3.5 IMPACT OF LEGACY PESTICIDE BIOACCUMULATION ON FISHING AND AQUATIC LIFE IN CALIFORNIA

3.5.1. Introduction

In regions of historic contamination, legacy pesticides (LPs) have influenced both aquatic food webs and human consumers. LPs, used for agriculture, pest control, and mosquito abatement, continue to enter the water and sediment of water bodies decades after their uses were banned (Table 3.5.1). Current inputs to water bodies include runoff from local watersheds, discharge of municipal and industrial wastewater, atmospheric deposition, erosion of historically contaminated sediment deposits, and dredging and disposal of dredged material. Known as legacy pesticides, LPs include: DDTs - the o,p'- and p,p'-isomers of dichlorodiphenyltrichloroethane (DDT), and its breakdown products, dichlorodiphenyldichloroethylene (DDE) and dichlorodiphenyldichloroethane (DDD); chlordanes - alpha-chlordane, gamma-chlordane, cis-nonachlor, trans-nonachlor, oxychlordane, heptachlor, and heptachlor epoxide; and dieldrin.

Table 3.5.1.
Use of legacy pesticides.

Pesticide	Start of Use	End of Use	Major Uses
DDT	1939	1972	Broad spectrum insecticide used on agricultural crops, for pestcontrol, and for mosquito abatement.
Chlordane	1948	1988	Originally used on agricultural crops, lawns, gardens, and as afumigating agent. Most uses banned in 1978, and after 1983, only used for termite control.
Dieldrin	1948	1987	Originally used on agricultural crops. After 1974, only used for termite control.

DDT was used in home and agricultural applications and for mosquito abatement beginning in the 1940s. A reduction of use in California began in 1963 due to ecological concerns and the potential health effects (e.g., Mischke et al. 1985). By 1972, the U.S. had banned DDT for all but emergency public health uses. However, California did not require reporting of DDT use prior to 1971, so there are no records of application rates. Nationally, more than 500 million kg were sold over a thirty-year period. Its presence as a manufacturing by-product in other pesticides was restricted to 0.1% in 1988.

Beginning in the late 1940s, chlordane was used in home and agricultural applications to control termites and other insect pests. Chlordane use was restricted in California in 1975 and throughout the U.S. in 1978. Production and sales ended in 1988.

In 1950, dieldrin began to be used for a variety of applications, including control of termites and other soil-dwelling insects, as a wood preservative, for moth-proofing clothing and carpets, and as a pesticide on cotton, corn, and citrus crops. Dieldrin was restricted in 1974, and most uses were banned in 1985. Use for underground termite control continued until 1987.

LPs remain environmental contaminants of concern due to their persistence in the environment and tendency to accumulate at the top of aquatic food webs. Resistance to abiotic and biotic chemical transformations has resulted in the continued presence of LPs in aquatic systems decades after their respective bans. Microbial degradation of pesticides is generally very slow, although it can be significant for certain isomers when concentrations are high. Furthermore, once such contaminants are buried in sediments they essentially do not degrade. Although some forms of these pesticides are metabolized by higher organisms, many pass largely un-metabolized through the food web. Furthermore, due to their resistance to metabolism and high affinity for lipid, LPs reach higher concentrations with increasing trophic levels; a process known as "biomagnification" (Gobas et al. 1993, Suedel et al. 1994). The overall significance is that LPs are neurotoxins and classified by USEPA as probable human carcinogens. Predatory fish, birds, and mammals (including humans that consume fish) at the top of the food web are particularly vulnerable to the toxic effects of this contamination.

Sections 3.5.2, 3.5.5, and 3.5.8 below and all the maps in this chapter are geared exclusively toward impact on fishing, with concentration categories related to human consumption of sport fish and human health concerns. Sections 3.5.3. 3.5.6, and 3.5.9 address how LPs may be affecting aquatic life in California, but sufficient data for aquatic life indicators were not available to create the same detailed maps. Maps geared toward impacts on aquatic life would have different species represented (e.g., small fish, such as Mississippi silversides, or bird eggs) and would apply different thresholds.

3.5.2. Impact of DDTs on Fishing in California

a. Current Status

Consumption Advisories

Fish consumption advisories issued by OEHHA are a key indicator of the risk that pollutant contamination poses to human health (http://www.oehha.ca.gov/fish/so_cal/index.html). However, as of May 2007, consumption advisories due at least partially to DDT were in place for only three regions: 1) San Francisco Bay and Sacramento-San Joaquin Delta, 2) Machado Lake (formally Harbor Park Lake) in Los Angeles County, and 3) coastal locations in southern California between Point Dume and Dana Point (Table 3.2.1). There is general agreement between the current fish consumption advisories and the 2002 303(d) listed water bodies. One exception is Region 5, where there are insufficient data to determine the possible threat that DDT may pose to human health in water bodies of the Central Valley other than those listed. A SWAMP project to address the lack of fish tissue organochlorine pesticide and PCB data to suport 303(d) listings and fish consumption advisories is currently underway for this Region (Robert Holmes, CVRWQCB, personal communication). DDT concentrations in California have been declining gradually since their peak in the 1960s. However, current fish consumption advisories in the state have all been issued since 1991. This most likely reflects an improved understanding of the behavior of DDT concentrations in sport fish, which has brought greater attention from water quality and human health managers. The low number of current advisories across the state suggests that DDT (in areas other than those listed) does not currently pose a significant threat to human health.

303(d) Listings

Section 303(d) of the Clean Water Act requires California to compile a list of impaired water bodies that do not meet water quality standards (the "303(d) List"). The 2002 303(d) List for California indicates that DDTs are a major contributor to impacts on water quality in the state. The 2002 303(d) List includes DDT listings (Appendix 3) for the following notable areas:

- Region 2 – San Francisco Bay (172,683 acres) and Sacramento-San Joaquin Delta (41,736 acres);
- Region 4 – Coastal water bodies in the Los Angeles area (many miles and acres, most notably Santa Monica Bay [146,645 acres]), and lakes (564 acres); and
- Region 5 – 127 miles of the San Joaquin River and Delta waterways (21,087 acres).

The majority of water bodies on the 303(d) List are bays and estuaries in highly urbanized regions. San Francisco Bay is listed as impaired by DDT pursuant to §303(d) of the U.S. Clean Water Act because of an interim fish consumption advisory developed by the California Office of Environmental Health Hazard Assessment (OEHHA) in 1994. The advisory was issued as a result of a 1994 pilot study in the Bay (SFRWQCB 1995), which indicated that legacy pesticides, as well as polychlorinated biphenyls (PCBs), mercury, and dioxins, were present at concentrations of potential concern for human health. This interim advisory for San Francisco Bay remains in place for DDT.

Recent Monitoring Data

Recent sport fish monitoring data (1998 – 2003) indicate that DDT concentrations in the vast majority (248 of 252, 98%) of the state were in the green < 800 ppb category (Table 3.5.2 and Figure 3.5.1). Four sites (2%) were in the yellow 800 – 3500 ppb category, and none were in the orange or red categories. Three of the four yellow sites were located in the Imperial Valley (Salton Sea) region of southern California. No current fish consumption advisories due to DDT exist for this region of the state. However, since the majority of recent sport fish DDT data indicate that locations around the state are in the green category, the limited number of fish advisories and 303 d listings for DDT appear appropriate at this time.

b. Long-term Trends in Impact of DDTs on Fishing in California

Management Actions

The primary use of DDT was in agriculture. During the 30 years prior to its cancellation, a total of approximately 1.35 billion pounds was used in the United States (USEPA 1975). The most significant management action to reduce DDT in California and the rest of the country was the federal restriction on DDT that began in 1972. The EPA's cancellation of DDT was in response to fears that continued use posed risks to both the environment and human health. The expectation is that DDT is currently declining gradually across the state, presumably in large part due to the ban. However, DDT is generally considered to be extremely persistent in the environment, exhibiting long residence times in polluted watersheds.

Since the ban, contaminant-research activities have responded to concerns over the degree of historic DDT contamination in California. In 1984, the California Assembly directed the Department of Food and Agriculture to investigate possible DDT sources (Mischke et al. 1985) due to suggestions that California rivers were significantly contaminated. The statewide survey investigated DDT concentrations in soil from agricultural areas. DDT residues were found wherever it was used historically. All 99 samples analyzed from 32 counties contained measurable DDT. The report concluded that residues from historic agricultural applications of DDT appeared to be the source of continuing contamination in California rivers.

Table 3.5.2. Total number of locations sampled for legacy pesticides and percentage in each concentration category for three different time intervals from 1978 to 2003.

Pollutant	Time Interval	Total Number of Locations Sampled	Low	Moderate	High	Very High
DDTs	Recent (1998 – 2003)	252	98%	2%	0%	0%
DDTs	1988 – 1997	241	94%	4%	2%	0%
DDTs	1978 – 1987	162	86%	11%	2%	1%
Dieldrin	Recent (1998 – 2003)	244	98%	2%	0%	0%
Dieldrin	1988 – 1997	237	97%	3%	0%	0%
Dieldrin	1978 – 1987	155	91%	7%	1%	1%
Chlordanes	Recent (1998 – 2003)	238	100%	0%	0%	0%
Chlordanes	1988 – 1997	237	99%	1%	0%	0%
Chlordanes	1978 – 1987	151	98%	1%	1%	0%

Local management actions to clean up historic DDT contamination have been ineffective in some areas. Lauritzen Canal, a portion of San Francisco Bay near Richmond, California, was heavily contaminated with DDT, as a result of releases from a pesticide-formulating plant. An EPA ecological risk assessment in 1991 and 1992 documented sediment contamination by DDT of up to 77,700 ppm (Lee et al. 1994). In response, EPA negotiated with the responsible parties to conduct remedial dredging of contaminated sediment. Despite the removal of 102,000 metric tons, representing 3 tons of DDT (based on average concentration in sediment), DDT concentrations in the food web were not reduced (Weston et al. 2002). In fact, sediment disturbance

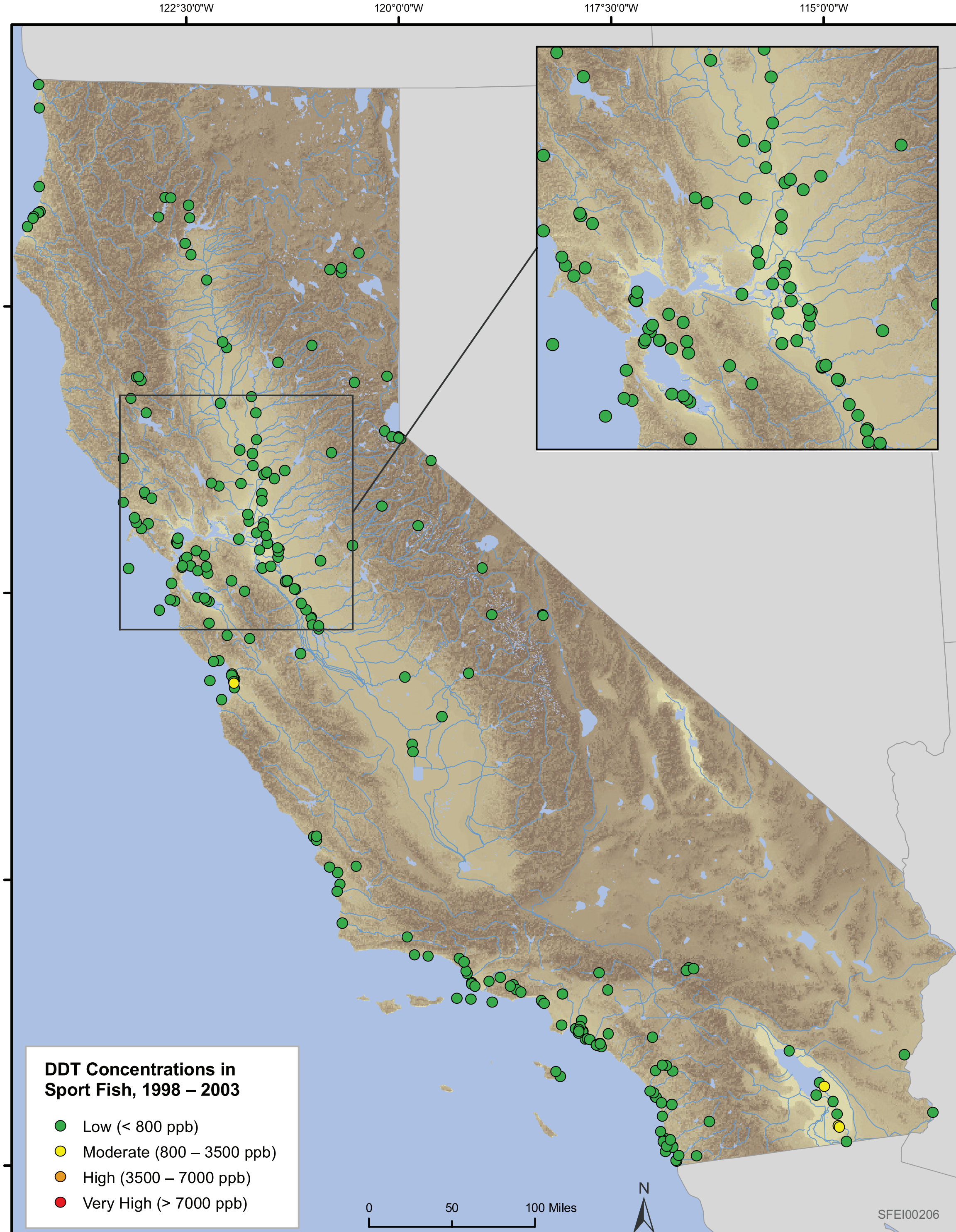

Figure 3.5.1. DDT concentrations in California sport fish, 1998 – 2003. Based on DDT measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

during dredging introduced a pulse of DDT into Lauritzen Canal that resulted in an increase in body burdens to fish and invertebrates of 2- to 76-fold, depending on the species. Approximately 18 months after remediation, 11 of 14 indicators showed contamination comparable with or worse than the contamination that preceded dredging. The Lauritzen Canal study demonstrates that despite effective methods to identify DDT contamination, successful cleanup can be challenging.

Similar management actions have been initiated in other areas in California with heavily contaminated sediments, with results yet to be determined. In Los Angeles County, DDT impacts have been reported since the late-1980s (Cross and Hose 1988). EPA began a pilot study in 2000 to investigate the feasibility of remediation of the historically contaminated sediments around the Superfund site in Los Angeles County (Montrose Chemical Company).

Long-term Trends

Sport Fish

Concentrations of DDT in aquatic food webs across the state have generally shown declines over the past 30 years in response to the use restrictions and federal ban. Concentrations over time (Figures 3.5.2, 3.5.3) indicate that sport fish DDT levels were higher prior to 1998 than currently (Figure 3.5.1). Fifteen (6%) locations monitored from 1988 – 1997 and 21 (13%) from 1978 – 1987 had concentrations in the yellow 800 – 3500 ppb and orange 3500 – 7000 ppb categories (Table 3.5.1). One location in Oxnard (1988 – 1997) was also in the red > 7000 ppb category. This is compared to recent data (1998 – 2003) showing only 2% of sites above 800 ppb. These data suggest the presence of a historical DDT hotspot in southern California (numerous yellow and one orange site in Figures 3.5.2, 3.5.3). The map of recent data (Figure 3.5.1) suggests that the Imperial Valley (Salton Sea) region of southern California consists of three locations with yellow concentrations. In previous time intervals, this region consisted of seven yellow or orange sites from 1988 – 1997 and six from 1978 – 1987. The number of yellow sites in the recent dataset suggests that DDT contamination continues to persist in the Imperial Valley (Salton Sea) region, but the severity of impact has decreased over time. Other regions of the state also appear to have similar or improved concentrations of DDT.

Long-term trend monitoring has provided further evidence for the recovery of California water bodies from DDT contamination. Compared to other contaminants (especially mercury), legacy pesticides have been monitored in the same species at a given location over long time periods (e.g., Toxic Substances Monitoring Program and Sacramento River Watershed Program), allowing for temporal trend comparison. Of ten locations examined for sport fish (Figure 3.5.4), four indicate a significant decline ($p < 0.05$) between years. The declining trends appear consistent across the state, as the four significantly declining locations are distributed widely. However, relatively high concentrations of DDT in channel catfish in New River at Westmorland and Alamo River at Calipatria indicate that DDT is still prevalent in the Imperial Valley (Salton Sea) region. These locations have demonstrated continuously high DDT concentrations in sport fish throughout the 26 years of monitoring when other sites have generally declined. Therefore, these data suggest that the ban on DDT to alleviate the historic contamination of this region has not been as effective as it has in other regions of the state. Stanislaus River is of note, as concentrations during the 1980s appeared to be increasing at this loca-

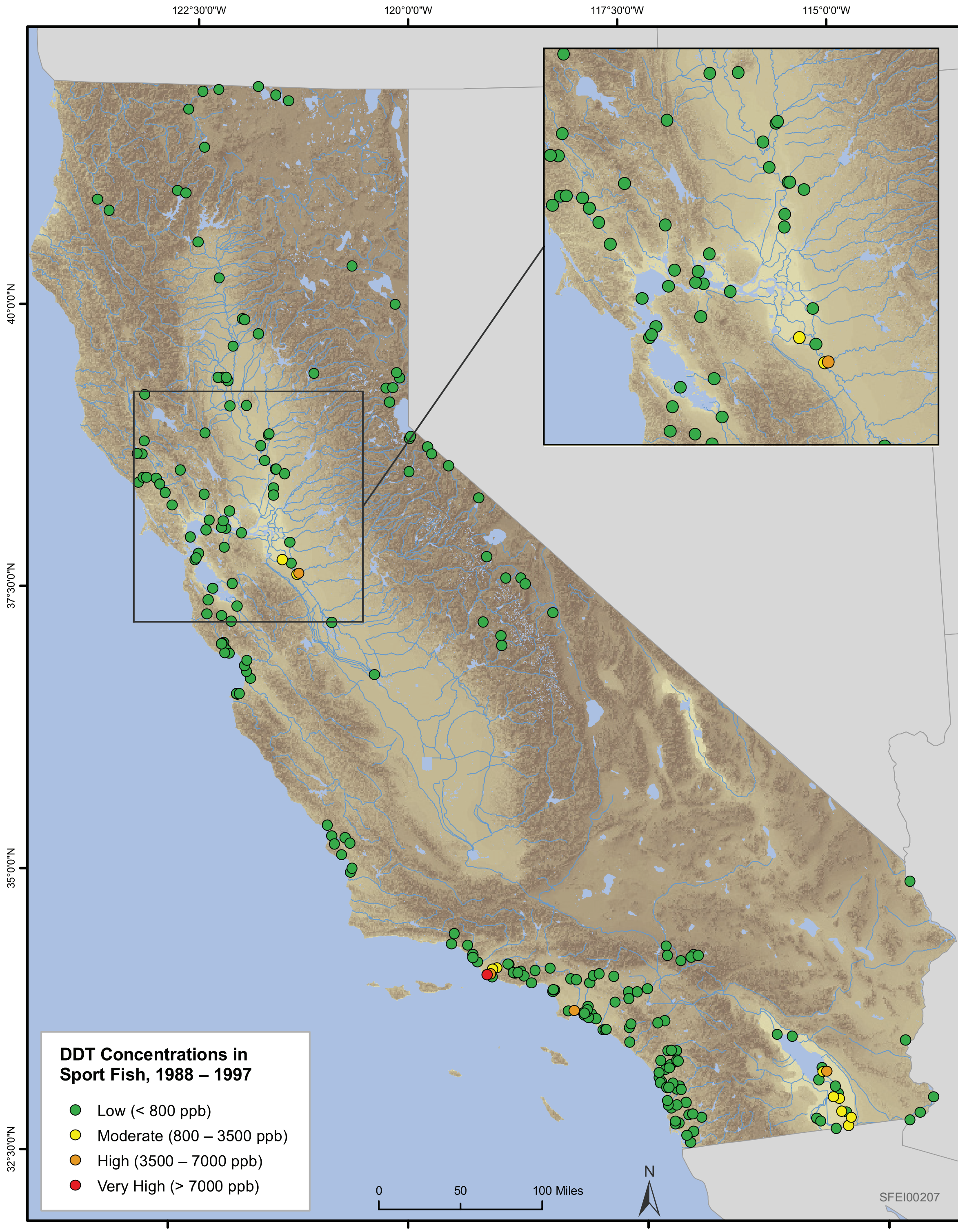

Figure 3.5.2. DDT concentrations in California sport fish, 1988 – 1997. Based on DDT measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

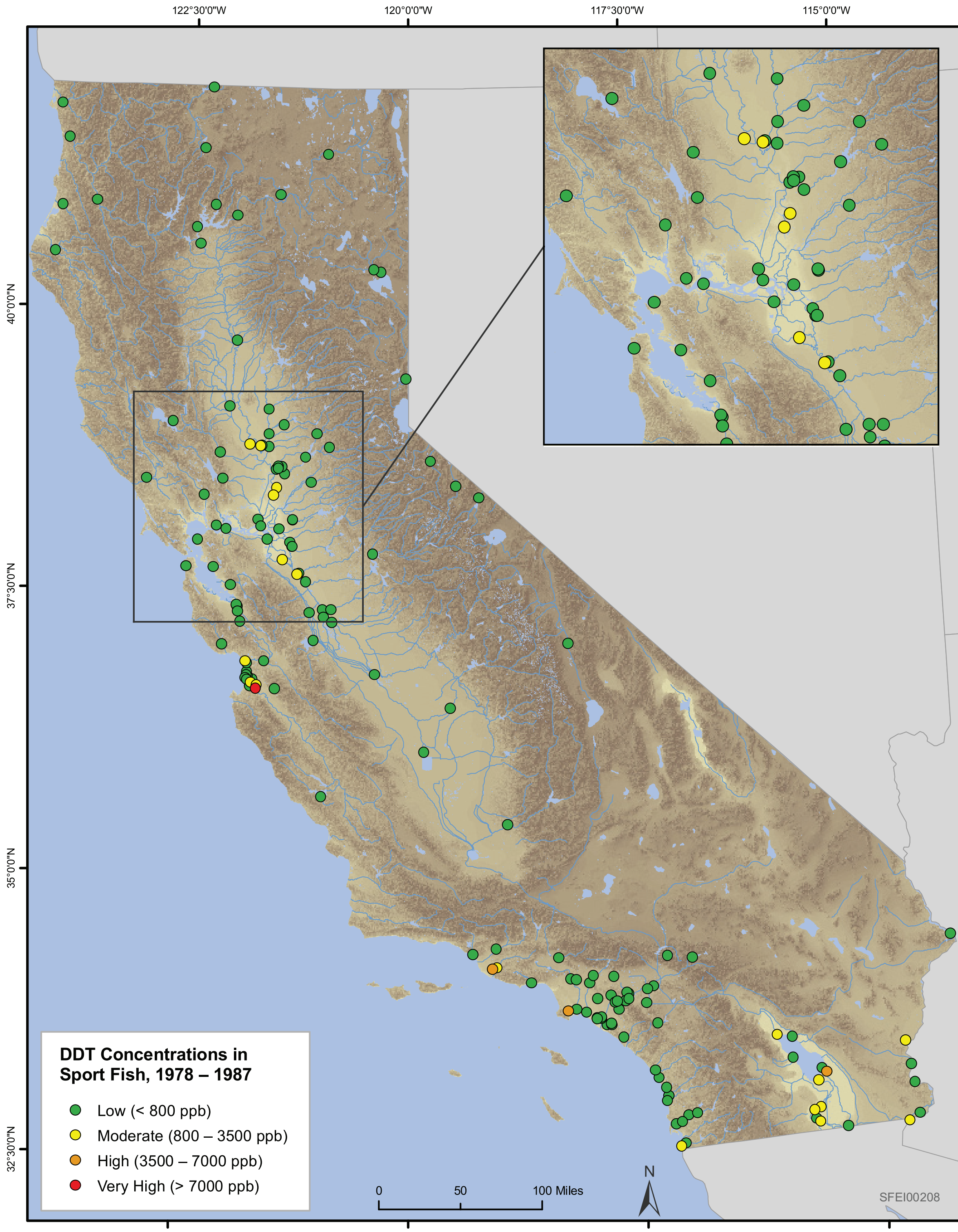

Figure 3.5.3. DDT concentrations in California sport fish, 1978 – 1987. Based on DDT measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

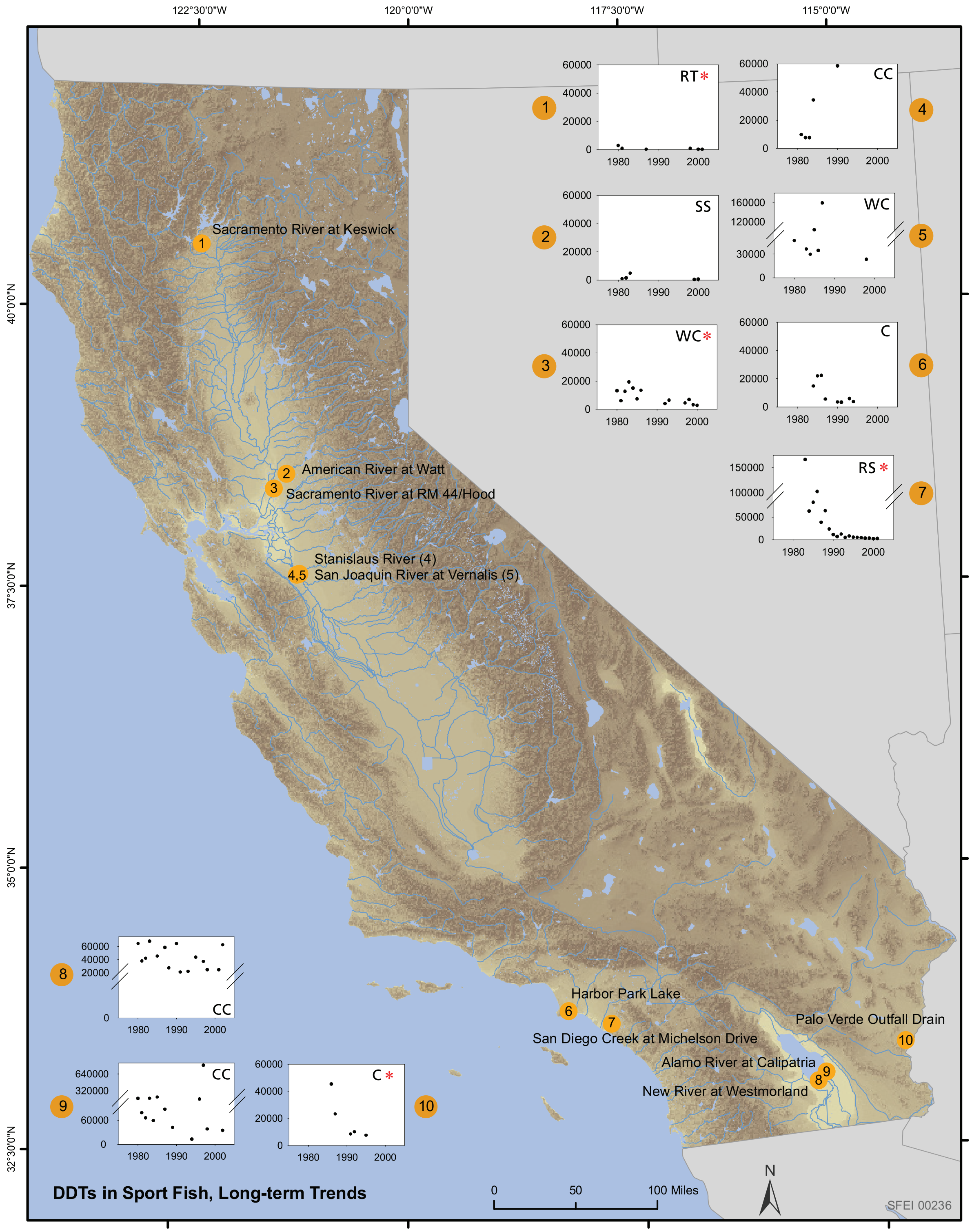

Figure 3.5.4. Long-term trends in DDT concentrations in California sport fish. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight. Species shown are rainbow trout (RT), Sacramento sucker (SS), white catfish (WC), channel catfish (CC), red shiner (RS), and common carp (C).

tion. However, the recent trend is unknown, as the site has not been monitored for channel catfish since then.

Bivalves

DDT monitoring of bivalves by the State Mussel Watch Program supports the overall trends of declining DDT concentrations in sport fish. Bivalves are important indicators for legacy pesticides, as they measure temporal and spatial variation in concentrations that are bioavailable to other organisms. Long-term datasets for mussels were available from 10 locations. All sites except for Sandholdt Bridge indicate a significant decline ($p < 0.05$) over time (Figure 3.5.5). At Sandholdt Bridge concentrations appear to have generally increased with time, though the most recent data are lower. Furthermore, the long-term trend is masked by considerable inter-annual variation. Some locations (e.g., Royal Palms) show a dramatic decline in DDT concentrations because they were historic DDT discharge points (Stephenson et al. 1995). The long-term monitoring results are consistent with previous interpretations of State Mussel Watch data (Stephenson et al. 1995) that have documented significant declines in DDT. These lines of evidence suggest that the sources of DDT to San Francisco Bay, the central coast, and southern California, have likely declined since the 1980s, which has resulted in a lessening of DDT bioaccumulation in bivalves.

Case Studies

Southern California Bight

Monitoring studies in southern California have provided considerable insight into the trends in fish contamination on a local scale. DDT contamination in southern California waters has been monitored extensively in sport fish as far back as the late 1960s and early 1970s. The most comprehensive studies were conducted by the Southern California Coastal Water Research Project (SCCWRP) from 1973 to 1981 (summarized by Mearns et al. 1991). Historical seafood contamination data was collected in 22 species of fish. Although some species were well sampled, the only species that was examined from the 1970s into the 1990s was white croaker. The average DDT concentration in white croaker (100.8 ppm wet wt) collected from 1980 – 1981 was the highest of all species sampled. By 1990, DDT concentrations had decreased in many species. Average concentrations in white croaker were significantly lower (18.3 ppm wet wt), with only locations near the Palos Verdes Shelf still showing relatively high concentrations (Figure 3.5.6). Historically deposited contaminated sediments on the Palos Verdes Shelf has been the primary source of DDT to waters near Los Angeles (Mearns et al. 1991), and is likely to be the primary source of the higher concentrations in white croaker (Allen and Cross 1992). Currently, white croaker consumption is still prohibited in waters of this region (e.g., White's Point – Table 3.2.1).

Relatively high concentrations of DDT in southern California have been linked to reproductive impacts to local fish populations. Declines in sport fish and commercial catches in southern California during the 1970s and 1980s were attributed to elevated contaminant burdens that may have reduced populations (Stull et al. 1987, Karpov et al. 1995). Cross and Hose (1988) showed that white croaker from San Pedro Bay in 1985 – 1986 had higher DDT concentrations compared to a southern California reference site (Dana Point). The

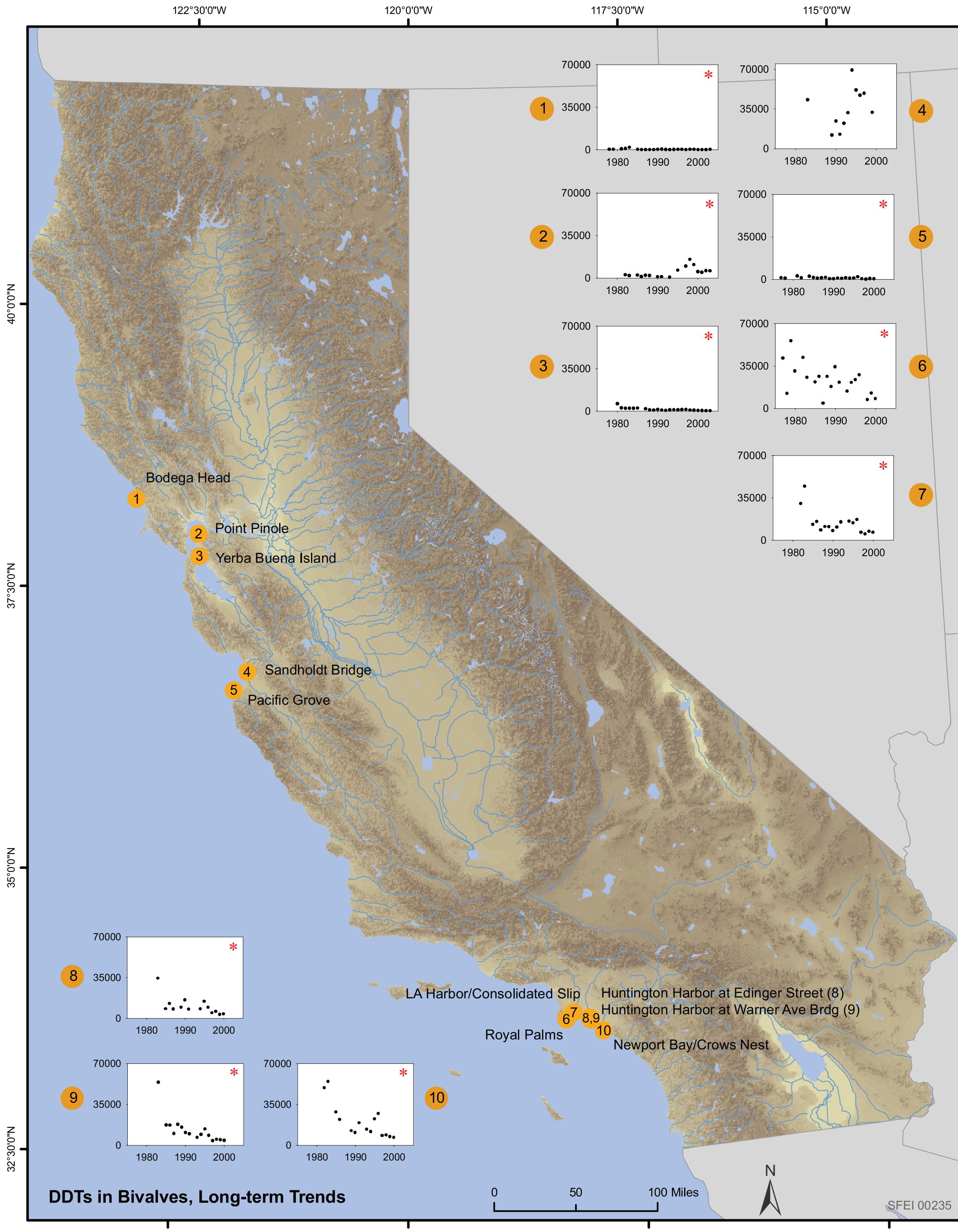

Figure 3.5.5. Long-term trends in DDT concentrations in California mussels measured by the State Mussel Watch Program. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight.

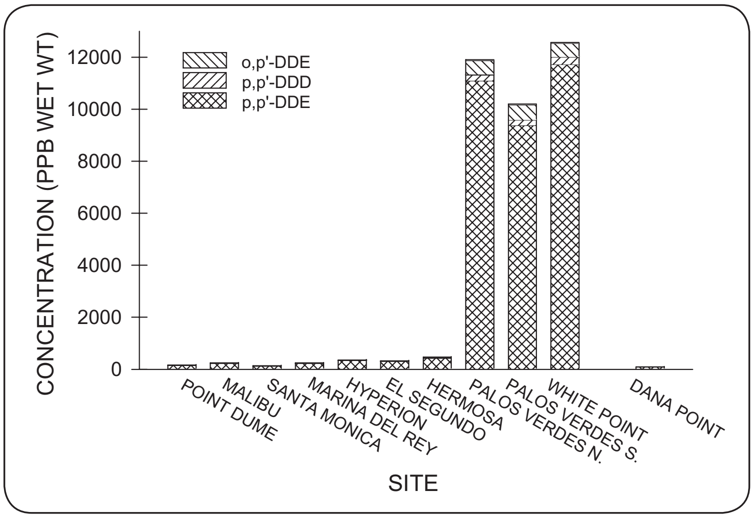

Figure 3.5.6. Mean concentrations of Total DDT in composites of white croaker collected from coastal Southern California in September 1990. From Allen & Cross (1992).

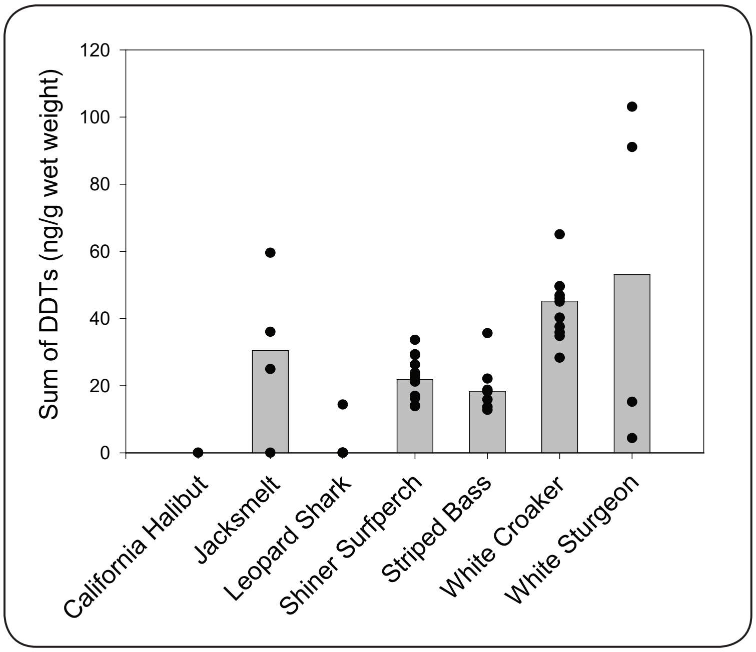

Figure 3.5.7. DDT concentrations (ppb) in San Francisco Bay fish, 2003. Points are concentrations in each composite sample analyzed. Bars indicate median concentrations. From RMP (2006).

elevated body burdens of DDT correlated well with a reduction in reproductive success. However, the reproductive effects could not be separated from the other organic contaminants (e.g., PCBs) present in San Pedro Bay. Other regions of southern California (e.g., Southern California Bight) have exhibited DDT concentrations in sediments in recent years (1995 – 1996, and 2000) that have shown the potential for impacts to local fish species (Allen et al. 2002b, Zeng and Tran 2002).

San Francisco Bay

San Francisco Bay has also had many contamination issues related to DDT use. Organic contaminants in Bay sport fish have been monitored since 1994, when the Bay Protection and Toxic Cleanup Program (BPTCP) conducted a pilot study to measure concentrations of contaminants in fish from the Bay (SFRWQCB 1995, Fairey et al. 1997). As a follow-up to the BPTCP, the San Francisco Estuary Institute's Regional Monitoring Program (RMP) began to monitor contaminants in sport fish from the Bay every three years (1997, 2000, 2003, and 2006). The RMP focuses on seven of the most popular sport fish species taken from the Bay and consumed by anglers (SFEI 2000). In 1997, 2000, and 2003, sport fish DDT samples did not exceed the human health screening level. However, fattier (higher lipid) species (e.g., white croaker and shiner surfperch) had higher concentrations (Figure 3.5.7). San Francisco Bay species have also indicated declines in DDT over time. Risebrough (1997) reported declines in shiner surfperch from 1000 – 1400 ppb (wet wt) in 1965 to 14 – 73 ppb in 1994. Furthermore, statistical analysis of RMP data (lipid normalized) detected significant DDT declines in leopard shark, striped bass, and white croaker concentrations from 1994 – 2003 (Connor et al. 2007, in press).

Monitoring of bivalves has generated the best evidence for long-term trends in DDT in San Francisco Bay. The State Mussel Watch (SMW) Program conducted bivalve monitoring in the Bay from 1980 to 1993. Subsequently, RMP continued this monitoring from 1993 to present. p, p'-DDE is commonly found in highest abundance compared to other DDT isomers. In the Central Bay, concentrations of p,p'-DDE were over 2000 ppb (lipid wt) in 1980, but declined throughout the 1980s (Gunther et al. 1999). Since 1988, concentrations have remained relatively constant.

Monitoring results have also indicated that DDT hotspots remain within the Bay. Fish tissue concentrations of DDT monitored in Lauritzen Canal from 1996 – 1998 were elevated compared to other San Francisco Bay locations. Shiner surfperch collected in Lauritzen Canal (pre- and post-dredging) contained DDT levels of 140 ppm (lipid wt) compared to an average concentration of 2.2 ppm (lipid wt) in the Bay as a whole. Results for bivalve species were more variable. The resident bivalve, Mytilus galloprovincialis, had DDT concentrations that declined from 280 ppm before dredging, to 94 ppm at 23 months after dredging. However, two years later, DDT concentrations had risen once again to 130 ppm. The levels of DDT reported in Lauritzen Canal suggest that two years after remediation, concentrations had yet to stabilize.

c. Sources and Pathways

The distribution of DDT concentrations across the state indicates the possible sources and pathways of DDT to California water bodies. Concentrations in sport fish from recent data (Figure 3.5.8) suggest that three sites (in Imperial Valley) had higher concentrations relative to the rest of the state. The historic applications of DDT to agricultural fields, as well as in agricultural drainage that feeds the Salton Sea are the most likely sources to this region. Large amounts of DDT-contaminated sediment likely remain in these watersheds, providing current and future sources of polluted sediment downstream. The other location exhibiting DDT concentration above 800 ppb was Old Salinas River, which is also a region of high agriculture practices. In the state as a whole, however, there do not currently appear to be large sources of DDT into aquatic ecosystems that have resulted in elevated concentrations among sport fish collected from 1998 – 2003.

Historic data indicate that the sources of DDTs were more widespread previously (Figure 3.5.9, 3.5.10). Elevated concentrations of DDT are indicated for agricultural (Imperial Valley, Oxnard, and Sacramento-San Joaquin Delta) and urban (San Francisco and Los Angeles) areas. The highest DDT concentration in each historic time interval was 19,270 ppb at Oxnard Drainage Ditch (1988 – 1997) and 17,188 ppb at Blanco Drain (1978 – 1987). The sources for historic DDT contamination at these hotspots are likely to be agricultural. In general, DDT sources had the largest impact (abundance of tall concentration bars) on sport fish in 1978 – 1987, compared to other time intervals. These results indicate a reduction of DDT in California waters, likely as a result of successful DDT management actions. DDT concentrations in aquatic food webs are not significantly elevated at present, and are greatly reduced from levels measured previously.

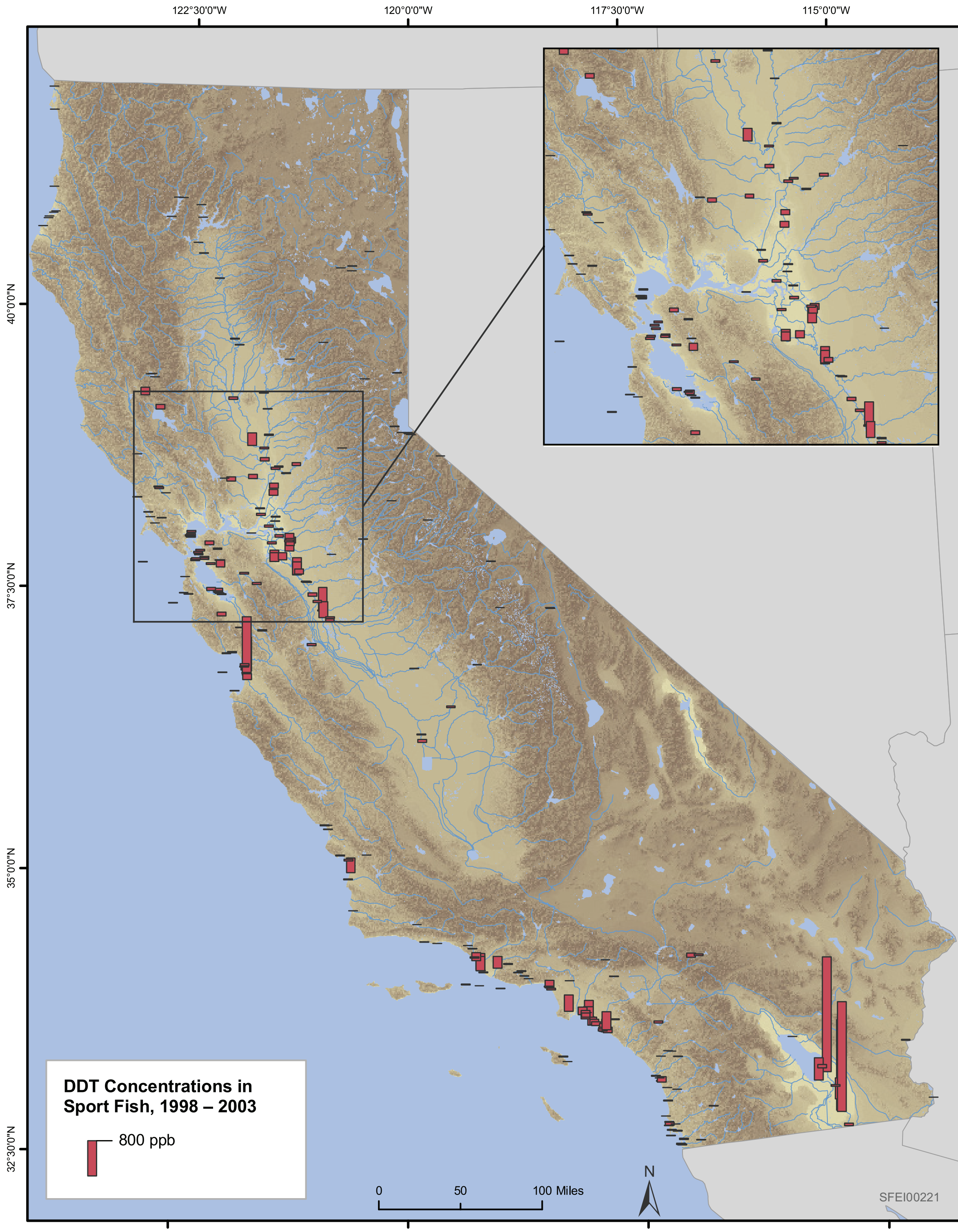

Figure 3.5.8. DDT concentrations in California sport fish, 1998 – 2003. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

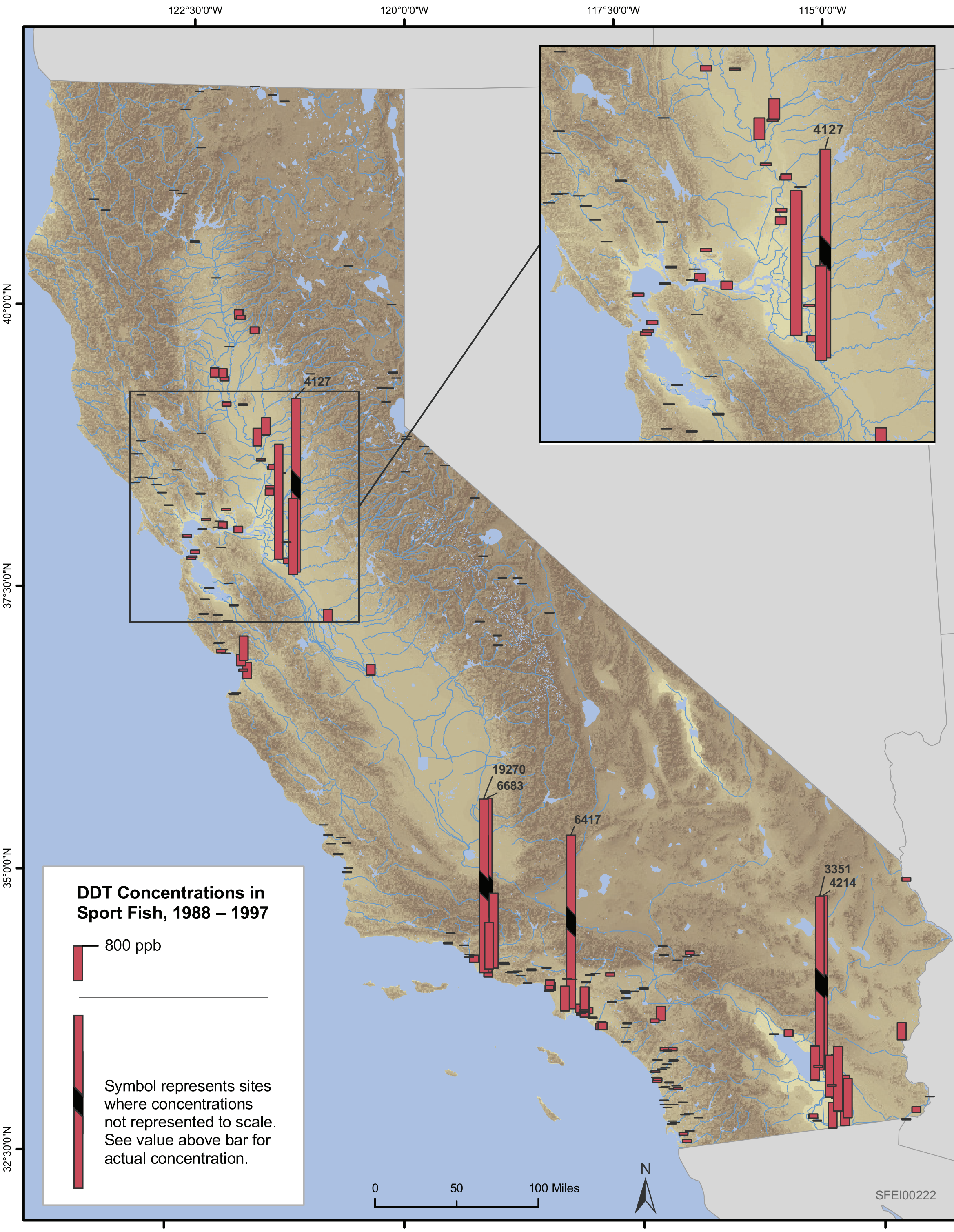

Figure 3.5.9. DDT concentrations in California sport fish, 1988 – 1997. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

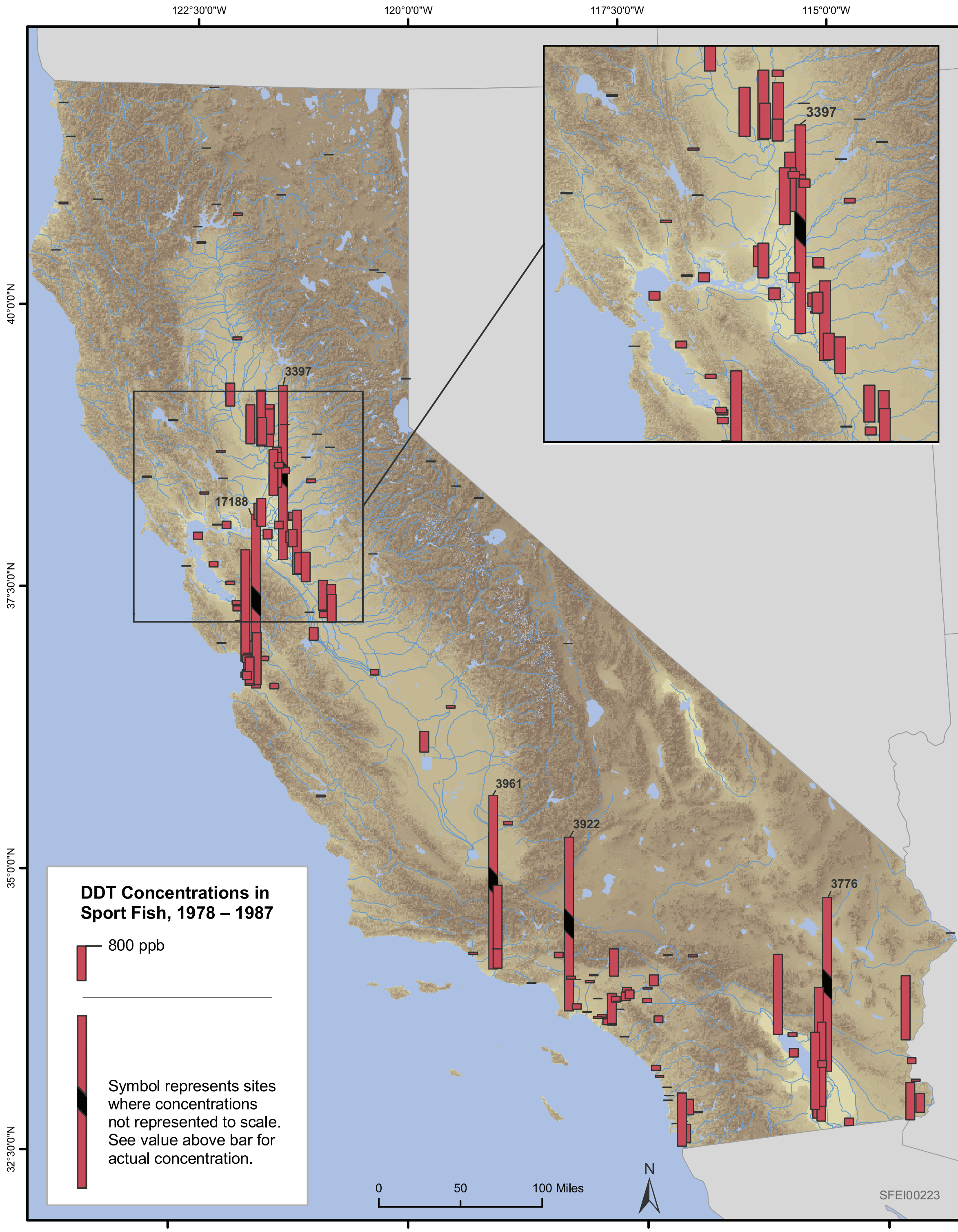

Figure 3.5.10. DDT concentrations in California sport fish, 1978 – 1987. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

3.5.3. Impact of DDTs on Aquatic Life in California

Historically high DDT concentrations have had a dramatic effect on aquatic life in California. With the steadily declining concentrations that have been documented over the past 20 years, impacts are now believed to be less significant. San Francisco Bay and southern California have provided classic examples of the recovery of wildlife populations. Therefore, a review of evidence for impacts to aquatic life in these ecosystems is presented below.

a. Statewide Assessment

Impacts to aquatic life due to pollutant bioaccumulation are often assessed using exposure to piscivorous wildlife. The bioaccumulation of toxic contaminants in prey organisms can pose a significant risk to predators, particularly those that reside at the top of the food web, such as birds and mammals. Dietary accumulation from aquatic prey can lead to potentially high levels of contaminants in predator species. In piscivorous birds, concentrations of pollutants in eggs and whole body prey fish are commonly used as indicators of wildlife risk. For piscivorous mammals, prey fish, blubber, or blood concentrations are used. Sport fish data are not as useful for this purpose as species are generally not consumed by wildlife and analysis is usually only performed on muscle tissue.

Statewide assessment of potential impacts of DDTs to aquatic life in California has yet to be performed. The sport fish data compiled in this report are not ideally suited for such an assessment. However, these data can be used to provide a preliminary, broad assessment of predator-risk. In the following discussion, piscivorous birds are used to examine the potential risks from DDT to aquatic predators in California water bodies.

Studies of DDT bioaccumulation through the food chain have shown that concentrations in bird eggs are commonly 20 to 60 times higher than in the fish that they consume. This ratio, between the concentrations of a chemical in an organism to that of its prey, is known as a biomagnification factor (BMF). Eggs of double-crested cormorants have shown ratios to fish prey of approximately 60 for p,p'-DDE (Weseloh et al. 1982). Braune and Nostrom (1989) also developed BMFs for DDE (the primary metabolite of DDT) and numerous other organochlorine compounds. Their ratios between prey fish and herring gull eggs predicted DDE to biomagnify to very high concentrations with a mean BMF of 85 $\pm$ 20 (s.d.). More recently, BMFs for DDE were predicted using an organochlorine bioaccumulation model for herring gulls of Lake Ontario (Norstrom et al. 2007). The predicted BMFs for fish:eggs were consistent with Braune and Nostrom (1989) ranging from 45 – 85, depending on whether the BMFs were calculated based on egg or whole body fish diets. However, in order to employ the sport fish dataset to evaluate risks to piscivorous birds, a BMF between sport fish and birds eggs is required. These ratios have not commonly been documented in the literature. DDTs in sport fish and cormorant eggs have been measured in San Pablo Bay. BMFs between sport fish:cormorant eggs ranged from 26 – 63 for seven sport fish species, with an average of 39 (Davis et al. 2004a, Davis et al. 2004b). Assuming these data are representative, a BMF of 40 would appear appropriate to examine the potential risk to birds based on a conversion between sport fish to bird egg concentrations.

Four locations sampled in the recent time period (1998 – 2003) had sport fish with concentrations above 800 ppb (or 0.8 ppm). Three of these locations were near the Salton Sea (Imperial Valley) in southern California. The other location was in Old Salinas River (Monterey County). Using a BMF of 40, bird eggs from these areas would be predicted to have concentrations of 32 ppm or higher. Typical concentrations of DDTs associated with 20% eggshell thinning have ranged from 1 – 5 ppm (Kiff et al. 1979, Blus 1982, Kiff 1994). Mean concentrations above 5 ppm have been associated with decreased hatching success (Blus 1996), while concentrations above 24 ppm have caused eggshell thinning sufficient to reduce populations in double-crested cormorants (Gress et al. 1973). Therefore, water bodies above 800 ppb might be expected to be well above thresholds for effects on piscivorous birds. Furthermore, conversion of the eggshell threshold of 24 ppm using a BMF of 40 would lead to a revised sport fish threshold of approximately 600 ppb (0.6 ppm). This would result in five locations in the recent sampling period to be above the threshold. This broad assessment suggests that some water bodies in California remain above the threshold for concern for risks to birds.

b. Impacts on Aquatic Birds

Southern California

The history of DDT impacts on aquatic birds in southern California is a classic example of the impact of toxic pollution on wildlife, management action, and recovery of the affected populations. A primary effect of DDT on seabirds is eggshell thinning and subsequent breeding failure resulting from egg breakage and reduced embryo survival (Cooke 1973, Blus 1982). A severe decline in the coastal populations of California least terns and brown pelicans (Hickey and Anderson 1968) prompted both state and federal governments to designate these as endangered species in 1970 (Massey 1974). The primary reason for the decline of brown pelicans was the use of DDT, which resulted in thin-shelled eggs that were easily broken during incubation. California least terns, double-crested cormorants and brown pelicans are particularly sensitive to egg-shell thinning caused by exposure to DDE (the primary metabolite of DDT). During the last few years of DDT use (1968 – 1972), many coastal estuaries were elevated markedly above background levels, especially in regions of high agricultural usage and industrial pesticide production (e.g., Los Angeles County; Fry 1995).

Concentrations of DDT in birds have generally declined in many areas of the United States since its use was banned (Jacknow et al. 1986). However, relatively high DDT levels have persisted in some regions of the state, including southern California (e.g., Gress et al. 1973, Ohlendorf and Miller 1984, Rasmussen et al. 1987, Sutula et al. 2005), San Francisco Bay (e.g. Schwarzbach 2001, Davis et al. 2004a), and the central Californian coast (e.g., Bacon et al. 1999, Kannan et al. 2004). The study by Sutula et al. (2005) suggested that DDTs may have still been causing some degree of reproductive impairment of clapper rails in Upper Newport Bay in the 2003 and 2004 breeding seasons.

Following the ban on DDT use, contaminant levels declined dramatically in coastal areas where agricultural runoff was the principal source (Fry 1995). However, in industrial and highly urbanized regions of southern California (e.g., San Pedro Harbor), DDT concentrations remained relatively high. DDT at these sites was bioavailable to marine organisms (fish, oysters, and mussels) in studies conducted in 1978 (Goldberg et al. 1978) and 1984 (NOAA 1987). Fry (1995) showed that the DDT concentrations in southern California seabirds remained high as recently as the 1990s. In fact, the average levels of DDT from every western gull colony sampled in the Southern California Bight in 1992 were higher than for any colony outside of this region (Figure 3.5.11). Regressions of DDE concentration to eggshell thickness in samples collected in 1992 demonstrated that half of the cormorant and brown pelican eggs collected at Anacapa Island (Southern California Bight) still had shell thinning greater than ten percent (Fry 1995), though this degree of thinning did not prevent recovery of these populations. Brown pelicans on islands of the Southern California Bight have increased in number throughout the 1990s, with 6380 nesting pairs reported by the US Fish & Wildlife Service (USFWS) in 1997 (DFG 2005). This is in stark comparison to the late 1960s, when the USFWS had found almost no brown pelican nesting pairs. Similar population responses to DDT exposure in recent years have been shown in caspian terns and black-crowned night herons in southern California (Henny et al. 1984, Ohlendorf et al. 1985, Roberts and Berg 2000).

Seabirds that inhabit the Southern California Bight on a seasonal basis have also shown higher DDT concentrations than birds from other regions. Double-crested cormorants, for example, are seasonally migratory, and feed almost exclusively on small fish in bays, estuaries, and the open ocean. Birds from northern California move as far south as the Southern California Bight for the non-breeding season. The highest DDT concentrations reported by Fry (1995) were found in eggs from double-crested cormorants in the Southern California Bight. Cormorants of the Bight are thought to predominantly forage around the highly contaminated locations of the Palos Verdes Shelf (Glaser and Connolly 2002). Lower residue levels were found in eggs from Morro Rock, Russian River Rocks, and Humboldt Bay. Elevated concentration in double-crested cormorants and other species, may relate closely to their exposure to highly contaminated areas during migration (Fry 1995).

The Salton Sea has become one of the most important nesting sites and stopovers for migrating birds. USFWS have stated that in some years, as many as 95% of the North American population of eared grebes, 90% of American white pelicans, 50% of ruddy ducks and 40% of Yuma clapper rails may use the Salton Sea. Exceptionally high concentrations of DDE were found in black-crowned night-herons (8.62 ppm wet wt) and great egret eggs (24.0 ppm wet wt) collected from the Imperial Valley (Salton Sea) region in 1985. In recent years, however, some of these species have shown recent improvement in populations. Roberts and Berg (2000) reported population recoveries in brown pelicans and peregrine falcons from the Salton Sea. From 1992 – 1993, peak numbers of brown pelicans reached up to 5,000 individuals.

San Francisco Bay

Historic DDT contamination in San Francisco Bay has previously shown the potential for causing adverse effects in birds, similar to southern California, though the available evidence for impacts is not nearly as compelling. In 1982, concentrations of DDE in 10% of black-crowned night heron eggs from Bair Island (south San Francisco Bay) were above threshold effect concentrations for egg hatchability, with eggshell thickness also reduced by 8 – 13% (Ohlendorf et al. 1988, Roberts and Berg 2000). The black-crowned night

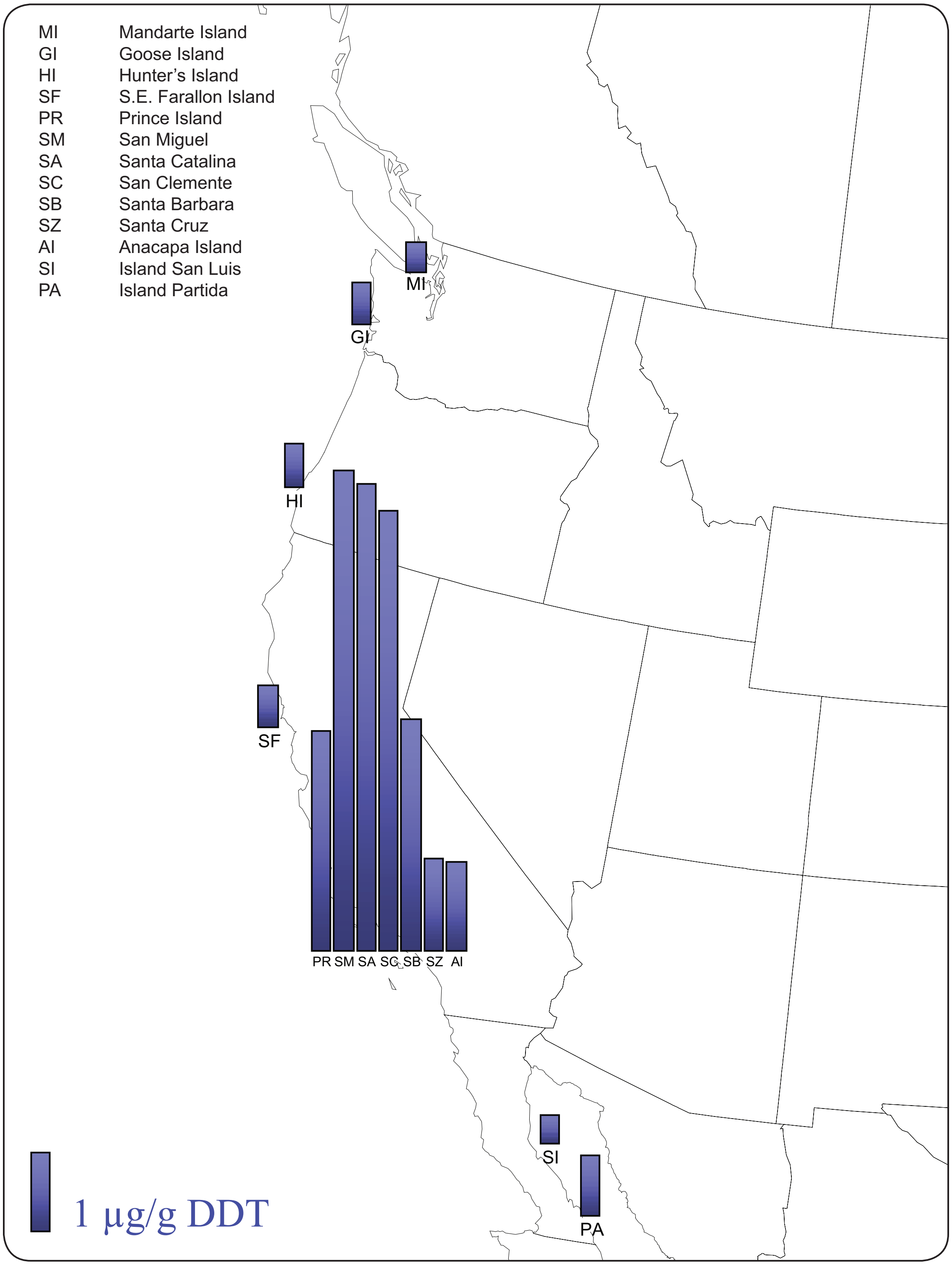

Figure 3.5.11. Total DDT in Western Gull eggs from island sampling locations along the Pacific Coast. From Fry (1995).

heron colonies that nest in San Francisco Bay are non-migratory, and thus more likely to acquire their contaminant loads locally (Gill and Mewaldt 1979). However, they are known to disperse randomly to regions of the central coast (e.g., Monterey Bay) and north-east to the Sacramento-San Joaquin Delta. The source of DDE and other contaminants found in these bird eggs may therefore have been through a diet of fish of the San Joaquin Valley or Monterey Bay (Ohlendorf et al. 1991). During the 1980s, fish from those regions exhibited DDE concentrations of up to 6 ppm (Ohlendorf et al. 1987, Rasmussen et al. 1987).

The historic discharge of pollutants to San Francisco Bay has been a potential threat to clapper rail populations. Eggs that were collected from several sites in San Francisco Bay in 1975 exhibited concentrations of DDE that ranged from 0.38 to 2.1 ppm (wet wt). These concentrations were relatively high, but below concentrations thought to induce reproductive defects (Lonzarich et al. 1992). Impacts on eggshell thickness have not commonly been reported in this species (Klaas et al. 1980, Goodbred et al. 1996). By the mid-1980s, the DDE concentration of clapper rail eggs in the Bay had shown a significant decline. Residues of DDE in eggs collected in 1987 ranged from 0.14 to 0.63 ppm. The eggshell thickness reported in this study were comparable to eggs collected prior to 1940 (Lonzarich et al. 1992).

Recent data has shown that cormorants and song sparrows in the Bay (Davis et al. 2004a) exhibit concentrations below the threshold levels for reproductive impairment, similar to clapper rails. DDE concentrations were significantly higher in 2000 and 2001, compared to 1999, but this was not considered high enough to cause eggshell thinning or embryo mortality. The threshold for eggshell thinning (24 ppm), that has been associated with reduced population declines in double- crested cormorants (Gress et al. 1973) is considered to be relatively high. The maximum concentration observed for cormorants in 2001 by Davis et al. (2004a) was 3 ppm. Results suggest that populations may be in a period of recovery after the peak in DDT contamination of San Francisco Bay in the 1960s, which led to severe population declines. Currently, clapper rail remains on the USFWS' Endangered Species List.

c. Impacts on Fish and Marine Mammals

The impact of chemical contamination in forage fish has been demonstrated to be a valuable indicator of the expected risk to predators in the ecosystem. A recent CISNET study (Davis et al. 2004b) was the first to describe organic contaminants in forage fish from San Pablo Bay. Juvenile striped bass and staghorn sculpin were analyzed in the study as they are residents in the Bay and likely to be consumed by wildlife. Both species exhibited average DDT (sum of DDD and DDE) concentrations that exceeded the DDT threshold for predator-risk. DDT accumulation in young striped bass (24 ppb wet wt) and sculpin (16 ppb wet wt) were above the predator-risk threshold of 14 ppb (wet wt) established by Environment Canada (CCME, 2002) for the protection of wildlife consumers of aquatic biota.

Two large local studies of contaminants in southern California forage fish were found in the literature. Both studies were conducted by SCCWRP to examine concentrations of DDTs and other pollutants in fish that are potential prey for piscivorous birds and mammals (Allen et al. 2002a, Allen et al. 2004a, Allen et al. 2004b).

The assessment in the Southern California Bight focused on five flatfish species (i.e., sanddab [three species], California halibut, and slender sole) that were common across the study area. Half of the stations sampled (119 of 225, 53%) were from the middle shelf zone (31 – 120 m), and the majority of samples consisted of age-1 fish (239 of 275, 89%). Total DDT concentrations were measured from whole body composite samples collected between June and September 1998, which ranged from non-detect to 10,462 ppb (wet wt). The highest concentrations were found on the Palos Verdes Shelf, and lowest values were on the inner shelf. Concentrations of DDT exceeded the predator-risk guideline of 14 ppb (CCME 2001) in 66% of samples. The proportion of fish exceeding the guideline was highest in the bays and estuaries (92%) and lowest in the inner shelf zone (0%).

The second SCCWRP study was performed in Newport Bay, sampling both recreational and forage fish species between 2000 and 2002 (Allen et al. 2004a). Nine forage fish species were collected from various sites within the upper and lower reaches of Newport Bay. Total DDT concentrations were measured from whole body composite samples collected in March – April and August – September 2002. All composites of the forage fish species had DDT concentrations above the predator-risk threshold of 14 ppb (CCME 2001). DDT concentrations ranged from 50 to 262 ppb (wet wt), with the highest average for cheekspot goby (195 ppb). Although the results from these studies suggest concern for wildlife species consuming fish from the region, the DDT concentrations are much lower than have been shown previously from similar southern California locations (Mearns et al. 1991), suggesting that the influence of DDT in the region is improving.

Marine mammals also accumulate significant amounts of legacy pesticides due to their long life spans, low capacity for metabolic degradation, and lipid rich blubber (Kannan et al. 2004). Sea otters are particularly good indicators as they remain relatively sedentary and reflect local conditions. From 1988 – 1992, Bacon et al. (1999) showed that DDT concentrations in the liver tissue of California sea otters (850 ppm wet wt) were over 20 times higher than in Aleutian otters (40 ppm) and over 800 times higher than California sea otters from southeast Alaska (1 ppm). DDT was also noted to be high in individuals residing in Morro Bay, Estero Bay, and Moss Landing Harbor. DDT concentrations in the liver tissue of California sea otters sampled from 1992 – 1996 ranged from 280 – 5900 ppb (wet wt), with individuals of Monterey Harbor having the highest concentrations (Nakata et al. 1998). The likely sources of DDT contamination at central California sites are agriculture and urban use (MacGregor 1974). Diseases in California sea otters have been attributed to immune suppression due to legacy pesticide contamination (Thomas and Cole 1996). The concentrations reported in California sea otters in 1988 – 1992 were below levels associated with reproductive problems (Jensen et al. 1977). However, the results of Nakata (1998) indicate the DDT may remain of concern in some local habitats.

Other marine mammals have shown similar concentrations and symptoms to the sea otters. For example, the liver tissue of adult California sea lions off the central California coast in 1970 exhibited average DDT concentrations of 6670 ppb (wet wt). The sea lions had infectious diseases and impaired reproduction (e.g., immature births and early termination of pregnancy) that may have been related to high body burdens of DDT (DeLong et al. 1973). Sea lions often inhabit industrialized areas, marinas, and harbors, where historic levels of DDT would have been high.

Recent data indicate that mammals in California may not be as impaired as indicated by earlier data (Lieberg-Clark et al. 1995, Neale et al. 2005). Since the 1970s, a decrease in DDT of more than two orders of magnitude has been reported for California sea lions. However, the impact of lower DDT concentrations on California sea lion populations requires further study. Despite reports of increased numbers in recent years, the cause-and-effect has yet to be established (O'Shea and Brownell 1998). The indications are, however, that the reduction in DDT body-burdens has led to a decrease in the reproductive impacts of the 1970s.

3.5.4. DDT Summary

Recent sport fish monitoring data (1998 – 2003) indicate that DDT concentrations in most areas of the state were in the low < 800 ppb concentration category. Long-term trend monitoring in sport fish and bivalves have generally shown declines over the past 30 years in response to the use restrictions and federal ban. Long-term trends in sport fish from the Imperial Valley (Salton Sea) region indicate consistently high DDT concentrations during the last 20 years. The DDT ban has not been as successful in reducing concentrations in sport fish of this region. A review of aquatic life studies indicates that DDT may be of significant concern to some bird and fish populations. The most likely historical sources of this DDT to California water bodies are agricultural and urban runoff.

3.5.5. Impact of Dieldrin on Fishing in California

a. Current Status

Consumption Advisories

Fish consumption advisories due at least partially to dieldrin only exist for San Francisco Bay and the Sacramento-San Joaquin Delta (Table 3.2.1). Therefore, numerous harbors, bays, and other water body types (e.g., Anaheim Bay and Big Bear Lake), which were 303d listed in 2002 do not currently have fish consumption advisories. However, in spite of this disparity and that dieldrin concentrations have been declining gradually since peaking during the 1960s and 1970s, the advisory for San Francisco Bay was only issued in 1994. This indicates that the current understanding of dieldrin contamination in sport fish has improved in the last 10 – 15 years, and that in areas other than San Francisco Bay and the Delta, dieldrin should not pose a significant threat to human health.

303(d) Listings

The 2002 303(d) List for California indicates that dieldrin is not a major contributor to impacts on water quality in the state. The 2002 303(d) List (Appendix 3) includes dieldrin listings for the following areas:

- Region 2 – San Francisco Bay (172,683,acres), San Pablo Bay (68,349 acres), and Sacramento-San Joaquin Delta (41,736 acres);
- Region 4 – Los Angeles Harbor (36 acres) and McGrath Lake (20 acres); and
- Region 8 – Anaheim Bay (402 acres) and Big Bear Lake (2865 acres).

The list includes major bays, estuaries, and lakes, mostly distributed in northern California. San Francisco Bay is the largest area affected, currently listed due in-part to dieldrin because of the interim fish consumption advisory developed by the California Office of Environmental Health Hazard Assessment (OEHHA) in 1994.

Recent Monitoring Data

Recent sport fish monitoring data (1998 – 2003) indicate that dieldrin concentrations in the vast majority (238 of 244, 98%) of the state were in the green < 25 ppb category (Table 3.5.2 and Figure 3.5.12). Five sites (2%) were in the yellow 800 – 3500 ppb category, and one site (< 1%) was in the orange category. No sites were in the red category. Three of the five yellow sites were located in the Imperial Valley (Salton Sea) region of southern California, where fish consumption advisories do not currently exist for dieldrin. The single orange site was for San Pablo Reservoir, an area where consumption advice for diedrin is currently in-place. Since the majority of recent sport fish data indicate that locations around the state are in the green category for dieldrin, the limited number of fish advisories and 303d listed water bodies appear appropriate at this time.

b. Long-term Trends in Impact of Dieldrin on Fishing in California

Management Actions

The primary use of dieldrin was in agriculture and structural termite control. The first significant management action to reduce dieldrin was the federal restriction on its application to food products that began in 1974. By 1985, EPA had imposed a ban on dieldrin throughout the United States for all but subsurface termite control, nonfood roots and tops, and moth-proofing in closed systems. Due to its widespread use throughout the United States, dieldrin contamination is found in both urban and agricultural areas. The expectation is that dieldrin is currently declining gradually across the state, presumably in large part due to the ban. However, this contaminant remains persistent in the environment due to long residence times in polluted watersheds. Specific management actions to cleanup historical residues of dieldrin were not found in the review conducted for this report.

Long-term Trends

Sport Fish

Concentrations of dieldrin in aquatic food webs across the state have generally shown declines over the past 30 years in response to the use restrictions and federal ban. Concentrations over time (Figures 3.5.13, 3.5.14) indicate that sport fish dieldrin levels were higher prior to 1998 than currently (Figure 3.5.12). Six (3%) locations monitored from 1988 – 1997 and 11 (7%) from 1978 – 1987 were in the yellow 25 – 100 ppb category (Table 3.5.2). No locations from 1988 – 1997 were greater than 100 ppb. However, from 1978 – 1987, one (1%) site was present in the orange 100 – 200 ppb category, and two (1%) sites were in the red > 200 ppb category. This is compared to recent data (1998 – 2003) showing only 2% of sites above 25 ppb. The higher concentrations (cluster of yellow, orange, and red sites) on the central coast in 1978 – 1987 appears to have diminished with time, as only a single yellow site remains in the most recent time interval.

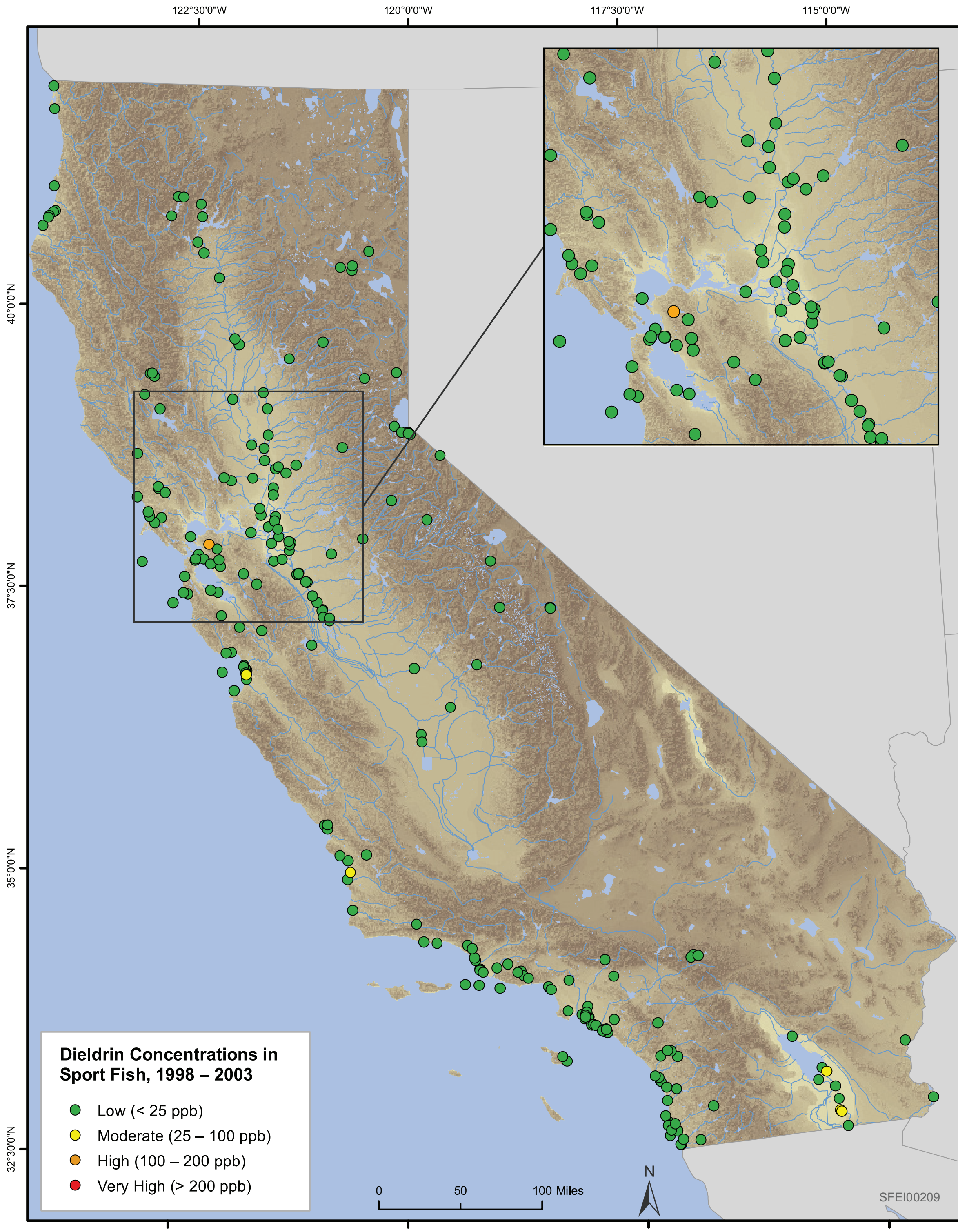

Figure 3.5.12. Dieldrin concentrations in California sport fish, 1998 – 2003. Based on dieldrin measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

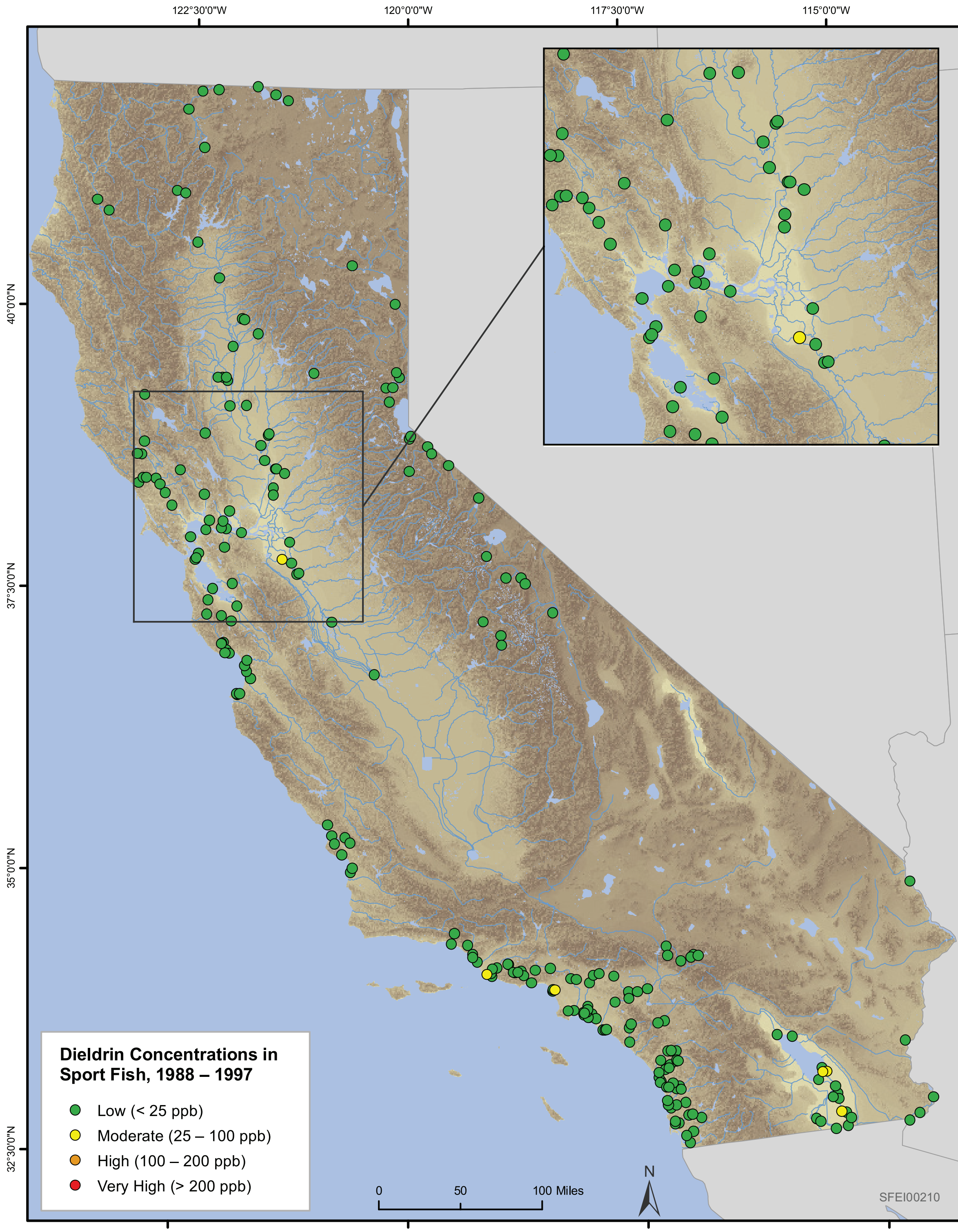

Figure 3.5.13. Dieldrin concentrations in California sport fish, 1988 – 1997. Based on dieldrin measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

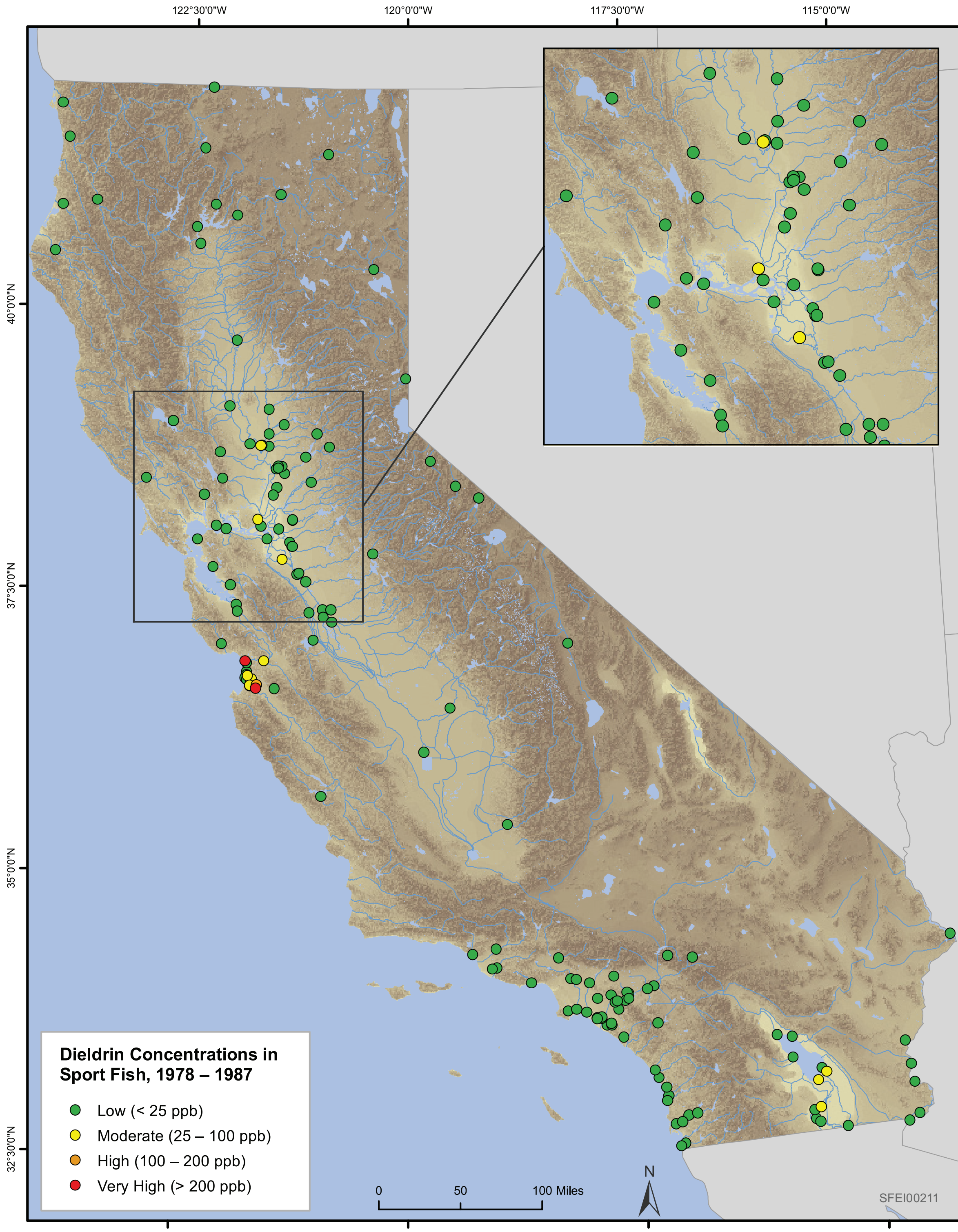

Figure 3.5.14. Dieldrin concentrations in California sport fish, 1978 – 1987. Based on dieldrin measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

As was observed for DDT, a hotspot for dieldrin contamination is located in the Imperial Valley (Salton Sea) region of southern California. Recent data (Figure 3.5.12) suggests that the Imperial Valley (Salton Sea) region includes three locations in the yellow category. Previous time intervals also included three yellow sites from 1988 – 1997 and 1978 – 1987. This trend in concentrations indicates that dieldrin has continued to persist in the Imperial Valley (Salton Sea) region.

Long term-trend monitoring in sport fish provides further evidence for decreased dieldrin concentrations in most California water bodies. Long-term time series data were available for 10 locations sampled for sport fish. A spatial pattern from these locations is apparent (Figure 3.5.15), as the data in the northern regions of the state indicate lower concentrations compared to sites in southern California (except Palo Verde Outfall Drain). However, of the ten sites examined, only San Diego Creek at Michelson Drive showed a significant long-term decline ($p < 0.05$). Historically high concentrations in southern California are evident from red shiner in San Diego Creek at Michelson Drive, and channel catfish in New River at Westmorland and Alamo River at Calipatria. These locations indicate high dieldrin concentrations throughout the 20 years of monitoring with considerable inter-annual variation. However, the most recent data from these sites indicate much lower concentrations than observed previously. These results suggest that the southern regions of the state were more historically affected, with respect to dieldrin, than the northern regions of the state. The most recent data are encouraging and indicate that these locations will improve in the long-term.

Bivalves

Monitoring of bivalves by the State Mussel Watch Program has provided further evidence for declining dieldrin concentrations in California water bodies (Figure 3.5.16). Of ten sites examined, five indicate a significant decline ($p < 0.05$) between years. At Point Pinole ($p < 0.05$) concentrations appear to have increased with time. Gunther et al. (1999) suggested that the State Mussel Watch data at Pinole Point were confounded by bivalves of low body condition. Many of the southern California sites (e.g., Huntington Harbor at Warner Avenue Bridge and Newport Bay at Crows Nest) show a dramatic decline in dieldrin concentrations throughout the 20 years of bivalve monitoring. These results are consistent with previous interpretations of State Mussel Watch data (e.g., Gunther et al. 1999) that have suggested declines in dieldrin at many locations across the state.

Case Studies

Monitoring by State Mussel Watch (SMW) and the RMP has provided important information on trends in dieldrin, particularly in San Francisco Bay. Sport fish collected by the RMP in 1997, 2000, and 2003 have exhibited concentrations that are well below 25 ppb. However, the fattier (higher lipid) species (e.g., white croaker and shiner surfperch) have shown relatively higher concentrations (Figure 3.5.17).

Monitoring of bivalves has also generated datasets on long-term trends in dieldrin. Concentrations in San Francisco Bay have varied greatly, with the highest concentrations being observed in 1980 and 1984 (Gunther et al. 1999). Since 1989, dieldrin has remained relatively constant (around 200 ppb, lipid wt). Evidence of a leveling off of dieldrin concentrations in the Bay is consistent with the slow degradation of this

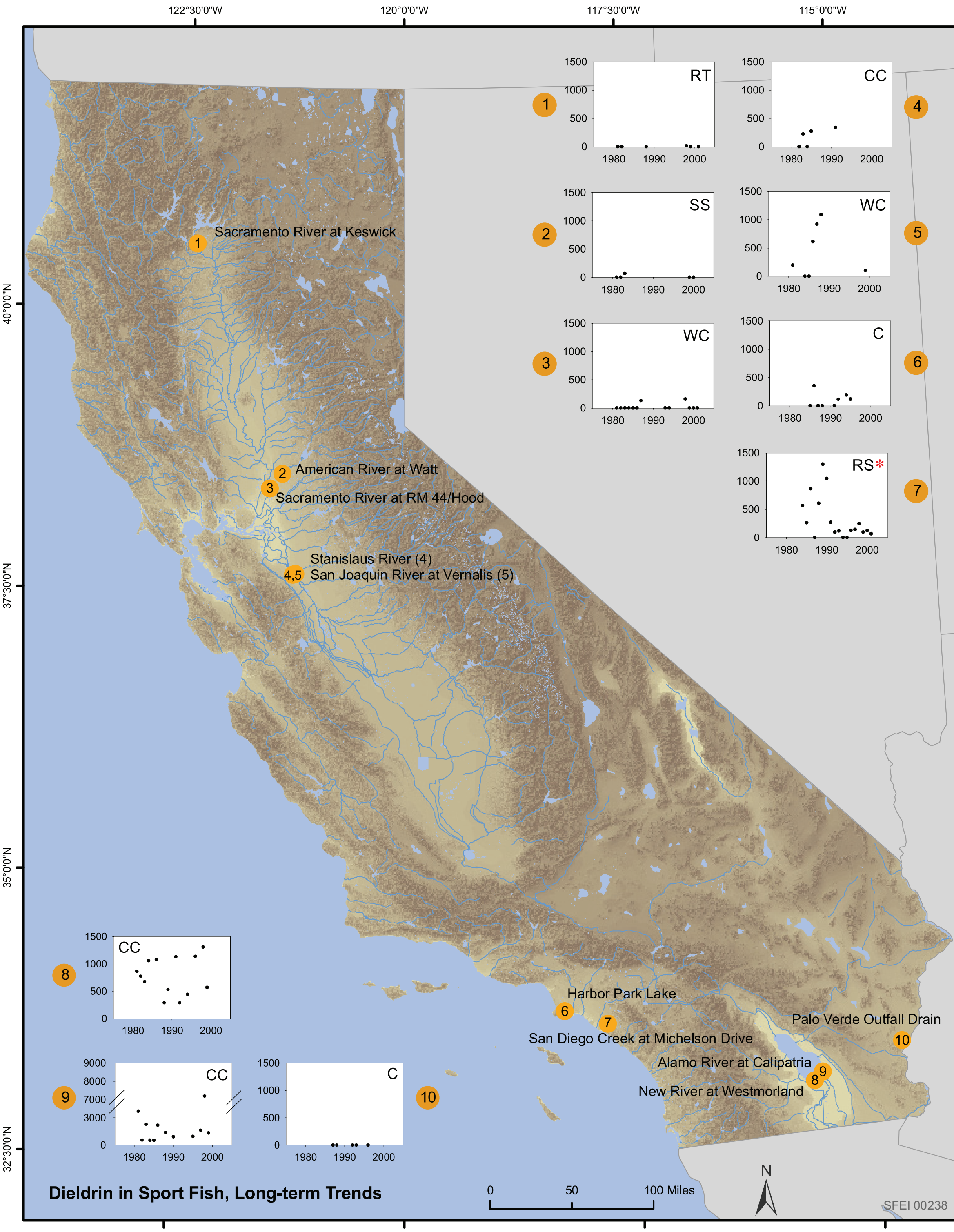

Figure 3.5.15. Long-term trends in dieldrin concentrations in California sport fish. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight. Species shown are rainbow trout (RT), Sacramento sucker (SS), white catfish (WC), channel catfish (CC), red shiner (RS), and common carp (C).

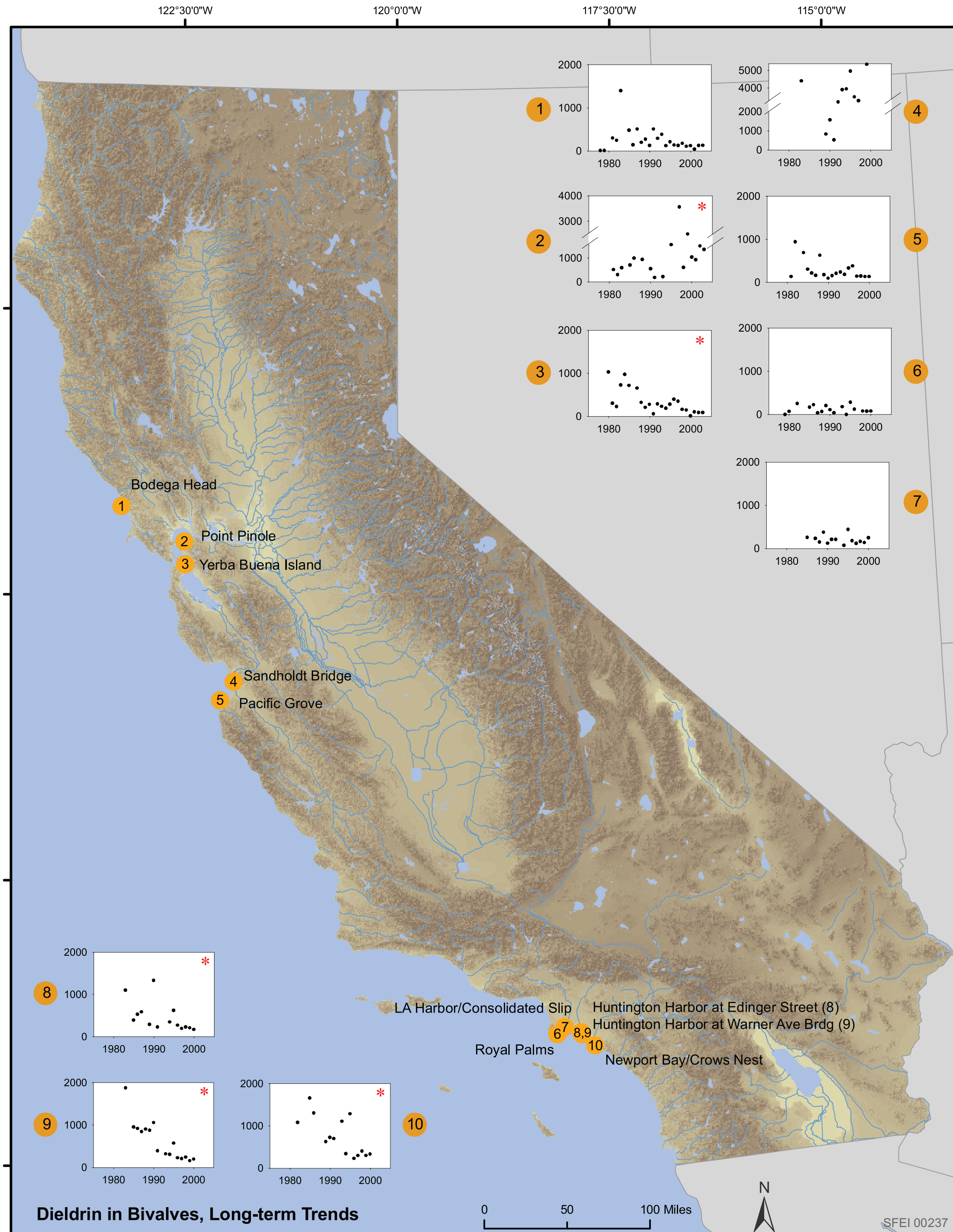

Figure 3.5.16. Long-term trends in dieldrin concentrations in California mussels measured by the State Mussel Watch Program. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight.

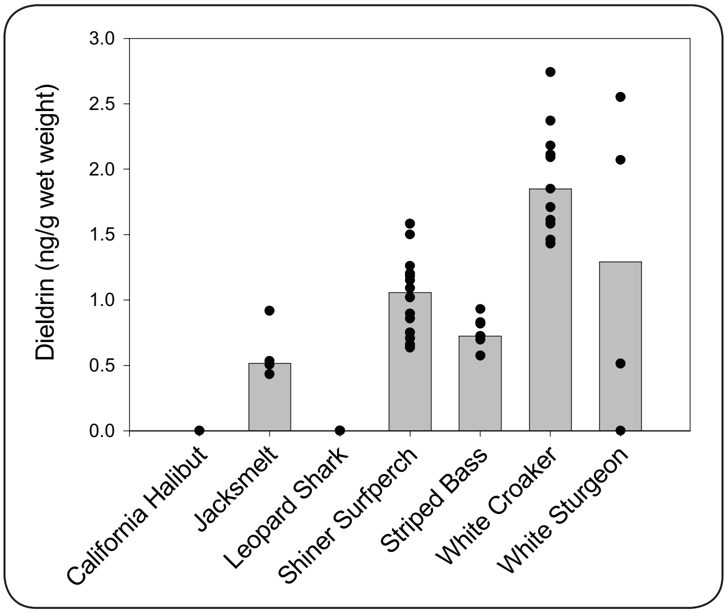

Figure 3.5.17. Dieldrin concentrations (ng/g) in Bay fish, 2003. Points are concentrations in each composite sample analyzed. Bars indicate median concentrations. From RMP (2006).

pesticide in the environment. Concerns over dieldrin contamination in sediments are, therefore, not expected to diminish in the near future.

c. Sources and Pathways

Dieldrin concentrations in the recent dataset (Figure 3.5.18) indicates that five sites distributed from San Francisco Bay to Imperial Valley (Salton Sea) had relatively higher concentration in sport fish compared to the rest of the state. Notably, a cluster of high concentrations is evident in the Imperial Valley (Salton Sea) region. However, the highest dieldrin concentration was shown for San Pablo Bay reservoir, where high DDT concentrations were also found. The source of pesticide contamination to the reservoir is unknown, but may be associated with Richmond Harbor where a superfund site was located. However, for the state as a whole, there do not appear to be sources of dieldrin to aquatic ecosystems that have resulted in elevated concentrations among sport fish collected from 1998 – 2003.

Historic datasets indicate that sources of dieldrin were more widespread previously (Figure 3.5.19, 3.5.20). Elevated concentrations of dieldrin were found in agricultural (Imperial Valley and San Joaquin River) and urban (Los Angeles and Oxnard) areas. The highest concentration in each historic time interval was 66 ppb at Pumice Drain near the Salton Sea (1988 – 1997) and 550 ppb at Blanco Drain near Oxnard (1978 – 1987). Both of these locations were in an area of numerous high concentrations along the high agriculture areas of the central coast and Imperial Valley. In these regions, watershed soils and inland water bodies may have slower degradation rates compared to adjacent marine and estuarine areas (Spencer et al. 1996), which has resulted in the higher local concentrations. Degradation rates are known to be higher in marine and estuarine areas due to high moisture content. In general, dieldrin sources had the largest impact (abundance of tall concentration bars) on sport fish in 1978 – 1987, compared to other time intervals. Dieldrin concentrations in aquatic food webs are not significantly elevated at present, and are greatly reduced from levels measured previously.

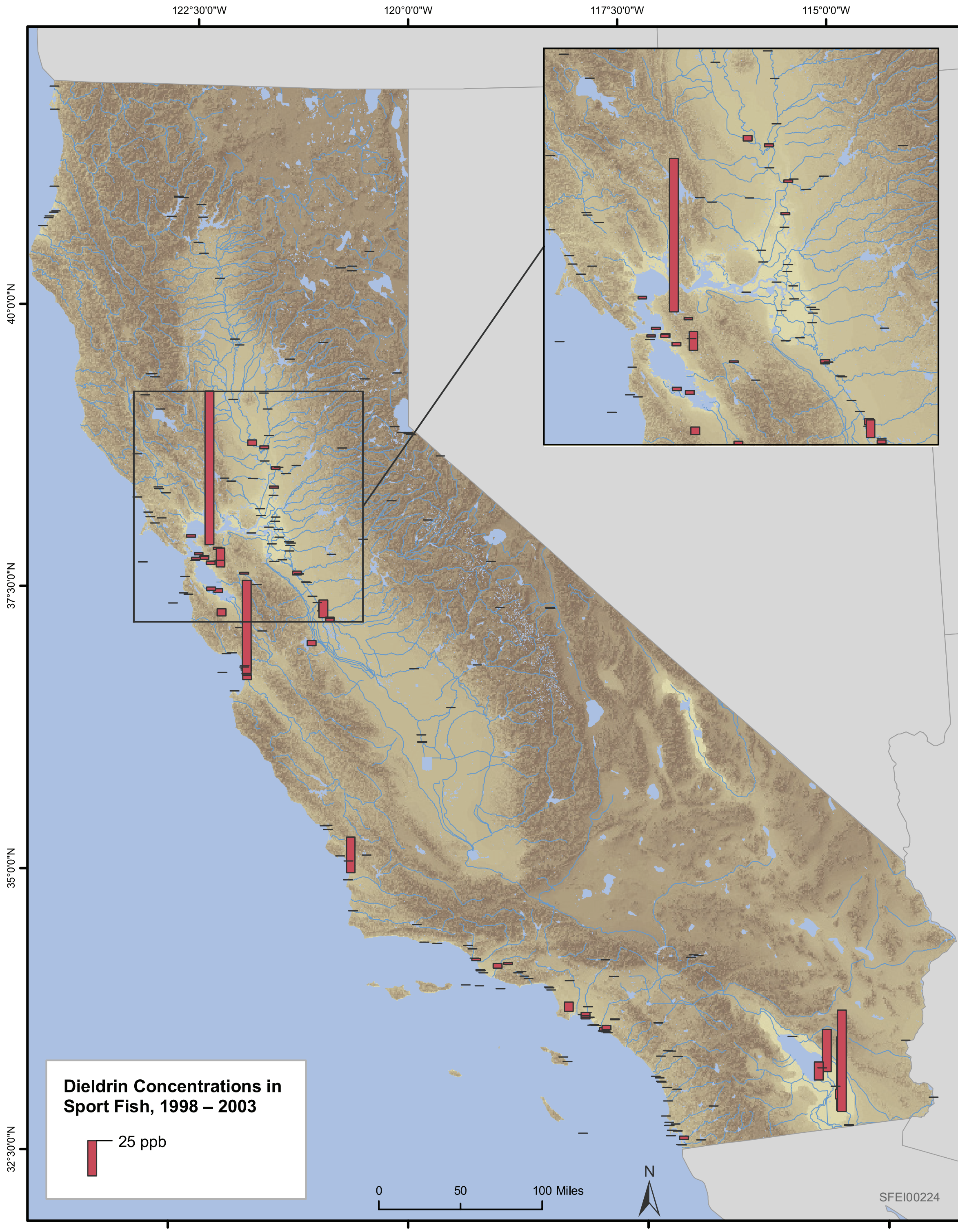

Figure 3.5.18. Dieldrin concentrations in California sport fish, 1998 – 2003. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

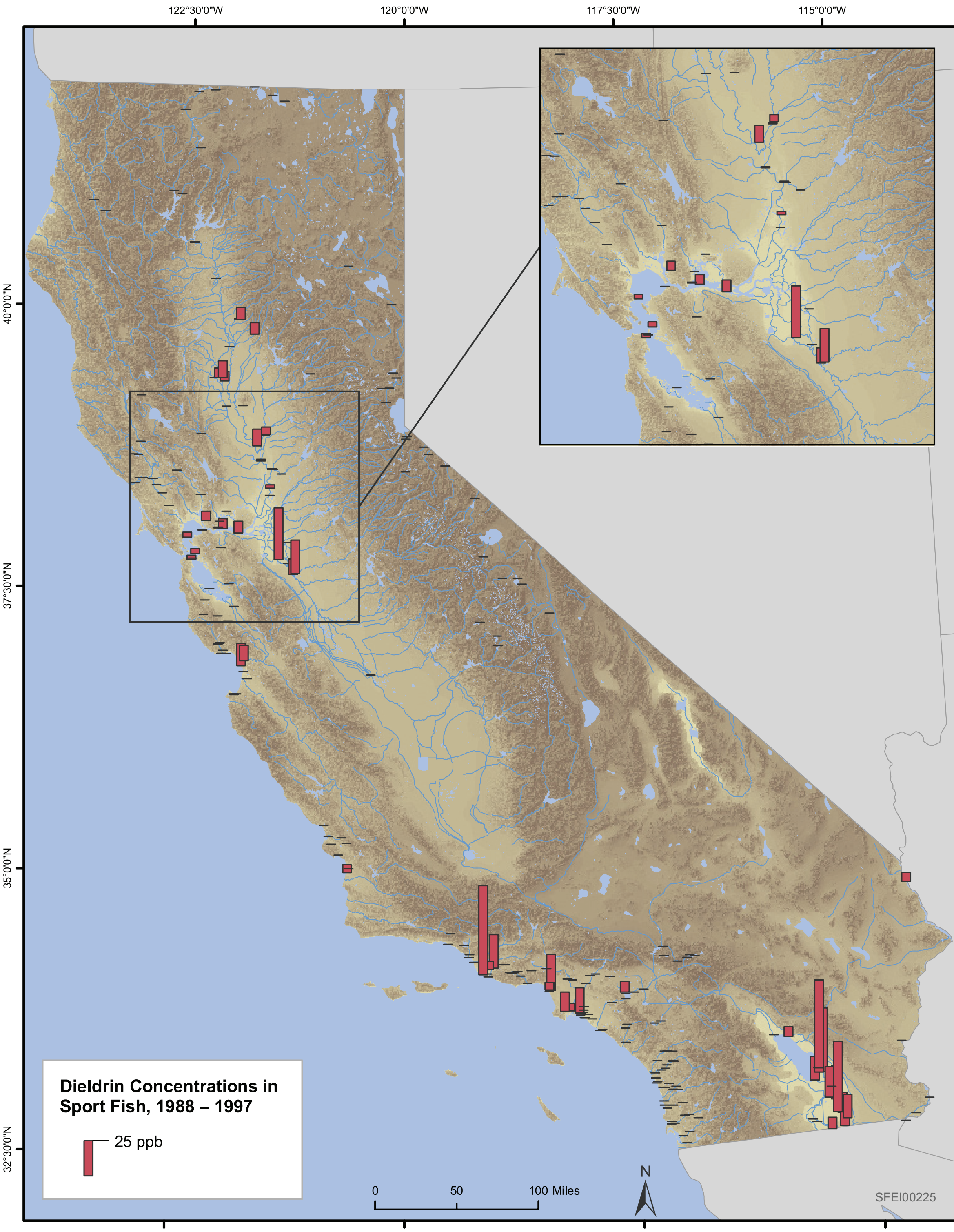

Figure 3.5.19. Dieldrin concentrations in California sport fish, 1988 – 1997. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

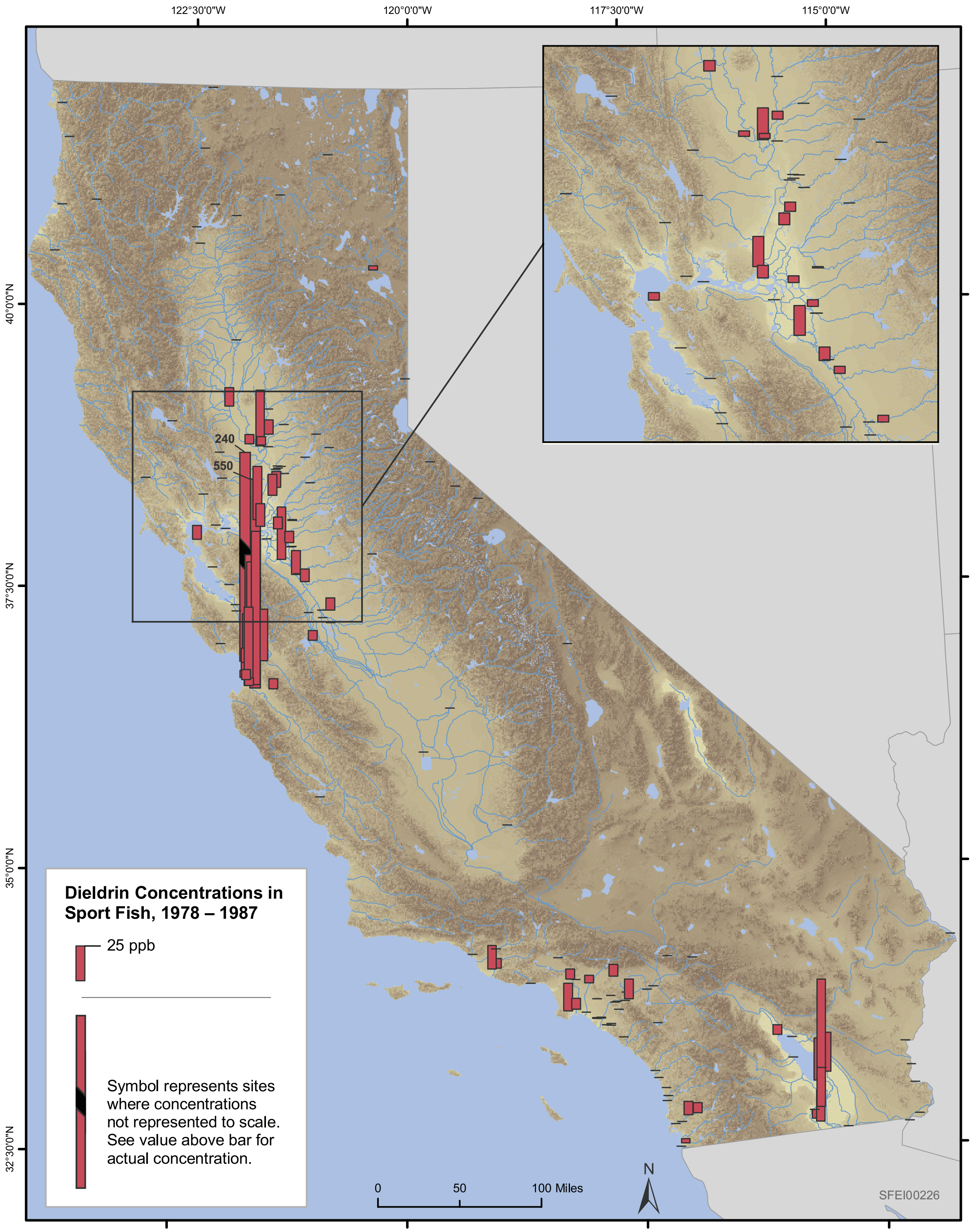

Figure 3.5.20. Dieldrin concentrations in California sport fish, 1978 – 1987. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

3.5.6. Impact of Dieldrin on Aquatic Life in California

a. Impacts on Aquatic Birds

The impact of dieldrin on aquatic life has most commonly been examined in seabirds. Compared to other contaminants (e.g., DDT and PCBs), however, dieldrin concentrations have generally been well below the threshold for toxic effects. Double-crested cormorants from San Pablo Bay, for example, contained PCBs approaching the tissue threshold, but only low concentrations of dieldrin (Davis et al. 2004a). Canadian tissue residue guidelines have not been developed for this contaminant. In San Francisco Bay, low concentrations have been shown for black-crowned night herons (Rattner et al. 1996), California least terns (Hothem and Zador 1995), and clapper rails (Schwarzbach 2001). Even surf scoters collected from the highly industrialized region of Richmond Harbor (San Francisco Bay) had concentrations of dieldrin that were unlikely to impact survival or reproduction (Ohlendorf et al. 1991). In the Salton Sea, where historic application of dieldrin has contributed significantly to contamination in the region, dieldrin was not suspected of eliciting toxic effects in night herons or great egrets (Ohlendorf and Marois 1990, Roberts and Berg 2000). Furthermore, temporal declines have been shown in San Francisco Bay for clapper rails (Lonzarich et al. 1992, Schwarzbach et al. 2001), and black-crowned night herons (Ohlendorf et al. 1988, Hothem et al. 1995). The declines reported in these studies suggest that adverse effects due to dieldrin should not be expected.

b. Impacts on Fish and Marine Mammals

Dieldrin has not been investigated in-depth for impacts to fish and marine mammals. This is likely due to other contaminants, such as DDT, being of higher concern. In the review conducted for this report, the only study found to investigate the effect of dieldrin to mammals was carried out using California sea otter liver tissue collected from 1988 – 1992 from various locations in California. Bacon et al. (1999) found higher concentration of DDT in California sea otters than in similar species from Alaska and the Aleutian Islands. Dieldrin concentrations in the same study were found at very low levels, and were not attributable to any ill-effects. In review of the limited data on dieldrin impacts to aquatic life, it appears that it is not considered a significant threat to species in California.

3.5.7. Dieldrin Summary

Recent sport fish monitoring data (1998 – 2003) indicate that dieldrin concentrations in most areas of the state were in the low < 25 ppb concentration category. Long-term trend monitoring in sport fish and bivalves have generally shown gradual declines over the past 30 years in response to the use restrictions and federal ban. Sport fish from the Imperial Valley (Salton Sea) region indicate only a recent decline. A review of dieldrin impacts to aquatic life species suggested that adverse affects should not be expected. The dieldrin ban appears to have been successful in reducing concentrations and impacts across the state, with locations of higher historical contamination recently improving. Agricultural runoff into California water bodies has been the primary historical source of this pollutant.

3.5.8. Impact of Chlordanes on Fishing in California

a. Current Status

Consumption Advisories

As of May 2007, consumption advisories due partially to chlordanes (Table 3.2.1) were only listed for San Francisco Bay and the Sacramento-San Joaquin Delta, and Machado Lake (formally Harbor Park Lake) in Los Angeles County. There is general agreement between the locations with advisories and water bodies on the 2002 303(d) List. Despite that chlordane concentrations have been declining gradually in California since their peak in the 1960s and 1970s, these consumption advisories have all been issued since 1994. Similar to DDT and dieldrin, this pattern indicates that the current understanding of chlordane contamination in sport fish has improved recently, which has resulted in greater attention from managers. Therefore, in areas other than those listed, chlordane are not predicted to pose a significant threat to human health due to fish consumption or water quality impairment.

303(d) Listings

The 2002 303(d) List for California indicates that chlordane is not a major contributor to impacts on water quality in the state. The 2002 303(d) List included chlordane listings (Appendix 3) for a few areas:

- Region 2 – San Francisco Bay (172,683 acres), Sacramento-San Joaquin Delta (41,736 acres), and San Pablo Bay (68,349 acres);
- Region 4 – Coastal water bodies in the Los Angeles area (many miles and acres, most notably Santa Monica Bay [146,645 acres]), lakes (411 acres), and 1.9 miles of the Oxnard Drain; and
- Region 9 – San Diego Bay shoreline (5.5 acres).

The majority of water bodies on the list are bays and estuaries in highly urbanized areas. San Francisco Bay is listed for chlordane due to an interim fish consumption advisory that was developed by the California Office of Environmental Health Hazard Assessment (OEHHA) in 1994. The advisory is based on a 1994 pilot study (SFRWQCB 1995), which indicated that the legacy pesticides and other chemicals were present at concentrations of potential concern. The interim advisory currently remains in place for chlordane.

Recent Monitoring Data

Recent sport fish monitoring data (1998 – 2003) indicate that chlordane concentrations in all areas (n = 238) of the state were in the green < 300 ppb category (Table 3.5.2 and Figure 3.5.21). Current fish consumption advisories and 303d listed water bodies due to chlordane, therefore, appear conservative compared to the current concentrations indicated from recent data.

b. Long-term Trends in Impact of Chlordanes on Fishing in California

Management Actions

The primary use of chlordane was in agriculture and structural termite control. The most significant management action to reduce chlordane was the federal restriction on its use that began in 1978. By 1983, EPA had imposed a ban on chlordane throughout the United States for all but underground termite control. Finally, chlordane sales were prohibited in the US after April 1988, though existing stocks were permitted for use by homeowners (USEPA 1995). Due to its widespread use throughout the US, chlordane contamination has been found in both urban and agricultural areas. The expectation is that chlordane is currently declining gradually across the state, presumably in large part due to the ban. Specific management actions to clean up historical residues of chlordane were not found in the review conducted for this report.

Long-term Trends

Sport Fish

Concentrations in sport fish across the state indicate that chlordanes have not been as persistent as other legacy pesticides over the past 30 years. Concentrations over time (Figures 3.5.22, 3.5.23) indicate that sport fish chlordane concentrations were higher at a small number of locations prior to 1998 (Figure 3.5.21). From 1988 – 1997, two (1%) sites were in the yellow 300 – 1400 ppb category, and one (< 1%) site was in the orange 1400 – 2400 ppb category. Similarly, from 1978 – 1987, two (2%) sites were yellow and one (1%) site was orange. No locations were above 2400 ppb in either time interval. Two of the three yellow and orange locations monitored from 1988 – 1997 were near Oxnard (Oxnard Drainage Ditch and Oxnard Drain). These data also indicate that chlordane at Machado Lake (formerly, Harbor Park Lake) has improved. This location was categorized as orange using data from 1978 – 1987, then yellow from 1988 – 1997, and finally green in the most recent data (1998 – 2003).

Long term-trend monitoring in sport fish also suggests declining chlordane concentrations. Of ten sites examined (Figure 3.5.24), four indicate a significant decline ($p < 0.05$) over time. Historically elevated concentrations in southern California were evident for red shiner in San Diego Creek at Michelson Drive, and channel catfish in New River at Westmorland and Alamo River at Calipatria. However, these locations are three of the four sites exhibiting significant declines. The improved status of Harbor Park Lake is also indicated, having the most dramatic decline of all locations examined. Differing from DDT and dieldrin, management action to reduce chlordane appears to have been particularly effective in southern locations of the state.

Bivalves

Long-term monitoring of bivalves by the State Mussel Watch program has provided further evidence for declining chlordane concentrations. Of ten sites examined, seven show a significant long-term decline ($p < 0.05$). Similar to sport fish, locations in the northern regions of the state demonstrate historically lower concentrations compared to sites in southern California (Figure 3.5.25). Bivalve locations in southern California (e.g., Huntington Harbor sites and Newport Bay at Crows Nest) indicate a significant and dramatic decline in chlordane concentrations throughout the 20 years of bivalve monitoring. Management action to reduce chlordane in California has been successful at nearly all sites examined across the state.

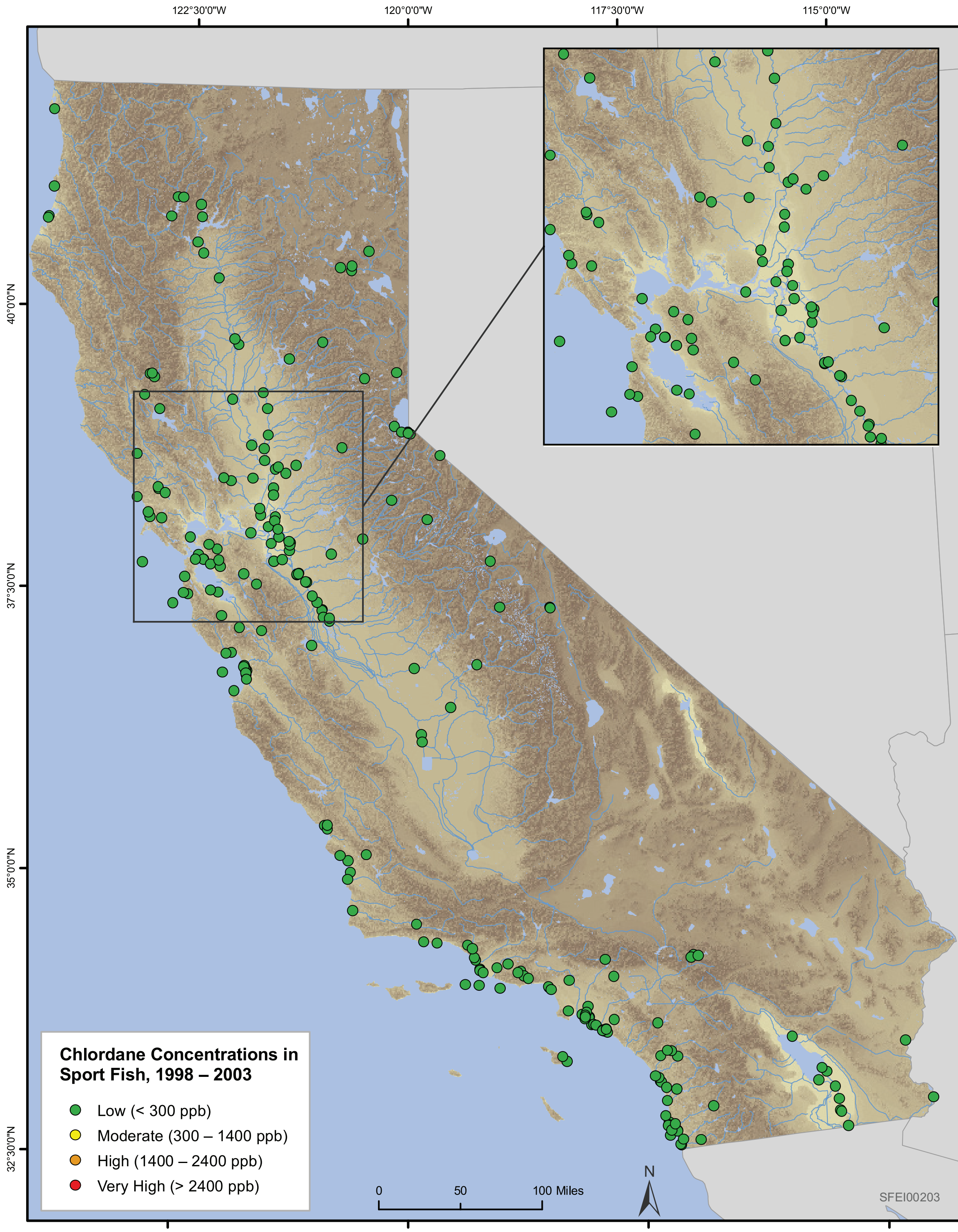

Figure 3.5.21. Chlordane concentrations in California sport fish, 1998 – 2003. Based on chlordane measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

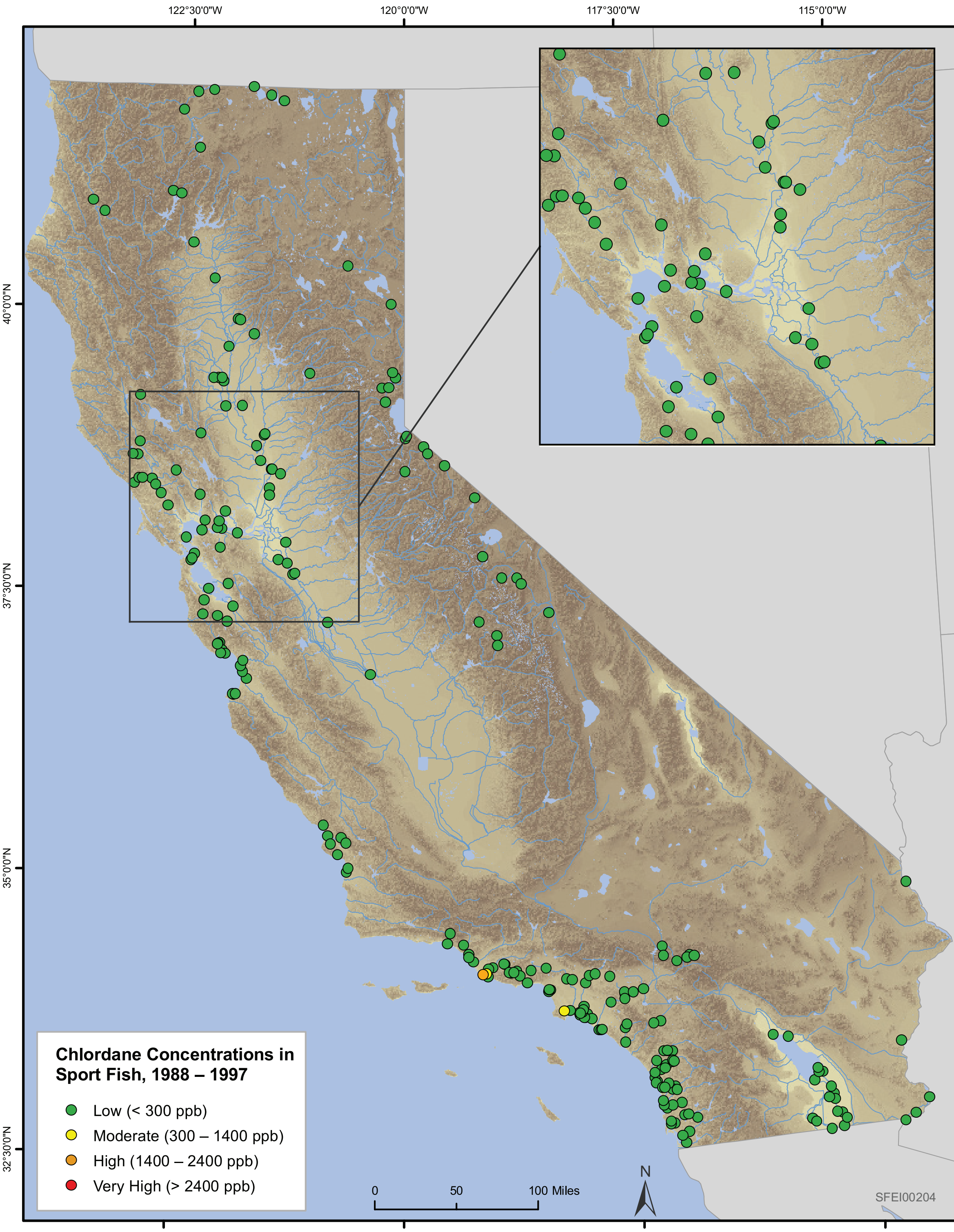

Figure 3.5.22. Chlordane concentrations in California sport fish, 1988 – 1997. Based on chlordane measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

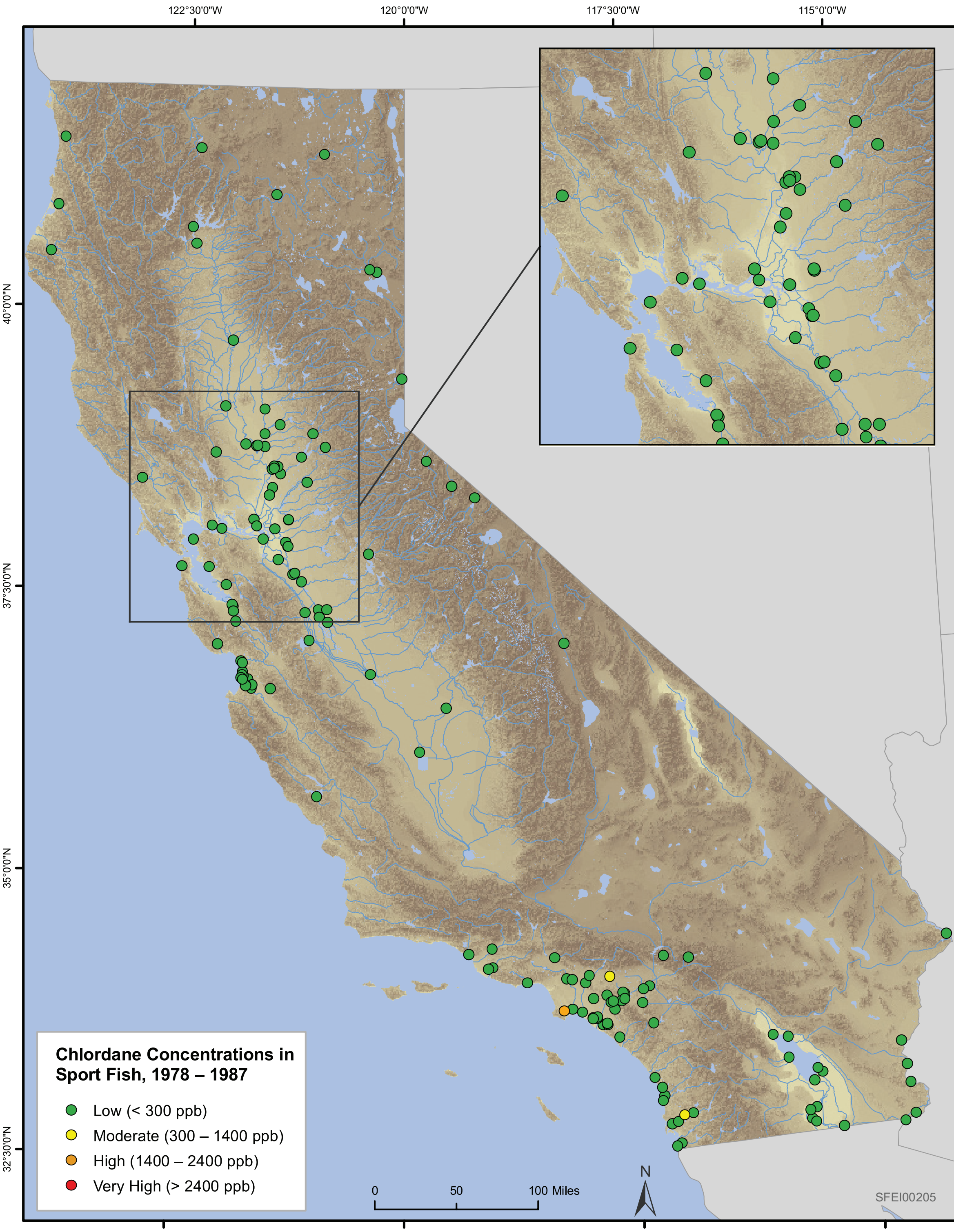

Figure 3.5.23. Chlordane concentrations in California sport fish, 1978 – 1987. Based on chlordane measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

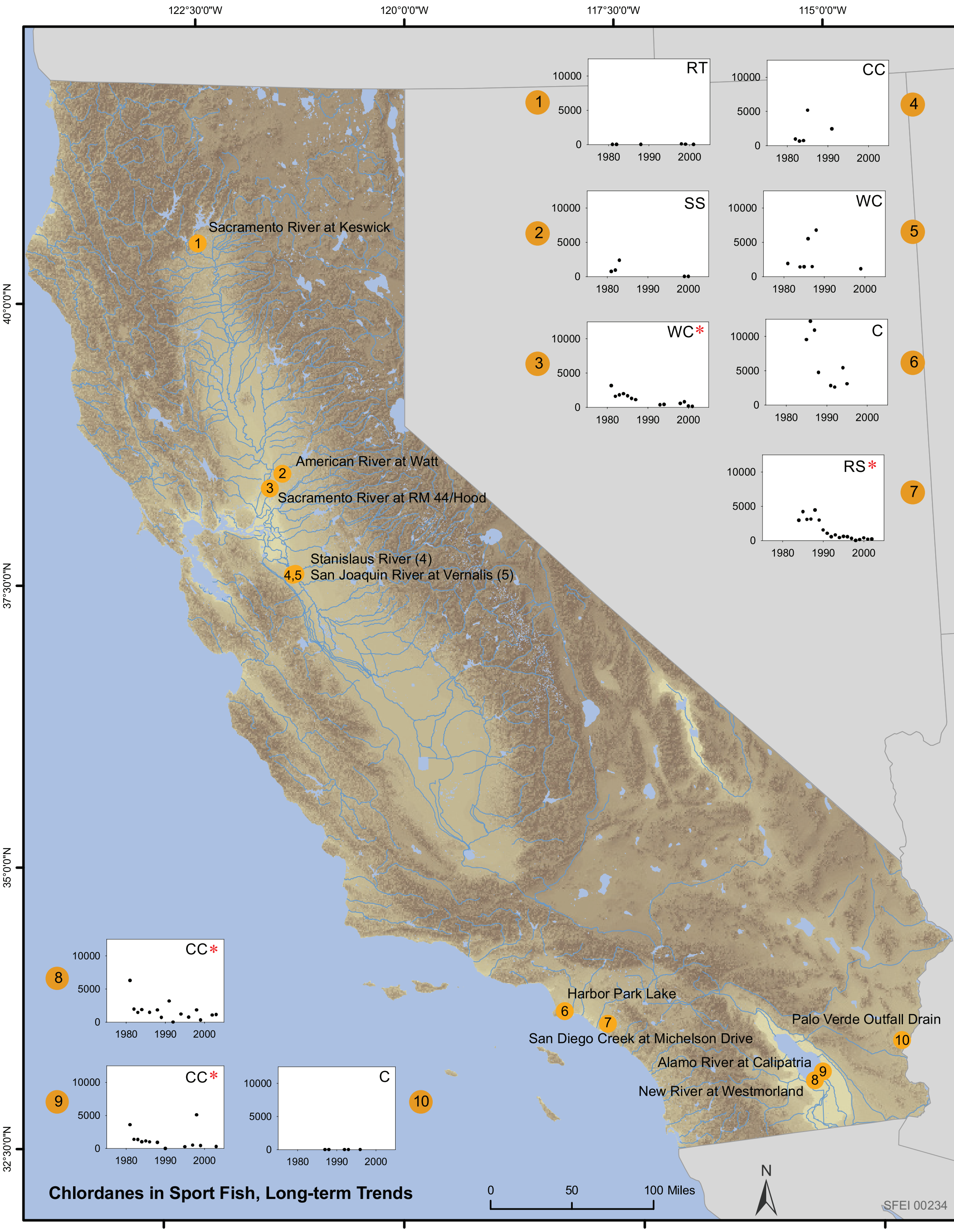

Figure 3.5.24. Long-term trends in chlordane concentrations in California sport fish. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight. Species shown are rainbow trout (RT), Sacramento sucker (SS), white catfish (WC), channel catfish (CC), red shiner (RS), and common carp (C).

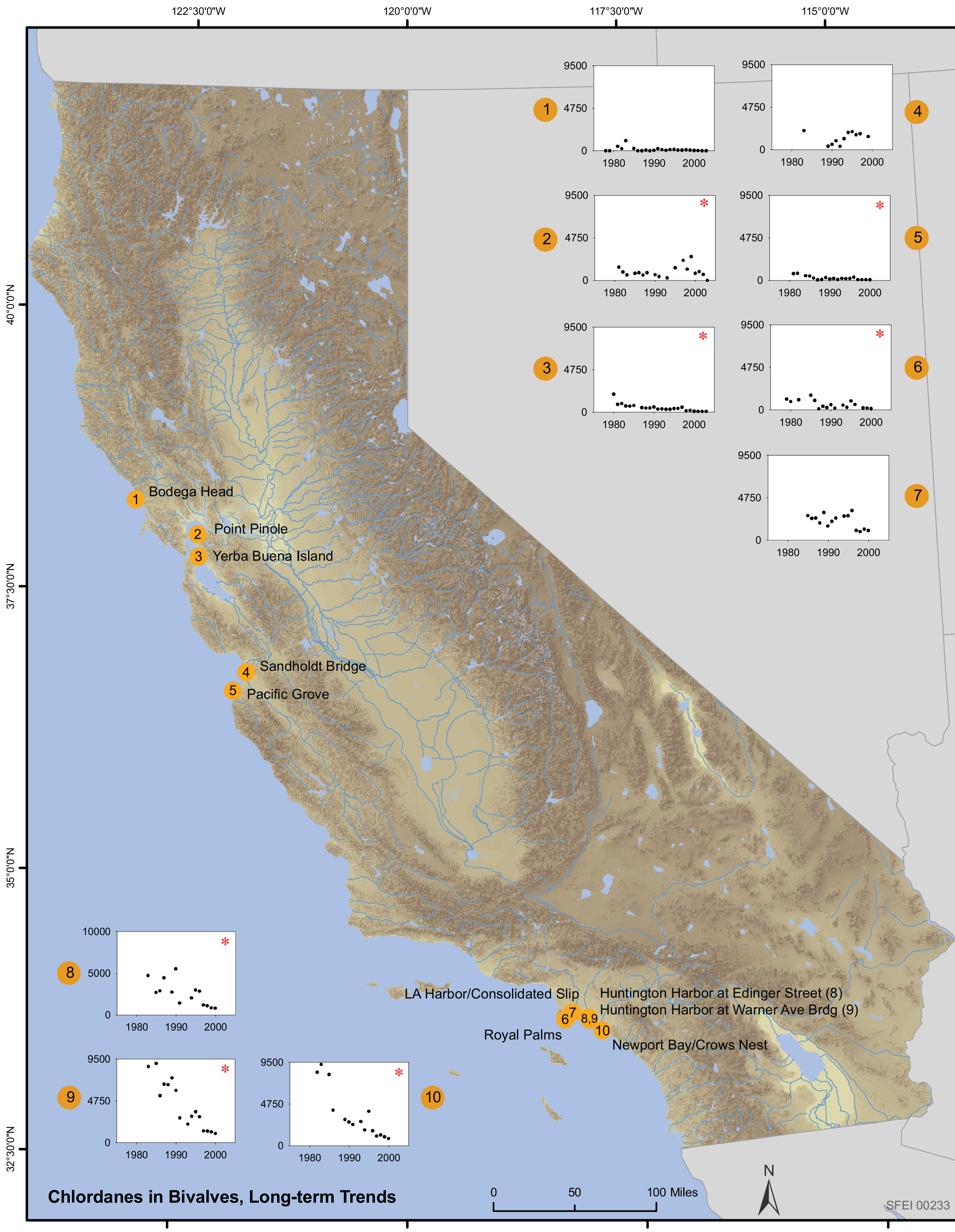

Figure 3.5.25. Long-term trends in chlordane concentrations in California mussels measured by the State Mussel Watch Program. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight.

Case Studies

Local monitoring studies of sport fish and bivalves have provided sufficient evidence for the widespread decline of chlordanes in aquatic food webs. RMP monitoring has shown that chlordane concentrations in San Francisco Bay sport fish have been consistently low. Comparison of data to the concentration categories applied in the report showed that all samples collected in 1997, 2000, and 2003 were below 25 ppb. In addition, statistical analysis of chlordane concentrations (lipid wt) detected significant declines in leopard shark, shiner surfperch, and white croaker from 1994 – 2003 (Figure 3.5.26; RMP 2006).

Monitoring of bivalves by the State Mussel Watch Program (SMW) and RMP in San Francisco Bay has generated evidence for long-term trends in chlordane. cis-chlordane is commonly found in highest abundance compared to other chlordane isomers (Dearth and Hites 1991, Gunther et al. 1999). Concentrations in San Francisco Bay were highest in 1980, but unlike other legacy pesticides, have shown a gradual decline until as recently as 1991 (Gunther et al. 1999). This trend was evident at many (50%) of the SMW stations analyzed by Stephenson et al. (1995). Since 1991, chlordane has remained relatively constant (around 100 ppb, lipid wt) at many locations across the state. These data suggest concentrations have declined as a result of the usage ban on chlordane.

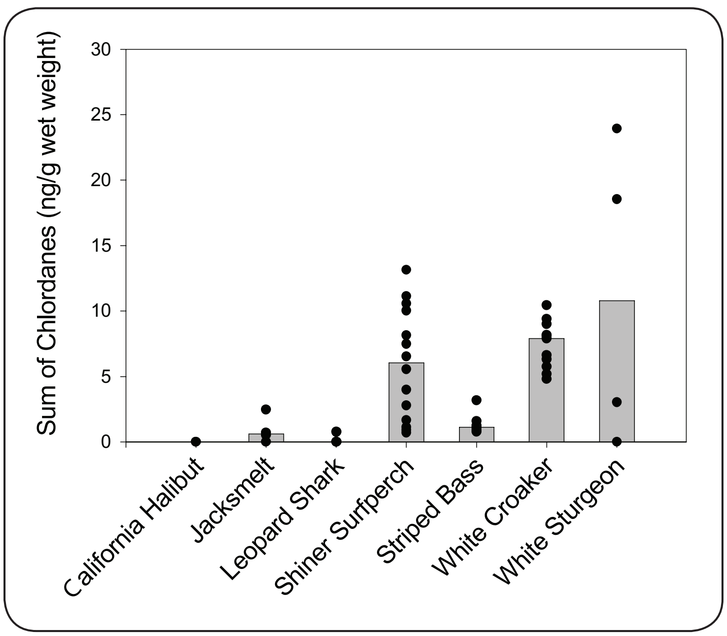

Figure 3.5.26. Chlordane concentrations (ng/g) in Bay fish, 2003. Points are concentrations in each composite sample analyzed. Bars indicate median concentrations. From RMP (2006).

c. Sources and Pathways

Chlordane concentrations in the recent dataset suggest that few locations have present sources of the pollutant (Figure 3.5.27). Only a few sites, located in San Francisco Bay (San Pablo Reservoir and Lake Chabot) and Los Angeles County had higher concentrations relative to the rest of the state. The highest chlordane concentration from 1998 – 2003 was shown at the urbanized Harbor Park Lake (213 ppb). However, in general, there do not currently appear to be sources to aquatic ecosystems that have resulted in elevated concentrations among sport fish collected from 1998 – 2003.

Historic data suggest that sources of chlordane were more widespread previously (Figure 3.5.28, 3.5.29). Elevated concentrations are indicated for agricultural (San Francisco Bay, Sacramento-San Joaquin Delta, and Oxnard) and urban (Los Angeles) areas. The highest chlordane concentration in each historic time interval was 1842 ppb at Oxnard Drain (1988 – 1997) and 2090 ppb at Harbor Park Lake in Los Angeles (1978 – 1987). In general, DDT sources had the largest impact (abundance of tall concentration bars) on sport fish in 1978 – 1987, compared to other

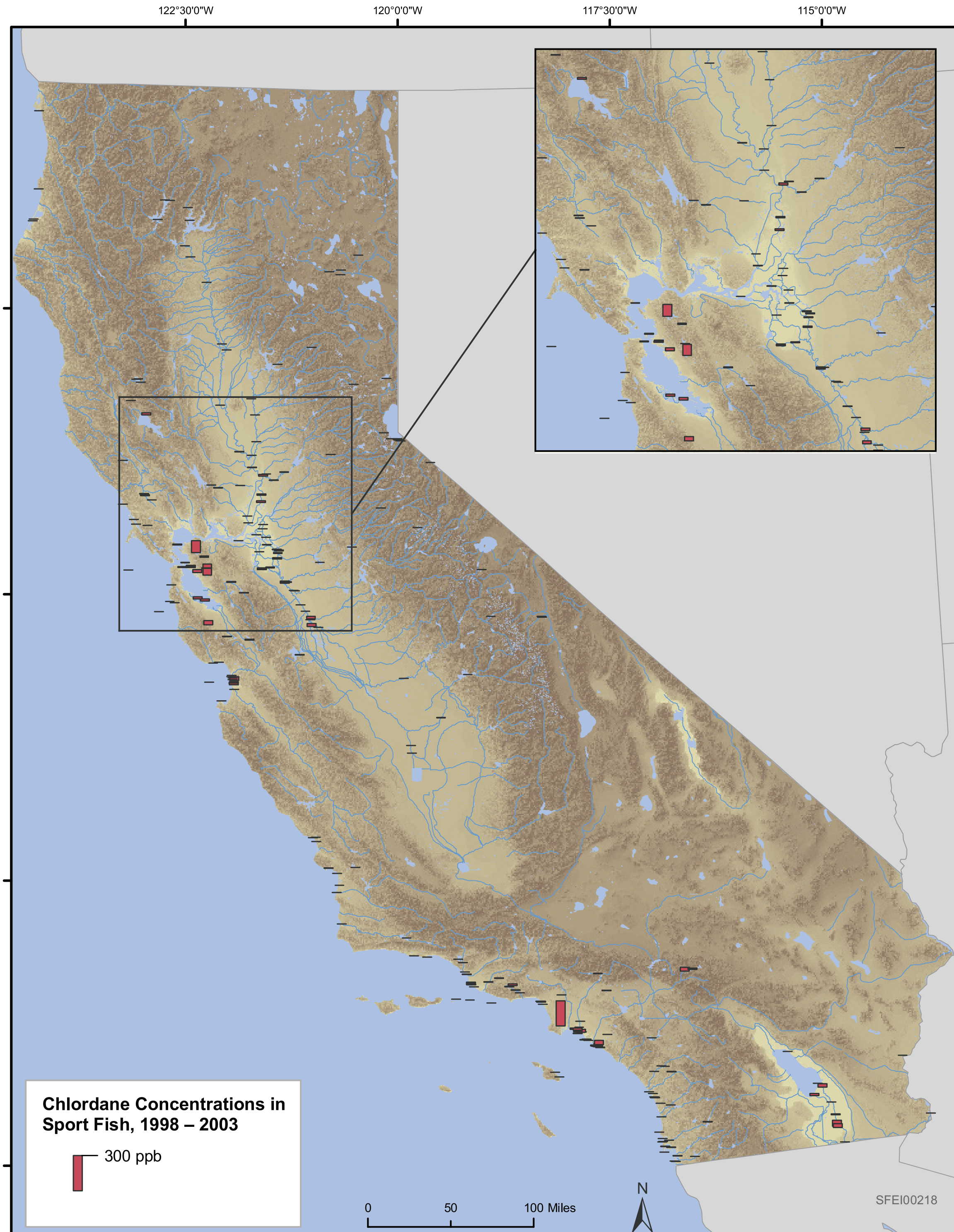

Figure 3.5.27. Chlordane concentrations in California sport fish, 1998 – 2003. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

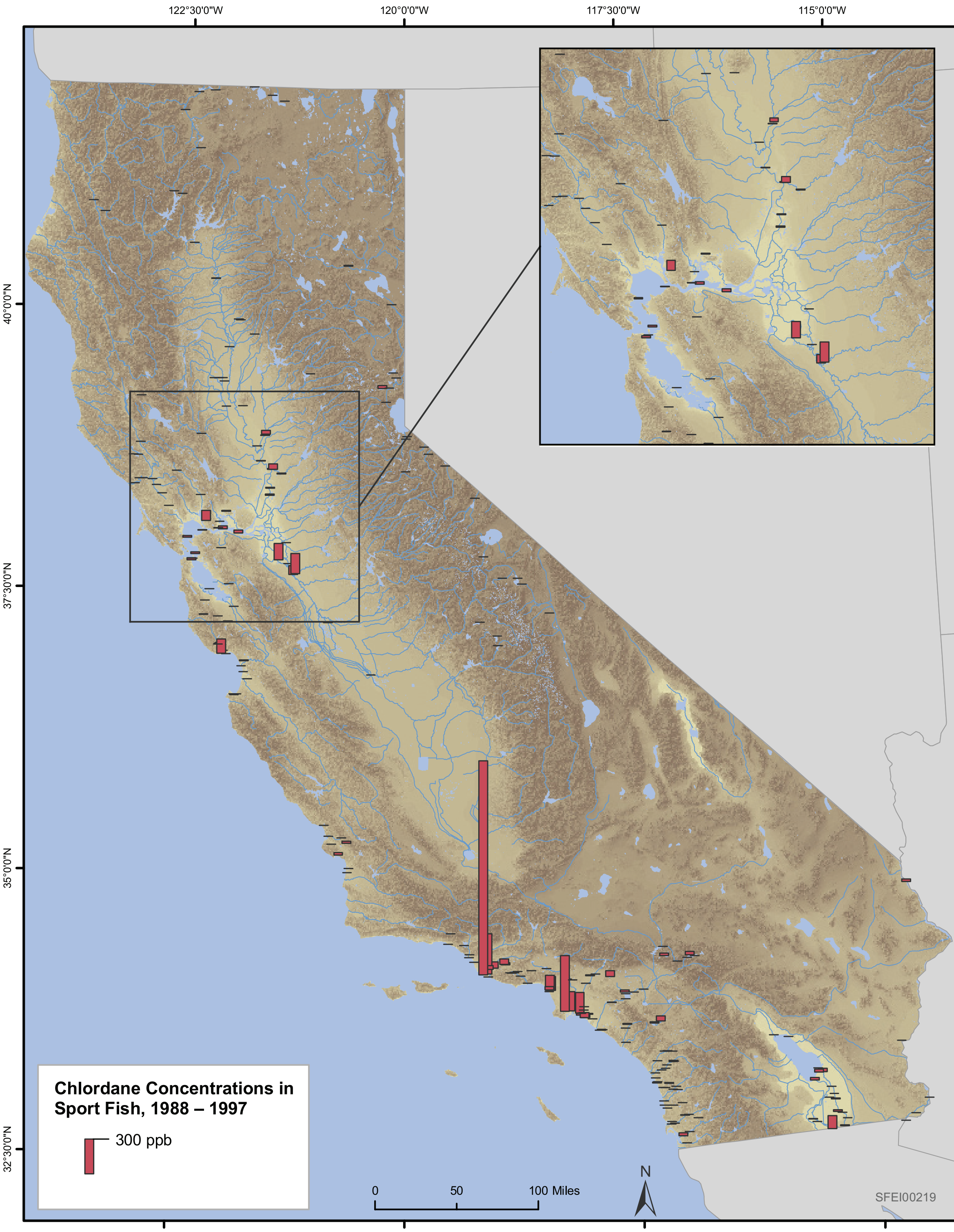

Figure 3.5.28. Chlordane concentrations in California sport fish, 1988 – 1997. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

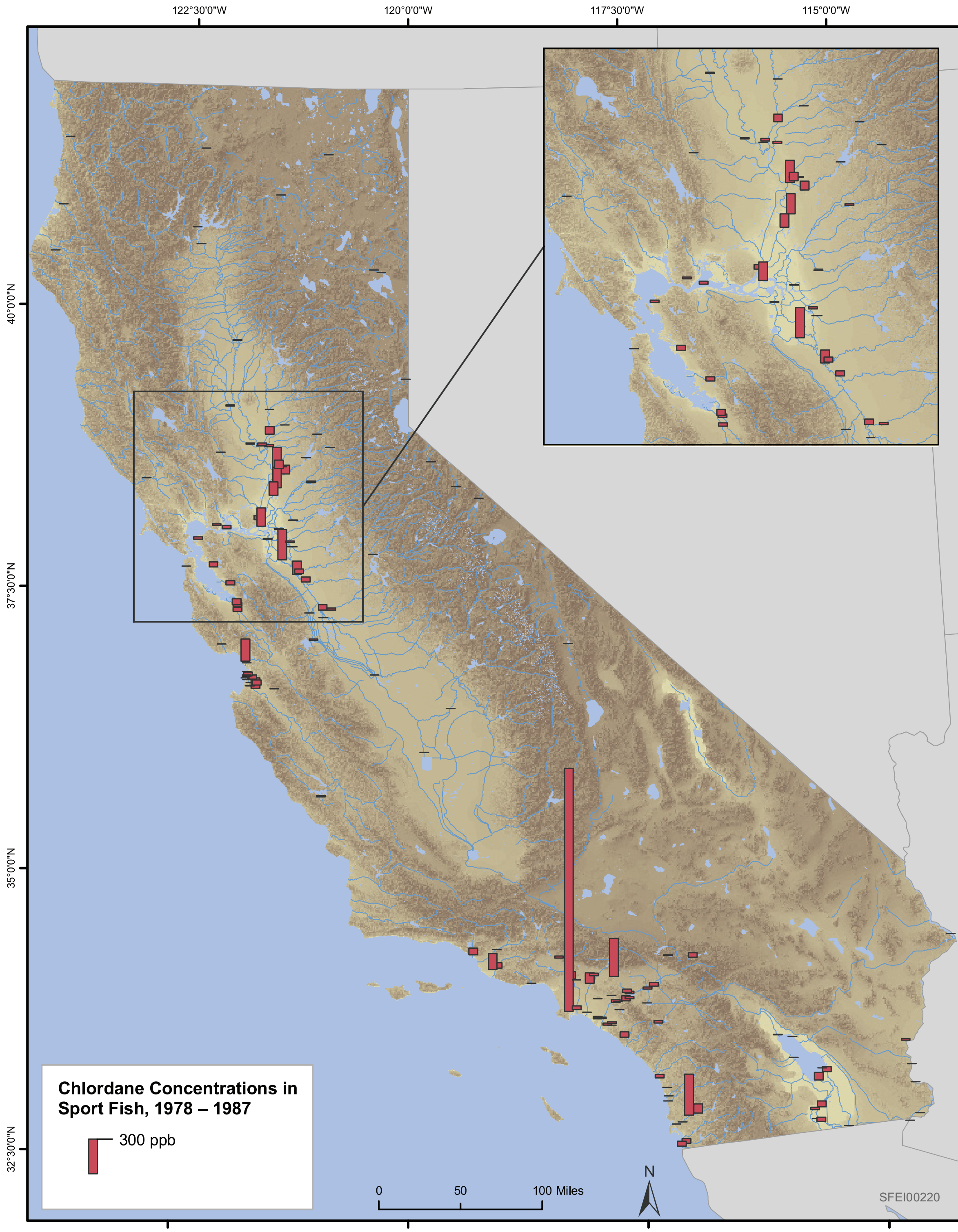

Figure 3.5.29. Chlordane concentrations in California sport fish, 1978 – 1987. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

time intervals. These results indicate a reduction of chlordane in California waters, likely as a result of successful management actions. Chlordane concentrations are not significantly elevated at present, and are greatly reduced from levels measured previously. The pathways for historic chlordane contamination at these locations are likely from urban and agricultural uses.

3.5.9. Impact of Chlordanes on Aquatic Life in California

a. Impacts on Aquatic Birds

The impact of chlordanes on aquatic life mirrors the discussion for dieldrin. Canadian tissue guidelines have not been developed for chlordane as they have been for DDT, though a relatively old screening level of 50 ppb (wet wt) is still used in some of the recent literature (NAS 1974). The lack of a newer screening level may be attributable to the relatively low concentrations of chlordanes in the aquatic life species where it has been measured. Chlordanes have generally been found at concentrations in bird eggs that were either undetectable or well below concentrations that would induce toxic effects (Hothem et al. 1995, Rattner et al. 1996). Night heron and great egret eggs collected in the 1980s from Imperial Valley (Salton Sea), San Francisco Bay and the Sacramento-San Joaquin River Delta, contained chlordanes that were barely detectable (Ohlendorf and Marois 1990). Low concentrations were found in both double-crested cormorants and song sparrows used to investigate bioaccumulation in the open water and marsh habitats of San Pablo Bay (Davis et al. 2004a), and in California least terns of San Francisco and San Diego Bays (Hothem and Zador 1995). More recently, chlordanes in San Francisco Bay clapper rails were at higher concentrations than dieldrin, yet neither was thought to have an impact on reproductive success (Schwarzbach 2001).

b. Impacts on Fish and Marine Mammals

Monitoring of chlordanes in forage fish species has not been widely performed. Monitoring by SCCWRP in the Southern California Bight and Newport Bay are the only local studies that the authors are aware (Allen et al. 2002a, Allen et al. 2004a). In the Bight, Total chlordane (sum of alpha- and gamma-chlordane) was analyzed in 275 flatfish composites from 225 stations. However, only 22 samples (8%) had detectable concentrations, which ranged from 0 to 15 ppb (wet wt), and were well below the National Academy of Sciences (NAS) threshold of 50 ppb (NAS 1974). Similarly, in Newport Bay forage fish (Allen et al. 2004a), chlordane ranged from non-detectable to 22 ppb (wet wt). The highest values in the study (maximum concentration of 14.6 ppb) were found near the LA County outfall. All composites of the nine species sampled were below the screening level of 50 ppb (NAS 1974) for wildlife fish consumption.

Chlordanes in marine mammals have generally been recorded at concentrations an order of magnitude lower than DDTs. Concentrations in the liver tissue of California sea otters has been shown to range from 14 – 310 ppb (wet wt), with individuals from Monterey Harbor having the highest concentrations (Nakata et al. 1998). All seven locations in California examined for chlordanes from 1992 – 1996 had concentrations that were secondary to that of DDT and PCBs. No sea otter deaths examined in that study were attributed to effects from chlordanes. Concentrations were also very low in California sea otters examined by Bacon et al. (1999) and Kannan et al. (2004). The ban on chlordane use in California appears to have eliminated any possibility of effects to aquatic life due to this contaminant.

3.5.10. Chlordanes Summary

Recent sport fish monitoring data (1998 – 2003) indicate that chlordane concentrations in all areas of the state were in the low < 300 ppb category. Long-term monitoring in sport fish and bivalves indicated dramatic declines in chlordane immediately after the ban, particularly at southern California locations. In bivalves, the declining trend was evident into the early 1990s, representing a longer period than has been reported for other legacy pesticides. A review of chlordane impacts to aquatic life species suggested that adverse affects should not be expected. Overall, concentrations across the state indicate that chlordane has not been as persistent as other legacy pesticides over the past 30 years. The chlordane ban has been quite effective in reducing concentrations across the state. Agricultural pollution was the most likely historical source of chlordanes in the environment.

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