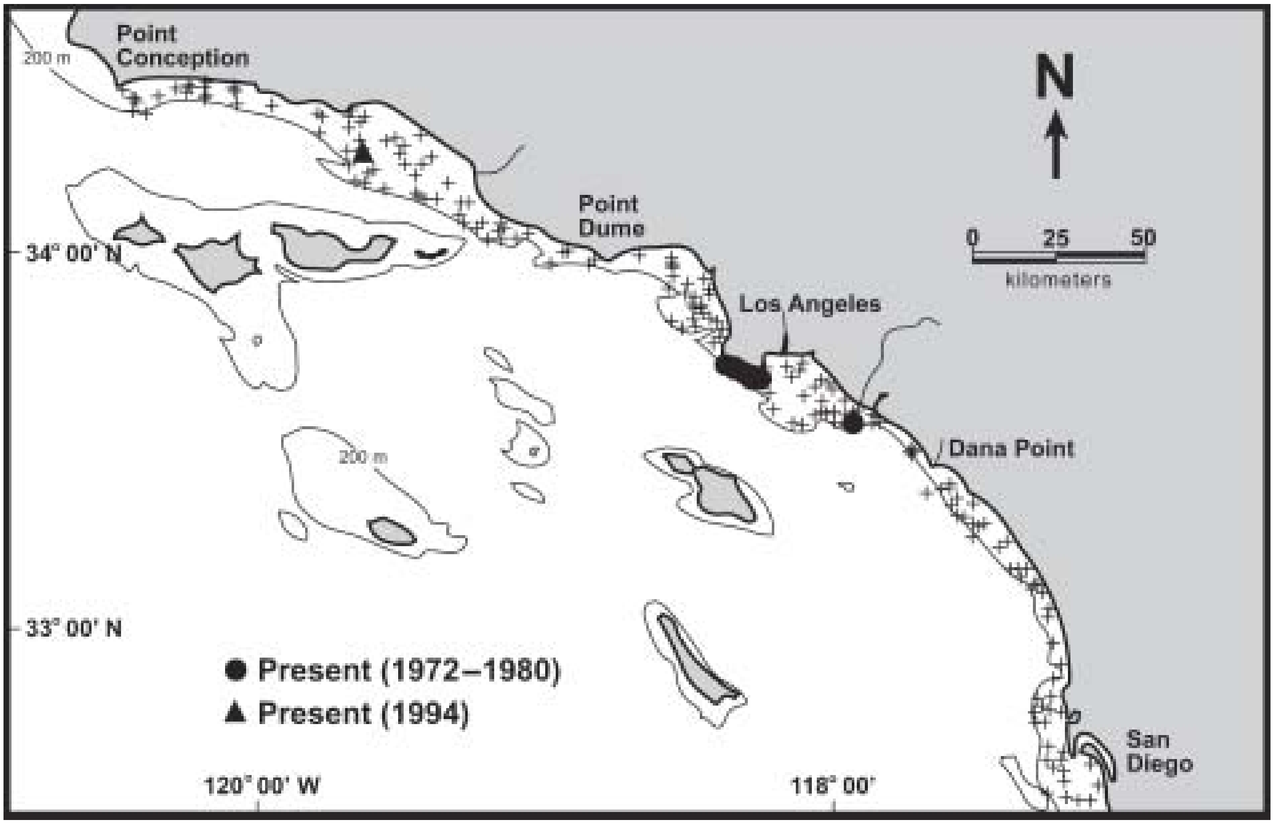

FIGURE 23-6 Distribution of fin erosion on the mainland shelf from 1973 to 1980 (only on the Palos Verdes Shelf and one record on south San Pedro Shelf) and in 1994 (one record in Santa Barbara Channel). From the mid-to-late 1990s to the present, fin erosion has been virtually absent from the Palos Verdes Shelf. + = trawl stations sampled in 1994 (after M. J. Allen et al., 1998)

Dover sole sampled from the SCB in the early 1970s (fig. 23-5) (Mearns and Sherwood, 1977). This disease was found almost entirely in small (6–12 cm in length) juveniles and the prevalence (percent fish affected) was similar among areas in the central mainland coast of the SCB. It was particularly noticeable on the Palos Verdes Shelf because of the large number of juvenile Dover sole found there. The prevalence of the disease on the Palos Verdes Shelf decreased with increasing distance from the wastewater outfall (Cross, 1988). The disease was found outside the SCB as far back as 1946 (Mearns and Sherwood, 1977). Incidences of this disease have decreased since the 1970s, and are presently at background levels (M. J. Allen et al., 1998, 2002a). These x-cell pseudotumors are probably the result of amoeboid parasitism (Dawe et al., 1979, Cross, 1988). Oral papillomas in white croaker and other species have been found near Los Angeles, but the prevalence of these abnormalities has been low (Young, 1964, Mearns and Sherwood, 1977). In contrast to Dover sole, the prevalence of these papillomas in white croaker increased with increasing size of the fish (Mearns and Sherwood, 1977). Histological analysis of the oral papillomas in white croaker determined that they were not malignant and were likely due to mechanical, chemical, or infectious irritation (Russell and Kotin, 1957, Young, 1964). Epidermal tumors have also been found in English sole in San Francisco Bay, with a prevalence of 12% and up to 33 tumors per fish in the northern part of the bay near industrial waste discharge (Cooper and Keller, 1969; Sindermann, 1979). English sole with tumors have not been found in southern California.

Other external anomalies, such as color anomalies (ambicoloration, albinism) in flatfishes and skeletal anomalies, have also been observed (Mearns and Sherwood, 1977). The most frequent color anomaly was ambicoloration (where eyed-side pigmentation occurs on parts of the blind side). Skeletal anomalies occur occasionally in a number of species, and include snub noses, no tail (hypural plates absent; fig. 23-5), bent fin rays, deformed spinal columns, and deformed gill rakers (Valentine, 1975, Sindermann, 1979). There is no clear evidence that the color anomalies are related to contamination. Some of these may occur during larval development in the water column. However, bent rays in fins in contact with sediments at Palos Verdes may be related to sediment contamination.

The relationship of fish diseases to marine pollution has been examined primarily in southern California, Puget Sound, Gulf of Saint Lawrence, and mid-Atlantic states (e.g. Mearns and Sherwood, 1977; Sindermann, 1979, Malins et al., 1984; Stein et al., 1993; Fournie et al., 1996). The fin erosion disease of benthic fishes found in southern California occurs in fins in contact with contaminated sediments whereas that of pelagic nearshore fishes erodes the caudal fin, which is not in contact with sediments (Mearns and Sherwood, 1977, Sindermann, 1979). Fin erosion similar (but not necessarily the same) to that found in southern California Dover sole has been found on the east coast in winter flounder (*Pseudopleuronectes americanus*), summer flounder (*Paralichthys dentatus*), in Puget Sound in starry flounder and English sole, in plaice (*Pleuronectes platessa*) and dab (*Limanda limanda*) in the Irish Sea, and in Japanese stargazers (*Uranoscopus japonicus*) off Japan (Perkins et al., 1972,

TABLE 23-1

Overall Prevalence of Anomalies in Demersal Fish Populations

Location	*Years*	*Percent Anomalies*	*Study*
Southern California	1969–1976	5.0	Mearns and Sherwood 1977
Southern California	1994	1.0	M. J. Allen *et al.* 1998
Southern California	1998	0.5	M. J. Allen *et al.* 2002a
Gulf Coast	1991–1992	0.7	Fournie *et al.* 1996
Mid-Atlantic Coast	1990	0.5	Fournie *et al.* 1996

Nakai et al., 1973, Murchelano, 1975, Wellings et al., 1976, Sindermann, 1979). Tail erosion similar to that found in white croaker has been found in Atlantic croaker (*Micropogonias undulatus*) (the ecological counterpart to white croaker on the East Coast) and spot (*Leiostomus xanthurus*) (Couch and Nimmo, 1974). Fin erosion likely results from chemical stress and low dissolved oxygen concentrations and possibly enhanced hydrogen sulfide, and in some cases secondary bacterial infection (Sindermann, 1979).

Epidermal tumors or neoplasms have been found in a variety of fishes on both coasts. In southern California, the primary occurrence is in Dover sole, and secondarily in white croaker (Mearns and Sherwood, 1977). Elsewhere in California, these have been found in English sole in San Francisco Bay (Cooper and Keller, 1969). Elsewhere on the Pacific Coast, epidermal tumors have been prevalent in Pacific cod (*Gadus macrocephalus*), starry flounder (Puget Sound), flathead sole (*Hippoglossoides elassodon*; Puget Sound), sand sole (*Psettichthys melanostictus*; British Columbia), and rock sole (*Lepidopsetta bilineata*; Bering Sea) (Nigrelli et al., 1965, McArn and Wellings, 1971, Miller and Wellings, 1971, Dawe et al., 1979, Sindermann, 1979). These tumors are likely caused by amoebic parasites or viruses, and in many cases may be a natural disease (Dawe, 1979, Sindermann, 1979).

Skeletal anomalies and genetic anomalies also occur in fish from other geographic areas (Sindermann, 1979). In 1998, the prevalence of external anomalies in 1998 in demersal fishes on the southern California shelf was similar to background anomaly rates in mid-Atlantic (0.5%) and Gulf Coast (0.7%) estuaries (table 23-1) (Fournie et al., 1996; M. J. Allen et al., 2002a). Toxicopathic hepatic lesions have been found in fishes from contaminated sites from southern California to Puget Sound (Malins et al., 1984, Myers et al., 1994, Roy et al., 2003), with prevalence of hepatic lesions in English sole being highest in Puget Sound, starry flounder in San Francisco Bay, and white croaker in San Francisco Bay and LA/LB Harbors (Myers et al., 1994).

Although parasitism in marine fishes is generally natural, the type and diversity of parasites and the prevalence (percent fish parasitized) and intensity (number of individual parasites) of parasites within a taxon may vary between contaminated and uncontaminated areas (Mearns and Sherwood, 1977, Perkins and Gartman, 1997, Kalman, 2001). Fish parasites may affect the health of their host fish or they may cause health problems in predators and humans that consume fish. Fish parasites consist of ectoparasites (external parasites; e.g., copepods, isopods, leeches) and endoparasites (internal parasites; e.g., nematodes, cestodes, trematodes). In the 1970s, the eye copepod (*Phrixocephalus cincinnatus*), which typically attaches to the eye of Pacific sanddab was not found in contaminated areas but was abundant in reference areas, perhaps a response to high chlorinated hydrocarbon concentrations in Pacific sanddab (Mearns and Sherwood, 1977). In 1989-1994, this parasite was more abundant on Pacific sanddab at Palos Verdes and Santa Monica Bay (Perkins and Gartman, 1997). Similarly, in 1998-1999 other ectoparasites (copepods, leeches, isopods) on flatfishes and scorpaeniform fishes were significantly more prevalent near the outfall than in non-outfall areas in Santa Monica Bay (Kalman, 2001). Other studies (M. J. Allen et al., 1998, 2002a; Hogue and Paris, 2002) did not find obvious response in the prevalence of fish parasites to outfall areas.

BIOMARKERS AND SUBLETHAL EFFECTS

Although contaminants accumulate in fish and pose health risks to consumers of fish, they can also affect the health of the fish itself. Effects include acutely toxic effects, which can cause a fish to die immediately, or chronic sublethal effects, which affect its health, behavior, or ability to reproduce. Although freshwater fish kills due to human activity occur occasionally, marine fish kills along the California coast are rare and generally related to red tides (Bongersma-Sanders, 1957). An extensive red tide in 1945 that extended from San Luis Obispo to Los Angeles Harbor killed sharks, stingrays, and California halibut in Santa Monica Bay (Sommer and Clark, 1946). Trawling sometimes result in dead by-catch fish floating at the surface, which might be mistaken for a toxic fish kill. Fish kills due to acute toxicity from contaminants are more likely to occur in shallow confined areas near shore than offshore, unless a broad area is affected (e.g., an oil spill).

High levels of total DDT and total PCB in white croaker from San Pedro Bay have been correlated with reproductive impairment, including decreased fecundity, decreased fertilization success, and decreased proportions of spawning females (Cross and Hose, 1988, Hose et al., 1989). Increased levels of these compounds were also correlated to subcellular damage in white croaker and kelp bass (Hose et al., 1987). An increased frequency of micronuclei, a by-product of DNA damage, was observed in blood cells of these two fish species, with increased concentrations of total DDT and total PCB. High levels of DDT and PCB of kelp bass had subtle differences in reproductive endocrine status (e.g., lower levels of maturational gonadotropin) (Spies and Thomas, 1995).

Biomarkers are biochemical compounds and histological effects produced in response to contaminant exposure. Whereas some contaminants bioaccumulate in fish tissue, others do not. Biomarkers provide a measure of an organism's exposure to potentially toxic contaminants. In the 1970s and

1980s white croaker had high levels of enlarged fatty-vacuolated livers containing degenerating cells and hepatic lesions in the Palos Verdes-Los Angeles Harbor area and in the Oakland Estuary where DDTs, PCBs, and/or PAHs were high (Malins et al., 1987, Myers et al., 1994). In a Pacific Coast survey of hepatic lesions in fish (Myers et al., 1994), prevalence of liver lesions of white croaker were highest in San Francisco Bay and LA/LB Harbor; in starry flounder these were highest in San Francisco Bay. Another biomarker is hepatic mixed-function oxidases (Spies et al., 1982). Elevated levels of these enzymes indicate exposure to oil and PCB. Pacific sanddab and speckled sanddab near wastewater outfalls (high PCB) and crude oil seeps in southern California had high levels of these enzymes relative to reference areas (Spies et al., 1982). Bile fluorescent aromatic compounds (FAC) concentrations (an indicator of PAH exposure) were high in juvenile California halibut from Marina del Rey, Long Beach Harbor, and Alamitos Bay relative to reference areas in 1989–1999 (Brown and Steinert, 2004). Of seven species of flatfishes examined, speckled sanddab had the highest levels of DNA damage, and hornyhead turbot (*Pleuronichthys verticalis*) the lowest. Although overall, no significant relationship was found between bile FAC concentrations and DNA damage, the incidence of DNA damage increased with bile FAC concentrations in Ventura Harbor and Marina del Rey. In contrast to biomarker studies in other coastal Los Angeles areas, examination of biomarkers (e.g., cytochrome P450 1A; vitellogenin levels, an indicator of endocrine disruption; bile FACS; DNA damage; and liver histopathology) in hornyhead turbot, English sole, and bigmouth sole (*Hippoglossina stomata*) near the wastewater outfall on the San Pedro Shelf in 2000 did not find adverse effects (Roy et al., 2003).

EFFECTS ON POPULATIONS AND ASSEMBLAGES

Fish assemblages in the SCB vary by habitat and depth, with distinct bay, rocky bottom, and soft-bottom assemblages (L. G. Allen, 1985). Demersal (soft-bottom) fish assemblages have been the focus of studies of pollution impact because wastewater outfalls in southern California are on soft-bottom habitat (M. J. Allen, 1982). Demersal fishes are relatively sedentary, easily collected by trawl, and respond to environmental stress. As there is a relatively steep bathymetric gradient along the southern California coast, demersal fish assemblages vary in species composition by depth, with different assemblages at different depths (see chapter 7).

Demersal fish assemblages have shown local shifts in abundance, biomass, species richness, diversity, and species composition in response to wastewater discharge in the SCB. In the early 1970s fish abundance, species richness, and diversity was depressed in the vicinity of wastewater outfalls in Santa Monica Bay and on the Palos Verdes Shelf (M. J. Allen, 1977). Depressed average fish abundance occurred over 27 km^2 and depressed species richness over 50 km^2 near the White Point Outfall on the Palos Verdes Shelf. From 1963 to 1974, fish abundance and diversity was low near the Hyperion 7-mile sludge pipe. When the Orange County Sanitation District moved its outfall from shallow (15 m) to deepwater (60 m) in the early 1970s, fish abundance increased 100% at the new outfall and decreased 50% at the old outfall (M. J. Allen, 1977).

In contrast to demersal fishes, the deepwater outfall pipes attracted large numbers of rockfishes and other species (M.J. Allen et al., 1976). These outfalls provide the only continuous hard-bottom substrate extending across the dominant soft-bottom habitat of the shelf in Santa Monica Bay, Palos Verdes Shelf, San Pedro Shelf, and Point Loma Shelf. Hence, they attract hard-bottom fishes characteristic of the outfall depth, and assemblages on these outfalls are quite different from those of the surrounding soft-bottom areas. Fish abundance and species richness along the Hyperion outfall pipes were much greater near the end of the outfalls in deepwater than in shallow water, in part due in part to dense populations of gigantic anemones (*Metridium farcimen*), which provided habitat complexity for some rockfishes, and the attraction of water-column fishes. Without comparison to assemblages on comparable natural hard-bottom habitats at the same depths, wastewater effects could not be determined (although it was apparent that the gigantic anemones and other sessile cnidarians were attracted to waste being discharged from diffuser ports) (M. J. Allen et al., 1976).

Among soft-bottom fishes, some species typical of reference assemblages were absent from outfall area on the Palos Verdes Shelf, whereas others were attracted to these areas (Allen, 1977). Species in reference assemblages that were missing in the outfall area included hornyhead turbot and California tonguefish. Species attracted to the outfall area included white croaker, shiner perch (*Cymatogaster aggregata*), and curlfin sole (*Pleuronichthys decurrens*). Changes in species composition were attributable to the feeding habits of fishes. Wastewater-related alteration of the sediments (primarily increased organic levels) favored infaunal polychaetes over crustaceans with concomitant shifts in the demersal fish community from crustacean feeding to polychaete-feeding species (M. J. Allen, 1977; M. J. Allen, 1982; Cross et al., 1985).

I developed a model of the functional organization of the soft-bottom fish communities that defined the community by the number and type of foraging guilds and dominant species within a guild at any depth (M. J. Allen, 1982). This model can be used to identify what species or guilds at a given depth are missing or added. Differences from the expected study can identify species or guilds on which to focus further studies to determine if altered components of the community are responding to a loss or gain in food (e.g., guild absent, food absent), response to a competitor (e.g., guild present, expected species different), or to other conditions (e.g., guild absent, food present—perhaps a direct response to contaminants). Application of this model to fish data from the Palos Verdes Shelf from 1972 to 1989 showed that nearest the outfall, benthic guilds (consisting of combfishes, sculpins, and poachers) that fed on small benthic gammaridean amphipods were absent (M. J. Allen unpublished data). Within the turbot (polychaete extractor) guild, the expected hornyhead turbot was replaced by curlfin sole at the outfall (M. J. Allen, 1982, Stull and Tang, 1996). White croaker and shiner perch, not consistently caught in deep water were consistently abundant near the discharge site. As conditions improved over time, these alterations generally went back to expected conditions (M. J. Allen unpublished data).

In the 1970s wastewater outfalls discharged high levels of suspended solids as well as contaminants, resulting in sediments with high levels of organic carbon near the outfalls. Sediments of high organic content are typically dominated by polychaetes whereas those with less organic carbon are dominated by crustaceans (e.g., amphipods). In the 1970s, the relative abundance of sanddab-guild fishes (predominantly crustacean-feeders; M. J. Allen, 1982, M. J. Allen et al., 2002b)

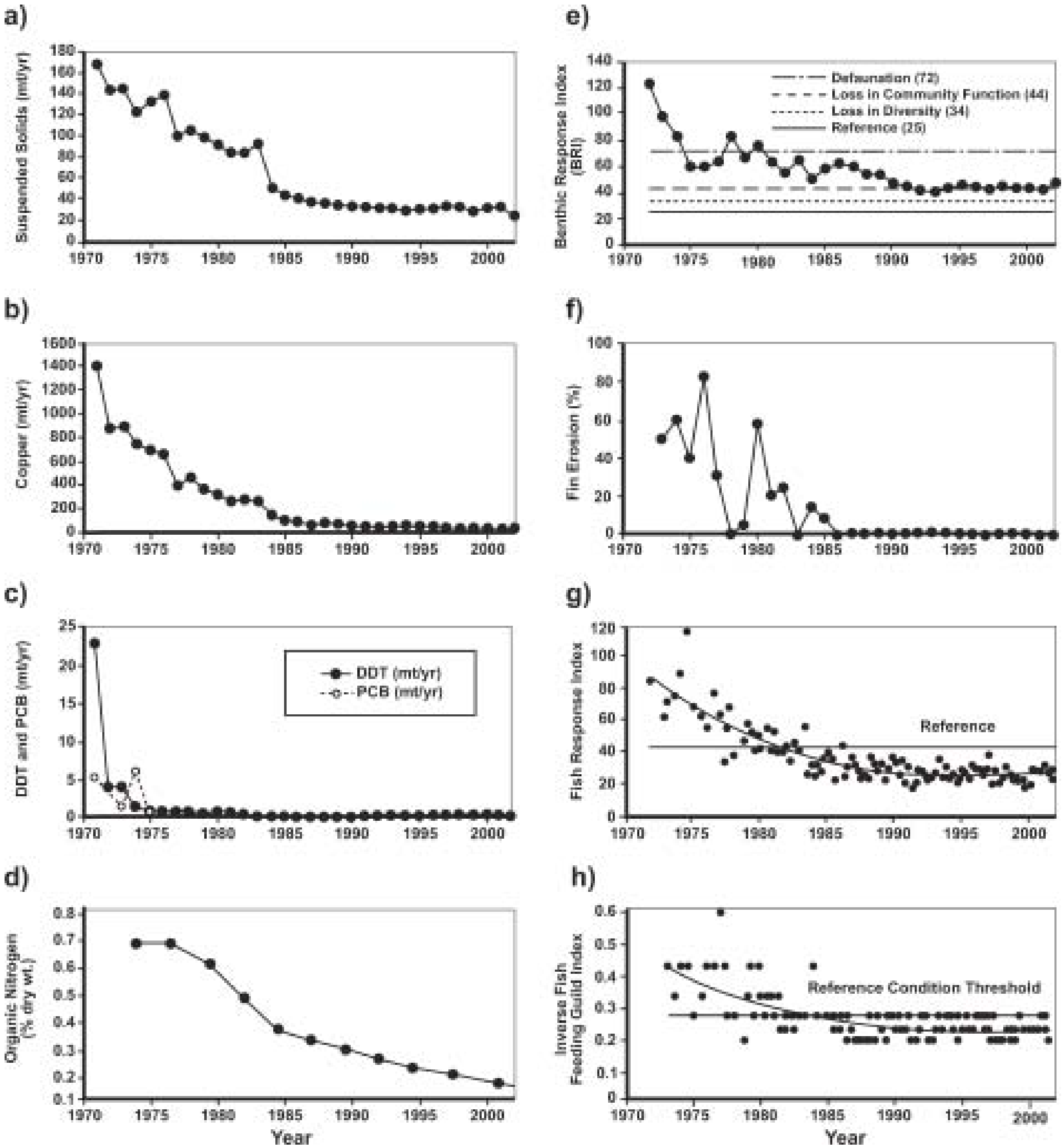

FIGURE 23-7 Effluent, sediment, and fish health metrics at the Palos Verdes Shelf wastewater discharge site from the early 1970s to 2002: a) mass emissions of effluent suspended solids; b) mass emissions of effluent copper; c) mass emissions of effluent DDT and PCBs; d) sediment organic nitrogen; e) Benthic Response Index; f) fin erosion in fish; g) Fish Response Index; and h) inverse of Fish Foraging Guild Index. Graphs modified from Montagne (2002a,b). Benthic biointegrity index after Smith et al. (2001); fish biointegrity indices after M. J. Allen et al. (2001).

was significantly lower on the Palos Verdes Shelf than that of turbot-guild species (polychaete-feeders; M. J. Allen et al., 2001). In contrast, the relative abundance of sanddab-guild species was significantly higher in reference areas (M. J. Allen et al., 2001). This relationship was used in developing a fish foraging guild biointegrity index (FFG) for the SCB (fig. 23-7h). In the same study, another biointegrity index (Fish Response Index, FRI; fig. 23-7g) was developed using the abundances of all species relative to the pollution gradient away from the Palos Verdes Shelf in the 1970s (as was done for infauna in the Benthic Response Index, BRI; Smith et al., 2001). Both indices, as well as prevalence of fin erosion (fig. 23-7e) identified a shift from impacted to reference assemblages on the Palos Verdes Shelf from the 1970s (when sediment organics and contamination was initially high) to the 1980s and later, as sediment conditions improved in response to improved effluent quality (fig. 23-7) (CSDLAC, 2002). The FFG index can be easily interpreted as shifting from an assemblage dominated by turbot-guild (benthic extracting benthivores; polychaete-feeding) fishes to sanddab-guild (benthic pelagobenthivore; epibenthic generalists) fishes in the early 1980s. Application of the FRI to the southern California shelf in 1998 showed reference-level fish assemblages throughout the SCB (except near the mouth of the Santa Clara River; M. J. Allen et al., 2002a). This area was severely disrupted by extremely high flow and a very large plume of suspended sediments from the Santa Clara River during the 1997–1998 El Niño, resulting in an assemblage with high abundance of shiner perch and white croaker, similar to that found on the Palos Verdes Shelf during the 1970s (M. J. Allen, 1977; M. J. Allen et al., 2002a). Other fish assemblage attributes (abundance, biomass, species richness, and diversity) did not differ at outfall areas from reference areas of similar depth on the shelf in region-wide assessments in 1994 and 1998 (M. J. Allen et al. 1998, 2002a).

Natural factors also play a role in changes to fish communities, and these must be considered as part of baseline reference conditions when assessing pollution or other anthropogenic effects on fish communities. If not considered, misleading interpretations and conclusions can result. For example, demersal fish communities changed somewhat from a coldwater to a warm water regime between 1972 and 1998, following a warming trend in SCB waters (chapter 7). This has made the assessment of fish populations on the Palos Verdes Shelf difficult as many coldwater species populations were decreasing and warm water species increasing at the same time that effluent quality and sediment contamination levels were improving.

Prospectus for Future Research

With rapidly increasing coastal populations in California (particularly in the south), pollution is likely to be a concern well into the future, even though the worst may be in the past. Thus there is need for much additional research to better protect fish, wildlife, and humans. Many of the obvious contaminant effects when contaminants were high have been well studied, but during that time, little effort was focused on sublethal and less obvious effects of contaminants. While work is continuing in this area, there is still a need to better understand how contaminants affect fish biology (and in particular, for species actually exposed to that contaminant in the wild). Although fin erosion has largely disappeared, the cause of this disease has not been determined. Further research as to the cause of this disease may provide insight into why it occurred, and what in the sediments should be monitored to prevent its reoccurrence. Specific, realistic food webs focused on links between fish that are contaminated, and their fish, bird, and mammal predators are needed to provide a better understanding of how contaminants of concern reach threatened predatory species. A statewide assessment of the health of fish communities along the coast and in bays and estuaries is needed to determine areas of assemblage degradation.

Acknowledgments

I thank the following people for their contribution to figures used in this study: Valerie Raco-Rands, Erica T. Jarvis, Dario Diehl (all of the Southern California Coastal Water Research Project), Dr. James A. Noblet (California State University, San Bernardino, Department of Chemistry), David Montagne and Chi-Li Tang (County Sanitation Districts of Los Angeles County; CSDLAC), and Dr. Alan J. Mearns (National Oceanic and Atmospheric Administration, National Ocean Service, Office of Response and Restoration). I also thank Larry G. Allen (California State University, Northridge) and David Montagne (CSDLAC) for advice on this paper.

Literature Cited

Allen, L.G. 1985. A habitat analysis of the nearshore marine fishes from southern California. Bull. So. Calif. Acad. Sci. 84(3):133–155.

Allen, L.G., and M.P. Franklin. 1992. Abundance, distribution, and settlement of young-of-the-year white seabass, *Atractoscion nobilis*, in the Southern California Bight, 1988-89. U.S. Fish. Bull. 90:633–641.

Allen, M.J. 1977. Pollution-related alterations of demersal fish communities. AFS, Cal-Neva Wildl. Trans. 1977:103–107.

———. 1982. Functional structure of soft-bottom fish communities of the southern California shelf. Ph.D. dissertation. Univ. Calif., San Diego. La Jolla, Ca. 577 p.

Allen, M.J., D.W. Diehl, and E.Y. Zeng. 2004. Bioaccumulation of contaminants in recreational and forage fishes in Newport Bay, California in 2000–2002. SCCWRP, Westminster, CA, Tech. Rep. 436, 69 pp.

Allen, M.J., A.K. Groce, D. Diener, J. Brown, S.A. Steinert, G. Deets, J.A. Noblet, et al. 2002a. Southern California Bight 1998 Regional Monitoring Program: V. Demersal Fishes and Megabenthic Invertebrates. SCCWRP, Westminster, CA., Tech. Rep. 380, 572 p.

Allen, M.J., and K.T. Herbinson. 1991. Beam-trawl survey if bay and nearshore fishes of the soft-bottom habitat of Southern California in 1989. CalCOFI Rep. 32:112–127.

Allen, M.J., S.L. Moore, K.C. Schiff, S.B. Weisberg, D. Diener, J.K. Stull, A. Groce, J. Mubarak, C.L. Tang, and R. Gartman. 1998. Southern California Bight 1994 Pilot Project: V. Demersal fishes and megabenthic invertebrates. SCCWRP. Westminster, CA, Tech. Rep 308, 365 p.

Allen, M.J., S.L. Moore, S.B. Weisberg, A.K. Groce, and M. Leecaster. 2002b. Comparability of bioaccumulation within the sanddab feeding guild in coastal southern California. Mar. Poll. Bull. 44:452–458.

Allen, M.J., H. Pecorelli, and J. Word. 1976. Marine organisms around outfall pipes in Santa Monica Bay. J. Water Poll. Contr. Fed 48: 1881–1893.

Allen, M.J., R.W. Smith, and V. Raco-Rands. 2001. Development of biointegrity indices for marine demersal fish and megabenthic invertebrate assemblages of southern California. EPA grant X-989186-01-0. Prep. for U.S. EPA, Office of Science and Technology, Washington, DC. SCCWRP. Westminster, CA. 233 p.

Allen, M.J., P.V. Velez, D.W. Diehl, S.E. McFadden, and M. Kelsh. 1996. Demographic variability in seafood consumption rates among recreational anglers of Santa Monica Bay, California in 1991–1992. U.S. Fish. Bull. 94(4):597–610.

Anderson, D.W., and J.J. Hickey. 1970. Ocological data on egg and breeding characteristics of brown pelicans. Wilson Bull. 82:14–28.

Anderson, J.W., D.J. Reish, R.B. Spies, M.E. Brady, and E.W. Segelhorst. 1993. Human impacts. pp. 682–766 In M.D. Dailey, D.J. Reish, and J.W. Anderson (eds.), Ecology of the Southern California Bight: a synthesis and interpretation. University Press, California Berkeley, CA.

Bongersma-Sanders, M. 1957. Mass mortality in the sea. pp. 941–1010 in J.W. Hedgpeth (ed.), Treatise on marine ecology and paleoecology, Vol. 1, Ecol. Geol. Soc. Am. Mem. 67.

Brown, D.A., S.M. Bay, and R.W. Gossett. 1985. Using the natural detoxification capacities of marine organisms to assess assimilative capacity. pp. 364–382. In R.D. Cardwell, R. Purdy, and R.C. Bahner (eds.), Aquatic toxicology and hazard assessment: 7th Symposium. Am. Soc. Test. Mat., Philadelphia, PA. ASTM STP 845.

Brown, J.S., and S.A. Steinert. 2004. DNA damage and billiary PAH metabolites in flatfish from southern California bays and harbors, and the Channel Islands. Ecol. Indic. 3:263–274.

Cardwell, R.D. 1991. Methods for evaluating risks to aquatic life and human health from exposure to marine discharges of municipal wastewaters. pp. 252–253. In: A.G. Miskiewicz (editor), Proceedings of a bioaccumulation workshop: Assessment of the distribution, impacts, and bioaccumulation of contaminants in aquatic environments. Austr. Mar. Sci. Assoc., Inc./Water Board.

Connolly, J.P., and D. Glaser. 2002. p, p′—DDE bioaccumulation in female sea lions of the California Channel Islands. Cont. Shelf Res. 22:1059–1078.

Cooper, R.C., and C.A. Keller. 1969. Epizootiology of papillomas in English sole, *Parophrys vetulus*. Nat. Cancer Inst. Monogr. 31:173–185.

Couch, J.A., and D.R. Nimmo. 1974. Detection of interactions between natural pathogens and pollutants in aquatic animals. pp. 261–268. In: R.L. Amborski, M.A. Hood, and R.R. Miller (eds.), Proceedings of the Gulf Coast regional symposium on diseases of aquatic animals. Louisiana State University, Wetland Res. Publ. No. LSU-SG-74-05.

Cross, J.N. 1988. Fin erosion and epidermal tumors in demersal fish from southern California. pp. 57–64. In: D.A. Wolfe and T.P. O'Connor (eds.), Ocean processes in marine pollution, Vol. 5, Urban wastes in coastal marine environments. Krieger Publishing Company. Malabar, FL.

Cross, J.N., J.T. Hardy, J.E. Hose, G.P. Hershelman, L.D. Antrim, R.W. Gossett, and E.A. Crecelius. 1987. Contaminant concentrations and toxicity of sea-surface microlayer near Los Angeles, California. Mar. Environ. Res. 23:307–323.

Cross, J.N., and J.E. Hose. 1988. Evidence for impaired reproduction in white croaker (*Genyonemus lineatus*) from contaminated areas off southern California. Mar. Environ. Res. 24:185–188.

Cross, J.N., J. Roney, and G.S. Kleppel. 1985. Fish food habits along a pollution gradient. Calif. Fish Game 71:28–39.

CSDLAC (County Sanitation Districts of Los Angeles County). 2002. Palos Verdes Ocean Monitoring Annual Report, 2002, Chapter 4: Invertebrate and Fish Trawls. County Sanitation Districts of Los Angeles County. Whittier, CA.

Dailey, M.D., J.W. Anderson, D.J. Reish, and D.S. Gorsline. 1993. The Southern California Bight: background and setting. pp. 1–18. In: M.D. Dailey, D.J. Reish, and J.W. Anderson (eds.), Ecology of the Southern California Bight: a synthesis and interpretation. University of California Press, Berkeley, CA.

Dawe, C.J., J. Bagshaw, and C.M. Poore. 1979. Amebic pseudotumors in pseudobranch of Pacific cod, *Gadus macrocephalus*. Proc. Am. Cancer Res. 20:245.

Eganhouse, R.P., and M.I. Venkatesan. 1993. Chemical oceanography and geochemistry. pp. 71–189. In: M.D. Dailey, D.J. Reish, and J.W. Anderson (eds.), Ecology of the Southern California Bight: a synthesis and interpretation. University of California Press, Berkeley, CA.

Environment Canada. 1997. Canadian tissue residue guidelines for DDT for the protection of wildlife consumers of aquatic biota. Final unpubl. draft. Environment Canada, Science Policy and Environmental Quality Branch, Guidelines and Standards Division, Hull, Quebec, CN. 278 p. (85 p + append)

Environment Canada. 1998. Canadian tissue residue guidelines for polychlorinated biphenyls for the protection of wildlife consumers of aquatic biota. Environment Canada, Environmental Quality Branch, Guidelines and Standards Division, Hull, Quebec, CN. 303 p. (101 p, append.)

Fournie, J.W., J.K. Summers, and S.B. Weisberg. 1996. Prevalence of gross pathological abnormalities in estuarine fishes. Trans. Am. Fish. Soc. 125:581–590.

Gadbois, D.F., and R.S. Maney. 1983. Survey of polychlorinated biphenyls in selected finfish species from United States coastal waters. U.S. Fish. Bull. 81(2):389–396.

Gress, F. 1994. Reproductive performance, eggshell thinning, and organochlorines in brown pelicans and double-crested cormorants breeding in the Southern California Bight. Report 18 of Southern California Bight natural resource damage assessment expert reports. U.S. Dept. Justice, Environment and Natural Resources Division, Washington, DC.

Hogue, C.C., and J.M. Paris. 2002. Macroparasites of Pacific sanddab, *Citharichthys sordidus* (Bothidae) from polluted waters of the Palos Verdes Shelf, southern California. Bull. So. Calif. Acad. Sci. 101(1): 36–41.

Hose, J.E., J.N. Cross, S.G. Smith, and D. Diehl. 1987. Elevated circulating erythrocyte micronuclei in fishes from contaminated sites off southern California. Environ. Res. 22:167–176.

———. 1989. Reproductive impairment in a fish inhabiting a contaminated coastal environment off southern California. Environ. Poll. 57:139–148.

Irukama, K. 1966. The pollution of Minamata Bay and Minamata disease. pp. 153–180. In: Advances in water pollution research: Proceedings of the Third International Conference, 1966, Vol. 1.

Isaacs, J.D. 1972. Unstructured marine food webs and "pollutant analogues." U.S. Fish. Bull. 70:1053–1059.

Kalman, J. 2001. Parasites of demersal fishes as potential indicators of wastewater discharge in Santa Monica Bay. MS Thesis. Calif. St. Univ., Long Beach, Long Beach, CA. 98 pp.

Kennish, M.J. 1998. Pollution impacts on marine biotic communities. CRC Press, Boca Raton, FL. 310 p.

MacGregor, J.S. 1974. Changes in the amount and proportions of DDT and its metabolites, DDE and DDD, in the marine environment off southern California, 1949–1972. U.S. Fish. Bull. 72(2):275–293.

———. 1976. DDT and its metabolites in the sediments off southern California, USA. U.S. Fish. Bull. 74(1):27–35.

Malins, D.C., B.B. McCain, D.W. Brown, S.L. Chan, M.S. Myers, J.T. Landahl, P.G. Prohaska, A.J. Friedman, L.D. Rhodes, D.G. Burrows, W.D. Gronlund, and H.O. Hodgins. 1984. Chemical pollutants in sediments and diseases of bottom fishes in Puget Sound, Washington. Environ. Sci. Tech. 18:705–713.

Malins, D.C., B.B. McCain, D.W. Brown, M.S. Myers, M.M. Krahn, and S.L. Chan. 1987. Toxic chemicals including aromatic and chlorinated hydrocarbons and their derivatives, and liver lesions in white croaker (*Genyonemus lineatus*) from the vicinity of Los Angeles. Environ. Sci. Tech. 21(8):765–770.

McArn, G.E., and S.R. Wellings. 1971. A comparison of skin tumors in three species of flounders. J. Fish. Res. Bd. Canada 28:1241–1251.

Mearns, A.J. 1993. Contaminant trends in the Southern California Bight: the coast is cleaner. Pages 1002–1016 *in* Coastal Zone '93: Proceedings, 8th Symposium on Coastal Management. Am. Shore Beach Pres/Am Soc. Civil Eng., New Orleans, LA.

Mearns, A.J., M. Matta, G. Shigenaka, D. MacDonald, M. Buchman, H. Harris, J. Golas, and G. Lauenstein. 1991. Contaminant trends in the Southern California Bight. United States Department of Commerce, NOAA, Seattle, WA. NOAA Tech. Mem. NOS ORCA 62. 413 pp.

Mearns, A.J., M.B. Matta, D. Simecek-Beatty, M.C. Buchman, G. Shigenaka, and W.A. Wert. 1988. PCB and chlorinated pesticide contamination in U.S Fish and Shellfish: A historical assessment report. U. S. Dep. Commerce, NOAA, Seattle, WA. NOAA Tech. Mem. NOS OMA 39. 140 p.

Mearns, A.J., and M.J. Sherwood. 1977. Distribution of neoplasms and other diseases in marine fishes relative to the discharge of waste water. pp. 210–224. In: H.F. Kraybill, C.J. Dawe, J.C. Harshbarger, and R.G. Tardiff (eds.), Aquatic pollutants and biological effects with emphasis on neoplasia. *Ann. New York Acad. Sci.* 298.

Melzian, B.D., C. Zoffman, and R.B. Spies. 1987. Chlorinated hydrocarbons in lower continental slope fish collected near the Farallon Islands, California. Mar. Poll. Bull. 18(7):388–393.

Miller, B.S., and S.R. Wellings. 1971. Epizootiology of tumors in flathead sole (*Hippoglossoides elassodon*) in East Sound, Orcas Island, Washington. Trans. Am. Fish. Soc. 100:247–266.

Montagne, D. 2002a. Palos Verdes Ocean Monitoring Annual Report, 2002, chapter 3: Sediments and infauna. County Sanitation Districts of Los Angeles County. Whittier, CA.

Montagne, D. 2002b. Palos Verdes Ocean Monitoring Annual Report, 2002, chapter 4: Invertebrate and fish trawls. County Sanitation Districts of Los Angeles County. Whittier, CA.

Moore, S.L., and M.J. Allen. 2000. Distribution of anthropogenic and natural debris on the mainland shelf of the Southern California Bight. Mar. Poll. Bull. 40(1):83–88.

Moore, S.L., D. Gregorio, M. Carreons, S.B. Weisberg, and M.K. Leecaster. 2001. Composition and distribution of beach debris in Orange County, California. Mar. Poll. Bull. 42(3):241–245.

Muir, D.C.G., R.J. Norstrom, and M. Simon. 1990. Organochlorine in Arctic marine food chains: accumulation of specific polychlorinated biphenyls and chlordane-related compounds. Environ. Sci. Tech. 22:1071–1079.

Murchelano, R.A. 1975. The histopathology of fin rot disease in winter flounder from the New York Bight. J. Wildl. Diseases 11:263–268.

Myers, M.S., C.M. Stehr, O.P. Olson, L.L. Johnson, B.B. McCain, S.L. Chan, and U. Varanasi. 1994. Relationships between toxicopathic hepatic lesions and exposure to chemical contaminants in English sole (*Pleuronectes vetulus*), starry flounder (*Platichthys stellatus*), and white croaker (*Genyonemus lineatus*) from selected marine sites on the Pacific Coast, USA. Environ. Health Persp. 102(2):200–214.

Nakai, Z., M. Kosaka, S. Kudoh, A. Nagai, F. Hayashida, T. Kubota, M. Ogura, T. Mizushima, and I. Uotani. 1973. Summary report on marine biological studies of Suruga Bay accomplished by Tokai University 1964–72. J. Fac. Mar. Sci. Tech., Tokai, University 7:63–117.

Nigrelli, R.F., K.S. Ketchen, and G.D. Ruggieri. 1965. Studies on virus diseases of fishes: Epizootiology of epithelial tumors in the skin of flatfishes of the Pacific Coast, with special reference to the sand sole (*Psettichthys melanostictus*) from northern Hecate Strait, British Columbia, Canada. Zoologica (NY) 50:115–122.

OEHHA (Office of Environmental Health Hazard Assessment). 2001. California sport fish advisories. California Environmental Protection Agency, Office of Environmental Health Hazard Assessment, Sacramento, CA. 11 pp.

Pastor, D., J. Boix, V. Fernandez, and J. Albaiges. 1996. Bioaccumulation of organochlorinated contaminants in three estuarine fish species (*Mullus barbatus*, *Mugil cephalus*, and *Dicentrarcus labrax*). Mar. Poll. Bull. 32:257–262.

Perkins, E.J., J.R.S. Gilchrist, and O.J. Abbott. 1972. Incidence of epidermal lesions in fish of the North East Irish Sea area, 1971. Nature (London) 238:101–103.

Perkins, P.S., and R. Gartman. 1997. Host-parasite relationship of the copepod eye parasite, *Phrixocephalus cincinnatus,* and Pacific sanddab (*Citharichthys sordidus*) collected from sewage outfall areas. Bull. So. Calif. Acad. Sci. 96(3): 87–132.

Puffer, H.W., M.J. Duda., and S.P. Azen. 1982. Potential health hazards from consumption of fish caught in polluted coastal waters of Los Angeles County. No. Am. J. Fish. Man. 2:74–79.

Risebrough, R.W., F.C. Sibley, and M.N. Kirven. 1971. Reproductive failure in brown pelican of Anacapa Island in 1969. Am. Birds 25:8–9.

Roy, L.A., J.L. Armstrong, K. Sakamoto, S. Steinert, E. Perkins, D.P. Lomax, L.L. Johnson, and D. Schlenk. 2003. The relationship of biochemical endpoints to histopathology and population metrics in feral flatfish species collected near the municipal wastewater outfall of Orange County, California, USA. Environ. Tox. Chem. 22(6):1309–1317.

Russell, F.E., and P. Kotin. 1957. Squamous papilloma in the white croaker. J. Nat. Cancer Inst. 18(6):857–861.

Schiff, K.C. 2000. Sediment chemistry on the mainland shelf of the Southern California Bight. Mar. Pollut. Bull. 40:268–276.

Schiff, K.C., and M.J. Allen. 2000. Chlorinated hydrocarbons in flatfishes from the Southern California Bight. Environ. Tox. Chem. 19(6): 1559–1565.

Schiff, K.C., M.J. Allen, E.Y. Zeng, and S.M. Bay. 2000. Southern California. Mar. Poll. Bull. 41(1-6):76–93.

SFEI (San Francisco Estuary Institute). 2000. San Francisco Bay seafood consumption study. San Francisco Estuary Institute, Richmond, CA., 310 pp.

Sherwood, M. 1982. Fin erosion, liver condition, and trace contaminant exposure in fishes from three coastal regions. pp. 359–377. In: G.F. Mayer (ed.). Ecological Stress and the New York Bight: Science and Management. NOAA, Stony Brook, NY.

Sindermann, C.J. 1979. Pollution-associated diseases and abnormalities of fish and shellfish: a review. U.S. Fish Bull. 76(4):717–749.

Smith, R.W., M. Bergen, S.B. Weisberg, D. Cadien, A. Dalkey, D. Montagne, J.K. Stull, and R.G. Velarde. 2001. Benthic response index for assessing infaunal communities on the southern California mainland shelf. Ecol. Appl. 11(4):1073–1087.

Sommer, H., and F.N. Clark. 1946. Effect of red water on marine life in Santa Monica Bay, California. Calif. Fish Game 32(2):100–101.

Spies, R.B., J.S. Felton, and L. Dillard. 1982. Hepatic mixed function oxidases in California flatfishes are increased in contaminated environments by oil and PCB ingestion. Mar. Biol. 70(2):117–127.

Spies, R.B., and P. Thomas. 1995. Reproductive and endocrine status of female kelp bass from a contaminated site in the Southern California Bight and estrogen receptor binding of DDTs. pp. 113–133. In: R.M. Rolland, M. Gilbertson, and R.E. Peterson (eds.), Chemically induced alterations in functional development and reproduction of fishes. SETAC Press, Pensacola, FL.

Stein, J.E., T.K. Collier, W.L. Reichert, E. Casillas, T. Hom, and U. Varanasi. 1993. Bioindicators of contaminant exposure and sublethal effects in benthic fish from Puget Sound, WA, USA. Mar. Environ. Res. 35:95–100.

Stull, J. 1995. Two decades of marine biological monitoring, Palos Verdes, California, 1972 to 1992. Bull. So. Calif. Acad. Sci. 94: 21–45.

Stull, J.K., K.A. Dreyden, and P.A. Gregory. 1987. A historical review of fisheries statistics and environmental and societal influences off the Palos Verdes Peninsula, California. Cal COFI Rep. 28:135–154.

Stull, J.K., and C.-L. Tang. 1996. Demersal fish trawls off Palos Verdes, southern California, 1973–1993. Cal COFI Rep. 37:211–240.

Ui, J. 1969. Minamata disease and water pollution by industrial waste. Rev. Intern. Oceanogr. Med. 13/14:5–35.

USCB (United States Census Bureau). 2001. Population change and distribution, 1999 to 2000. U. S. Dep. of Commerce, Census Bureau, Washington, DC. 7 pp.

USEPA (United States Environmental Protection Agency). 1995. Guidance for assessing chemical contaminant data for use in fish advisories. Volume 1-Fish Sampling and Analysis. 2nd edition. U. S. Environmental Protection Agency, Office of Water, Washington, DC. EPA 823-R-95-007.

Valentine, D.W. 1975. Skeletal anomalies in marine teleosts. pp. 695–718. In: W.E. Ribeline and G. Migaki (eds.), The pathology of fishes. University Wisconsin Press, Madison, WI.

Wellings, S.R., C.E. Alpers, B.B. McCain, and B.S. Miller. 1976. Fin erosion disease of starry flounder (*Platichthys stellatus*) and English sole (*Parophrys vetulus*) in the estuary of the Duwamish River, Seattle, Washington. J. Fish. Res. Bd. Canada 33:2577–2586.

Young, D.R., A.J. Mearns, T.K. Jan, T.C. Heesen, M.D. Moore, R.P. Eganhouse, G.P. Hershelman, and R.W. Gossett. 1980. Trophic structure and pollutant concentrations in marine ecosystems of southern California. CalCOFI 21:197–206.

Young, P.H. 1964. Some effects of sewer effluent on marine life. Calif. Fish Game 50(1):33–41.

Zeng, E.Y., and M.I. Venkatesan. 1999. Dispersion of sediment DDTs in the coastal ocean off southern California. Sci. Total Environ. 229: 195–208.