SWAMP
Surface Water Ambient Monitoring Program

# BIOACCUMULATION OF POLLUTANTS IN CALIFORNIA WATERS: A REVIEW OF HISTORIC DATA AND ASSESSMENT OF IMPACTS ON FISHING AND AQUATIC LIFE


**J. A. Davis**
**J. L. Grenier**
**A. R. Melwani**
**S. N. Bezalel**
**E. M. Letteney**
**E. J. Zhang**
**M. Odaya**

**San Francisco Estuary Institute**


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## DEDICATION

This report is dedicated to the memory of Del Rasmussen, who passed away in February 2008. Del worked for the State Water Resources Control Board for more than 25 years. Del managed the State Mussel Watch Program, Toxic Substances Monitoring Program, and Coastal Fish Contamination Program. These programs generated most of the information summarized in this report, and represent one of the world's best datasets on bioaccumulation. The programs sampled hundreds of sites across the state and identified many cases of severe contamination, leading to cleanup actions and fish advisories to protect humans and wildlife. They also documented the successful management of many pollutants that had posed serious threats to wildlife and human health in the 1970s and 1980s. Del also contributed to the early development of the Surface Water Ambient Monitoring Program. The wealth of information generated by the programs under Del's stewardship provides an essential foundation for future monitoring and continuing efforts to improve the health of California's coast, estuaries, lakes, rivers, and streams.

Del Rasmussen

## ACKNOWLEDGEMENTS

Preparation of this review was a formidable task, and thanks are due to many people for their direct or indirect contributions to this document.

Many colleagues graciously provided data that formed the basis for this report.  The primary contacts are listed in Tables 2.1 and 2.2 of the report.  Thanks are also owed to many others not listed here that helped direct us to these sources of information.

A draft of this document was reviewed and improved by comments received from Rainer Hoenicke (San Francisco Estuary Institute), Bob Brodberg (California Office of Environmental Health Hazard Assessment), Pavlova Vitale (Santa Ana Regional Water Quality Control Board), Kathy Rose (Santa Ana Regional Water Quality Control Board), Tom Kimball (State Water Resources Control Board), Robert Holmes (Central Valley Regional Water Quality Control Board), Jim Wiener (University of Wisconsin-LaCrosse), Ross Norstrom (Carleton University, Ottawa, Canada), and Chris Schmitt (U.S. Geological Survey, Columbia, Missouri).

Valerie Connor and Emilie Reyes of the State Water Resources Control Board guided the project as SWAMP Unit Chiefs.

Rusty Fairey and Max Puckett of Moss Landing Marine Lab managed the subcontract with SFEI to perform this work.

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Jay Davis would like to note his appreciation for the extraordinary patience and support of his wife, Nicole David, during the production of this report.

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# EXECUTIVE SUMMARY E

This report was written for the State Water Resources Control Board's Surface Water Ambient Monitoring Program (SWAMP) as a step toward the development of an improved bioaccumulation monitoring program for California. The report provides a review of bioaccumulation monitoring data generated under three historic State Board programs (the Toxic Substances Monitoring Program, the State Mussel Watch Program, and the Coastal Fish Contamination Program) and other major bioaccumulation studies since 1970. Future monitoring will be guided by assessment questions developed for the SWAMP. The objective of this report was to evaluate how well the historic data from the State Board programs and from other major monitoring efforts since 1970 address these questions. This exercise has provided a substantial amount of information about present and historical impacts of pollutant bioaccumulation on fishing and aquatic life in California, and has also highlighted areas where improved sampling approaches can better address the assessment questions.

## NET IMPACT OF POLLUTANTS ON FISHING

Present concentrations of pollutants in many California water bodies are high enough to cause concern for possible effects on human health and to have a significant impact on the fishing beneficial use. Consumption advisories, 303(d) listings, and the bioaccumulation database as a whole provide three indices of the status of this impact. Consumption advisories exist for an increasing number of water bodies, but these represent only a fraction of the areas likely to need them. Lack of suitable data is a major impediment to developing advice for additional water bodies. A USEPA evaluation of the 2002 303(d) List indicated that large portions of the state had not been assessed, especially rivers and coastline. Most of the lake area in the state (61%) had been assessed, and a relatively small percentage of the total area (6%) was classified as impaired. Assessment of lakes, however, has focused primarily on the largest lakes, leaving the vast majority of smaller lakes unsampled. Many of these small lakes are near population centers and are popular for fishing. Bays and estuaries had been thoroughly assessed (98% of the area) and 93% of the total area was impaired.

Evaluation of the most recent monitoring data (collected from 1998 – 2003) indicates that, for the locations sampled, 32% had low concentrations of pollutants, 42% had moderate concentrations, 18% had high concentrations, and 8% had very high concentrations (Figure 1). Concentrations in the low category are in a range where consumption is generally encouraged by the California Office of Environmental Health Hazard Assessment (OEHHA) (Klasing and Brodberg 2006). OEHHA is the agency responsible for managing health risks due to contaminated sport fish in California. Concentrations in the very high category are in a range where OEHHA discourages consumption (Klasing and Brodberg 2006). Lakes assigned to the moderate, high, or very high concentration categories were primarily affected by mercury, with PCBs also playing a lesser role.

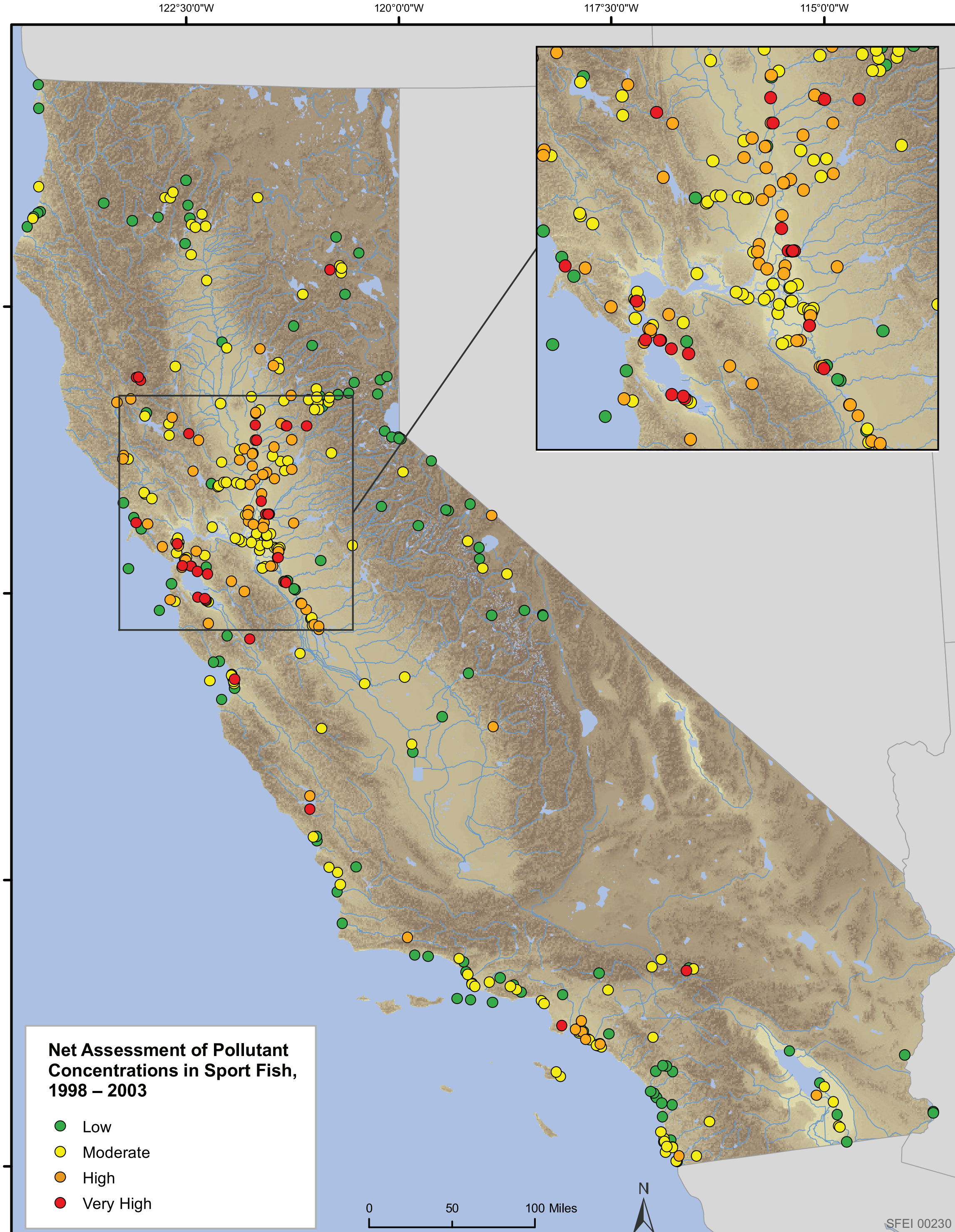



Figure 1. Net assessment of pollutant concentrations in California sport fish, 1998 – 2003. Based on measurements of several chemicals (mercury, PCBs, DDTs, dieldrin, and chlordanes) in muscle tissue from a variety of fish species. Size limits were applied for evaluation of mercury data (Appendix 1). Dots represent sampling locations. Dot colors correspond to degrees of contamination (low, moderate, high, very high) defined for each pollutant and represent the species with the highest degree of contamination at each location.

# IMPACTS OF SPECIFIC POLLUTANTS ON FISHING AND AQUATIC LIFE

## Mercury

Mercury contamination is common in California aquatic food webs, affecting both the fishing and aquatic life beneficial uses in many areas of the state, with long-term trends indicating little change over the past few decades. Large regions of the state contain fish with moderate, high, or very high concentrations of mercury. Twenty-three of the 294 locations (8%) sampled from 1998 – 2003 had a species with a median mercury concentration above 0.9 ppm, placing these sites in the very high category. Another 68% of the locations sampled from 1998 – 2003 had mercury concentrations in the moderate and high categories. Only 24% of the locations had concentrations in the low category (Figure 2). The number of locations with high or very high concentrations was greatest in the San Francisco Bay-Delta, Central Valley, and surrounding areas. The few good time series available for mercury in sport fish showed no clear trends over the past three decades. Thus, the available evidence supports the hypothesis that the mercury problem may take decades to be resolved. TMDL implementation actions, mine clean-ups, and consumption advisories are important management actions that may improve the situation over different time-scales. Large-scale wetland restoration has the potential to exacerbate the mercury problem by increasing production of methylmercury, the most toxic and readily accumulated form. In the region with the most data regarding impacts on aquatic life, the San Francisco Bay-Delta, impacts on wildlife populations, including endangered species, from mercury contamination appear likely.

## PCBs

Polychlorinated biphenyl (PCB) bioaccumulation in aquatic food webs in California has declined significantly since production was banned in the 1970s, but this persistent pollutant continues to have a negative impact on fishing and aquatic life in many parts of the state. Sport fish monitoring at 251 locations from 1998 – 2003 found that 4% of the locations had a species with median concentrations above 270 ppb, placing them in the very high concentration category (Figure 3). Thirty percent of the locations sampled had PCB concentrations in the moderate or high concentration categories. Most (66%) of the locations sampled had concentrations in the low category, with median concentrations for all species analyzed below 30 ppb. PCB concentrations in some areas also appear to be high enough to cause adverse impacts in wildlife. Concentrations are highest in water bodies near major urban centers, including the Bay Area, Sacramento, Los Angeles, and San Diego. PCB concentrations in San Francisco Bay are particularly high and appear to be unusually persistent. In general, PCB concentrations are steadily declining across the state (Figure 4). The 1979 ban on PCB sale and production and other regulations relating to disposal of PCBs appear to have generally been effective at reducing the impact of PCBs in California water bodies. In some locations, however, particularly San Francisco Bay, recovery from PCB contamination may take many decades.

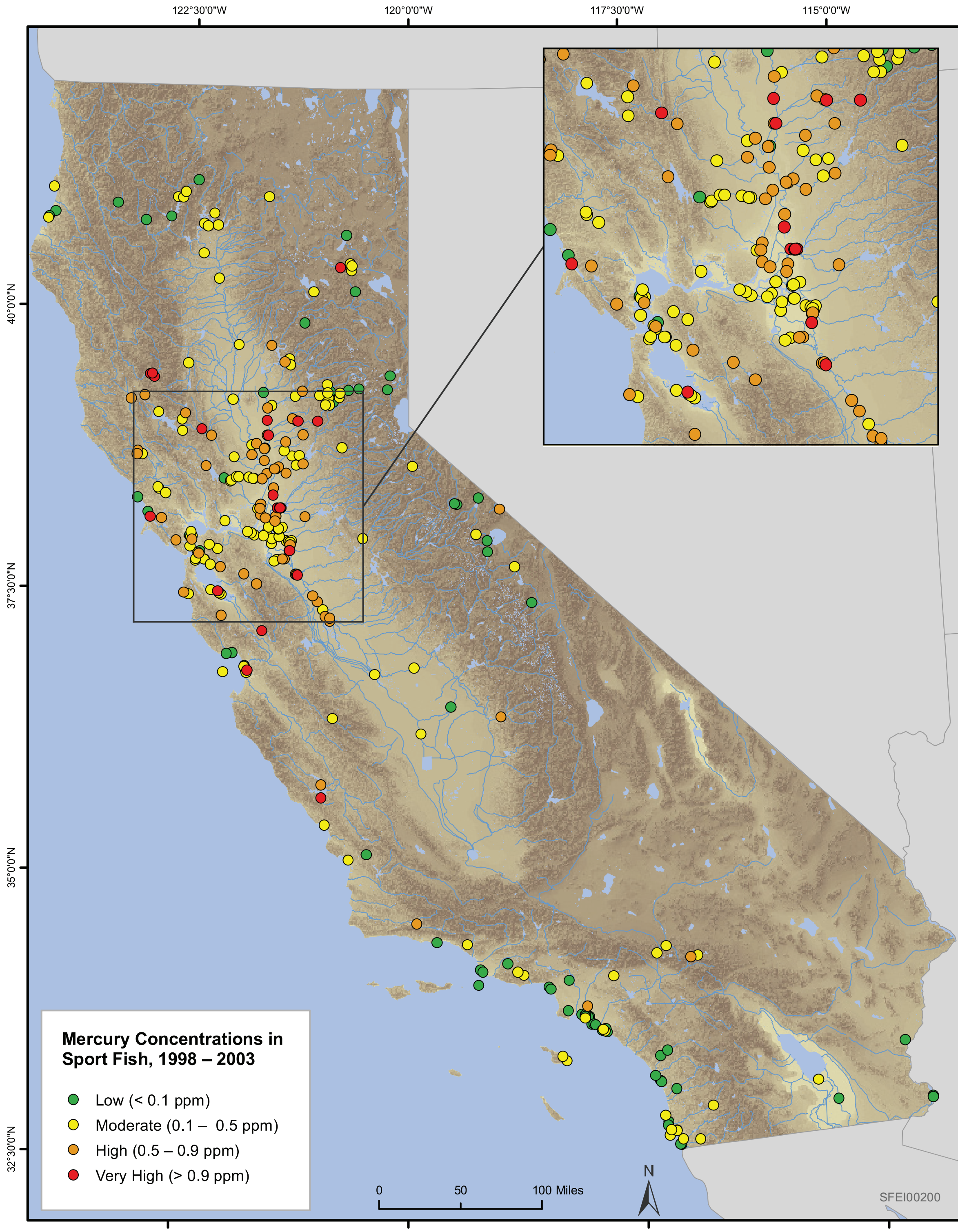



Figure 2. Mercury concentrations in California sport fish, 1998 – 2003. Based on mercury measurements (ppm wet wt) in muscle tissue from a variety of fish species. Size limits for each species were applied (Appendix 1). Dots represent sampling locations. Dot colors are based on the species with the highest median concentration at a location.

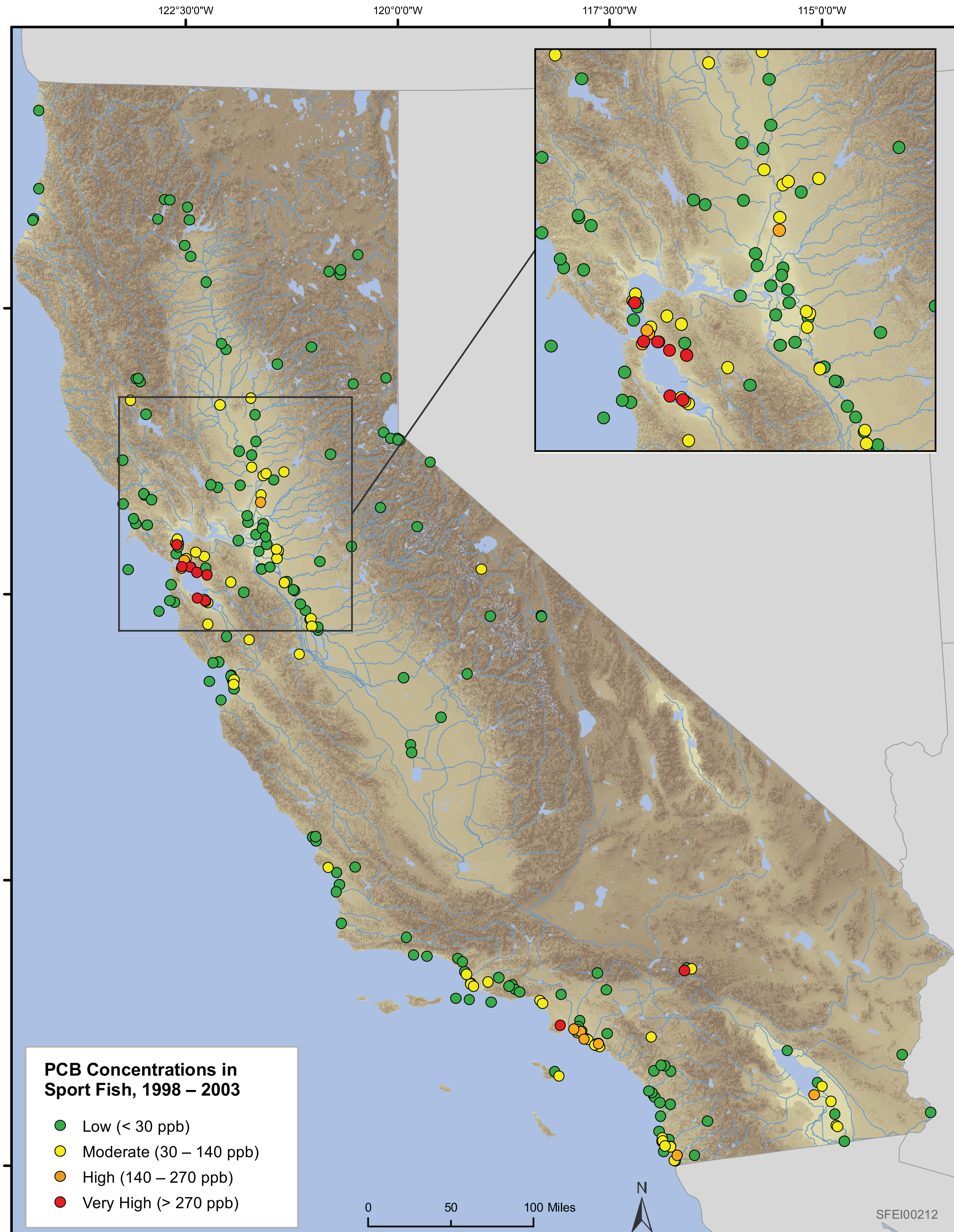



Figure 3. PCB concentrations in California sport fish, 1998 – 2003. Based on PCB measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

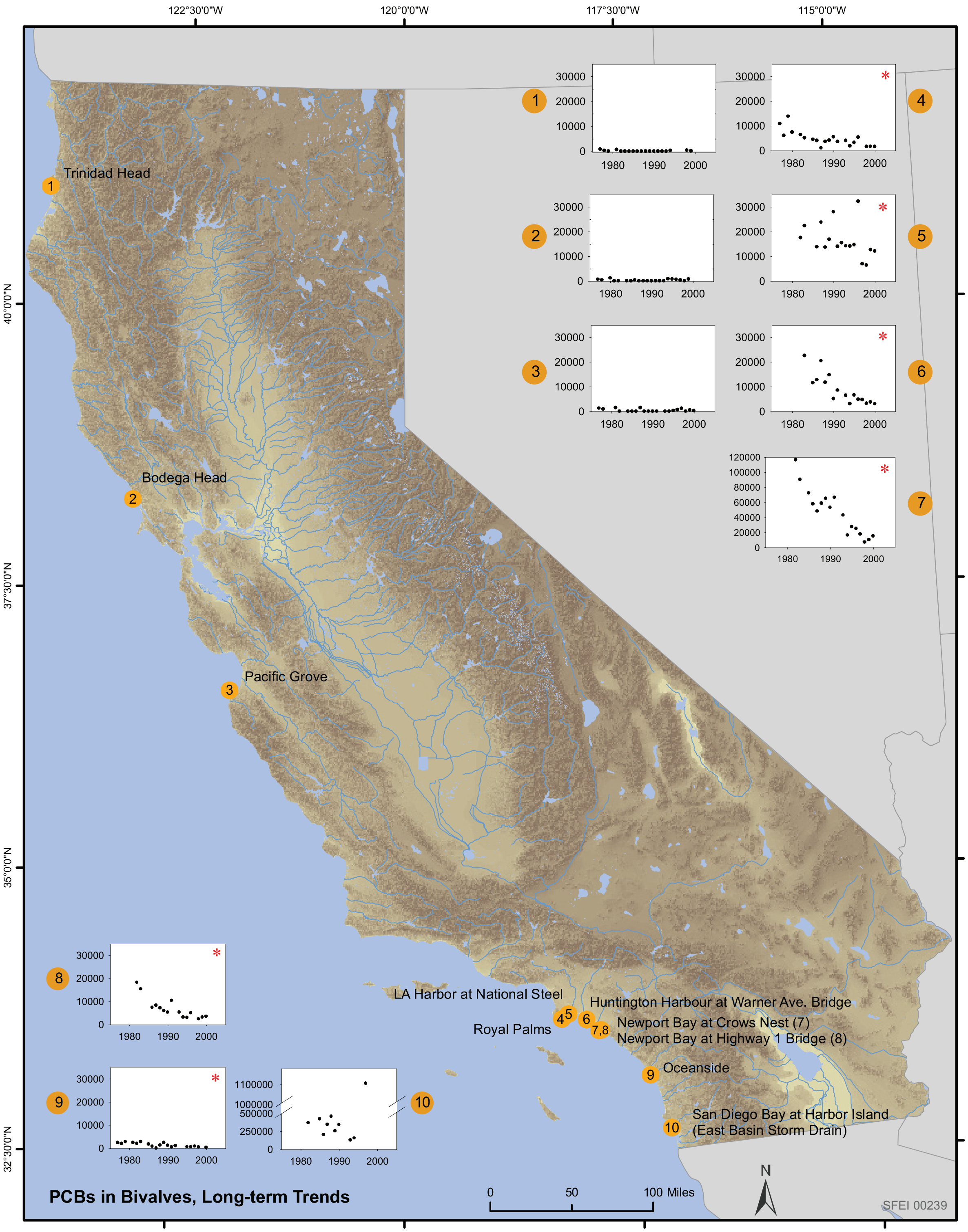



Figure 4. Long-term trends in PCB concentrations in California mussels measured by the State Mussel Watch Program. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight.

## DDT

Recent sport fish monitoring data (1998 – 2003) indicated that DDT concentrations in the vast majority of the state (248 of 252 locations sampled) were in the low concentration category, and thus, are having little impact on fishing. Concentrations of DDT in aquatic food webs across the state have generally shown significant declines over the past 30 years in response to the use restrictions and federal ban in 1972. Prior to these management actions, DDT had severe impacts on populations of aquatic birds on the California coast, including brown pelicans and double-crested cormorants. These populations have rebounded in response to the decline in DDT contamination, though concentrations still remain above thresholds for concern in some cases. 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 this region. Agricultural and urban runoff were the primary historical sources to California water bodies.

## Dieldrin

Recent sport fish data indicated that dieldrin concentrations in most areas of the state (238 of 244 locations sampled) were in the low category and having little impact on fishing. Concentrations of dieldrin in aquatic food webs across the state have generally shown gradual declines over the past 30 years in response to use restrictions and the federal ban in 1987. Dieldrin concentrations in food webs have also generally been below thresholds for concern for impacts on aquatic life. Long-term trend monitoring in sport fish from the Imperial Valley (Salton Sea) region indicates only a recent decline. Overall, the dieldrin ban has been successful in reducing concentrations and impacts across the state, with locations of higher historical contamination improving more recently. Agricultural runoff into California water bodies has been the primary historical source of this pollutant.

## Chlordane

Chlordane concentrations in all areas of the state (238 locations sampled) were low in recent sport fish sampling, and thus, not impacting fishing. Chlordane concentrations measured in food webs have also been below thresholds for concern for impacts on aquatic life. Chlordanes have not been as persistent as other legacy pesticides. Dramatic declines in chlordanes were evident immediately after the 1988 ban. Long-term trend monitoring in sport fish across the state also indicates declines in chlordane concentrations. The chlordane ban has been quite effective in reducing impacts of this insecticide. Agricultural and urban runoff were the most prominent pathways for transport into California water bodies.

## SUMMARY AND RECOMMENDATIONS

The State Board bioaccumulation monitoring programs documented the successful management of many pollutants that posed serious threats to wildlife and human health in the 1970s and 1980s. These programs were instituted just in time to document the rapid improvements in water quality that resulted from bans on PCBs and legacy pesticides, reductions in metals due to wastewater treatment, and other improvements. Many instances of severe contamination were identified, leading to cleanup actions and fish advisories to reduce exposure of humans and wildlife. These programs and other studies greatly advanced scientific understanding of bioaccumulation in California.

However, the dataset generated by the State Board bioaccumulation monitoring programs has several limitations with regard to answering the questions that are currently high priorities for water quality managers:

- many areas were not sampled adequately, including areas with significant fishing activity;
- the distribution of sampling locations varied over time;
- most of the sampling, though focused on sport fish, was not tailored to the development of consumption advice;
- the dataset was also not tailored to evaluation of risks to piscivorous wildlife through monitoring of prey species;
- long-term time series for detecting trends in sport fish or other wildlife contamination were lacking; and
- much of the sampling was biased toward characterization of polluted areas.

The evaluation performed in this report makes it evident that a sampling design that includes spatial randomization would be better suited to answering the SWAMP assessment questions related to statewide condition. Such a design would allow for an unbiased overall assessment of the condition of California water bodies. Indices of net impact during different time intervals would be directly comparable, since all areas would be sampled in a representative manner. A randomized design could be developed that samples different locations in proportion to the amount of fishing activity, an important feature with regard to development of consumption advice. A randomized design could also be augmented by other approaches, such as targeted sampling for long-term trends in particular locations or focused efforts to sample water bodies of particularly high interest. A combination of randomized and targeted sampling would provide an optimal approach for providing the information that water quality managers need from a bioaccumulation monitoring program in California.

# SECTION 1 INTRODUCTION

In the 1970s, the California State Water Resources Control Board (State Water Board) initiated two statewide monitoring programs employing the new technique of "bioaccumulation monitoring" – measuring the concentrations of pollutants in fish and bivalves residing in California water bodies. Bioaccumulation monitoring offers several advantages over monitoring of water or sediment, including:

- Measuring the degree to which pollutants are actually entering the food web, which for some pollutants can be quite different from the total concentrations present in water and sediment;
- Yielding a strong signal of contamination, since many pollutants reach concentrations that are much higher and easier to measure than concentrations in water and sediment;
- Providing an integrative measure of pollutant concentrations over time and a cost-effective tool for obtaining information on average concentrations; and
- Especially for fish, providing information that is directly linked to the impacts of pollutants on human and wildlife health.

The Toxic Substances Monitoring Program (TSMP), initiated in 1976, was a statewide program that employed a uniform approach for monitoring pollutants in fish and invertebrates in freshwater and estuarine habitats (SWRCB 1986, Rasmussen 1995, 1997). The TSMP primarily targeted water bodies with known or suspected water quality impairments, and successfully identified and documented many hotspots of contamination.

The State Mussel Watch Program (SMWP) was initiated in 1977 to provide information on long-term trends in water quality in coastal marine waters and to identify specific areas with elevated concentrations (Hayes et al. 1985, Hayes and Phillips 1986, Rasmussen 2000). Bivalves have some advantages compared to fish as indicator species: they are less mobile than fish and therefore good indicators of conditions at specific locations, and they can be transplanted into locations where bioaccumulation monitoring is desired.

Over the years, these two programs yielded a wealth of information on water quality in California. The chemical analyses were performed by top laboratories with excellent quality assurance and the data they generated are considered to be highly reliable. Hundreds of locations were sampled. Many instances of severe contamination were identified, leading to cleanup actions and fish advisories to reduce exposure of humans and wildlife. In addition, many areas with low concentrations (below past or present thresholds of concern) were identified. As described in this report, these programs have documented the successful management of many pollutants that posed serious threats to wildlife and human health in the 1970s and 1980s. These programs were instituted just in time to document the rapid improvements in water quality that resulted from bans on PCBs and legacy pesticides, reductions in metals due to wastewater treatment, and other improvements.

In 1998, a third statewide bioaccumulation monitoring program, the Coastal Fish Contamination Program (CFCP), was implemented (Gassel et al. 2002). This program was developed to assess the health risks of consumption of sport fish and shellfish from nearshore waters along the entire California coast. The CFCP was considered to be a critical component of a comprehensive coastal water quality protection program, and an important opportunity to build a long-term coastal monitoring database for water quality and contaminants in fish.

In 2000, the State Water Board, responding to a bill passed by the California legislature, developed a plan to restructure their existing water quality monitoring programs (including TSMP, SMWP, and CFCP) and create a Surface Water Ambient Monitoring Program (SWAMP) for water quality that addresses all hydrologic units of the state using consistent and objective monitoring, sampling and analytical methods; consistent data quality assurance protocols; and centralized data management (SWRCB 2000). Sampling under the three monitoring programs ended in 2003, as SWAMP began to take shape.

This report was written for the SWAMP as a step toward the development of an improved bioaccumulation monitoring program for California. This report provides a review of bioaccumulation monitoring data generated under the three State Board programs. Future monitoring will be guided by assessment questions developed for the SWAMP (Table 1.1). The objective of this review was to evaluate how well the historic data from the State Water Board programs and from other major monitoring efforts since 1970 address these questions. This exercise has provided a substantial amount of information about present and historical impacts of pollutant bioaccumulation on beneficial uses in California, and also highlights areas where different sampling approaches can better address the assessment questions of current interest.

**Table 1.1**
**Draft objectives and assessment questions for the SWAMP.**

| | | | |
|---|---|---|---|
| **FISHING BENEFICIAL USE SUPPORT** | | | |
| D.1. Determine the status of the fishing beneficial use throughout the state without bias to known impairment<br><br>D.1.1 What is the extent and location of water bodies not supporting any fishing beneficial use?<br><br>D.1.2 What is the extent and location of water bodies partially supporting the fishing beneficial use?<br><br>D.1.3 What is the extent and location of water bodies fully supporting the fishing beneficial use?<br><br>D.1.4 What is the proportion of water bodies in the state and each region falling within the three levels of support of the fishing beneficial use? | D.2. Assess trends in the fishing beneficial use throughout the state<br><br>D.2.1 Are water bodies improving or deteriorating with respect to the fishing beneficial use?<br><br>D.2.2 Have water bodies fully supporting the fishing beneficial use become impaired?<br><br>D.2.3 Has full support of the fishing beneficial use been restored to previously impaired water bodies? | D3. Evaluate sources and pathways of factors impacting the fishing beneficial use<br><br>D3.1 What is the relative importance of different pollutant sources and pathways in terms of impact on the fishing beneficial use on a regional and statewide basis? | D4. Evaluate effectiveness of management actions in improving the fishing beneficial use<br><br>D4.1 How is the fishing beneficial use affected by remediation, source control, or pollution prevention actions and policies regionally and statewide? |
| **AQUATIC LIFE BENEFICIAL USE SUPPORT** | | | |
| A.1. Determine the status of aquatic life use support throughout the state without bias to known impairment<br><br>A.1.1 What is the extent and location of water bodies with limited support of the aquatic life beneficial use?<br><br>A.1.2 What is the extent and location of water bodies fully supporting the aquatic life beneficial use?<br><br>A.1.3. What is the proportion of water bodies in the state and each region in each level of support of the aquatic life beneficial use? | A.2. Assess trends in support of the aquatic life beneficial use throughout the state<br><br>A.2.1 Are water bodies improving or deteriorating with respect to aquatic life?<br><br>A.2.2 Have water bodies fully supporting the aquatic life beneficial use become impaired?<br><br>A.2.3 Has full support of the aquatic life beneficial use been restored to previously impaired water bodies? | A.3. Evaluate sources and pathways of factors impacting the aquatic life beneficial use<br><br>A.3.3 What is the relative importance of different pollutant sources and pathways in terms of impact on the aquatic life beneficial use? | A.4. Evaluate effectiveness of management actions improving the aquatic life beneficial use<br><br>A.4.1 How is the aquatic life beneficial use affected by remediation, source control, or pollution prevention actions and policies regionally and statewide? |

## *Literature Cited*

*Gassel, M., R. K. Brodberg, and S. Roberts. 2002. The Coastal Fish Contamination Program: Monitoring of Coastal Water Quality and Chemical Contamination in Fish and Shellfish in California. Pages 977-990 California and the World Ocean '02: Revisiting and Revising California's Ocean Agenda.*

*Hayes, S. P., and P. T. Phillips. 1986. California State Mussel Watch Marine Water Quality Monitoring Program 1984-1985. Water Quality Monitoring Report 86-3WQ, California State Water Resources Control Board, Sacramento, CA.*

*Hayes, S. P., P. T. Phillips, M. Martin, M. Stephenson, D. Smith, and J. Linfield. 1985. California State Mussel Watch: Marine Water Quality Monitoring Program, 1983 -1984. Water Quality Monitoring Report 85-2WQ, Water Resources Control Board, Sacramento, CA.*

*Rasmussen, D. 1995. Toxic Substances Monitoring Report, 1994-5 Data Report. Water Quality Monitoring Report 95-1WQ, State Water Resources Control Board, Sacramento, CA.*

*Rasmussen, D. 1997. Toxic Substances Monitoring Program 1992-93 Data Report. State Water Resources Control Board, Sacramento, CA.*

*Rasmussen, D. 2000. State Mussel Watch Program 1995-97 Data Report. State Water Resources Control Board, Sacramento, CA.*

*SWRCB. 1986. Toxic Substances Monitoring Program 1984. Water Quality Monitoring Report No. 86-4-WQ, State Water Resources Control Board, Sacramento, CA.*

*SWRCB. 2000. Proposal for a Comprehensive Ambient Surface Water Quality Monitoring Program: Report to the Legislature. State Water Resources Control Board, Sacramento, CA.*

# SECTION 2 METHODS

## 2.1 CREATING THE DATABASE

To assess statewide impairment from mercury and organic contaminants, a bioaccumulation database was assembled. The 'Bioaccumulation Database' (BD) was created using tissue-contaminant data summarized from numerous studies and sources (Table 2.1). We consulted with colleagues in agencies and universities to obtain data from bioaccumulation studies of regional or statewide importance that were in their final version, having gone through all internal and external QA and review processes required by the data authors. We included all such data sets that we were able to obtain and that passed our internal SFEI review as described below. Those data sets that either were unavailable or did not pass our internal review are listed in Table 2.2.

Data were received in varying formats and designs, and imported into individual Microsoft Access tables for review. Data sets were assessed by internal data QA and metadata review (see description below), with acceptable data sets subsequently arranged into a consistent format. The initial BD format was based on the San Francisco Estuary Institute's Regional Monitoring Program for Water Quality in the San Francisco Estuary (RMP) Fish Bioaccumulation Database. Fields used to store data reflected sample attributes (i.e., collection date, site name, species, contaminant name, fish length, etc.). Fields were populated with a code indicating that data were unavailable ('-77') if information needed for format consistency was not provided (e.g., method detection limits). Once a consistent format was attained for each dataset, fields were standardized between studies (e.g., scientific and common names, length units in mm, latitude/longitude in decimal degrees).

Data uncertainties were addressed with a consistent approach across data sets. Data that were reported as below detection (i.e., non-detect) were treated as zero. This approach was necessary, because many data sets did not contain MDLs, which are necessary for more sophisticated interpolation of values below detection limit. The geographic locations of records with inaccurate or missing coordinates were estimated using Terraserver (http://terraserver.microsoft.com). Identical site names with differing coordinates were evaluated on a case-by-case basis. Where the difference in distance was less than 1 km, the first set of coordinates was used. Where the difference was greater than 1 km, the site names were differentiated (e.g., Belmont Pier 1 and Belmont Pier 2). Where site names differed and coordinates were identical, a single site name was chosen. The final unique combinations of site name and coordinates were compiled in a lookup list.

**Table 2.1.**
**Studies included in the review data set.**

| Short Name | Agency | Contact | Recent Report |
|---|---|---|---|
| CalFed | CalFed | Jay Davis | Davis, J.A., B.K. Greenfield, G. Ichikawa, and M. Stephenson. 2004. Mercury in Sport Fish from the Delta Region (Task 2A). Final Report submitted to the CALFED Bay-Delta Program for the Project: An Assessment of the Ecological and Human Health Impacts of Mercury in the Bay-Delta Watershed. 63 pp. |
| CDFG-Clear Lake | OEHHA & CDFG | Margy Gassel (report), David Crane (data) | Gassel, M., S. Klasing, R.K. Brodberg, and S. Roberts. 2005. Health Advisory: Fish Consumption Guidelines for Clear Lake, Cache Creek, and Bear Creek (Lake, Yolo, and Colusa Counties). Office of Environmental Health Hazard Assessment, Sacramento, CA. |
| CFCP | OEHHA & SWRCB | Margy Gassel (report), Emilie Reyes (data) | Gassel, M., R.K. Brodberg, and S. Roberts. 2002. The Coastal Fish Contamination Program: Monitoring of Coastal Water Quality and Chemical Contamination in Fish and Shellfish in California in California and the World Ocean '02: Revisiting and Revising California's Ocean Agenda. |
| Delta98 Organics | SFEI | Jay Davis | Davis, J.A., M.D. May, G. Ichikawa, and D. Crane. 2000. Contaminant Concentrations in Fish from the Sacramento-San Joaquin Delta and Lower San Joaquin River, 1998. San Francisco Estuary Institute, Richmond, CA. |
| DWR Reservoir | DWR | Glen Pearson | Boles, J. 2004. Mercury Contamination in Fish from Northern California Lakes and Reservoirs. Department of Water Resources. |
| EMAP West | EPA | Dan Guzman | report not available |
| NFTS | EPA | Michael Walsh (data), Leanne Stahl (report) | CSC Environmental. 2005. Quality Assurance Report for the National Study of Chemical Residues in Lake Fish Tissue: Analytical Data for Years 1 through 4. US EPA. 57 pp. |
| RMP | SFEI | Jay Davis | Greenfield, B.K., J.A. Davis, R. Fairey, C. Roberts, D. Crane, and G. Ichikawa. 2005. Seasonal, inter-annual, and long-term variation in sport fish contamination, San Francisco Bay. Science of the Total Environment 336:25-43. |
| Schmitt | CDFG | Christopher Schmitt | Saiki, M. K. and C.J. Schmitt. 1986. Organochlorine chemical residues in bluegills and common carp from the irrigated San Joaquin Valley floor, California. Arch. Environ. Con. Tox. 15: 357-366. |
| SMWP | SWRCB | Emilie Reyes | Rasmussen, D. 2000. State Mussel Watch Program 1995-1997 Data Report. State Water Resources Control Board, California Environmental Protection Agency. |
| SRWP | SRWP | Claus Suverkropp | LWA, 2004. Sacramento River Watershed Program. Annual Monitoring Report 2002-2003. |
| TSMP | SWRCB | Emilie Reyes | Crane, D.B, K. Regaldo, G. Munoz, L. Smith, D. Gilman, M. Hicks, G. Ichikawa, J. Goetzl, A. Bonnema, and W. Heim. 2004. Environmental Chemistry Quality Assurance and Data Report for the Toxic Substances Monitoring Program 2001–2002. |
| UCDavis1 | UC Davis | Darell Slotton | Slotton, D.G., S.M. Ayers, J.E. Reuter, and C.R. Goldman. 1999. Lower Putah Creek 1997-1998 Mercury Biological Distribution Study. Dept. of Environmental Science and Policy, University of California, Davis. |

| | | | |
|---|---|---|---|
| UCDavis2 | UC Davis | Darell Slotton | Slotton, D.G., S.M. Ayers, J.E. Reuter, and C.R. Goldman. 1997. Cache Creek Watershed Preliminary Mercury Assessment, Using Benthic Macro-Invertebrates. Division of Environmental Studies, University of California, Davis. Final Report. |
| UCDavis3 | UC Davis | Darell Slotton | Slotton, D.G., S.M. Ayers, T.H. Suchanek, R.D. Weyand, A.M. Liston, C. MacDonald, D.C. Nelson, and B. Johnson. 2002. The Effects of Wetland Restoration on the Production and Bioaccumulation of Methylmercury in the Sacramento-San Joaquin Delta, California. CALFED Mercury Program Draft Final Project Report. |
| UCDavis4 | UC Davis | Darell Slotton | Slotton, D.G., S.M. Ayers, J.E. Reuter, and C.R. Goldman. 1997. Gold mining impacts on food chain mercury in northwestern Sierra Nevada streams (1997 revision), Appendix B in Larry Walker Associates, 1997, Sacramento River watershed mercury control planning project—report for the Sacramento Regional County Sanitation District. 74 pp. |
| UCDavis5 | UC Davis | Darell Slotton | Slotton, D.G., S.M. Ayers, T.H. Suchanek, R.D. Weyand, and A.M. Liston. 2002. Mercury Bioaccumulation and Trophic Transfer in the Cache Creek Watershed, California, in Relation to Diverse Aqueous Mercury Exposure Conditions. CALFED Mercury Program Draft Final Project Report. |
| UCDavis6 | UC Davis | Darell Slotton | Slotton, D.G., S.M. Ayers, and J.E. Reuter. 1996. Marsh Creek Watershed 1995 Mercury Assessment Project, Final Report. Conducted for Contra Costa County, California. 66 pp. |
| UCDavis7 | UC Davis | Darell Slotton | Slotton, D.G., S.M. Ayers, and J.E. Reuter. 1998. Marsh Creek Watershed Mercury Assessment Project: Third Year (1997) Baseline Data Report with 3-yr Review of Selected Data. Report for Contra Costa County, June 1998. 62 pp. |
| UCDavis9 | UC Davis | Darell Slotton | Slotton, D.G., and Ayers, S.M. 2001. Cache Creek Nature Preserve Mercury Monitoring Program, Yolo County, California. Second Semi-Annual Data Report (Spring - Summer 2001). Study and report prepared for Yolo County, California. |
| UCD Clear Lake | UC Davis | Darell Slotton | not available |
| USGS Natoma | USGS | Michael Saiki | Saiki, M.K., D.G. Slotton, T.W. May, S.M. Ayers, and C.N. Alpers. 2004. Summary of Total Mercury Concentrations in Fillets of Selected Sport Fishes Collected during 2000-2003 from Lake Natoma, Sacramento County, California: USGS Data Series 103. 21 pp. |
| USGS NAWQA | USGS | Dorene E. MacCoy | MacCoy, D. E. and J.L. Domagalski. 1999. Trace elements and organic compounds in streambed sediment and aquatic biota from the Sacramento River Basin, California, October and November 1995. USGS, Sacramento, CA. 37 pp. |
| USGS Sacramento | USGS | Larry R. Brown | Brown, L. R., 1998. Concentrations of chlorinated organic compounds in biota and sediments in streams of the lower San Joaquin River drainage, California. USGS, Sacramento, CA. 23 pp. |
| USGS Trinity | USGS | Jason May | May, J.T., R.L Hothem, and C.N. Alpers. 2005. Mercury concentrations in fishes from select water bodies in Trinity County, California, 2000-2002. USGS Open-File Report 2005-1321. |
| USGS1 | USGS | Jason May | May, J.T., R.L. Hothem, C.N. Alpers, and M.A. Law.. 2000. Mercury Bioaccumulation in Fish in a Region Affected by Historical Gold Mining: The South Yuba River, Deer Creek, and Bear River Watersheds, California, 1999: USGS Open-File Report 00-367. 30 pp. |
| USGS2 | USGS | Joseph L. Domagal-ski | Domagalski, J.L., P.D. Dileanis, D.L. Knifong, C.M. Munday, J.T. May, B.J. Dawson, J.L. Shelton, and C.N. Alpers. 2000. Water-Quality Assessment of the Sacramento River Basin, California: Water-Quality, Sediment and Tissue Chemistry, and Biological Data, 1995-1998. USGS Open-File Report 2000-391. |

**Table 2.2. Studies excluded from the review data set, and the reasons for their exclusion.**

| Study | Full Study Name | Year | Agency | Contact | Reason for Exclusion |
|---|---|---|---|---|---|
| CDFG - Berryessa | Regional Mercury Assessment of Putah Creek and Lake Berryessa | 1982 – 1984 | California Dept. of Fish & Game | Jerry Bruns (RWQCB) | No length or compositing methods given |
| DWR1 | DWR Special Tributary Project - 1998 Fish Sampling | 1998 | California Dept. of Water Resources | Larry Walker Associates | No length or compositing methods given |
| DWR1 | DWR Special Tributary Project - 1999 Fish Sampling | 1999 | California Dept. of Water Resources | Larry Walker Associates | No length or compositing methods given |
| NOAA S/T Benthic Survey | NOAA Status and Trends | 1984 – 1992 | NOAA | Ed Johnson | No length or compositing methods given |
| NOAA S/T Mussel Watch | NOAA Status and Trends | 1986 – 2003 | NOAA | Ed Johnson | No length or compositing methods given |
| CCAMP | Central Coast Ambient Monitoring Program | 1977 – 2001 | CCRWQCB | Karen Worcester | Incomplete dataset provided |
| Delta98 | Delta Fish 1998 | 1998 | SFEI | Jay Davis | Mismatched site and length data for mercury only |
| Guadalupe Fish | Guadalupe River Watershed Mercury TMDL Project | 2003 | Tetra Tech | Dave Drury | Data not released to SFEI |
| Oroville Reservoir | Contaminant Accumulation in Fish, Sediments, and the Aquatic Food Chanin, Study Plan W2 | 2003 | California Dept. of Water Resources | Scott McReynolds | Data not released to SFEI |
| SCCWRP | 1994 Pilot Project & Bight 1998 Survey | 1994, 1998 | Southern California Coastal Water Research Project | Ken Schiff | Data errors not fixed by source |
| SWAMP Fish | Surface Water Ambient Monitoring Program: Chemical Concentrations in Fish Tissues from Selected Reservoirs and Coastal Areas in the San Francisco Bay Region | 2000 – 2002 | State Water Resources Control Board | Karen Taberski | Data duplicated in TSMP and CFCP datasets |

Internal data quality QA consisted of checking for inconsistencies in reporting and communicating our concerns to original sources for clarification. A review of metadata for each study was also conducted to assess quality. This consisted of an evaluation of data collection methodology (e.g., compositing and clean techniques), lab methods (e.g., duplicates, spikes, and blanks), and data quality (QA/QC). Data were approved for analysis based on the documentation of QA procedures by the study authors, compositing methods (e.g., smallest fish length must be greater than 75% of the largest fish length, as recommended by the EPA), and the collection of sufficient samples for inclusion (i.e., consistent field and lab methods for multiple species, samples, and sites).

## 2.2 DATA ANALYSIS

DDT, chlordane, and PCB records reflect the summation of individual compound values for each sample. Total mercury and dieldrin records reflect single compounds, not sums. Nearly all (> 90%) of the mercury present in edible fish tissue is methylmercury. Total mercury therefore represents a valid, cost effective estimate of methylmercury concentration in fish tissue. Summation procedures were based on methods from the RMP. Total DDTs were calculated by summing the concentrations of p,p'-DDT, o,p'-DDT, p,p'-DDE, o,p'-DDE, p,p'-DDD, and o,p'-DDD. Total chlordanes were calculated by summing the concentrations of cis-chlordane, trans-chlordane, heptachlor, heptachlor epoxide, cis-nonachlor, trans-nonachlor, and oxychlordane. Total PCBs were calculated by summing Aroclor concentrations (if available) of 1248, 1254, and 1260. If Aroclor data were unavailable, the forty PCB congeners measured by RMP were summed (for example, see Gunther et al. 1999). Data points were excluded if they did not include the individual parameters that RMP designates as being significant portions of a sum (e.g., sum of DDTs must include p,p'-DDE and p,p'-DDD).

Sport fish analyses were conducted on individual and composite muscle tissue samples. For the mercury impairment analyses, size limits for each species were applied to limit the variation in mercury concentration due to fish length (Appendix 1). Average total length size limits were applied to samples analyzed as composites; otherwise individual total length data were used. The size limits were chosen to include a large proportion of the available data and ensure that the smallest fish were at least 75% the length of largest fish within a species. Fork and total length measurements were included in the same analyses, because excluding either one would have drastically reduced the sample size for analysis.

In the three time intervals examined (1998 – 2003, 1988 – 1997, and 1978 – 1987), the median wet weight concentration for each species at a given location was calculated. Medians were chosen, rather than means, because they reduced the influence of the many non-detects in the database. The maximum median concentration among species at a given location was then used for comparison to OEHHA Guidance Tissue Levels (GTLs) to assess impairment. Any species with at least one sample at a given site was included in the analysis. OEHHA's thresholds were used because they trigger consumption advice and are applied consistently across the state. The GTLs are draft values from Klasing and Brodberg (2006).

Long-term time trends were analyzed in sport fish and bivalves using lipid-normalized concentrations of the organic contaminants. Previous studies have documented a significant relationship between tissue lipid content and organochlorine concentrations (Larsson et al. 1993). Statistical evaluations of long-term trends were performed by computing the Spearman rank correlation coefficient (rs) of year versus the lipid-normalized average concentration (Greenfield et al. 2005). Fish length/age is another important factor for accumulation of DDTs and PCBs, and to a lesser extent dieldrin and chlordane (R. Norstrom, personal communication), but the analyses of trends in organics in this report did not attempt to adjust for this factor.

For the long-term time trend analysis of mercury in sport fish, the effect of fish length on mercury concentration was addressed in the following manner. Mercury concentration was regressed on fish length, and the residuals were analyzed as a time series of length-adjusted mercury data. For the one site with sufficient years in the time series, the residual mercury concentrations were regressed on year.

## 2.3 GIS MAPPING

The map figures were designed using ESRI ArcInfo 9.1 software. All maps are in a California Teale Albers NAD 83 projection. A connection to the GIS from a Microsoft Access 2003 database was established in order to display the results of queries that calculated median concentrations.

After median concentrations were calculated in Access, the highest median concentration among species for each sampling location was used in generating the map figures. Sampling locations were displayed on the maps using latitude and longitude coordinates from the Access database.

We displayed the concentration values in two different ways – categorized by concentration categories and simply as median concentrations. For maps depicting categories, pollutant concentrations are presented in a four-color graduated scheme (green, yellow, orange, and red), representing low, moderate, high, and very high concentrations (Table 3.2.2). The lowest concentrations (low category) are in a range where consumption is strongly encouraged by OEHHA (Klasing and Brodberg 2006). OEHHA is the agency responsible for managing health risks due to contaminated sport fish in California. Locations with concentrations in this category are colored green. The highest concentrations (very high category) are in a range where OEHHA discourages consumption for women of childbearing age and children 17 and younger (Klasing and Brodberg 2006). Locations with concentrations in this category are colored red. Locations with concentrations between these endpoints are colored either yellow (moderate category) or orange (high category). For maps representing the median concentrations directly, we used a single-color bar chart to represent the values.

We created these concentration category and median concentration maps for each contaminant across three different time periods to represent trends over time. To represent the net impact of all pollutants together, we compared the concentration categories for each pollutant at a given site. The worst impairment level from among the pollutants was chosen to represent the site on the net impact map figure. For example, if dieldrin, DDT and chlordanes fell into the low category, while mercury fell in the moderate category and PCBs fell into the very high category at a particular site, then very high was chosen as the concentration category for that site in the net impact analysis.

It should be noted that this analysis of net impact did not attempt to evaluate the potential synergistic or antagonistic interactions of the pollutants under consideration. These potential interactions are a concern, but have not been studied adequately to support such an assessment.

Long-term time trends were mapped separately by site to depict any spatial differences that may have existed in these data.

## *Literature Cited*

*Greenfield, B. K., J. A. Davis, R. Fairey, C. Roberts, D. Crane, and G. Ichikawa. 2005. Seasonal, interannual, and long-term variation in sport fish contamination, San Francisco Bay. Sci. Tot. Environ. 336:25-43.*

*Gunther, A. J., J. A. Davis, D. Hardin, J. Gold, D. Bell, J. Crick, G. Scelfo, J. Sericano, and M. Stephenson. 1999. Long term bioaccumulation monitoring with transplanted bivalves in San Francisco Estuary. Mar. Pollut. Bull. 38:170-181.*

*Klasing, S., and R. Brodberg. 2006. DRAFT Report: Development of guidance tissue levels and screening values for common contaminants in California sport fish: chlordane, DDTs, dieldrin, methylmercury, PCBs, selenium, and toxaphene. California Office of Environmental Health Hazard Assessment, Sacramento, CA.*

*Larsson, P., L. Okla, and L. Collvin. 1993. Reproductive status and lipid content as factors in PCB, DDT, and HCH contamination of a population of pike (Esox lucius L.). Environ. Toxicol. Chem. 12:855-861.*

# SECTION 3
# RESULTS AND DISCUSSION

## 3.1 SUMMARY INFORMATION

The data set assembled included samples representing much of California's geographic scope (Figure 3.1.1). Biases in the spatial coverage are reflected in areas with fewer studies. The scope of the individual studies varied considerably, and a breakdown by study of the contaminants, sample sizes, number of sampling sites, and sampling years is detailed in Appendix 2. We summarized which sport fish, small fish, and bivalve species were most commonly sampled for each contaminant in Table 3.1.1.

## 3.2 THE NET IMPACT OF POLLUTANTS ON FISHING IN CALIFORNIA

### 3.2.1. Introduction

Present concentrations of pollutants in many California water bodies are high enough to cause concern for possible impacts on human health and to have a significant impact on the fishing beneficial use. This section evaluates the "net impact" of pollutants on fishing. Fish in California water bodies are exposed to multiple pollutants, and multiple pollutants are passed on to humans with each fish consumed. "Net impact" refers to the comprehensive consideration of all pollutants in a sample. Maps are presented in this section displaying the locations sampled in recent and historic monitoring. For each location, if the median concentration of any pollutant exceeded thresholds delineating the concentration categories used in this report (low, moderate, high, very high – see Methods for full description), this is indicated on the map. The existence of a comprehensive set of thresholds for human health risks (Klasing and Brodberg 2006) and a relatively uniform population (humans) makes this type of assessment possible. There are a very limited number of cases for which thresholds for certain wildlife species and contaminants can reasonably be estimated. The lack of established thresholds and the taxonomic diversity of wildlife populations make it impossible to perform this type of statewide assessment for impacts on the aquatic life beneficial use.

This chapter focuses exclusively on contamination issues relating to the fishing beneficial use (i.e., sport fish and human health concerns). Sufficient small fish data and assessment thresholds were not available to support a parallel assessment of impacts on aquatic life. Maps geared toward impacts on wildlife would have different species represented (i.e., small fish, such as Mississippi silversides) and would apply different thresholds.

It should also be noted that this analysis of net impact did not attempt to evaluate the potential synergistic or antagonistic interactions of the pollutants under consideration. These potential interactions are a concern, but have not been studied adequately to support such an assessment.

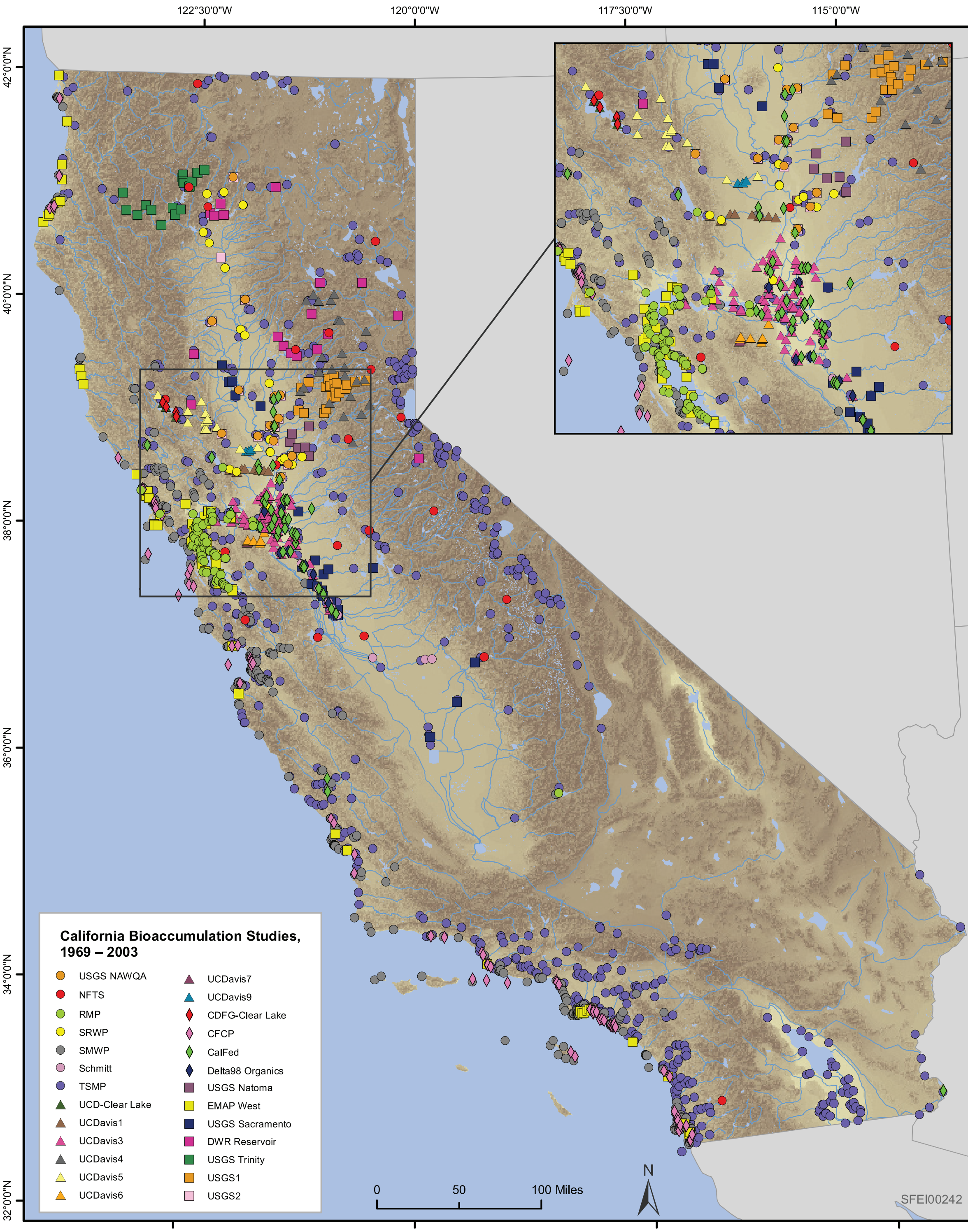



Figure 3.1.1. California Bioaccumulation Studies, 1969 – 2003. Sampling locations for bioaccumulation studies included in the database compiled for this review. Studies conducted from 1969 to 2003. Full titles for each study and additional information are provided in Table 2.1.

**Table 3.1.1.**
**Most widely sampled species in the review data set.**

| POLLUTANT | BIOTA TYPE | NUMBER OF SAMPLES | SPECIES COMMON NAME | POLLUTANT | BIOTA TYPE | NUMBER OF SAMPLES | SPECIES COMMON NAME |
|---|---|---|---|---|---|---|---|
| Chlordanes | Sport | 136 | Channel Catfish | Dieldrin | Small | 22 | Sculpin |
| Chlordanes | Sport | 182 | Common Carp | Mercury | Small | 45 | Bigscale Logperch |
| Chlordanes | Sport | 274 | Largemouth Bass | Mercury | Small | 39 | Sculpin |
| DDTs | Sport | 143 | Channel Catfish | Mercury | Small | 39 | Goldfish |
| DDTs | Sport | 209 | Common Carp | Mercury | Small | 49 | Pacific Staghorn Sculpin |
| DDTs | Sport | 283 | Largemouth Bass | PCBs | Small | 45 | Pacific Staghorn Sculpin |
| Dieldrin | Sport | 143 | Channel Catfish | PCBs | Small | 22 | Sculpin |
| Dieldrin | Sport | 205 | Common Carp | PCBs | Small | 57 | Goldfish |
| Dieldrin | Sport | 277 | Largemouth Bass | Chlordanes | Bivalve | 1029 | California Mussel |
| Mercury | Sport | 1482 | Largemouth Bass | Chlordanes | Bivalve | 205 | Freshwater Clam |
| Mercury | Sport | 656 | Rainbow Trout | Chlordanes | Bivalve | 96 | Pacific Oyster |
| Mercury | Sport | 361 | Bluegill | DDTs | Bivalve | 1172 | California Mussel |
| PCBs | Sport | 146 | Channel Catfish | DDTs | Bivalve | 205 | Freshwater Clam |
| PCBs | Sport | 212 | Common Carp | DDTs | Bivalve | 96 | Pacific Oyster |
| PCBs | Sport | 296 | Largemouth Bass | Dieldrin | Bivalve | 1005 | California Mussel |
| Chlordanes | Small | 54 | Goldfish | Dieldrin | Bivalve | 205 | Freshwater Clam |
| Chlordanes | Small | 45 | Pacific Staghorn Sculpin | Dieldrin | Bivalve | 96 | Pacific Oyster |
| Chlordanes | Small | 22 | Sculpin | Mercury | Bivalve | 377 | Asiatic Clam |
| DDTs | Small | 52 | Goldfish | Mercury | Bivalve | 1795 | California Mussel |
| DDTs | Small | 45 | Pacific Staghorn Sculpin | Mercury | Bivalve | 196 | Freshwater Clam |
| DDTs | Small | 22 | Sculpin | PCBs | Bivalve | 99 | Bay Mussel |
| Dieldrin | Small | 49 | Goldfish | PCBs | Bivalve | 1416 | California Mussel |
| Dieldrin | Small | 45 | Pacific Staghorn Sculpin | PCBs | Bivalve | 196 | Freshwater Clam |

### 3.2.2. Impact of Pollutants on the Fishing Beneficial Use

#### a. Current Status of Net Impact of Pollutants on Fishing in California

**Consumption Advisories**

The existence of consumption advisories issued by OEHHA is one important indicator of the impact of pollutants on the fishing beneficial use in California. As of April 2007, consumption advisories were in place for the Trinity River watershed, several lakes and reservoirs in the northern California Coast Range, a region in the northern Sierra Nevada foothills, Lake Natoma and the lower American River, Tomales Bay, San Francisco Bay, the Sacramento-San Joaquin Delta, the Grassland Area, Lake Nacimiento, coastal areas around Santa Monica Bay, Harbor Park Lake, Newport Pier, and the Salton Sea (Figure 3.2.1, Table 3.2.1). In northern California, most of the advisories were triggered by mercury. PCBs also contributed to the need for advisories for San Francisco Bay and Bay Area reservoirs. In contrast, advisories in the Los Angeles area were prompted by organic chemicals (PCBs and legacy pesticides). Advisories due to selenium have been issued for the Grassland Area and the Salton Sea.

The status of consumption advisories is an inaccurate indicator of the status of impact of pollutants on the fishing beneficial use in California because advisories presently exist for only a fraction of the water bodies that are likely to need them. Resource limitations are the primary reason for the lack of more extensive advice. OEHHA has a small staff assigned to advisory development. OEHHA has accelerated the pace of advisory development in recent years. With a larger staff, OEHHA could develop and update advice for the areas in need in a much more timely manner. Limited resources have also constrained the amount of monitoring that has been conducted. Monitoring of many water bodies has been incomplete or nonexistent, making it impossible to issue consumption advice. A lack of comprehensive data for more species (for both metals and organics) has also had a significant role in limiting development of advisories. Also contributing to the lack of more extensive consumption advice is the past inconsistency of monitoring methods. The TSMP database, for example, contains many inconsistencies in the species sampled at each location and the number and size of fish in composites. Recent studies, with guidance from OEHHA, have employed consistent methods that are better suited to development of advisories. Advisories that will cover a large portion of the state (specifically, much of the Central Valley) are currently being developed as part of the CALFED-funded Fish Mercury Project.

**303(d) Listings**

Inclusion of water bodies on the 303(d) Lists compiled by the SWRCB and the Regional Boards is another important indicator of the impact of pollutants on beneficial uses, including fishing (SWRCB 2003). The 2002 303(d) List included many water bodies that were listed for the pollutants included in this report, including 72 water bodies for mercury, 69 for DDT, 68 for PCBs, 21 for dieldrin, and 27 for chlordanes (Appendix 3). (It should be noted, however, that most, but not all, of these listings are for impacts on fishing.)

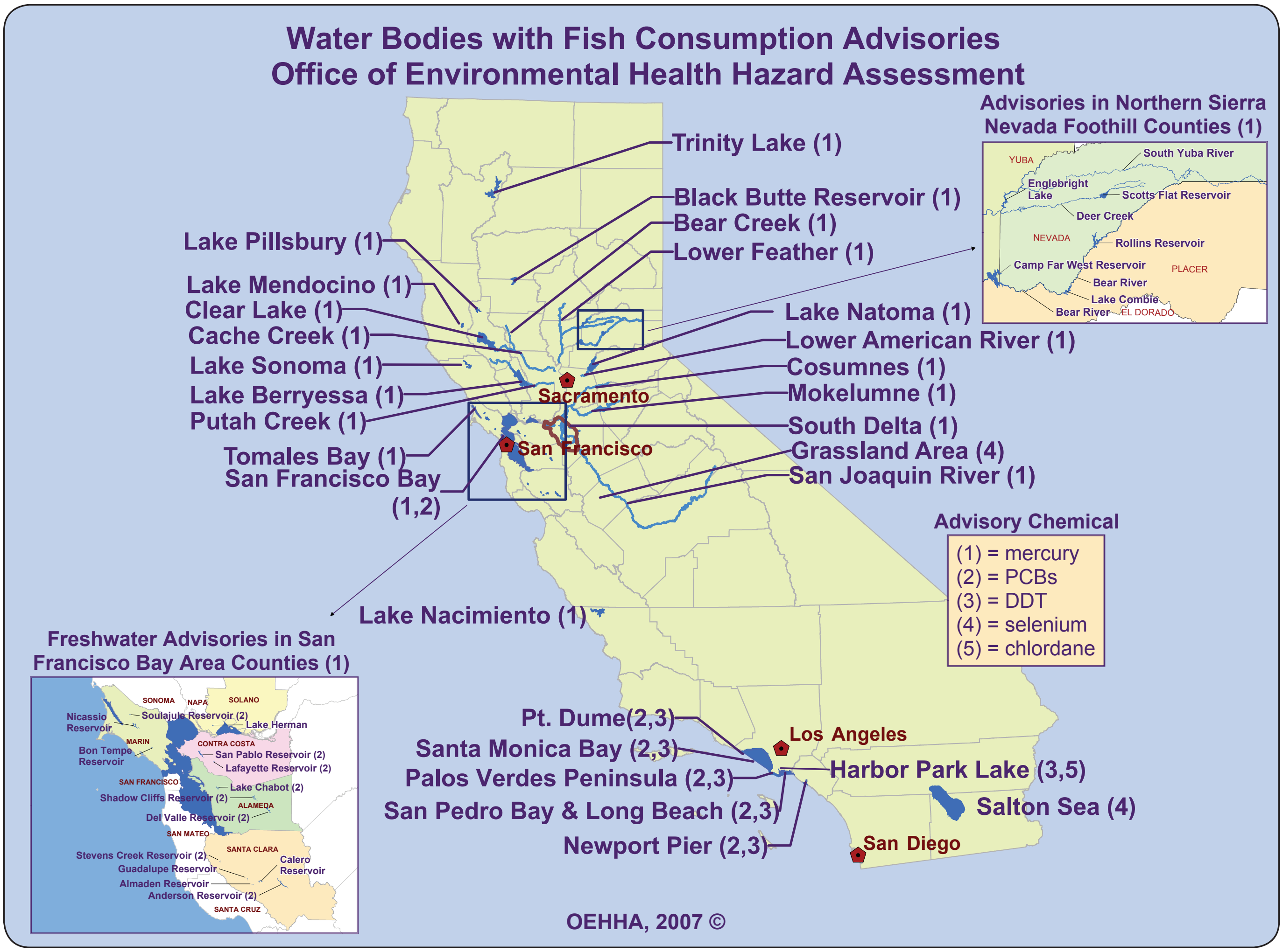
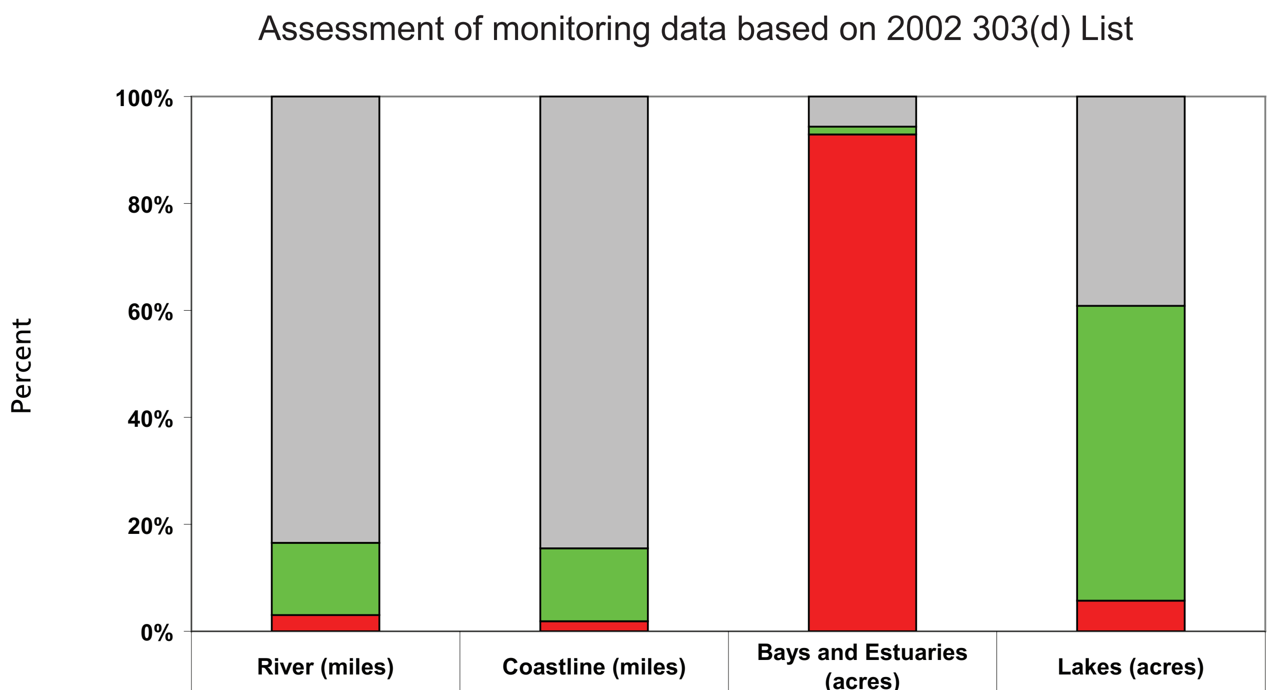



Figure 3.2.1. Consumption advisories in California, 2007. From OEHHA (http://www.oehha.ca.gov/fish/pdf/fishmap2007.pdf).

**Table 3.2.1.**
**Consumption advisories in California in place as of April 2007.**

| LOCATION | POLLUTANT | YEAR ISSUED |
|---|---|---|
| Trinity River Watershed (Trinity County) | Mercury | 2005 |
| Black Butte Reservoir (Glenn and Tehama Counties) | Mercury | 2003 |
| Lower Feather River (Butte, Yuba And Sutter Counties)* | Mercury | 2006 draft |
| Lake Pillsbury (Lake County) | Mercury | 2000 |
| Clear Lake, Cache Creek, and Bear Creek (Lake, Yolo, and Colusa Counties) | Mercury | 2005 |
| Putah Creek (Yolo and Solano Counties) | Mercury | 2006 |
| Lake Sonoma (Sonoma County) and Lake Mendocino (Mendocino County)* | Mercury | 2006 draft |
| Lake Berryessa (Napa County) | Mercury | 2006 |
| Lake Herman (Solano County) | Mercury | 1987 |
| San Francisco Bay and Delta Region** | Mercury, PCBs, DDT, dieldrin, chlordane and dioxins | 1995 |
| Northern Sierra Nevada Foothills (Nevada, Placer, and Yuba Counties) | Mercury | 2003 |
| Lake Natoma and the Lower American River (Sacramento Counties) | Mercury | 2004 |
| Lower Cosumnes and Lower Mokelumne Rivers (Sacramento and San Joaquin Counties)* | Mercury | 2006 draft |
| San Joaquin River and South Delta (Contra Costa, San Joaquin, Stanislaus, Merced, Madera, and Fresno Counties)* | Mercury | 2007 draft |
| Tomales Bay (Marin County) | Mercury | 2004 |
| Guadalupe Reservoir, Calero Reservoir, Almaden Reservoir, Guadalupe River, Guadalupe Creek, Alamitos Creek, and the associated percolation ponds along the river and creeks (Santa Clara County) | Mercury | Not available |
| 10 Bay Area Reservoirs (Alameda, Contra Costa, Marin, and Santa Clara Counties)** | PCBs and Mercury | 2004 |
| Grassland Area (Merced County) | Selenium | Not available |
| Lake Nacimiento (San Luis Obispo County) | Mercury | 2004 |
| Harbor Park Lake (Los Angeles County) | Chlordane and DDT | Not available |
| Point Dume/ Malibu off shore | PCB and DDT | 1991 |
| Malibu Pier | PCB and DDT | 1991 |
| Short Bank | PCB and DDT | 1991 |
| Redondo Pier | PCB and DDT | 1991 |
| Point Vicente Palos Verdes-Northwest | PCB and DDT | 1991 |
| Whites Point | PCB and DDT | 1991 |
| Los Angeles/Long Beach Harbors (especially Cabrillo Pier) | PCB and DDT | 1991 |

| | | |
|---|---|---|
| Los Angeles/Long Beach Breakwater (ocean side) | PCB and DDT | 1991 |
| Belmont Pier (Pier J) | PCB and DDT | 1991 |
| Horseshoe Kelp | PCB and DDT | 1991 |
| Newport Pier | PCB and DDT | 1991 |
| Salton Sea (Imperial and Riverside Counties) | Selenium | 2004 |
| * draft advisory ** interim advisory | | |




| | River (miles) | Coastline (miles) | Bays and Estuaries (acres) | Lakes (acres) |
|---|---|---|---|---|
| Unassessed | 176,569 | 2,895 | 33,873 | 654,424 |
| Unimpaired | 28,494 | 467 | 8,832 | 922,668 |
| Impaired | 6,450 | 65 | 560,000 | 95,592 |

Figure 3.2.2. Assessment of monitoring data based on the 2002 303(d) List. From USEPA (unpublished).

Based on the final 2002 303(d) List, USEPA (Terry Fleming, USEPA, personal communication) prepared an overall tally of the extents of assessment and impairment relative to the fishing beneficial use for several classes of California water bodies: rivers, coastline, bays and estuaries, and lakes (Figure 3.2.2). "Bays and estuaries" is the category that has been assessed most completely, with 94% of the total area in California assessed. Bays and estuaries are relatively highly impacted by pollutants – of the assessed portion of "bays and estuaries", almost all (99%) were classified as impaired. The listing of San Francisco Bay due to concentrations of multiple pollutants in fish tissue accounts for a large percentage of the impaired area. "Lakes" was the next most completely assessed category, with 61% of total lake area assessed, but only a relatively small percentage (6%) of the lake area was impaired. Large percentages of the total miles of river and coastline in the State were not assessed (83% and 84%, respectively), and 18% and 12% of the assessed miles were impaired.

For lakes, these figures based on the proportion of total area assessed are misleading, however, because they are skewed by the small number of very large lakes that has been sampled. There are 9379 lakes in California. Of these, 5297 are very small (less than 4 ha) – too small to be of much value for fishing. A small proportion of the remaining 4082 lakes larger than 4 hectares have actually been sampled in recent years. Based on numbers of lakes sampled, lakes have not been thoroughly assessed. Based on the data compiled for this study, approximately 127 lakes were sampled in the period 1998 – 2003, or only 3% of the lakes in California larger than 4 hectares. Furthermore, most of the lakes that were sampled were not thoroughly assessed. Many of the lakes that are near population centers and are popular for fishing (Stienstra 2004) have not been sampled in recent years. These lakes that have been studied were not sampled in a representative manner that might allow inference about the large number of unsampled lakes. Overall, therefore, the status of California lakes with respect to impacts on the fishing beneficial use is a major information gap.

Past 303(d) listings are also an inappropriate indicator of the status of the fishing beneficial use. The primary shortcoming is the incomplete coverage of the waters of the state, particularly for rivers and coastline. Another problem with 303(d) listings as an indicator is that they are based on sampling that was biased toward characterization of high-risk areas. A third problem is that 303(d) listings are done by the nine Regional Boards in a manner that is not entirely consistent from region to region.

**Recent Monitoring Data**

A third index of the status of the fishing beneficial use can be obtained by comparing the most recent monitoring data for the state to current thresholds for human health concern. The principal advantages of this approach are that it provides a consistent statewide assessment based on recently established risk thresholds ("guidance tissue levels", or GTLs) developed by OEHHA (Table 3.2.2) (Klasing and Brodberg 2006), and provides a clear representation of the data that are available. The GTLs are thresholds that will be directly linked to the development of consumption advice and are therefore a useful tool for communicating to the public.

Figure 3.2.3 provides a summary of the impact of pollutants on fishing in California based on the most recent (1998 – 2003) monitoring data available. Locations where at least one of the pollutants included in the analysis (mercury, PCBs, DDT, dieldrin, and chlordane) were monitored in at least one sample are shown on the map. Pollutant concentrations were evaluated using a four-color graduated scheme (green, yellow, orange, and red), representing low, moderate, high, and very high concentrations (Table 3.2.2 – see Methods for a more complete description). The color assigned to each location in Figure 3.2.3 represents the highest concentration category for any pollutant based on median concentrations for all of the species sampled. For example, if the highest median concentration for mercury at a location fell into the very high (red) category (> 0.9 ppm) and other pollutants were each in the ranges corresponding to the low (green) category, the location was given a red dot. The Figure is intended to provide an initial overview of the extent of impact of pollutants on fishing by depicting an exposure scenario for each location based on the species with the highest concentrations. It is important to note that at many of these locations there are other species present with much lower concentrations of pollutants (data not shown).

**Table 3.2.2.**
**Pollutant concentration categories used in this report. See Methods for description of categories.**

| Pollutant | Low | Moderate | High | Very High |
|---|---|---|---|---|
| Chlordane (ppb) | < 300 | 300 – 1400 | > 1400 – 2400 | > 2400 |
| DDT (ppb) | < 800 | 800 – 3500 | > 3500 – 7000 | > 7000 |
| Dieldrin (ppb) | < 25 | 25 – 100 | > 100 – 200 | > 200 |
| PCBs (ppb) | < 30 | 30 – 140 | > 140 – 270 | > 270 |
| Mercury (ppm) | < 0.1 | 0.1 – 0.5 | > 0.5 – 0.9 | > 0.9 |

For the studies included in this analysis, a total of 390 locations in California was sampled from 1998 – 2003 (Table 3.2.3). Using the most polluted species at each location, 32% of the locations sampled fell into the low category, 42% in the moderate category, 18% in the high category, and 8% in the very high category.

Most (23) of the 33 locations with at least one species in the very high category were placed there because of high mercury concentrations. Another 10 locations were a result of PCB contamination. None of the very high designations were caused by legacy pesticides. The high mercury sites were primarily located in San Francisco Bay, the Delta, historic mercury mining areas in the northern California Coast Range, and historic gold mining areas in the northern Sierra Nevada. Sites classified as very high due to PCBs were primarily in San Francisco Bay, but also in one Bay Area reservoir (Lake Chabot), and two southern California lakes (Big Bear Lake and Harbor Lake).

A majority (60%) of the locations sampled from 1998 – 2003 had species in the moderate and high categories. Mercury and PCBs were again the primary causes for concern at these locations. Intensive

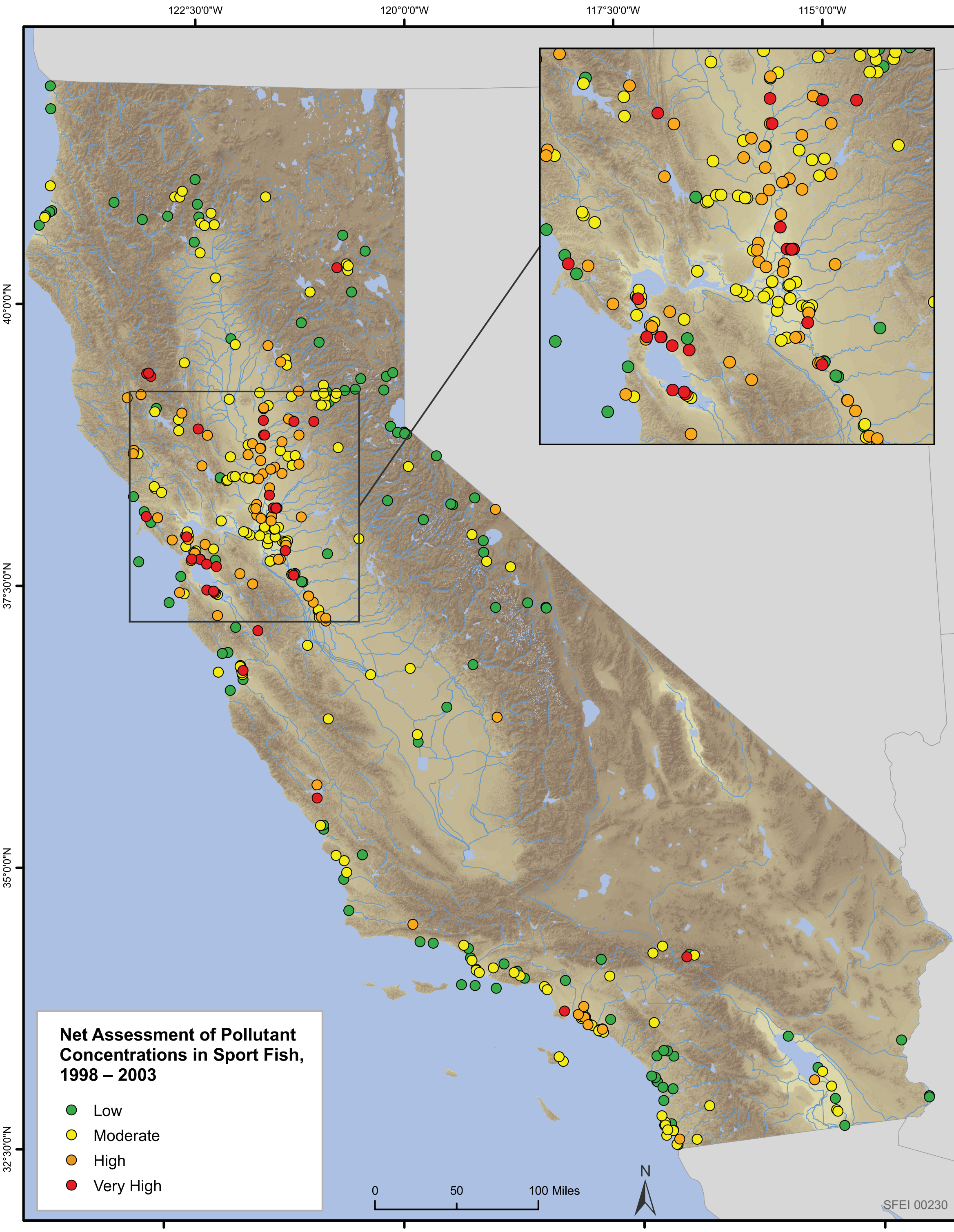



Figure 3.2.3. Net assessment of pollutant concentrations in California sport fish, 1998 – 2003. Based on measurements of several chemicals (mercury, PCBs, DDTs, dieldrin, and chlordanes) in muscle tissue from a variety of fish species. Size limits were applied for evaluation of mercury data (Appendix 1). Dots represent sampling locations. Dot colors correspond to degrees of contamination (low, moderate, high, very high) defined for each pollutant and represent the species with the highest degree of contamination at each location.

sampling focused on mercury was conducted during this period in the Delta region (Davis et al. 2000, 2003), resulting in identification of a dense cluster of moderate and high locations in this area.

Thirty-two percent of the locations sampled fell into the low category. These locations were scattered throughout the state. Areas with a particular prevalence of low concentration locations were the higher elevation water bodies in the Sierra Nevada and a cluster of water bodies in the area north of San Diego. Low concentration locations were relatively scarce in the Delta region.

Figure 3.2.3 also illustrates that some areas of the state were not sampled thoroughly in recent years. Only 33 locations north of Chico were sampled, and many of these were clustered in Humboldt Bay, Trinity Lake, Shasta Lake, and the Susan River. Many areas of the northern part of the state were not sampled at all. Sampling in the portion of California between Chico and Monterey was relatively intense, with especially thorough coverage of the Delta and its nearby tributaries, San Francisco Bay and nearby reservoirs, and Monterey Bay. Most of this sampling, however, has included only mercury analysis, leaving significant information gaps concerning other pollutants. One part of this portion of the state that has received little attention is the central Sierra Nevada and its foothills which encompass many reservoirs and streams. Sampling of reservoirs in general has been insufficient – reservoirs are often large and heterogeneous ecosystems with considerable variation from one arm to the next, and require multiple samples for an adequate representation of condition. Very few samples were collected in the region between Monterey and Santa Barbara, with some concentrated sampling near Morro Bay and San Luis Obispo, and only nine other locations sampled. Sampling in the portion of the state from Santa Barbara south was relatively thorough, especially along the coast near population centers, but many inland reservoirs and streams were not sampled. Overall, the distribution of sampling effort across the state was uneven and non-systematic, often focusing on problem areas. This has resulted in a dataset that provides a skewed assessment at the statewide scale of the impact of pollutants on fishing.

In addition to areas that were not sampled at all, it should be noted that many of the dots shown in Figure 3.2.3 represent very small sample sizes. Of the 390 locations sampled from 1998 – 2003, 139 (36%) were represented by only one sample of one species.

It should also be noted that the analysis presented in Figure 3.2.3 did not include a few other pollutants of concern. Selenium and dioxin were not included and could have had a minor influence on the display. Inclusion of selenium could have resulted in a few more moderate or more contaminated locations in the Salton Sea and San Joaquin Valley. Very few dioxin data exist for 1998 – 2003. Dioxin concentrations in San Francisco Bay have been measured and were above thresholds for concern (Greenfield et al. 2003), but a GTL does not exist for dioxin, and inclusion of dioxins in the San Francisco Bay data would have had a minor influence on the net degree of impairment in that region, since PCBs already place the Bay into the "very high" contamination category. Polybrominated diphenyl ethers (PBDEs) are another class of pollutants of increasing concern, but few data exist for the time period of interest and thresholds for human health concern have not yet been developed.

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

Trends in the overall impact of pollutants on fishing in California can be evaluated by comparing historic data to the same concentration thresholds applied to the recent monitoring data in the previous section (Table 3.2.3, Figures 3.2.4 and 3.2.5). Sampling intensity was highest in the most recent interval (390 locations sampled) in spite of this interval being shorter than the others. This was principally due to significant studies by USGS and CALFED in northern California during this period. A total of 223 locations were sampled in the 1978 – 1987 interval, and 304 in the 1988 – 1997 interval.

**Table 3.2.3. Total number of locations sampled for all pollutants and percentage in each concentration category for three different time intervals from 1978 to 2003.**

| Time Interval | Total Number of Locations Sampled | Low | Moderate | High | Very High |
|---|---|---|---|---|---|
| Recent (1998 – 2003) | 390 | 32% | 42% | 18% | 8% |
| 1988 – 1997 | 304 | 57% | 28% | 9% | 6% |
| 1978 – 1987 | 223 | 39% | 38% | 9% | 14% |

The percentages of locations falling into each concentration category varied across the three time intervals. The primary causes of these changes were probably decreases in concentrations of organic pollutants and biases caused by shifts in the geographic emphasis of sampling. The percentage of locations in the low category was highest (57%) in the 1988 – 1997 interval. This was at least partially due to geographic shifts in sampling. As mentioned previously, sampling in the most recent interval was particularly concentrated in the Delta region, which had a high prevalence of locations in the moderate and high categories. In the 1988 – 1997 period, a relatively large proportion of relatively clean locations were sampled near the Oregon border, in the upper Sacramento River watershed, in the Sierra Nevada, and southern San Diego County.

Concentrations of organics have generally declined across the state, and this probably contributed to the lower numbers of locations in the very high category in the 1988 – 1997 and 1998 – 2003 intervals relative to the 1978 – 1987 interval. In 1978 – 1987, 23 locations fell into the very high category based on PCB concentrations. This number fell to 14 in 1988-1997 and to 10 in 1998 – 2003, in spite of an increased emphasis on San Francisco Bay (with its persistent PCB problem). In the most recent sampling, seven of the red dots attributable to PCBs in the recent period were in San Francisco Bay. The earlier time intervals also included a few locations that could be classified as very high due to concentrations of DDT and dieldrin, while none were observed in the recent interval. Trends in the impact of specific pollutants (mercury, PCBs, and legacy pesticides) on fishing in California are evaluated in more detail, including analysis of time series at selected locations, in later sections of this report.

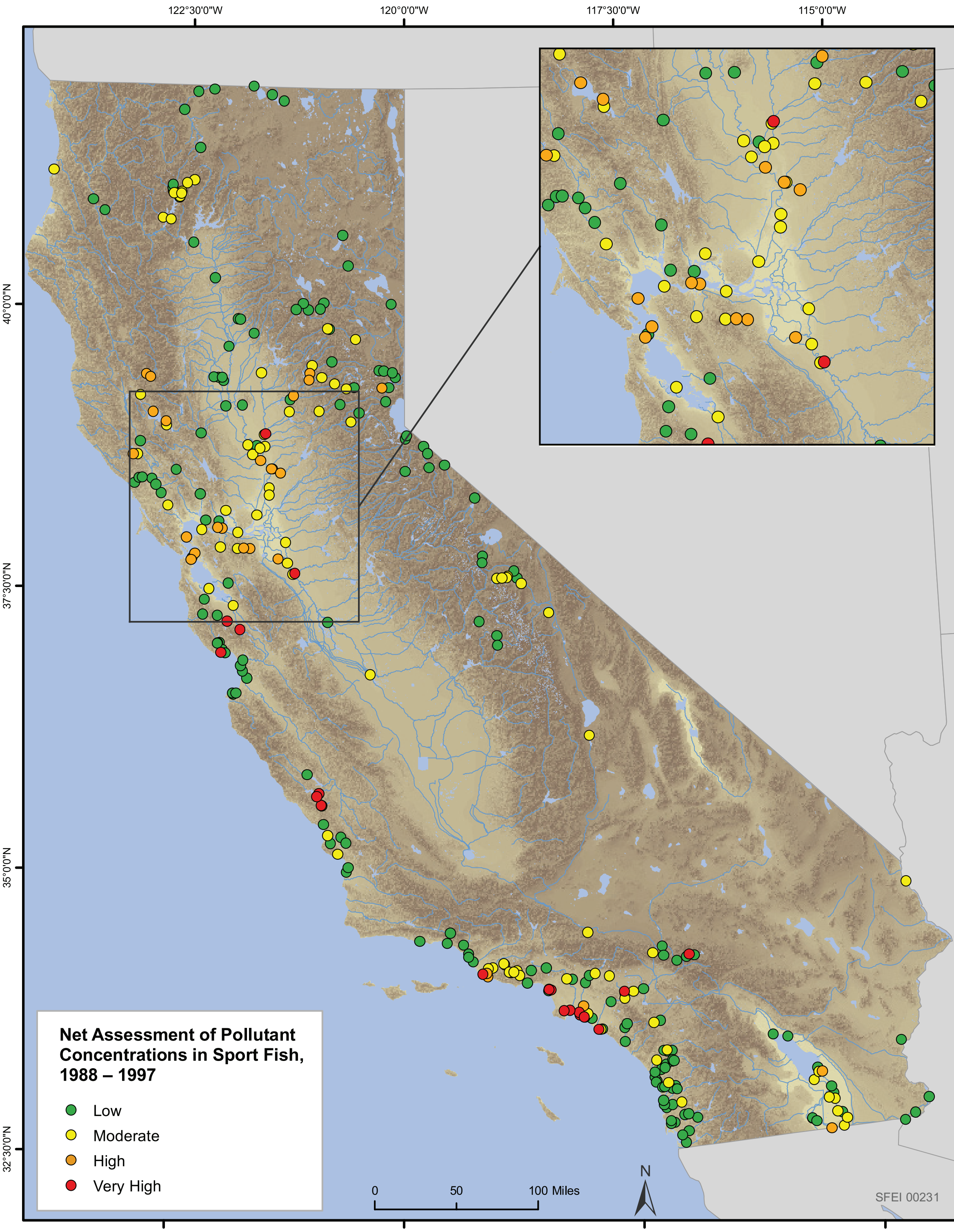



Figure 3.2.4. Net assessment of pollutant concentrations in California sport fish, 1988 – 1997. Based on measurements of several chemicals (mercury, PCBs, DDTs, dieldrin, and chlordanes) in muscle tissue from a variety of fish species. Size limits were applied for evaluation of mercury data (Appendix 1). Dots represent sampling locations. Dot colors correspond to degrees of contamination (low, moderate, high, very high) defined for each pollutant and represent the species with the highest degree of contamination at each location.

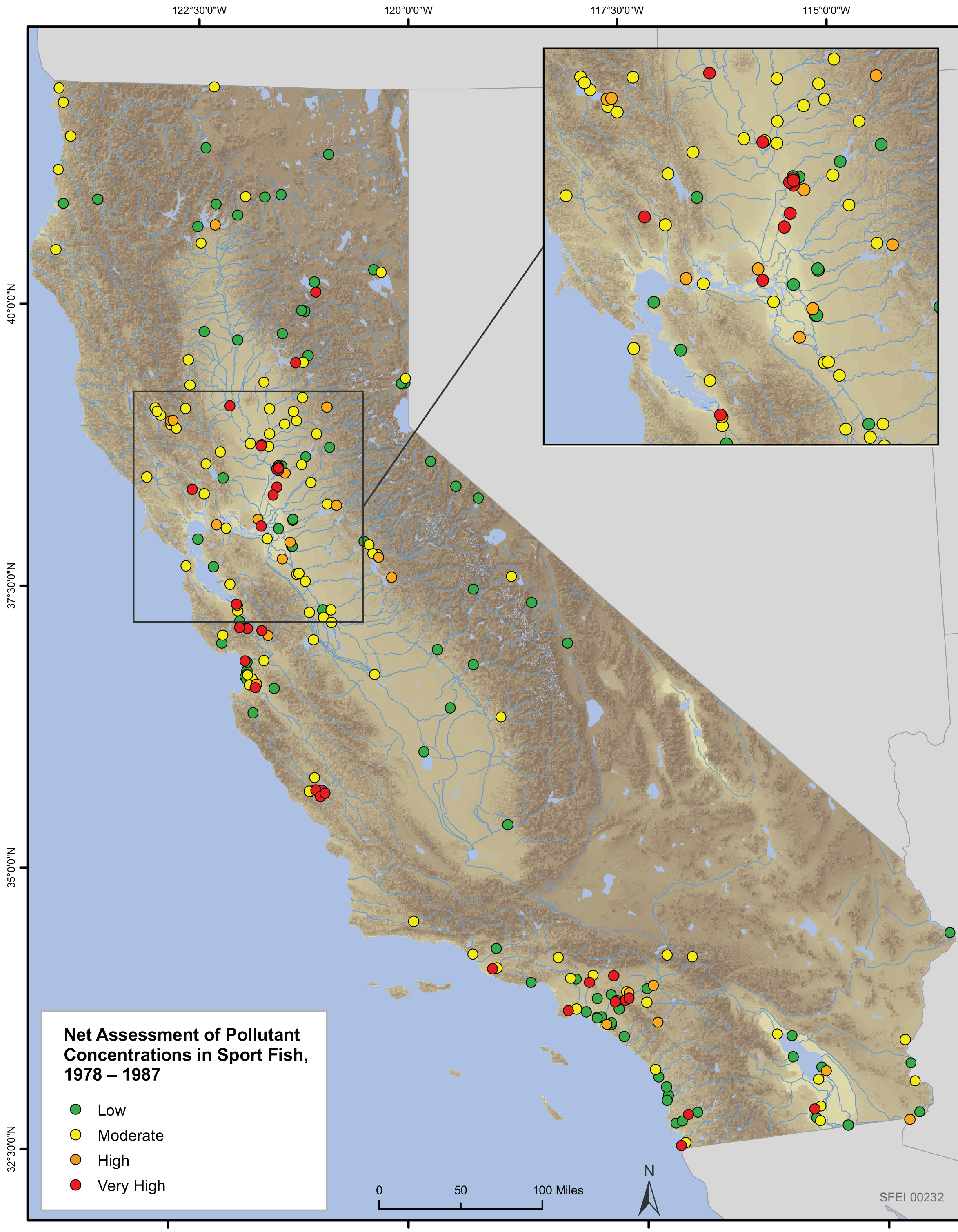



Figure 3.2.5. Net assessment of pollutant concentrations in California sport fish, 1978 – 1987. Based on measurements of several chemicals (mercury, PCBs, DDTs, dieldrin, and chlordanes) in muscle tissue from a variety of fish species. Size limits were applied for evaluation of mercury data (Appendix 1). Dots represent sampling locations. Dot colors correspond to degrees of contamination (low, moderate, high, very high) defined for each pollutant and represent the species with the highest degree of contamination at each location.

In summary, the available data provide a weak basis for evaluation of long-term trends in the impact of pollutants on the fishing beneficial use in California. The geographic focus of sampling has shifted significantly over time, causing apparent but spurious shifts in the impact of pollutants on fishing.

### 3.2.3. Summary and Recommendations

Pollutants are having a significant impact on fishing in California water bodies. Consumption advisories, 303(d) listings, and the bioaccumulation database as a whole provide three indices of the status of impact. Consumption advisories exist for an increasing number of water bodies, but only a fraction of the areas likely to need them. Lack of suitable data is a major impediment to developing advice for additional water bodies. The 2002 303(d) List indicates that large portions of the state have not been assessed, especially for rivers and coastline. On an area basis, most of the lake area in the state has been assessed, and a relatively small percentage of the total area (6%) is classified as impaired. However, based on numbers of lakes sampled, only 3% of the lakes in California larger than 4 hectares have been sampled in recent years, and these lakes were not sampled in a representative manner that might allow inference about the large number of unsampled lakes. Overall, therefore, the status of California lakes with respect to impacts on fishing is a major information gap. Bays and estuaries have been thoroughly assessed (98% of the area) and are highly impacted (93% of the total area). Evaluation of the most recent monitoring data indicates that, for the locations sampled, 32% have low concentrations of pollutants, 42% have moderate concentrations, 18% have high concentrations, and 8% have very high concentrations. Mercury is the pollutant responsible for the majority of locations assigned to the very high category, with PCBs having a secondary role.

The dataset available for these evaluations, however, has several limitations:

- many areas have not been sampled adequately;
- the distribution of sampling locations has varied over time;
- much of the sampling has not been tailored to the development of consumption advice; and
- much of the sampling has been biased toward characterization of polluted areas.

The evaluation of recent data in this section makes it evident that a sampling design with spatial randomization would be better suited to answering the SWAMP assessment questions related to statewide condition. Such a design would allow for an unbiased statewide assessment of the condition of California water bodies. Indices of net impact during different time intervals would be directly comparable since all areas would be sampled in a representative manner. A randomized design could be developed that samples different locations in proportion to the amount of fishing activity, an important feature with regard to development of consumption advice. A randomized design could also be complemented by other approaches, such as targeted sampling for long-term trends in particular locations or focused efforts to sample lakes of particularly high interest. A combination of randomized and targeted sampling would be an optimal approach for providing the information that water quality managers need from a bioaccumulation monitoring program in California.

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## 3.3 IMPACT OF MERCURY BIOACCUMULATION ON FISHING AND AQUATIC LIFE IN CALIFORNIA

### 3.3.1. Introduction

Mercury contamination is common in aquatic food webs in California, with long-term trends indicating little change over the past few decades. Mercury exists in the environment in several different chemical forms, the most problematic of which is methylmercury. Methylmercury accumulates in aquatic food webs, with most of the contaminant being efficiently passed up the food chain at every trophic level. Depuration of methylmercury in vertebrates is very slow.

Mercury is present in California from many different sources. The main source by mass is historical mining operations for mercury and for gold (where mercury was used to extract the precious metal). Combustion emissions result in atmospheric deposition of mercury. Some sources are urban-related, including fluorescent light bulbs, electrical switches, dental fillings, medical instruments, and vaccines.

Mercury becomes a problem when it bioaccumulates in food webs to concentrations that may harm humans or wildlife (Wiener et al. 2003a). Mercury is neurotoxic to vertebrates and other animals, causing deformities, impairing the nervous system, and altering metabolism. The greatest effects occur during early development, and the concentrations at which subtle effects arise are not well understood for many species, including humans. Even less is known concerning the synergistic effects with other contaminants and stressors. The main management actions that affect significant mercury source loading to the environment are currently TMDLs and mine clean-ups. Consumption advisories, when effective, can help protect human populations. This chapter addresses the assessment questions, which are summarized in the Introduction.

The following section (3.3.2) and all the maps in the mercury chapter are geared exclusively toward concentration categories relating to human consumption of sport fish and human health concerns. Section 3.3.3 addresses how mercury may be affecting aquatic life in California, but sufficient small fish data were not available to create the same detailed maps. Maps geared toward impacts on aquatic life would have different species represented (i.e., small fish, such as Mississippi silversides) and would have much lower thresholds (see the TMDL wildlife threshold discussed in section 3.3.3).

### 3.3.2. Impact of Mercury on Fishing in California

#### a. Current Status

**Consumption Advisories**

Consumption advisories issued by OEHHA are one key indicator of the impact of mercury on fishing in California. As of January 2006, consumption advisories due at least partially to mercury were in place for four general groups of water bodies: 1) estuaries near San Francisco (Tomales Bay and San Francisco

Bay-Delta), 2) lakes and reservoirs in the Coastal Range (from Lake Nacimiento in San Louis Obispo County to Black Butte Reservoir in Glenn and Tehama Counties), 3) lakes and rivers in the northern Sierra Nevada foothills and northern Central Valley, and 4) the Trinity River Watershed in the Klamath/Trinity Range (Figure 3.2.1, Table 3.2.1). All of the consumption advisories involving mercury are north of Morro Bay and all but one of the advisories north of Morro Bay include mercury (the Grassland Area advisory for selenium is the exception). Most of these advisories are exclusive for mercury, save for the two in the most urbanized areas: San Francisco Bay-Delta and the Bay Area reservoirs.

Most of the mercury advisories have been issued in the last decade. This pattern reflects a trend toward increasing availability of information on mercury contamination in sport fish and increasing awareness of the mercury problem, not a trend of increasing concentrations. Mercury contamination is extremely persistent, particularly in northern California where substantial loads are still moving down the watersheds toward estuaries, so these advisories are likely to be in place for quite some time. It is possible and even likely that, with increased spatial coverage in monitoring of water bodies in California, other areas may be identified where mercury concentrations persist above the threshold for concern, as happened recently with Bay Area reservoirs. Advisories that will cover a large portion of the state (specifically, much of the Central Valley) are currently being developed as part of the CALFED-funded Fish Mercury Project.

**303(d) Listings**

The 2002 303(d) List for California indicates that mercury is a major contributor to pollutant impact on the fishing beneficial use in the state (Appendix 3). The 2002 303(d) List included mercury listings for the following general areas:

- Region 1 – Lake Pillsbury, Lake Mendocino, Sonoma Lake (6054 acres);
- Region 2 – San Francisco Bay (276,698 acres), Sacramento-San Joaquin Delta (41,736 acres), Tomales Bay (8545 acres) and Walker Creek (16 miles), Guadalupe water bodies (63 acres and 26 miles), and other Bay Area reservoirs and creeks (1226 acres and 16 miles);
- Region 3 – Clear Creek (10 miles) and Hernandez Reservoir (626 acres);
- Region 4 – A few coastal water bodies (380 acres and 1 mile), creeks (6 miles), and lakes (413 acres);
- Region 5 – Many lakes and reservoirs (79,652 acres), rivers and creeks (421 miles), and the Delta waterways (21,087 acres);
- Region 8 – Big Bear Lake (2865 acres); and
- Region 9 – San Diego Bay shoreline (55 acres).

Most of the impacted areas are in central and northern California, although there are a few in the south. The list includes major estuaries, rivers, and lakes as well as many smaller creeks and reservoirs. Some impacted areas may not yet have been assessed for 303d listing, as discussed in section 3.2.

There is general agreement between areas on the 303(d) List and those with consumption advisories in Regions 1, 2, and 5, with exceptions as follows. In Regions 3, 8, and 9 the water bodies on the 303(d) List do not have consumption advisories. These water bodies may have been listed as impaired due to data, such as sediment or small fish mercury concentrations, which are insufficient to produce a consumption advisory. At least nine sport fish must have been sampled in the defined area for an advisory to be issued. Notable areas with advisories that are not on the 303(d) List include the Trinity River Watershed and Lake Nacimiento. In the former case, this discrepancy is probably due to the advisory being issued after the 2002 303(d) List was finalized.

**Recent Monitoring Data**

Sport fish monitoring data collected from 1998 – 2003 indicate that mercury concentrations are above thresholds for concern for human health in many areas of the state (Figure 3.3.1, Table 3.3.1). A total of 294 locations were sampled for mercury during this period. Intensive sampling focused on mercury occurred in the Delta region (Davis et al. 2003a), resulting in the aggregation of dots in this area on the map.

**Table 3.3.1. Total number of locations sampled for mercury and percentage in each concentration category for three different time intervals from 1978 to 2003.**

| Time Interval | Total Number of Locations Sampled | Low | Moderate | High | Very High |
|---|---|---|---|---|---|
| Recent (1998 – 2003) | 294 | 24% | 47% | 21% | 8% |
| 1988 – 1997 | 162 | 38% | 47% | 12% | 3% |
| 1978 – 1987 | 113 | 16% | 67% | 10% | 7% |

Twenty-three of these locations (8% of the total) had a species with a median concentration above 0.90 ppm, placing these sites in the red very high mercury category. All these locations are north of Morro Bay. These hotspots are clustered in central California: in the Sierra Nevada, Coast Range, Central Valley and surrounding foothills, Sacramento-San Joaquin Delta, and San Francisco Bay. Most of the very high mercury sites are within the Golden Gate watershed, with a few others along the coast near the Golden Gate and Lake Nacimiento notably farther south. The Golden Gate watershed and most of the other hotspots (Tomales Bay and Lake Nacimiento) are areas that have been known or suspected to have mercury contamination for decades.

Around one-fifth (21%) of the locations sampled from 1998 – 2003 had mercury concentrations in the orange high category (> 0.5 – 0.9 ppm), and nearly half (47%) fell in the yellow moderate grouping (0.1 – 0.5 ppm). These moderate and high locations were primarily concentrated in the same central California cluster as the hotspots, with the addition of several yellow and orange sites in the Klamath/Trinity Range, southern Sierra Nevada, southern Central Valley, and in southern California along the urbanized coast, Los Angeles Basin, and Salton Sea.

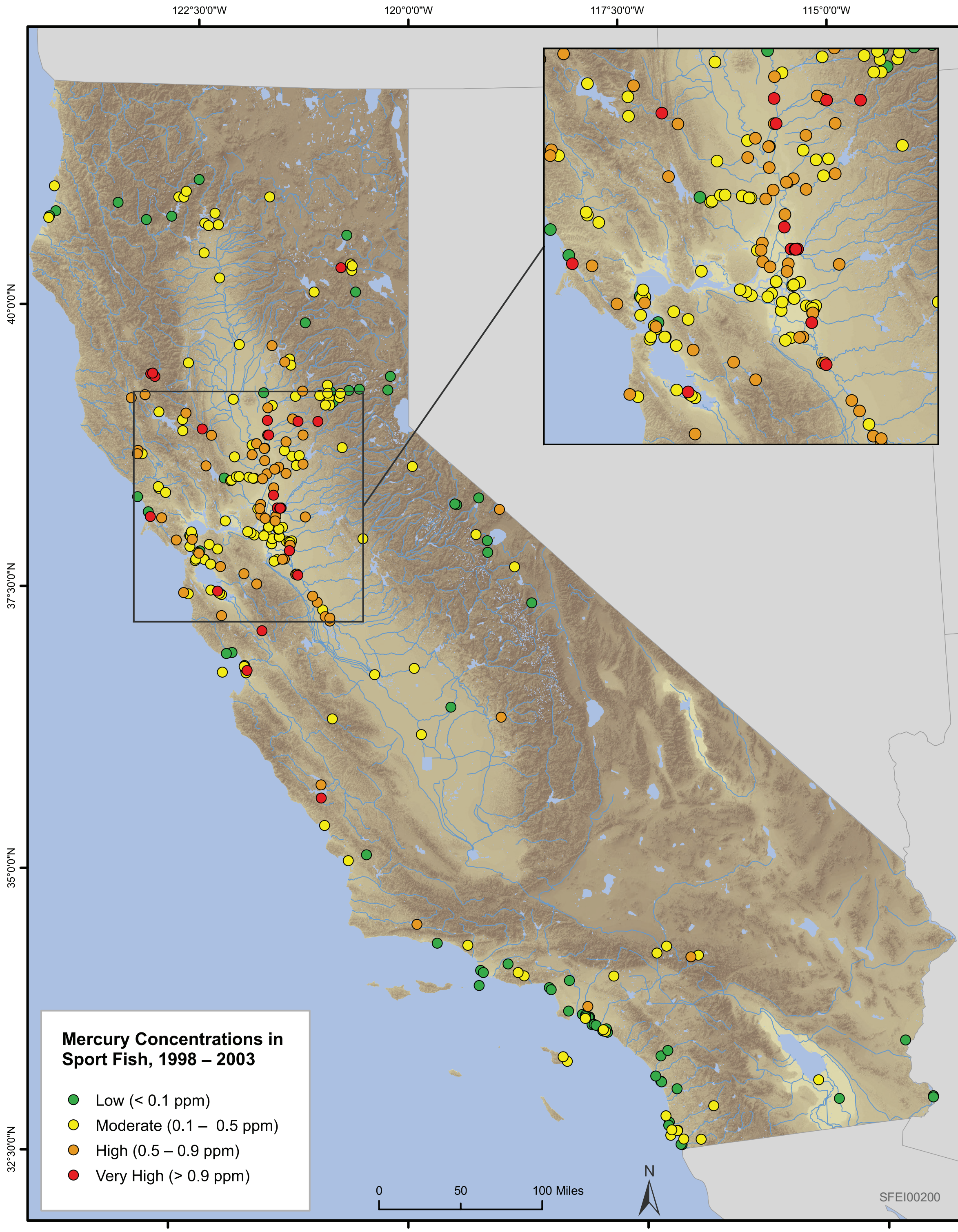


Figure 3.3.1. Mercury concentrations in California sport fish, 1998 – 2003. Based on mercury measurements (ppm wet wt) in muscle tissue from a variety of fish species. Size limits for each species were applied (Appendix 1). Dots represent sampling locations. Dot colors are based on the species with the highest median concentration at a location.

One-quarter (24%) of the locations sampled from 1998 – 2003 had concentrations in the green low mercury category, indicating that median concentrations for all species analyzed at these locations were below 0.1 ppm. Mountainous areas of the state (Sierra Nevada and Klamath/Trinity Range) and southern California had the preponderance of green locations, but there were no regions with sites exclusively in this category.

The geographic patterns of more impacted areas in the state from the recent monitoring data correspond well with the location of consumption advisories and 303(d) listed water bodies.

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

**Management Actions**

Mercury has proven to be the most pervasive and problematic trace metal pollutant in the aquatic environment of California. Data for assessing trends in mercury over several decades are sparse. The expectation is that mercury is neither increasing nor declining across the state as a whole, because this metal is extremely persistent in the environment and has a long residence time in polluted watersheds. Due to the complex cycling of mercury species in aquatic environments, and the small percentage of total mercury that the toxic form methylmercury represents, even a significant reduction in total mercury may not greatly affect food web contamination by methylmercury. In many cases in California, total mercury currently does not appear to be a limiting factor for food web methylmercury concentrations.

Cessation of mercury and gold mining decades ago was an important change in human activities that reduced the production and use of mercury in California but also left hundreds of mine sites in need of remediation. Current management actions of importance include TMDLs, wetland restoration, mine clean-up activities, and consumption advisories. TMDLs have been established for 16 of the 72 water bodies on the 303(d) List, and more are in development, including a large and complex effort for the San Francisco Bay and another for the Guadalupe River watershed. TMDLs aim to reduce the load of mercury entering a water body. Although current concentrations of total mercury may not be limiting food web bioaccumulation in some areas, improvement over the long run is expected if TMDLs reduce total mercury concentrations significantly to the point where it is limiting.

Another management action that could affect mercury contamination of food webs is wetland restoration. There is concern that wetlands may have the potential to increase bioaccumulation of mercury, due to environmental conditions that promote the activity of sulfur-reducing bacteria that methylate mercury and because wetlands can export dissolved organic matter laden with methylmercury (Davis et al. 2003b, Wiener et al. 2006). Previous studies have shown that on a regional level, watersheds with greater wetland acreage have higher rates of mercury bioaccumulation (Krabbenhoft et al. 1999). Large projects, such as the South Bay Salt Pond Restoration in San Francisco Bay and CalFed-sponsored efforts in the Delta, as well as numerous smaller individual restoration projects, have raised a flag of concern that tidal marshes and other wetlands may return to the San Francisco Bay-Delta on a scale that could affect mercury concentrations in biota throughout the region. More research on this topic is needed to understand the specifics of methylmercury production and bioaccumulation in California's wetlands and adjacent ecosystems, and many such projects are underway.

Mine clean-up projects are a third management action of importance for mercury contamination. Several projects at mine sites have removed mercury-laden sediment and tailings from particular areas and reduced total mercury loads downstream. Examples include the Empire Mine near Grass Valley (clean-ups in 1980s and another effort planned for the future), the Sulphur Bank Mercury Mine near Clear Lake, the Polar Star Mine near Dutch Flat, and the Buena Vista Mercury Mine. Remediation of more mine sites will likely be funded by CalFed. Mine clean-up has proved expensive relative to the scope of the problem, and hundreds of mines, tailing sites, and sediment deposits in streams and rivers have yet to be cleaned up.

Given that even under optimistic scenarios of TMDL implementation and remediation projects, mercury concentrations may not decline significantly in the next ten or twenty years, the most effective short-term management option for human health concerns is to communicate consumption advice. Clearly this approach will not be effective for wildlife, but educating human populations about quantities of fish to eat could prevent harm to those who rely heavily on fish and shellfish in their diet. The CalFed-funded Fish Mercury Project has communication of consumption advice – from target sampling locations to analyzing fish tissue to calculating consumption levels to communicating risk – as one of the project's major goals.

A program designed to address bioaccumulation issues should include provisions for all these management actions. Consumptions advisories should be included as the fastest potential method of reducing mercury bioaccumulation in humans from sport fish in California. TMDLs and clean-up actions would contribute to the longer-term improvement of mercury loading into aquatic habitats. Finally, monitoring and process studies should be undertaken to track whether wetland restoration affects mercury bioaccumulation in the local and regional food webs and whether specific design features or management approaches for wetlands could minimize mercury impacts.

**Long-term Trends**

*Sport Fish*

Mercury impairment of the fishing beneficial use has changed little since the 1970s (Table 3.3.1, Figures 3.3.2 and 3.3.3). In each time period, the lowest percentage of sites is in the red category (3 – 8%), while the yellow category contains the largest proportion of sites (47 – 67%). The low green and high orange groupings each contain a significant proportion of sites, with more in the green (16 – 38%) than the orange (10 – 21%). The small changes in percentages in each category among time periods are likely an artifact of changing projects and sampling locations rather than true alteration of mercury bioaccumulation. Spatial patterns in the more and less impacted areas of the state remain very similar over time. Sampling intensity increased over time, which exposed more contaminated areas, but the general location of less and more contaminated areas is remarkably consistent. The New Almaden and Lake Nacimiento sites have been hotspots from the earliest time period through the latest one. The San Francisco Bay-Delta region is

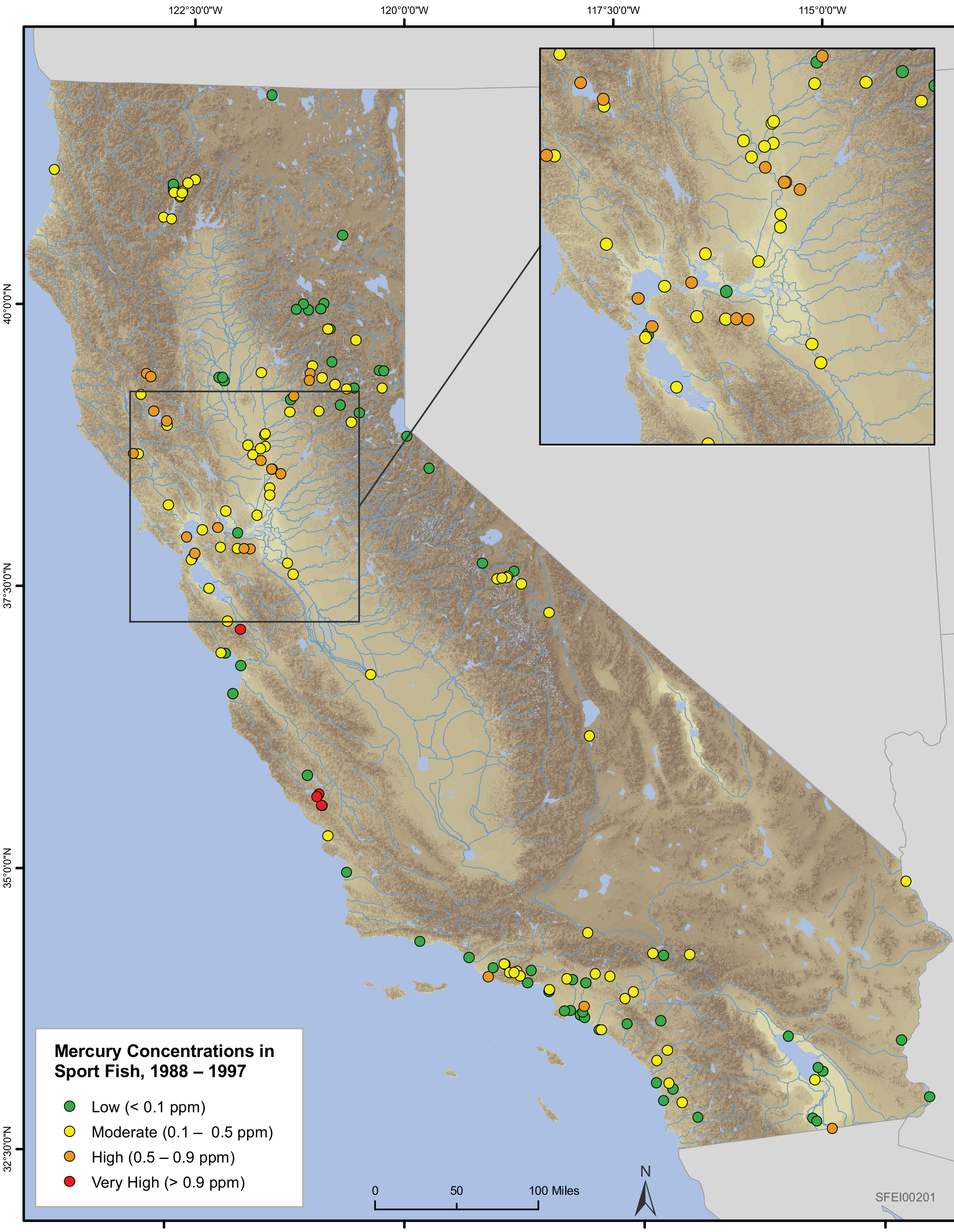


Figure 3.3.2. Mercury concentrations in California sport fish, 1988 – 1997. Based on mercury measurements (ppm wet wt) in muscle tissue from a variety of fish species. Size limits for each species were applied (Appendix 1). Dots represent sampling locations. Dot colors are based on the species with the highest median concentration at a location.

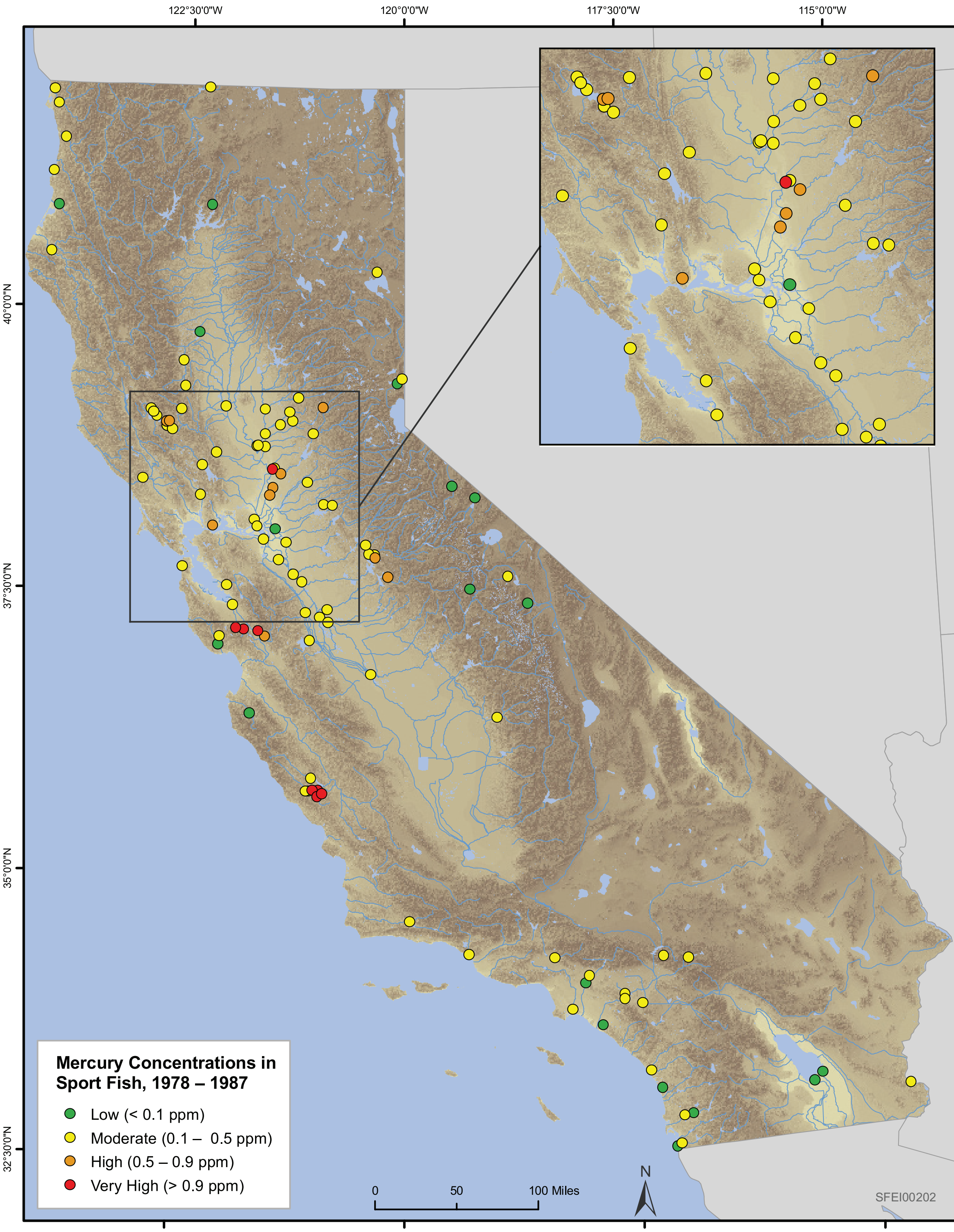


Figure 3.3.3. Mercury concentrations in California sport fish, 1978 – 1987. Based on mercury measurements (ppm wet wt) in muscle tissue from a variety of fish species. Size limits for each species were applied (Appendix 1). Dots represent sampling locations. Dot colors are based on the species with the highest median concentration at a location.

consistently dominated by moderate and high sites with more very high hotspots becoming apparent as sampling increased. Southern California, the Sierra Nevada, and the extreme northwestern part of the state have had a mix of green and yellow sites throughout the 25 years that data were collected. This lack of temporal variability in mercury contamination of food webs is consistent with the expectation of long residence time, as previously discussed.

The data set analyzed in this report is not ideal for the analysis of time trends, because it comprises a variety of studies with differing goals, sampling designs, and methods. These studies form a large body of previous work that advanced our understanding of mercury in the environment, but they were not designed to measure long-term trends. The method detection limits vary over time and between studies, as do the fish species, compositing regimes, and fish-length ranges. Since the data were not designed for this type of analysis, there are few locations that have long time series of comparable data. In general, data from older time periods are sparse. Imperfect as they may be, these time series are nevertheless worth analyzing as the only source of long-term information currently available. Time series of monitoring data that are designed for the purpose of detecting long-term trends should be collected in the future.

The most robust time series were examined, where data from the same species at the same site were available over a span of several years (Figure 3.3.4). Mercury concentration is known to increase with fish size, so the effect of fish length was removed from the analysis (Figure 3.3.5). Fish length was regressed on mercury concentration, and the residuals were analyzed as a time series. The amount of variation in mercury explained by fish length varied from 0.27 to 0.69. Lower correlations may have been partly due to changing study techniques. Despite being the best series available, the Feather, American, and San Joaquin River sites did not have enough data to analyze for trends. Short-term time trends may appear to be decreasing at the Feather and American River sites and increasing at the San Joaquin site in the plots provided, but conclusions should not be drawn from the limited data available. Concentrations in the plots may vary over time due to small sample sizes that do not adequately characterize the population. In general, this data set is weak for analyzing time trends.

At one site on the Sacramento River, data were sufficient to analyze trends over time. These data suggest that mercury in white catfish has declined since 1980, while concentrations in largemouth bass show no significant trend. Davis et al. (2003a) first studied this time series in catfish and noted the same trend. The causes for the decline are unknown. However, the pattern of a steep decrease followed by a leveling off, which is suggested by the data (Figure 3.3.4), is consistent with declines observed in several studies that tracked the response of fish populations following the abatement of an industrial point source (Wiener et al. 2003a, Wiener et al. 2003b). The bass data from the same site are biased toward more recent years and are fewer, so no strong conclusions may be drawn regarding incongruence in the trends between the two species.

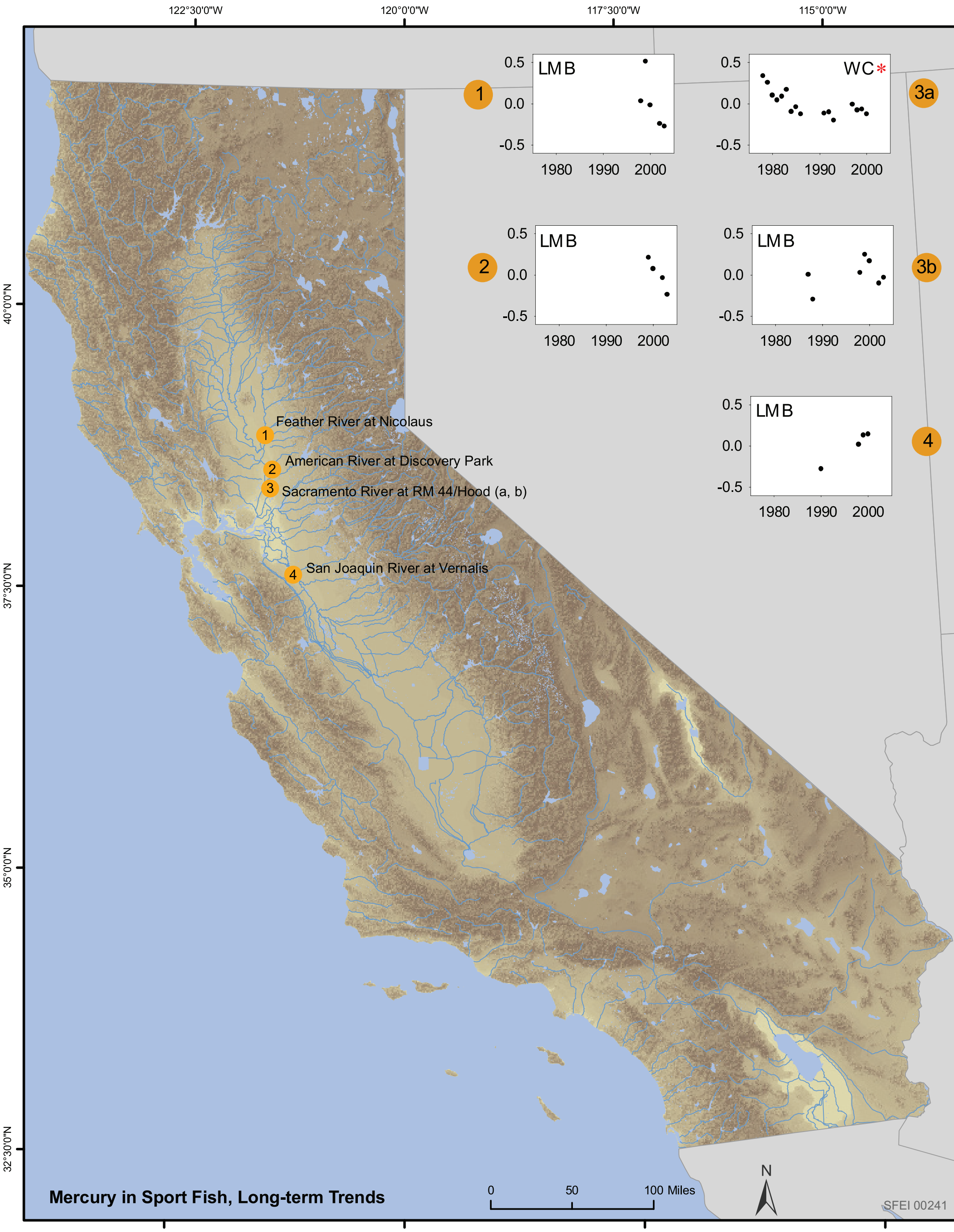


Figure 3.3.4. Long-term trends in mercury concentrations in California sport fish. Locations shown represent the best time series available for different parts of the state. Concentrations (ppm wet wt) presented are the are the residuals of a length versus mercury regression (Figure 3.3.5) for each location to remove the effect of variation in fish size. The red asterisk indicates a significant trend. Species shown are largemouth bass (LMB) and white catfish (WC).

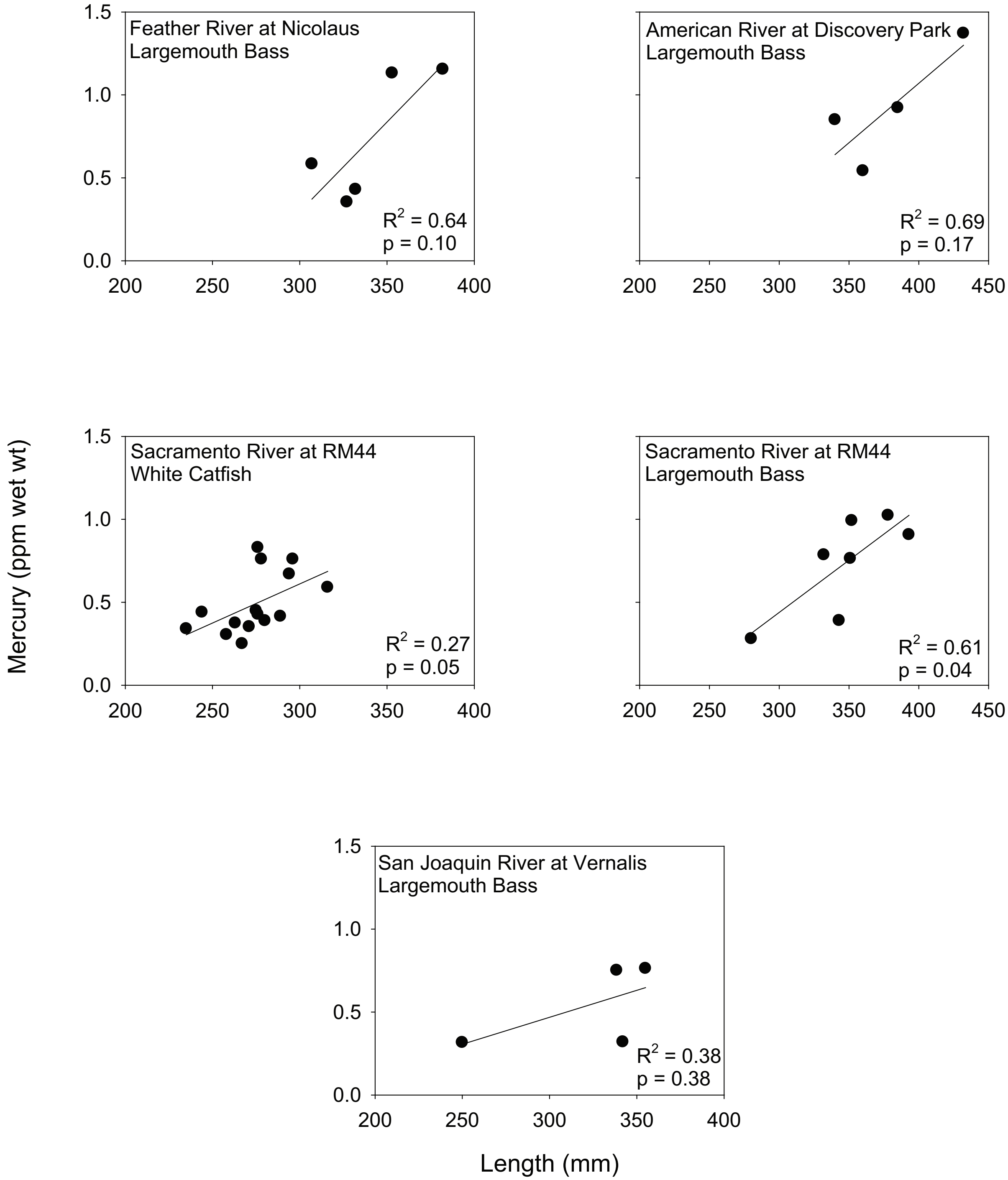


Figure 3.3.5. Regressions of mercury concentration on fish length by site. This analysis was conducted to remove the effect of variation in mercury due to fish size. Residuals were used to assess long-term time trends.

*Bivalves*

Long-term trends in bivalves cannot be analyzed here, as the necessary data are not in existence. Bivalves are not a good indicator for mercury, unless methylmercury is measured. Methylmercury in bivalves generally has not been a component of the large statewide monitoring programs reviewed in this report.

**Case Studies**

*Sacramento-San Joaquin Delta*

The two largest efforts to characterize mercury in sport fish from the Sacramento-San Joaquin Delta were both CalFed-sponsored. The first (Davis et al. 2003a) was completed in 2003, and the other is the Fish Mercury Project which was launched in 2005 and is currently ongoing. The earlier study concluded that concentrations exceeded a 0.3 ppm (wet wt) OEHHA screening value for human health in use at that time (Brodberg and Pollock 1999) in several species (including largemouth bass, striped bass, Sacramento pikeminnow, channel catfish, and white catfish), frequently reaching 1 ppm. The authors identified alternate species, bluegill and redear sunfish, as being safer for consumption with values that rarely exceeded the screening level. This approach of obtaining data on safer species to which anglers may redirect their attention is an important component of sport fish monitoring. Striped bass, an indicator of broad-scale trends in time and space, showed no declines in mercury concentration since the 1970s. Striped bass often have large home ranges, so data from this species represent a large spatial area, such as the Delta and even the adjacent main-stem rivers and Suisun Bay. An important finding was that bioaccumulation in the central Delta was relatively low – a pattern that repeated across several species (Figure 3.3.6). This result – that mercury bioaccumulation in a highly connected ecosystem can vary over short distances – suggested that certain habitats or environments may inhibit methylmercury bioaccumulation relative to adjacent habitats. Current research is pursuing this possibility. The Fish Mercury Project continues and expands upon the earlier study. The Fish Mercury Project has a larger scope and includes components for developing and communicating consumption advisories, as well as a biosentinel component to assess the effects of wetland restoration.

*San Francisco Bay*

Data from San Francisco Bay sport fish monitoring were reviewed in Greenfield et al. (2005) and updated with the latest round of monitoring by the Regional Monitoring Program for Water Quality in 2003 (RMP 2006). Like the Delta, San Francisco Bay has fish that frequently exceed screening levels in a variety of species. In 2003, 69% of the sampled fish (from species chosen for their value as indicators or popularity for consumption), had concentrations exceeding the screening value of 0.2 ppm, which is the proposed value for the San Francisco Bay Mercury TMDL (RMP 2006). Species with the most frequent exceedances were leopard shark (100%), striped bass (93%), California halibut (100%), and white sturgeon (86%). Slightly more than half (58%) of the white croaker samples exceeded the screening value. These problematic concentrations apparently have neither increased nor declined from 1970 through 2000 (Figure 3.3.7; Greenfield et al. 2005). The time trend analysis by Greenfield et al. (2005) took into account DFG data from the early 1970s, Regional Monitoring Program data, and data from the earlier CalFed study discussed in the paragraph above. The authors found no evidence of any change in mercury values over the 30-year period. These

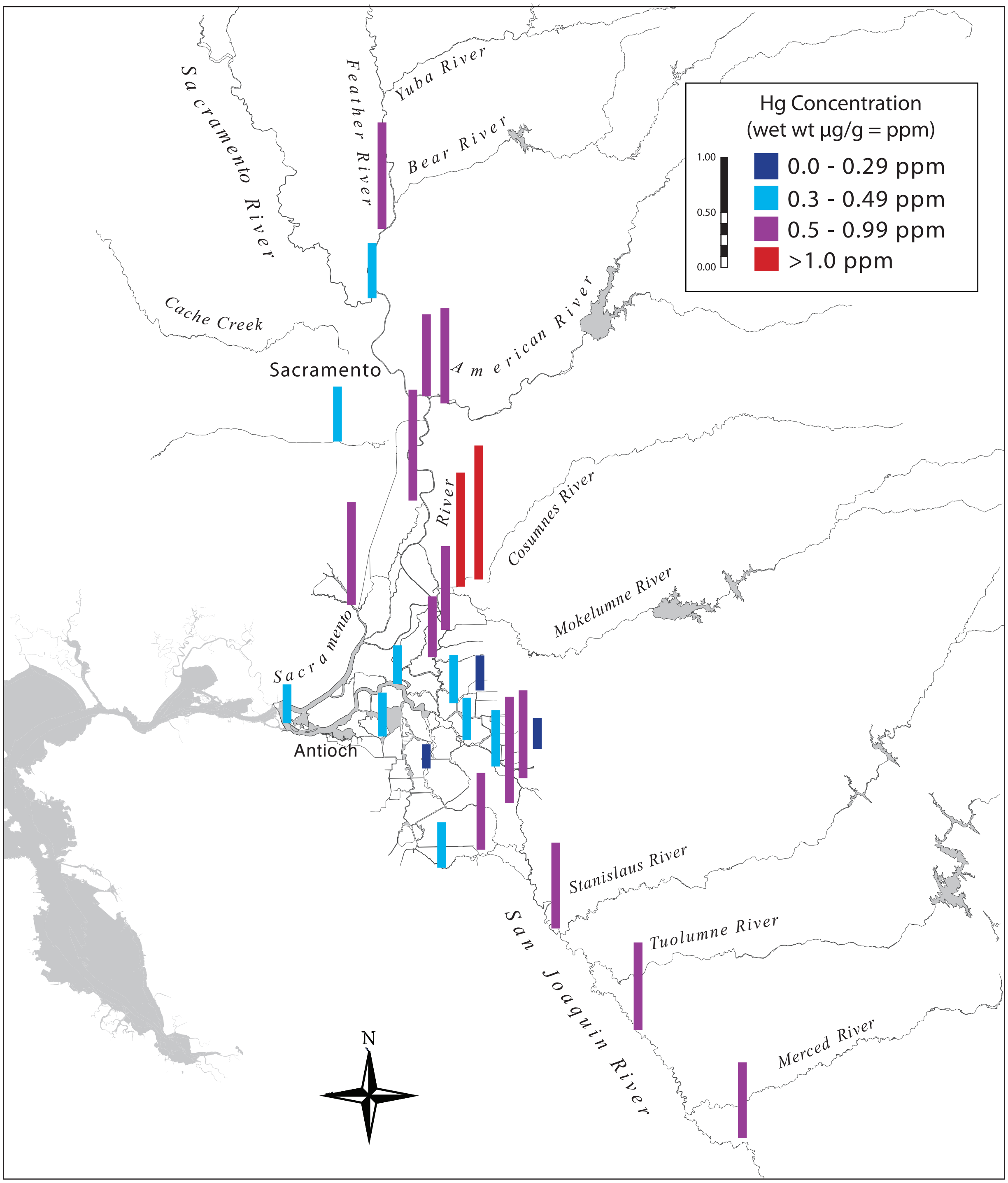


Figure 3.3.6. Average mercury concentrations in largemouth bass from sampling locations in the Sacramento-San Joaquin Delta, 1999. From Davis et al. (2003a). Note the lower concentrations in the central Delta, despite being downstream from two major rivers with high bioaccumulation.

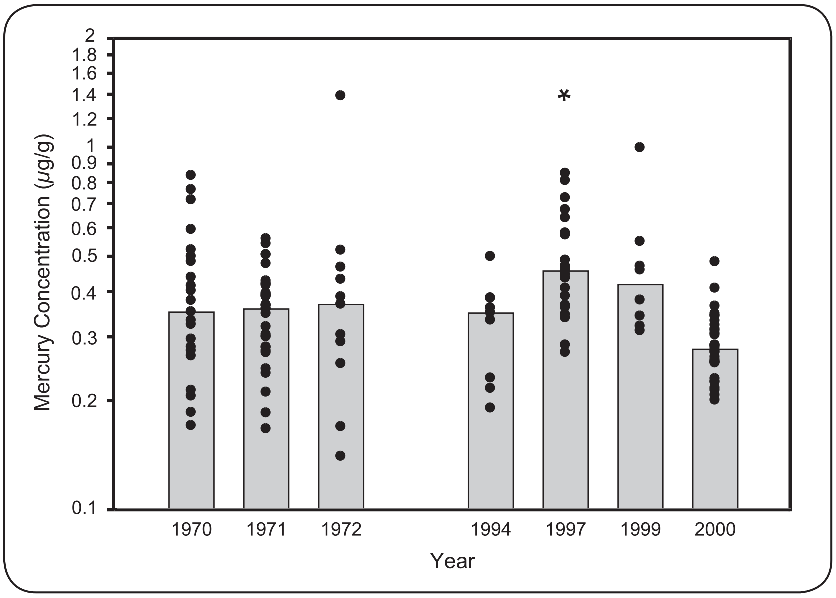


Figure 3.3.7. Mercury concentrations in striped bass in the 1970s and 1990s. From Greenfield et al. (2005). Gray bars indicate annual median concentrations. To correct for variation in fish length, all plotted data were calculated for a 55 cm fish using the residuals of a length vs. log(Hg) relationship. Asterisk above 1997 indicates significant difference from overall length vs. mercury regression. Data were obtained from CDFG historical records (1970 – 1972), a CalFed-funded collaborative study (1999), and the Regional Monitoring Program (1994, 1997, and 2000). Note log scale on the y-axis.

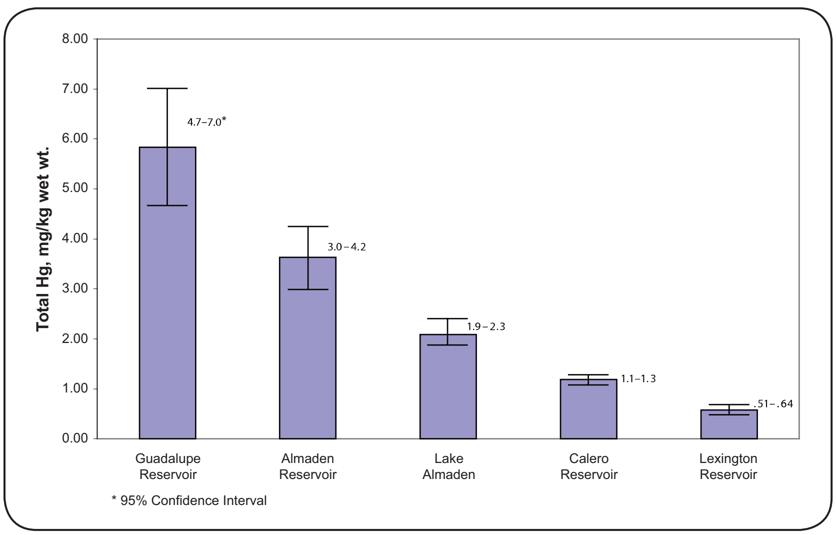


Figure 3.3.8. Mercury concentrations for standardized 40 cm largemouth bass from different reservoirs in the Guadalupe Watershed. From Tetra Tech (2005).

studies indicate that mercury is a significant problem in the San Francisco Bay and the Sacramento-San Joaquin Delta that has neither improved nor worsened appreciably overall during the past few decades.

### *Guadalupe Watershed*

The Guadalupe River Watershed at the south end of San Francisco Bay contains fish with some of the highest mercury concentrations in California, which were studied in depth as preparation for the mercury TMDL in that region (Tetra Tech 2005). Largemouth bass had distinct variation in mercury concentrations between reservoirs, with comparatively little variation within reservoirs (Figure 3.3.8). Differences in mean concentration in each reservoir were related to how closely each site was associated with the New Almaden mining district. Reservoirs close to the historic mining activity (Guadalupe and Almaden) had some of the most contaminated fish in the state, while those farther away (Lexington) had relatively typical concentrations for the larger northern California region. This study identified historic mercury mines as a primary source of mercury in highly contaminated fish in the Guadalupe River basin.

## c. Sources and Pathways

The geographic patterns in mercury concentrations are as previously described for mercury impairment and are consistent across the three time periods (Figures 3.3.9, 3.3.10, and 3.3.11). Specifically, the San Francisco Bay-Delta and Central Valley have a cluster of relatively high concentrations, with the Lake Nacimiento area also being quite high, and parts of southern California and the Klamath/Trinity Range showing significant but lower contamination. The concentration-based maps are useful for showing how much greater the concentrations are in the Golden Gate watershed compared to nearly all other sites in the state. Furthermore, Cache Creek in the Coastal Range and Bear Creek to the east can be identified as the most contaminated sites from the recent time period, each having concentrations greater than 2 ppm in sport fish. Few significant changes in the pattern of mercury concentrations are apparent over time.

The predominant geographic correlation between mercury contamination in fish and possible sources in the environment is historical mining. Nearly all the hotspots (red locations in Figures 3.3.1, 3.3.2, and 3.3.3) are associated with a mine nearby or upstream (Figure 3.3.12). Most of these hotspots are in the region of highest mercury concentrations in the Sierra Nevada, Coastal Ranges, and the parts of the Central Valley that they drain into. The central and northern Sierra Nevada and the central Coastal Range were sites of intensive mining for gold and mercury, respectively. Sediment from these operations, particularly placer and hydraulic mining, has moved down through the watersheds, affecting the Sierra Nevada foothills, Central Valley, Delta, and San Francisco Bay. Large amounts of mercury-laden sediment and mine tailings remain in the watersheds, providing current and future sources of contaminated sediment downstream and opportunities for remediation. In addition to those in the Central Valley watershed, most of the other hotspots are also geographically correlated with mining, for example, Lake Nacimiento and the Buena Vista Mercury Mine, south San Francisco Bay and the New Almaden Mine, Tomales Bay and the Gambonini Mercury Mine, Susanville and gold mines in the watershed. The red site in Elkhorn Slough is not as easily attributed to mining. However, historic mines in the Pajaro River watershed could have delivered contaminated sediment to the Slough during episodic periods of hydrologic connectivity. Many sites in the moderate and high range also fall inside the main mining belts, and the mercury consumption advisories are in these areas as well.

Exceptions to the association between the mining areas and high mercury concentrations may include Lake Pillsbury and several Sierra Nevada locations. Lake Pillsbury is a hotspot for mercury in sport fish but has no identified mines in the watershed. Nevertheless, the fact sheet put out by OEHHA regarding mercury in fish from Lake Pillsbury states that "the surrounding area is likely to be rich in mercury, and physical and chemical conditions in the lake may be very suitable for mercury that has settled in the bottom to be converted to methylmercury" (OEHHA 2000, p. 2). Furthermore, a comment letter from the North Coast Regional Water Board to the California State Water Board regarding the 303(d) List indicates that mining in the watershed may have contributed to the concentrations found in fish (Kuhlman 2006). Conversely, the Sierra Nevada region has several low mercury concentration locations, despite the multitude of historic gold mines in that area. The low mercury levels in fish at these locations may occur because the trout in high-mountain streams are at a low trophic level and do not tend to bioaccumulate high concentrations of mercury, and/or the sites are at high elevations above mines.

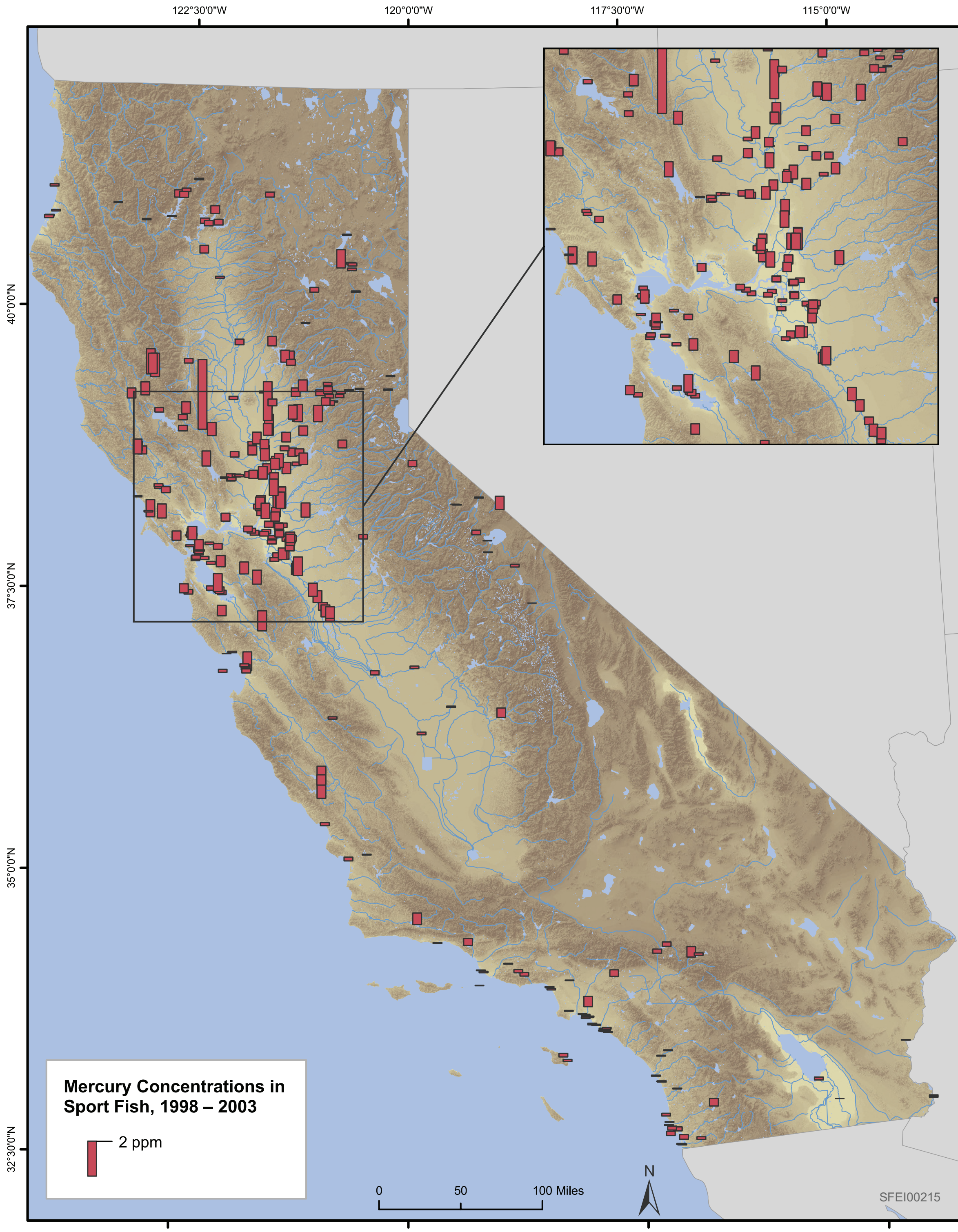


Figure 3.3.9. Mercury concentrations in California sport fish, 1998 – 2003. Bars represent the highest median concentration (ppm wet wt) among species sampled at each location. Size limits for each species were applied (Appendix 1).

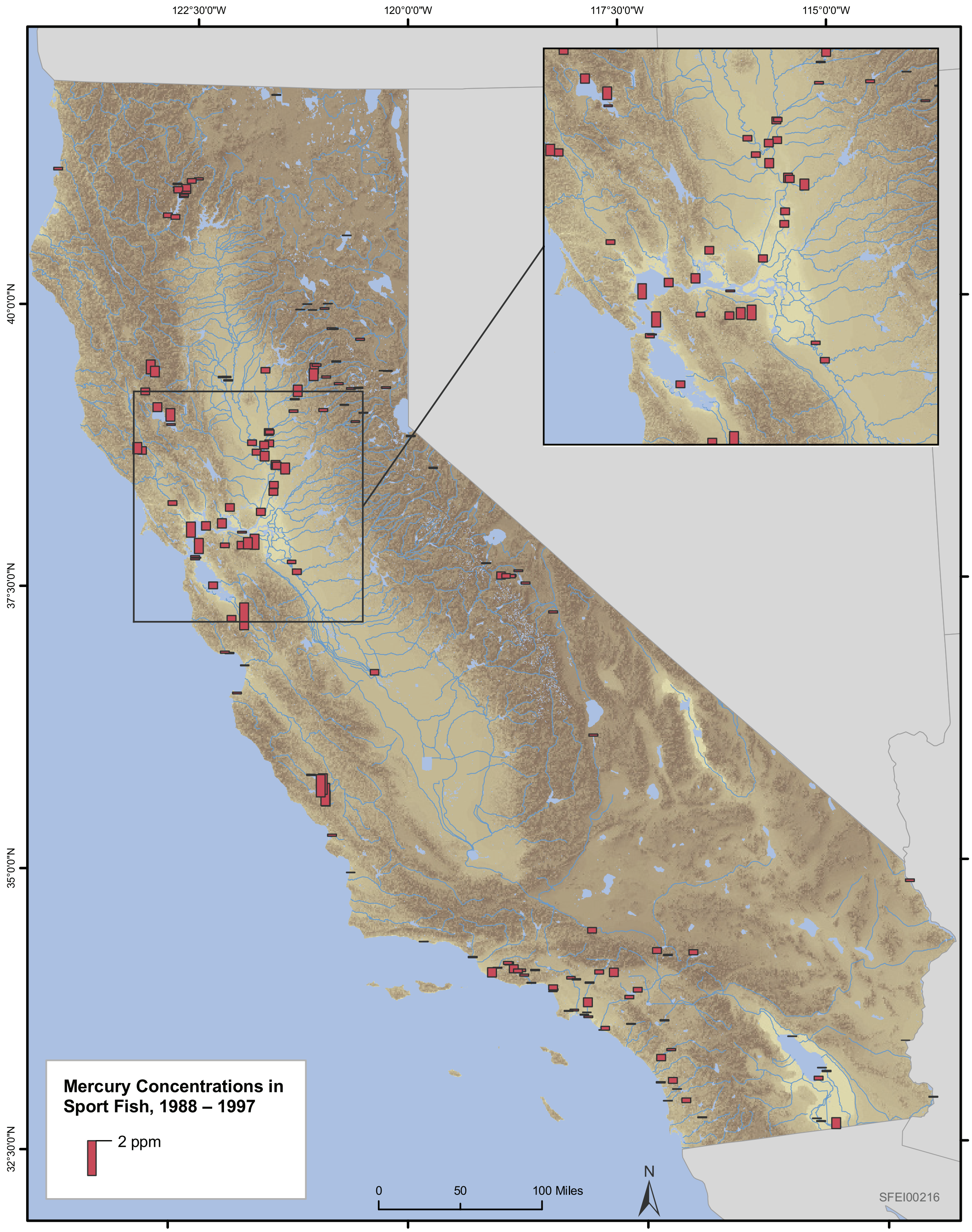


Figure 3.3.10. Mercury concentrations in California sport fish, 1988 – 1997. Bars represent the highest median concentration (ppm wet wt) among species sampled at each location. Size limits for each species were applied (Appendix 1).

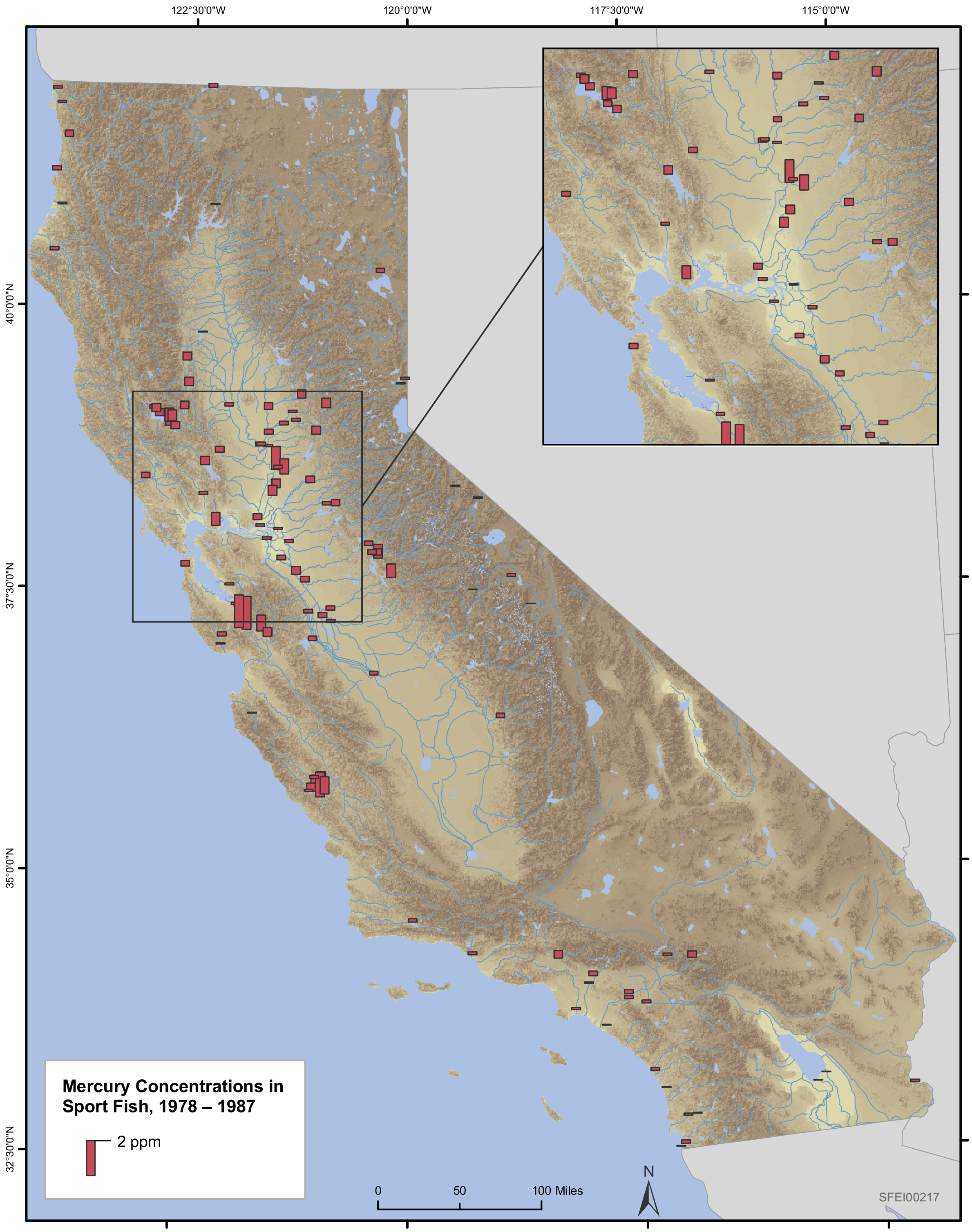


Figure 3.3.11. Mercury concentrations in California sport fish, 1978 – 1987. Bars represent the highest median concentration (ppm wet wt) among species sampled at each location. Size limits for each species were applied (Appendix 1).

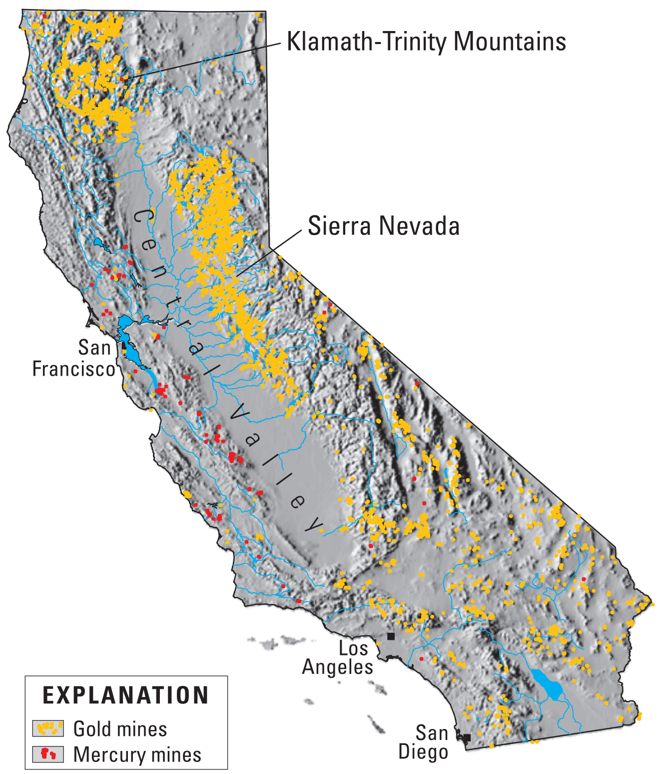


Figure 3.3.12. Locations of past and present gold and mercury mines in California. From Alpers and Hunerlach (2000). Data source: MAS/MILS (Minerals Availability System/Mineral Information Location System) database compiled by the former U.S. Bureau of Mines, now archived by the USGS.

The 303(d) List provides further evidence that mines are an important source. Mining-related potential sources are listed for most of the mercury-impaired water bodies. In the potential sources for the San Francisco Bay-Delta, mining is coupled with urban and industrial sources, as well as atmospheric deposition.

In southern California, mercury concentrations are lower and historic mines are much fewer. Sites in the yellow category are frequent along the highly urbanized coast and around the Los Angeles basin (Figure 3.3.1). This pattern suggests urban and industrial sources may be important in southern California, especially because the moderate (and few high) sites are interspersed with low sites, indicating very local impacts in the small watersheds than run perpendicular to the coast. Agricultural sources are the most likely explanation for the moderate site at the southern edge of the Salton Sea. The Salton Sea is fed by agricultural wastewater that is known to be high in selenium and DDTs. Mining cannot be ruled out as a source for many areas in southern California, however, because some historical mines are present.

Finally, atmospherically deposited mercury may be a significant source (Orihel et al. 2006, Wiener et al. 2006). The 303(d) listing for San Pablo Reservoir cites atmospheric deposition as the sole potential source. Atmospheric deposition is listed as one of several potential sources for the San Francisco Bay-Delta as well. While small in mass relative to other mercury sources in California, atmospheric sources may be sufficient to cause much of the observed bioaccumulation and may be more available for methylation than mercury bound to sediment (Hintelmann et al. 2002).

### 3.3.3. Impact of Mercury on Aquatic Life in California

#### a. Overview of the Mercury Issue for Wildlife

Mercury contamination can have essentially the same health and population consequences for wildlife that it has for humans, with an important difference being that some wildlife species are obligate consumers of aquatic prey, meaning their entire diet of fish or invertebrates may be contaminated with mercury from the aquatic or wetland food web. Mercury can impair behavior, reproduction and early development of fish (Latif et al. 2001, Hintelmann et al. 2002, Drevnick and Sandheinrich 2003), mammals (Barron et al. 2003), and birds (Spalding et al. 2000, Kenow et al. 2003), and probably other vertebrate groups as well. At higher concentrations, organic mercury poisoning can result in acute toxicity and death. Comprehensive reviews of the effects of mercury on wildlife are provided in Wolfe et al. (1998) and Scheuhammer et al. (2007), including classic feeding studies, which provide some of the best effects data. The specific studies mentioned above are just a sampling of newer research.

The TMDLs in preparation for San Francisco Bay and the Sacramento-San Joaquin Delta will employ 0.03 ppm wet weight from whole body analyses of small fish (less than 5 cm) as the threshold for protection of fish-eating wildlife (USFWS 2003, Johnson and Looker 2004, Wood et al. 2005). This relatively low concentration is likely to be exceeded in many instances (see San Francisco Bay-Delta Case Study). The threshold was calculated to be inclusive of protection for the California least tern, which is on the Federal Endangered Species List. All other wildlife species were deemed to be protected at this threshold.

This TMDL threshold is remarkably similar to the Canadian Tissue Residue Guidelines for the Protection of Wildlife Consumers of Aquatic Biota, which is 0.033 ppm wet weight for methylmercury in prey tissue, such as small fish (CCME 2001). Like the TMDL value above, this Canadian guideline refers to a methylmercury concentration in prey items that is not expected to result in adverse effects in predaceous wildlife.

## b. Possible Impairment of Wildlife

**San Francisco Bay-Delta Case Study**

*Open Water Habitats*

In the San Francisco Bay-Delta, several lines of evidence from different taxonomic groups indicate that mercury may be adversely impacting wildlife populations. In open water habitats, mercury exposure and effects have been best studied in birds, yet exposure data are not available for all species, and effects thresholds remain largely unknown. The CISNET study (Davis et al. 2004) indicated that concentrations in double-crested cormorant eggs are probably below concentrations of concern, and the same result was found by Schwarzbach and Adelsbach (2003) for a variety of other Bay and Delta species, including herons, egrets, gulls, and cormorants. Forster's and Caspian tern eggs, however, exceeded threshold effects levels, and avocet, stilt and snowy plover exposures were higher than expected for species that mainly consume invertebrates (Figure 3.3.13; Schwarzbach and Adelsbach 2003). The CalFed-sponsored Mercury Effects in San Francisco Bay-Delta Birds Project, a three-year study currently in progress, aims for much more comprehensive understanding of mercury bioaccumulation and effects in three guilds of Bay birds: diving ducks, terns, and recurvirostrids (black-necked stilt and American avocet). This project is likely the most large-scale and in-depth study of the effects of mercury on birds in California ever conducted and should produce a wealth of information. Older studies on diving ducks suggest mercury effects are present in diving ducks in the Bay. Adult male greater scaup, surf scoter, and ruddy ducks from San Francisco and Tomales

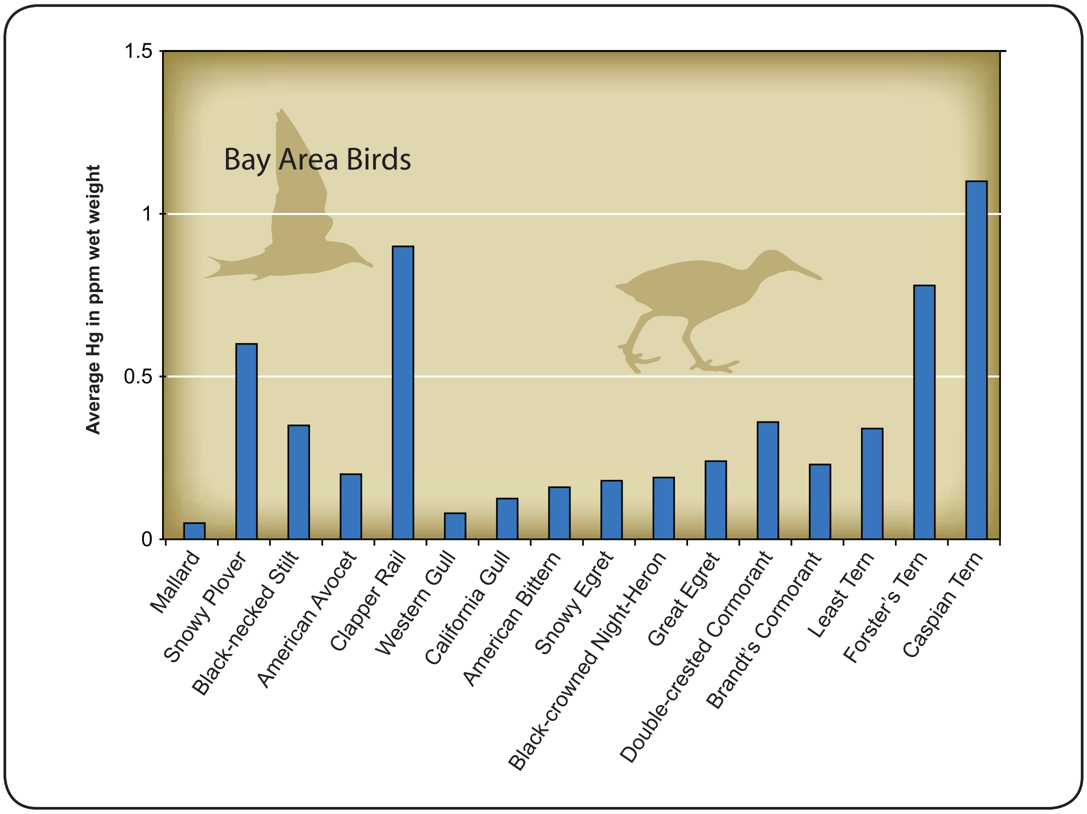


Figure 3.3.13. Mercury concentrations in the eggs of Bay-Delta birds. Mercury concentrations in eggs of the endangered California clapper rail as well as piscivorous Forster's and Caspian terns exceeded the concentration (0.5 – 0.8 ppm) of observed effects in birds. Adapted from Schwarzbach and Adelsbach (2003).

Bays with higher liver mercury had lower body and organ weights (heart, liver) and altered liver metabolism (Ohlendorf et al. 1991, Hoffman et al. 1998). Exceedances of threshold concentrations in small fish, which are important avian prey items for many species in the Bay-Delta, are discussed in the next paragraph.

Less information is available for non-avian wildlife exposure and effects in open-water habitats. Harbor seals in San Francisco Bay have elevated mercury in blood and blubber (Kopec and Harvey 1995). Regarding effects on fish, exposure has been documented in some areas, but effects research is rare. Unpublished data from San Francisco Bay for a study to develop small fish as a monitoring tool for the Regional Monitoring Program show that small gobies and atherinopsids contain mercury at concentrations below effects thresholds for fish (0.2 ppm wet weight, whole body; Beckvar et al. 2006) but above the TMDL target (0.03 ppm) and Canadian guideline (0.033 ppm) for piscivorous wildlife protection (Greenfield, unpubl. data). Extensive studies of small fish in the Delta region by Dr. Darell Slotton (e.g., Slotton et al. 2002) have documented concentrations in inland silversides and other small fish species well above the fish effects screening level of 0.2 ppm, particularly in the north Delta and Suisun. Furthermore, California roach in the Guadalupe watershed exceed the fish effects concentration in some areas and are at or above the 0.03 ppm TMDL target in all locations (Tetra Tech 2005).

*Wetlands*

Mercury exposure and effects in the wetlands of San Francisco Bay are important for two main reasons. First, these wetlands harbor endangered species, one of which may be inhibited from recovery to some degree by mercury contamination. Second, wetland restoration projects on a large scale are planned for the Bay, and there is concern that restored wetlands will increase mercury bioaccumulation (reviewed in Slotton et al. 2002, Davis et al. 2003b).

California clapper rails are the primary species of concern for mercury effects in Bay wetlands. Clapper rail eggs frequently exceeded the concentrations at which mercury become toxic to avian embryos (Lonzarich et al. 1992), averaging 0.81 ppm wet weight in a recent study of fail-to-hatch eggs (Schwarzbach and Adelsbach 2003). Egg-injections have showed that the clapper rail is more sensitive to mercury than the pheasant and mallard from which the threshold of 0.5 – 0.8 ppm was developed (Heinz 2003). Therefore, negative effects on reproduction seem likely, and this conclusion was supported by a field study indicating repressed fecundity in populations around the Bay (Schwarzbach et al. 2006). This endangered species is a rare case where the evidence points toward population-level effects from mercury contamination.

Other marsh species with poorly known mercury exposure and effects include the endangered salt marsh harvest mouse and three unique subspecies of tidal marsh song sparrow, which are California Species of Special Concern. A small study of egg mercury showed low concentrations in song sparrows (Davis et al. 2004). However, the sensitivity to mercury of this species is unknown, and egg-injection results indicated high sensitivity in other songbirds (Heinz 2003). The only published study of mercury in tidal marsh rodents found a marsh-scale relationship between high mercury concentrations in voles and rats and an absence of salt marsh harvest mice (Clark et al. 1992). More research is needed on the exposure and effects of mercury for wetland species. Two current projects aim to study bioaccumulation from water and sediment through wetland birds: the CalFed Petaluma River Mercury Study and the South Baylands Mercury Project.

Concerns about increased mercury bioaccumulation following wetland restoration have prompted the use of biosentinel species, particularly small fish and birds, as indicators of mercury patterns in space and time. Restoration-oriented mercury monitoring, such as the biosentinel component of the Fish Mercury Project and the South Baylands Mercury Project, will use biosentinel species to test hypotheses about wetland restoration. These studies will also provide important information about wildlife exposure to mercury.

**Other Areas**

TMDL documents generally include wildlife targets and discuss wildlife impacts. In general, direct studies of wildlife impacts are rare in the watersheds where TMDLs will be implemented, yet tissue concentrations for prey fish and wildlife often exceed screening values. In one of the more contaminated areas, for example, the Clear Lake TMDL states that current concentrations in wildlife are high enough to cause adverse impacts (Cooke 2002), including reduced hatching success and survival of young, as well as behavioral abnormalities (Elbert 1993). Mercury concentrations have been observed to exceed a toxic risk level (20 ppm dry wt; Scheuhammer 1991) for great blue herons, osprey, and double-crested cormorant and may be affecting western grebe nesting success (Elbert and Anderson 1998). Furthermore, river otter and kingfisher may not be protected, even after the TMDL is implemented (Cooke 2002).

Major gaps exist in our understanding of mercury exposure and effects in wildlife, which can only be filled with further research and monitoring. Long-term trends and effectiveness of management actions for wildlife cannot be assessed, because no such data are available. Use of museum skins to document historical patterns of contaminants in wildlife may be useful, and one such study is underway for clapper rails (Schwarzbach, personal communication). It is important to remember that some areas of the state, such as Mud Slough and certain tidal marshes, are not important human fishing areas, but are teeming with wildlife and have high mercury loads. Sites like these will require monitoring that is targeted toward characterizing the status of and trends in wildlife exposure. Top predators in aquatic ecosystems, such as osprey, northern harriers, and otters, are an important wildlife guild that is likely to have high mercury concentrations. However, very few studies have examined mercury exposure and effects for top predators in California.

### 3.3.4. Mercury Summary

The impact of mercury bioaccumulation on fishing and aquatic life in California waters is significant. Regarding mercury impact on sport fishing, large regions of the state contain fish in high and very high mercury categories (> 0.5 ppm). The impact is generally greatest in the San Francisco Bay-Delta, the Central Valley, and at higher elevations in the watershed, with sites downstream of abandoned mercury mines containing the most highly mercury-contaminated fish. The very few good time series available for mercury in sport fish show no clear trends over the past three decades. Thus, the available evidence supports the hypothesis that the problem may take decades to be resolved. TMDL implementation actions, mine clean-ups, and consumption advisories are important management actions that may improve the situation over different time scales. The effect of large-scale wetland restoration in the San Francisco Bay-Delta on mercury bioaccumulation in this region is unknown. The area with the most data for wildlife, also the San Francisco Bay-Delta, shows that impacts on wildlife populations, including endangered species, from mercury contamination are likely.

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# 3.4 THE IMPACT OF PCB BIOACCUMULATION ON FISHING AND AQUATIC LIFE IN CALIFORNIA

## 3.4.1. Introduction

Polychlorinated biphenyl (PCB) bioaccumulation in aquatic food webs in California has declined significantly since PCB production was banned in the 1970s, but this persistent pollutant continues to have a negative impact on fishing and aquatic life in many parts of the State.

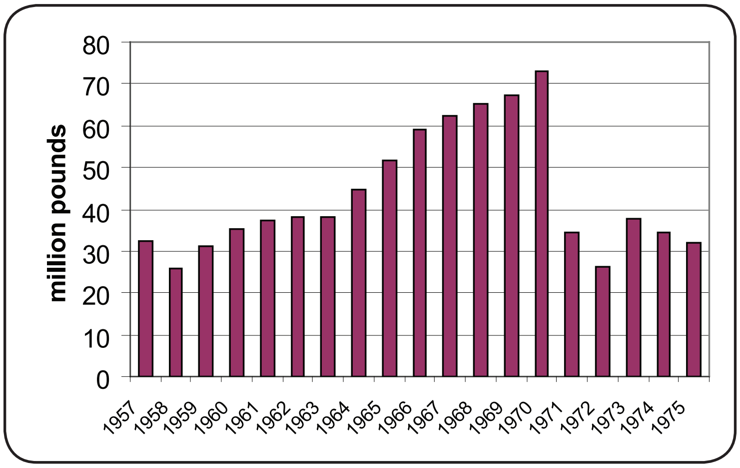


Figure 3.4.1. PCB production in the U.S., 1957 – 1975. From Brinkmann and de Kok (1980).

PCBs are a family of chemicals that were widely used for many decades, are extremely stable in the environment, have a strong tendency to accumulate in living organisms, and continue to pose health risks to humans and wildlife. The term "polychlorinated biphenyl" refers to a family of 209 individual chemicals (called "congeners"). In the U.S., PCBs were sold as mixtures of many congeners known as "Aroclors". Due to their resistance to electrical, thermal, and chemical processes, PCBs were used in a wide variety of applications from the time of their initial commercial production in 1929 (Brinkmann and de Kok 1980). PCBs were most commonly used as insulators in electrical equipment such as transformers and capacitors. Electrical utilities and industries consuming large quantities of electricity used the greatest quantities of PCBs. PCBs were also used in many other applications, including hydraulic fluids, lubricants, inks, and as a plasticizer. U.S. production peaked in 1970 at 39 million kg (Figure 3.4.1). Trends in PCB release to the environment approximately matched trends in PCB production.

U.S. production of PCB-containing capacitors and transformers ended in January 1979. However, the use of PCBs in some totally enclosed applications remains legal to this day. The life-expectancy of capacitors and transformers is decades. In-place capacitors, transformers, and other PCB-containing equipment may still be significant potential sources of PCBs to the environment. A U.S. EPA voluntary transformer registration database showed significant ongoing use, almost 200,000 kg, in the San Francisco Bay Area (the entries in the database were reported between 1998 and 2001) (USEPA 2004). PCBs are extremely persistent in the environment. Leakage from or improper handling of PCB-containing equipment over many decades has led to contamination that persists today, and stormwater continues to wash tainted soils from contaminated sites into California water bodies.

The 1979 ban resulted from a growing appreciation of the health risks of PCBs. In spite of the fact that their use has been restricted for almost two decades, PCBs remain among the environmental contaminants of greatest concern because they are potent toxicants that are resistant to degradation and have a strong tendency to accumulate in biota. PCBs can cause toxic symptoms including developmental abnormalities and growth suppression, disruption of the endocrine system, impairment of immune function, and cancer. U.S. EPA classifies PCBs as a probable human carcinogen. PCBs and other similar organochlorines reach higher concentrations in higher levels of aquatic food chains in a process known as “biomagnification”. Consequently, predatory fish, birds, and mammals (including humans that consume fish) at the top of the food web are particularly vulnerable to the effects of PCB contamination.

The following section (3.4.2) and 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. Section 3.4.3 addresses how PCBs 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.

### 3.4.2. Impact of PCBs on Fishing in California

#### a. Current Status

**Consumption Advisories**

Consumption advisories issued by OEHHA are one key indicator of the impact of PCBs on fishing in California. As of April 2007, consumption advisories due at least partially to PCBs were in place for three general groups of water bodies: 1) San Francisco Bay, 2) reservoirs in the San Francisco Bay Area, and 3) coastal locations in southern California between Point Dume and Dana Point (Figure 3.2.1, Table 3.2.1). In spite of the fact that PCB concentrations in fish in California were probably at their peak in the 1960s and 1970s and have declined gradually since that time, these advisories have all been issued since 1991. This reflects a trend toward increasing availability of information on PCBs in sport fish, not a trend of increasing concentrations. PCBs are extremely persistent and in some cases are well above the threshold for concern, so some of these advisories may be in place for quite some time. In San Francisco Bay, for example, it is expected that it will take 50 to 100 years for PCB concentrations in white croaker and shiner surfperch to fall below the applicable threshold for human health concern (Davis et al. 2007). It is possible that with increased spatial coverage in monitoring of water bodies in California, other areas may be identified where PCB concentrations persist above the threshold for concern, as happened recently with Bay Area reservoirs.

**303(d) Listings**

The 2002 303(d) List for California indicates that PCBs are a major contributor to pollutant impact on fishing in the state (Appendix 3). The 2002 303(d) List included PCB listings for the following general areas:

- Humboldt Bay (16,075 acres);
- San Francisco Bay (318,417 acres);
- Coastal water bodies in the Los Angeles area (many miles and acres, most notably Santa Monica Bay [146,645 acres]);
- Two inland lakes in the Los Angeles area (256 acres);
- 15.5 miles of drainage canal in the Sacramento area;
- 3.3 miles of channel near Stockton;
- 623 acres at Anaheim Bay; and
- 55 acres of San Diego Bay.

Most of the area impacted lies in major bays and estuaries – PCBs are a major contributor to the high degree of impact of pollutants on this class of water body as discussed in Section 3.2.

There is general agreement between areas on the 303(d) List and those with consumption advisories. Major exceptions to this, where water bodies are listed but no consumption advisory including PCBs is in place, are Humboldt Bay, Anaheim Bay and Huntington Harbor, and San Diego Bay. The Bay Area reservoirs are also an exception, where consumption advice is in place but the reservoirs do not appear on the 303(d) List, probably due to the advisories being issued after the 2002 303(d) List was finalized.

**Recent Monitoring Data**

Sport fish monitoring data collected from 1998 – 2003 indicate that PCB concentrations are elevated in many areas of the state (Figure 3.4.2, Table 3.4.1). A total of 251 locations were sampled for PCBs during this

**Table 3.4.1. Total number of locations sampled for PCBs and percentage in each concentration category for three different time intervals from 1978 to 2003.**

| Time Interval | Total Number of Locations Sampled | Low | Moderate | High | Very High |
|---|---|---|---|---|---|
| Recent (1998 – 2003) | 251 | 66% | 26% | 4% | 4% |
| 1988 – 1997 | 237 | 82% | 9% | 3% | 6% |
| 1978 – 1987 | 186 | 66% | 14% | 8% | 12% |

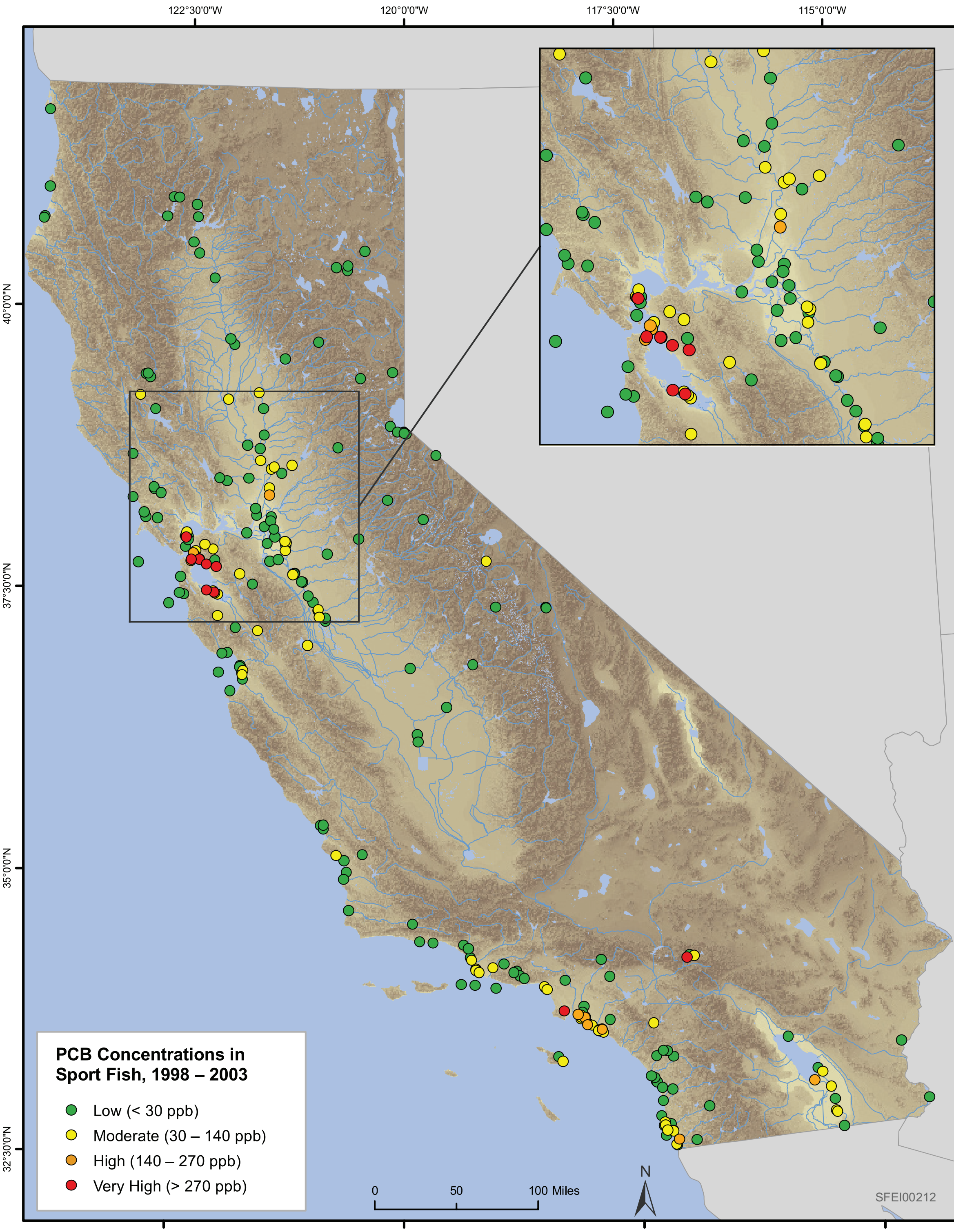


Figure 3.4.2. PCB concentrations in California sport fish, 1998 – 2003. Based on PCB measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

period. Ten of these locations (4% of the total) had a species with median concentrations above 270 ppb, placing them in the very high concentration category. Seven of these locations were within San Francisco Bay, which has a well-documented, persistent PCB problem. The other three locations in the very high category were Lake Chabot in the Bay Area, Machado Lake (formerly Harbor Park Lake) in Los Angeles County, and Big Bear Lake in San Bernardino County. Very high PCB concentrations in carp from Lake Chabot (up to 406 ppb) were first discovered in 2001. Elevated concentrations in multiple species from Machado Lake (formerly Harbor Park Lake) have been measured repeatedly since 1984. Very high concentrations have been observed in carp from Big Bear Lake since 1988.

Thirty percent of the locations sampled in 1998 – 2003 had PCB concentrations in the moderate and high categories. These locations were primarily concentrated near highly urban and industrial areas in the Bay-Delta region, the Los Angeles area, near San Diego, and in the Imperial Valley. However, a few isolated locations in parts of the state removed from dense urbanization had moderate or high concentrations.

Most (66%) of the locations sampled in 1998 – 2003 had concentrations in the low category, indicating that median concentrations for all species analyzed at these locations were below 30 ppb. Areas of the state away from extensive urban and industrial development, such as the northern Sacramento Valley, the Sierra Nevada and foothills, and northern San Diego County, had a preponderance of locations with concentrations below 30 ppb.

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

**Management Actions**

PCBs have proven to be among the most persistent organic pollutants in the aquatic environment. Concentrations in aquatic food webs across the state have generally shown gradual declines over the past 30 years in response to the use restrictions and federal ban in the 1970s. However, PCBs are declining at a much slower pace than the legacy pesticides, apparently due to their greater resistance to degradation in the environment. Without drastic action, PCB concentrations in highly polluted ecosystems like San Francisco Bay are likely to remain above thresholds for concern for many decades to come.

The most important management actions ever taken to reduce PCB pollution in California and the rest of the country were the phaseout during the 1970s and the 1979 federal ban on sale and production of PCBs (Figure 3.4.1) (Brinkmann and deKok 1980). These actions led to a rapid decline in the open-ended uses of PCBs (e.g., as a pesticide and paint additive, in carbonless copy paper), and a gradual decline in the inventory of PCBs used in electrical equipment and other applications in the watersheds. However, as mentioned above, despite the 1979 ban, a considerable amount of PCBs remains in use today. The PCB ban has had a significant positive long-term impact, but without further action it appears that the general recovery of California water bodies from PCB contamination will take many more decades.

In the 1980s and 1990s, additional management of PCBs in the state was largely driven by regulations pertaining to the cleanup of highly contaminated sites. PCB hotspots have been remediated under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), or "Superfund". Cleanup of these hotspots undoubtedly has reduced PCB loading to California water bodies, but a review of these actions was beyond the scope of this report.

**Long-term Trends**

*Sport Fish*

If the state's sport fish monitoring program applied a consistent sampling design over the years, trends in the impact of PCBs on fishing in California could be evaluated by comparing historic data to the same concentration categories applied to the recent monitoring data in the previous section (Table 3.4.1, Figures 3.4.3 and 3.4.4). While this type of comparison provides a general picture of PCB impact over the long-term, inconsistencies over the years interfere with finer scale comparisons.

Sampling intensity is one factor that varied over the period of record. Sampling intensity was highest in the most recent interval (251 locations sampled) in spite of this interval being shorter than the others. A comparable number of locations (237) was sampled in the 1988 – 1997 interval, but sampling was less intense during the 1978 – 1987 interval (186 locations).

The percentage of locations in the very high concentration category declined from 12% in the 1978 – 1987 interval to 4% in the most recent interval. In contrast, the proportion of locations in the moderate category was highest (26%) in the recent period. The percentage of locations in the low category was highest in the 1988 – 1997 (82%), but still a majority of the samples in the other periods (66% for both).

These changes in percentages of locations in the four categories were influenced by a combination of gradual declines in PCB concentrations over the 26-year period and the shifting geographic emphasis of sampling during the different periods. In the earliest interval, very high locations (red dots) were present in several parts of the state, including clusters of locations in the Sacramento River watershed, the northern Delta, and inland water bodies in the Los Angeles area. In contrast, in the most recent interval the only cluster of red dots was in San Francisco Bay. Declining concentrations are illustrated by trends in the Delta region, which had a cluster of red dots in the earliest period, but no red dots in the recent period in spite of thorough sampling. The influence of changing geographic emphasis is illustrated by the prominent cluster of red locations in San Francisco Bay in the recent interval and the influence of these locations on the overall statistics, compared to the lack of any points in this area in the earliest interval. If the Bay had been sampled in the 1978 – 1987 period there surely would have been more red dots on the map.

A more precise analysis of long-term trends in PCB concentrations in sport fish can be made at locations where sampling was performed consistently over the years. Unfortunately, this type of sampling was performed in very few cases. The best time series generated from the late 1970s to the present are illustrated

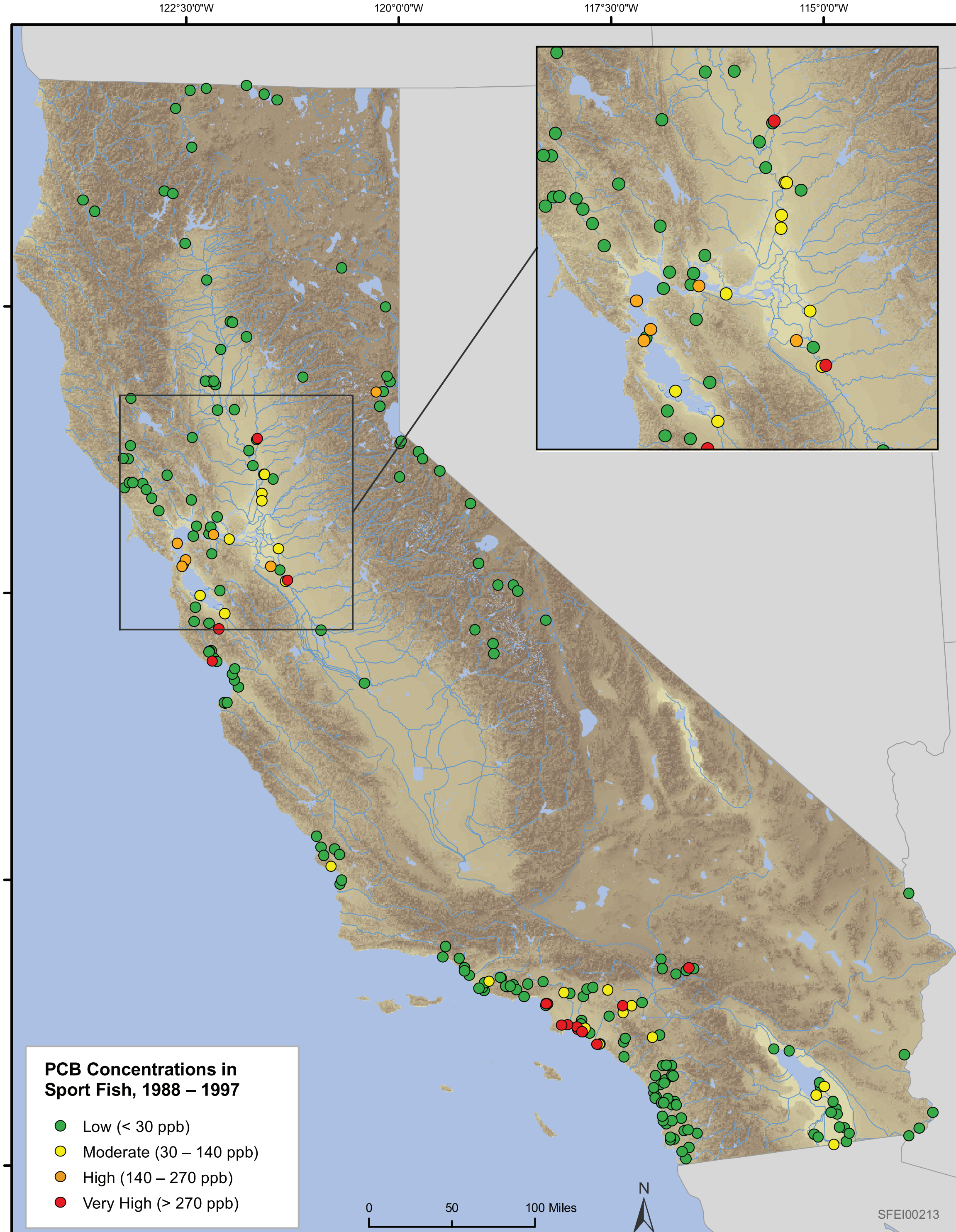


Figure 3.4.3. PCB concentrations in California sport fish, 1988 – 1997. Based on PCB measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

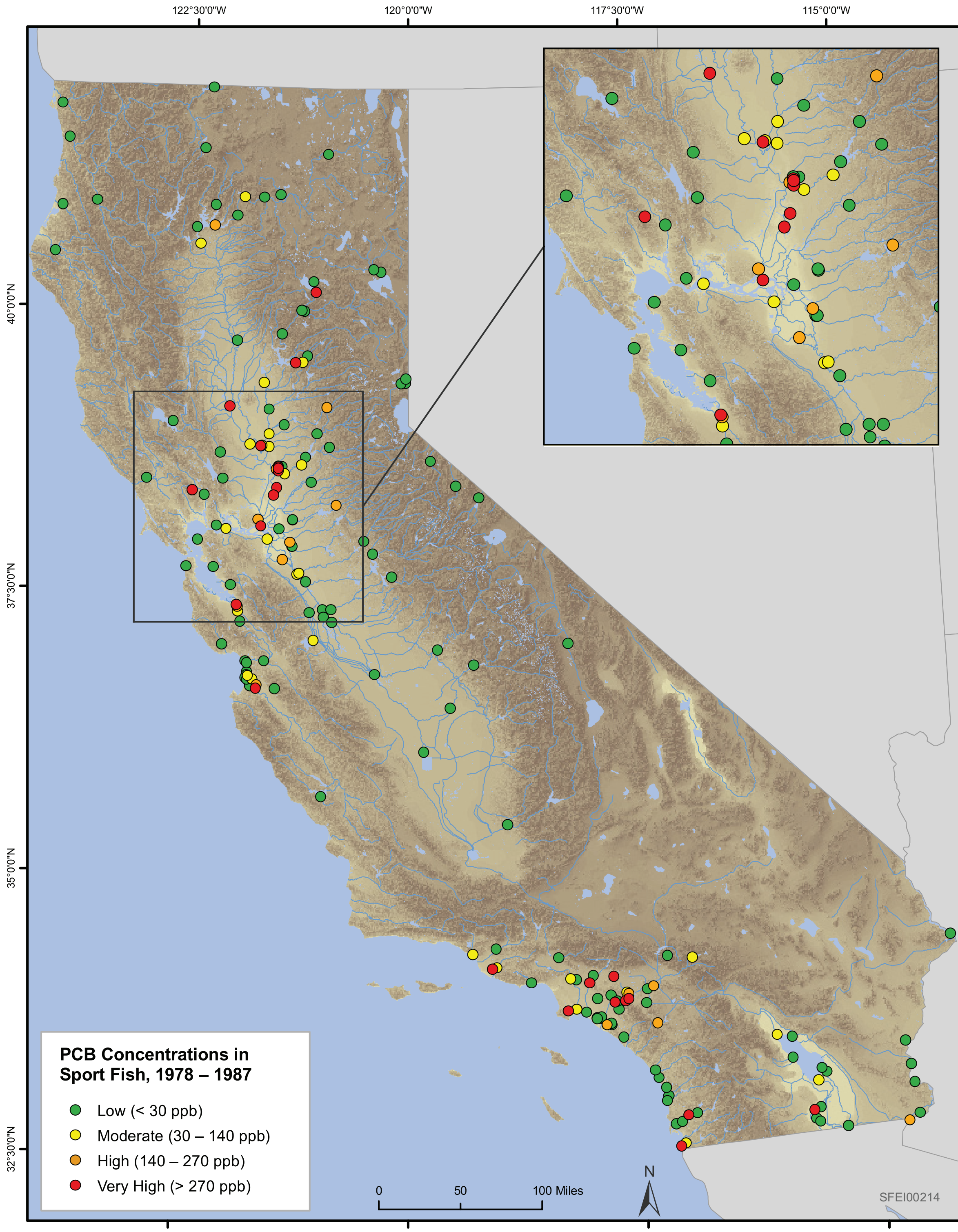


Figure 3.4.4. PCB concentrations in California sport fish, 1978 – 1987. Based on PCB measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

in Figure 3.4.5. This trend dataset has many shortcomings. First, there are very few decent time series with consistent sampling over the period of record. The best time series are for white catfish at the Sacramento River at RM44/Hood (13 observations), red shiner at San Diego Creek at Michelson Drive (19 observations), and channel catfish at New River at Westmorland (14 observations). The other five locations were not sampled adequately to characterize long-term trends. Other problems plaguing these datasets include high MDLs (causing the many "zero" values shown on the graphs) and inconsistent compositing and size ranges in the samples. The mobility of fish populations is another factor that increases interannual variability and decreases the power of fish monitoring as a trend-detection tool.

In spite of these problems with the dataset, the three locations with reasonable time series illustrate what appear to be common scenarios for PCBs. At some locations, concentrations have declined considerably. San Diego Creek at Michelson Drive is an excellent example of a progressive, statistically significant ($p < 0.05$) decline, with over a ten-fold reduction from 1983 to 2001 (Figure 3.4.5). Sacramento River at RM44/Hood is another time series suggesting a decline, but interannual variability was higher at this location and the relationship was not statistically significant. In contrast, the channel catfish time series from New River at Westmorland is characterized by high variability and persistent high concentrations, with high concentrations in recent years suggesting a possible increase in food web PCBs. This scenario seems to apply in other places such as San Francisco Bay, where concentrations in sport fish persist with no obvious indication of decline (discussed below).

*Bivalves*

Bioaccumulation monitoring with bivalves conducted by the State Mussel Watch (SMW) Program and other regional programs is another valuable source of information on long-term trends in pollutant concentrations in California water bodies. Bivalves are an indirect indicator of pollutant impact on the fishing beneficial use, but complement fish monitoring by providing a powerful tool for detecting long-term trends in bioavailable pollutants at precise locations. In contrast to the time series for sport fish, the bivalve data include many robust time series that document statistically significant declines from the late 1970s to the present (Figure 3.4.6). Statistically significant ($p < 0.05$) declines were observed at six of the ten locations included in Figure 3.4.6. The Figure shows the best time series available for different parts of the state. PCB concentrations at long-term monitoring locations in the northern part of the state were generally low, and the trends are obscured by a prevalence of below detection limit results. These sites provide a useful indication of conditions in the California water bodies unaffected by local significant PCB sources. Trends within San Francisco Bay are not shown in Figure 3.4.6 because they are discussed in detail below. Many long-term monitoring sites in southern California exhibited a pattern of decline in PCBs. Six of the seven sites included in Figure 3.4.6 had statistically significant declines. One important exception was San Diego Bay at Harbor Island, which had the highest PCB concentrations of any location in the SMW Program. The most recent sample analyzed at this location had the highest concentration (on a lipid weight basis), indicating that PCBs in this water body are persisting at high concentrations. Two sites are shown for Newport Bay, both of which showed a progressive, significant, ten-fold reduction, though the initial concentrations at each location were quite different.

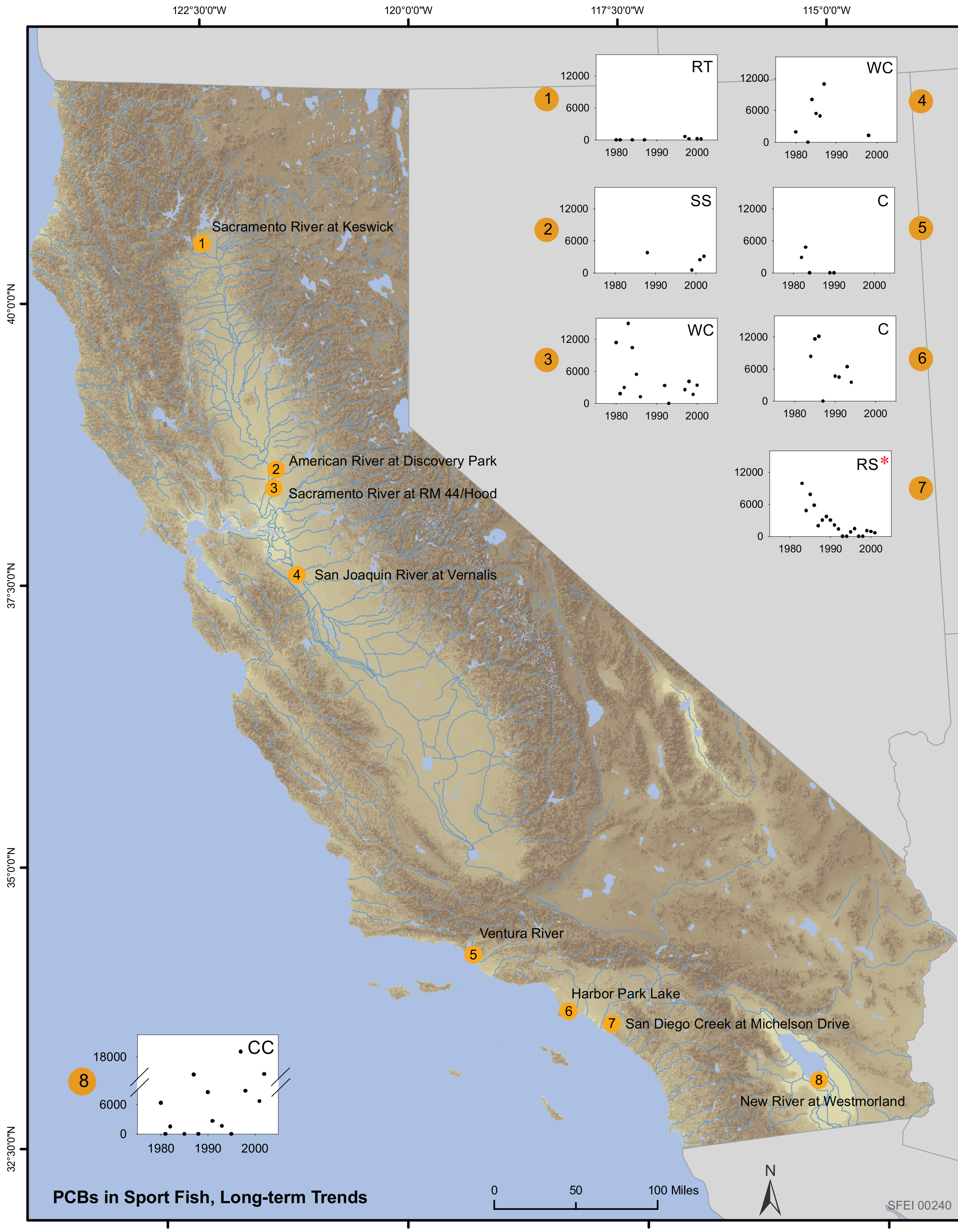


Figure 3.4.5. Long-term trends in PCB concentrations in California sport fish. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight. Species shown are rainbow trout (RT), Sacramento sucker (SS), white catfish (WC), channel catfish (CC), red shiner (RS), and common carp (C).

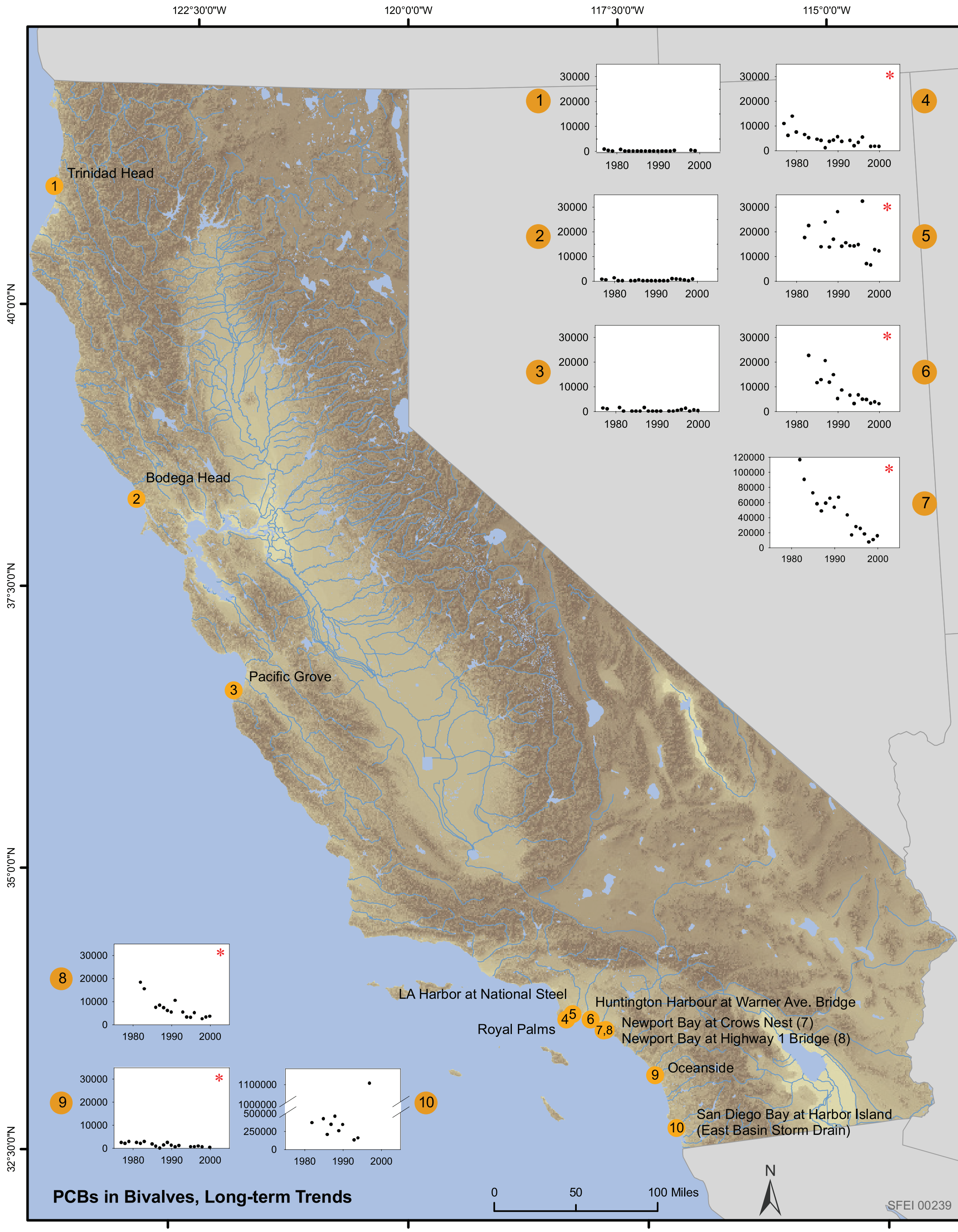


Figure 3.4.6. Long-term trends in PCB concentrations in California mussels measured by the State Mussel Watch Program. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight.

**Case Studies**

*San Francisco Bay*

Long-term trends in the impact of PCBs on fishing are of particular interest in San Francisco Bay due to the persistence of relatively high concentrations in this ecosystem and the relatively thorough monitoring that has been performed over the past decade by the Regional Monitoring Program. The phaseout of PCBs during the 1970s and the 1979 federal ban on sale and production appear to have led to relatively rapid declines in Bay PCBs during the 1970s and early 1980s, followed by a slower trajectory of decline from 1982 to the present. Without further management action it appears that the general recovery of the Bay from PCB contamination will take many more decades. In response to this persistent problem, water quality managers are currently developing a PCB Total Maximum Daily Load (TMDL) and implementation plan to accelerate the recovery of the Bay.

In San Francisco Bay, seven locations sampled by the SMW Program were continued by the Regional Monitoring Program (Davis et al. 2007), and represent the best dataset available on trends in the Bay over the past 20 years (Figure 3.4.7) (Stephenson et al. 1995, Gunther et al. 1999, SFEI 2005b). The trend signals are obscured to some extent by the use of different analytical laboratories and methods. Two distinct general patterns are evident in these data. For the northern Estuary locations (Pinole Point, Richmond Bridge/Red Rock, and Fort Baker/Horseshoe Bay), concentrations declined from approximately 4000 ng/g lipid in 1982 to 1000 ng/g lipid in 2003. For the southern Estuary locations (Treasure Island/Yerba Buena Island, Hunter's Point/Alameda, Redwood Creek, and Dumbarton Bridge), concentrations declined from approximately 6000 ng/g lipid in 1982 to 2000 ng/g lipid in 2003.

Extrapolating these regression lines into the future for southern Estuary locations indicates that a twenty-fold reduction in concentration (the magnitude of reduction needed to bring fish concentrations down below the threshold for concern) will take approximately another 40 years at Yerba Buena Island and Alameda, 80 years at Redwood Creek, and 70 years at Dumbarton Bridge. For the northern Estuary locations, where present concentrations are lower, it will take approximately 45 years at Pinole Point, 40 years at Richmond Bridge/Red Rock, and 25 years at Fort Baker/ Horseshoe Bay to reach 100 ng/g lipid. These are uncertain estimates, based on extrapolation of noisy datasets far into the future.

These estimates are also likely to be lower-bound estimates of time to recovery (in other words, actual recovery is likely to take longer). Food web monitoring data from the Great Lakes indicate that exponential declines with half-lives of a few years are usually good descriptors of PCB trends immediately after active sources have ceased. However, over the long term, processes such as runoff from the watershed, remobilization from sediments, and atmospheric deposition (local urban sources as well as global) begin to dominate the mass budget. This results in a tendency for losses to become balanced with inputs, and the initial rate of decline begins to slow at some point (Devault et al. 1996, Stow et al. 2004). Such processes are likely to reduce the long-term rate of recovery in San Francisco Bay. In addition, as described below, sport fish have not shown similar declines.

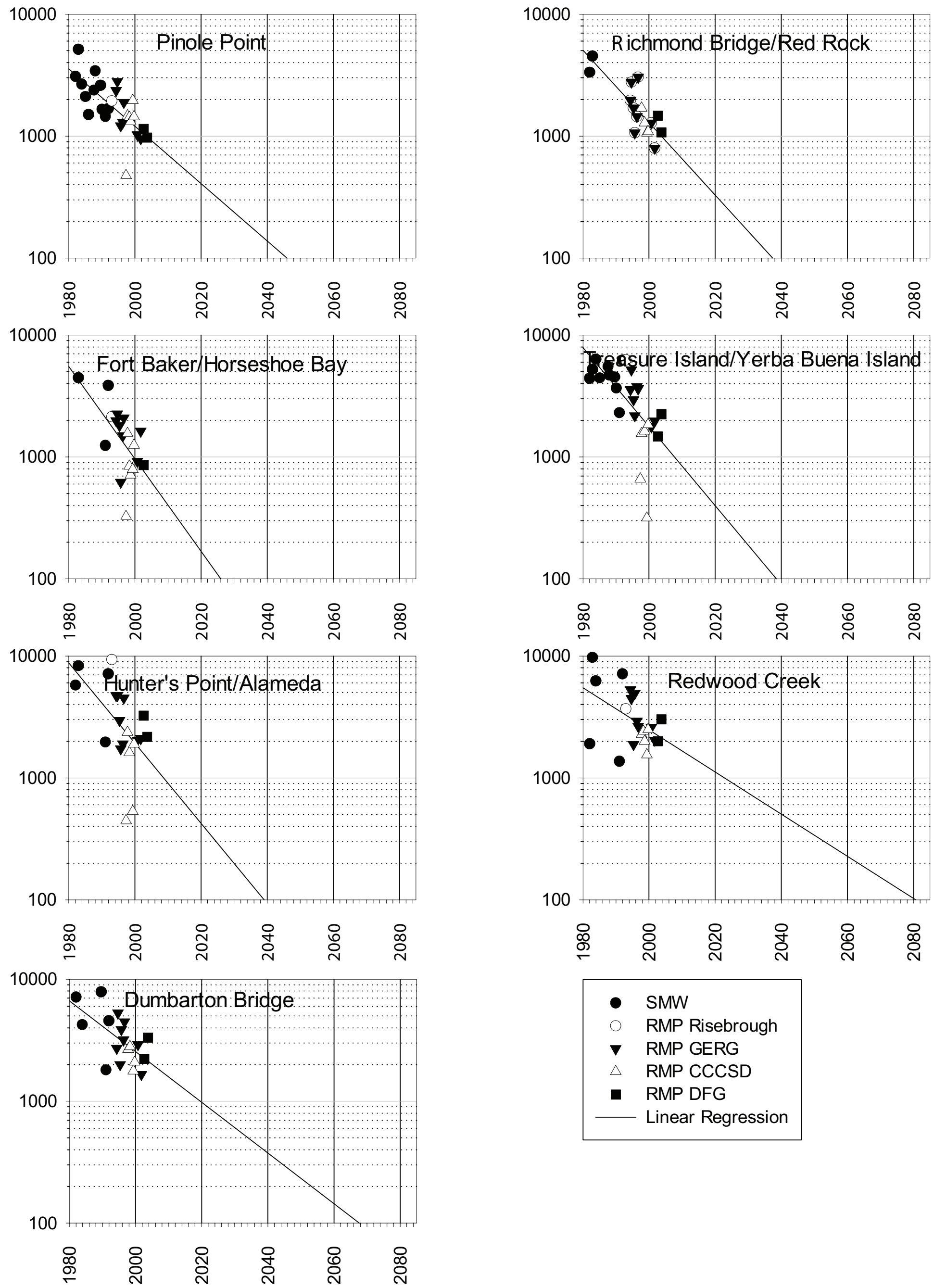


Figure 3.4.7. PCB concentrations in transplanted mussels, 1982 – 2003 (ppb lipid weight). Data from the State Mussel Watch Program as sum of Aroclors and the RMP as sum of congeners. The RMP has used four different analytical labs: Bodega Bay Institute (BBI), Geochemical and Environmental Research Group at Texas A&M (GERG), Central Contra Costa Sanitation District (CCCSD), and Department of Fish and Game (DFG).

Relatively thorough sampling of sport fish has also been conducted in the Bay, primarily over the past decade. The first measurements of PCBs in samples from the Bay were made by Risebrough in shiner surfperch collected in 1965 (Risebrough 1997). Regular sampling of this species on a three-year cycle has been conducted in recent years by the RMP. The mean concentration measured in three composite samples (10 – 15 fish in each) in 1965 was 832 ng/g wet (as Aroclors). In comparison, the Bay-wide median concentration measured in 2003 was 217 ng/g wet (as Aroclors), suggesting a reduction of approximately 74% over this 38 year span. Concentrations in shiner surfperch over the past nine years have shown no clear pattern of decline (Figure 3.4.8 – expressed as sums of congeners). Expressed as sums of congeners on a wet weight basis – most appropriate as an indicator of the status of impairment – Bay-wide medians were nearly identical in 1997, 2000, and 2003 (Figure 3.4.8). Expressed on a lipid weight basis – providing a better index of trends in PCB concentrations in the Bay – Bay-wide medians were highest in 1994 and 2003 (12600 and 10900 ng/g lipid, respectively), and exhibited considerable interannual variation with much lower concentrations in 1997 and 2000 (5200 and 5000 ng/g lipid, respectively). A relatively long time series (data not shown) also exists for white sturgeon in the Bay (1986 – 2003), but sample sizes have been small and relatively high concentrations were observed in the 2003 sampling. Time series for other sport fish species are limited to the 1994 – 2003 period. Concentrations in white croaker, another key indicator species, have also shown no clear pattern of decline from 1994 to 2003. On a wet weight basis, concentrations in white croaker have been quite consistent since 1994, ranging from 191 ng/g wet to 225 ng/g wet (sum of congeners), with the highest median observed in 2003 (Figure 3.4.8). Lipid weight medians have been more variable, ranging from 3800 ng/g lipid in 2003 to 6700 ng/g lipid in 1994 (Figure 3.4.8). Trends in sport fish are a crucial indicator of trends in impairment, but seasonal and interannual variation in fish physiology make them a somewhat unreliable indicator of general trends in Bay contamination, as suggested by the high interannual variance in the lipid-normalized data.

### *Newport Bay*

Newport Bay provides an interesting contrast to San Francisco Bay. Like San Francisco Bay, Newport Bay is a highly urbanized water body that supports a substantial amount of fishing activity and had elevated concentrations of PCBs in the late 1970s (Allen et al. 2004). However, as discussed above, SMW data for bivalves and a recent study of PCBs in Newport Bay fish (Allen et al. 2004) both suggest that concentrations in Newport Bay biota have declined significantly. Of 50 composite samples of sport fish collected in 2000 and 2001, only two samples had concentrations above the 30 ppb threshold used in this report. The maximum concentration observed for any species was 58 ppb. This is much lower than concentrations typically found in white croaker from San Francisco Bay, most of which were above 200 ppb in 2003. In the late 1970s, six species of sport fish had average concentrations above the 30 ppb threshold. It should be noted, however, that in spite of these apparent general declines, recent TSMP sampling in Newport Bay has still found some fish samples with high PCBs – including one with 172 ppb from Upper Newport Bay in 2002. In the TMDL for Newport Bay (USEPA 2002), it is hypothesized that PCB spills at Air Stations and hazardous waste sites in the watershed were the historic sources of PCBs to Newport Bay. The long-term trend data suggest that the PCB ban and the regulations that have resulted in the reduction or elimination of spills have been sufficient to allow a relatively rapid rate of recovery of Newport Bay.

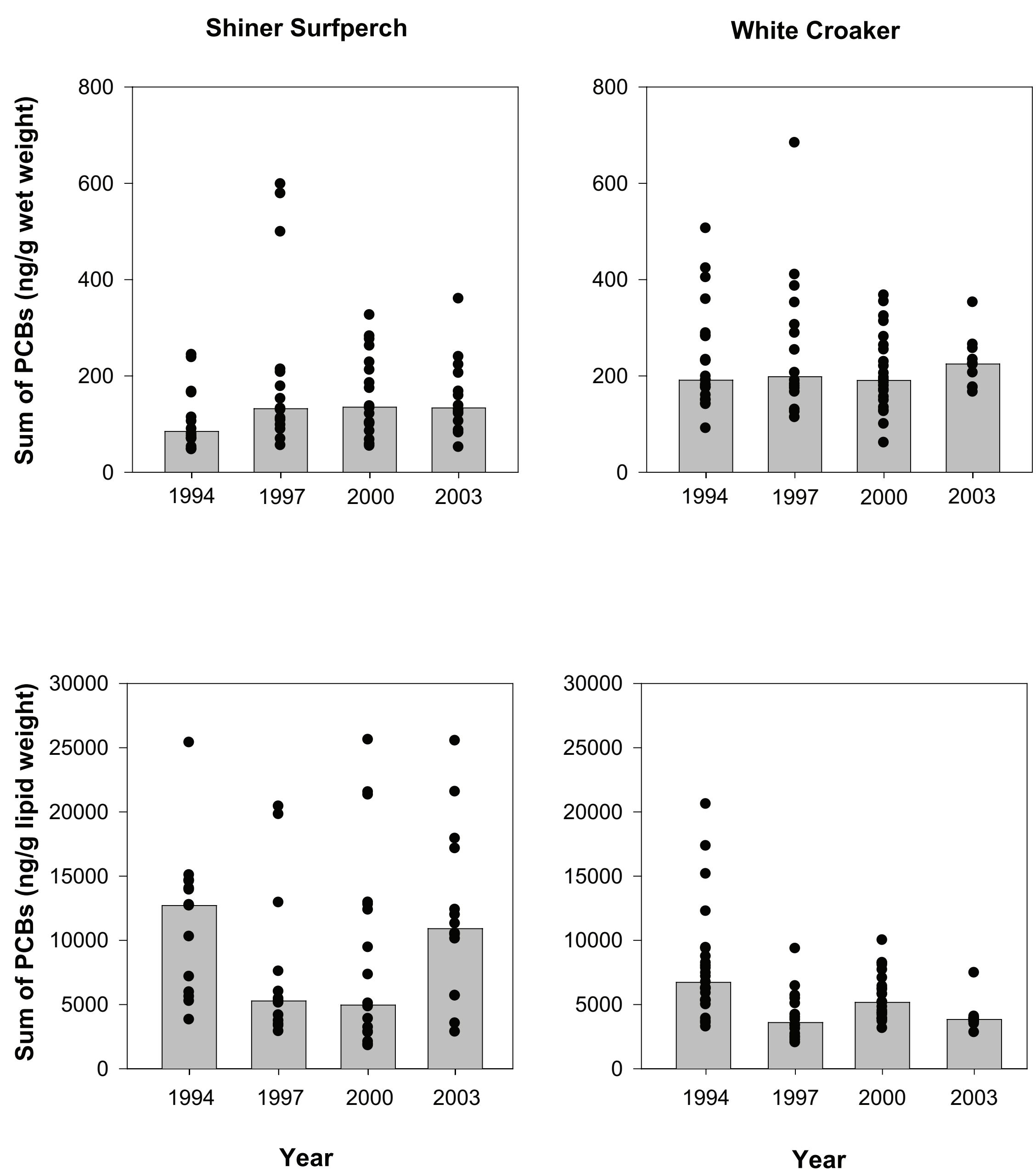


Figure 3.4.8. PCB trends in shiner surfperch and white croaker, 1994 – 2003. Expressed as sum of congeners on a wet weight basis (upper plots) and a lipid weight basis (lower plots). From Davis et al. (2006).

## c. Sources and Pathways

The geographic distribution of PCBs measured in California sport fish provides an indication of the location and nature of the principal sources of these chemicals (Figures 3.4.9, 3.4.10, 3.4.11). High concentrations of PCBs are typically found in areas where historic use or maintenance of electrical equipment that contained PCBs occurred. These areas tend to be concentrated in urban centers with high amounts of industrial activity, but also occur in scattered areas across the landscape where electrical equipment or other PCB-containing equipment was used. PCBs were also used as a vehicle in pesticide mixtures, so in some cases their appearance in agricultural areas may be related to that practice. PCBs are additionally transported around the globe through atmospheric processes, leading to a low level background of contamination even in remote areas.

In the earliest time period (1978 – 1987), high concentrations of PCBs occurred primarily near urban areas (e.g., Sacramento, Los Angeles, and San Diego) but there were also elevated concentrations observed in rural areas. Most prominent in Figure 3.4.11 was an extremely high concentration (7,700 ppb wet wt) measured on the south fork of the Feather River at Forbestown, where a PCB spill from a hydroelectric facility (Forbestown Powerhouse) occurred (CVRWQCB 1987). Sampling at this location was only conducted in 1980, so the recovery of this area has not been documented. PCBs were commonly used in electrical equipment, so the many hydroelectric facilities in the state are potential sites of past or even present PCB contamination. Notably absent in this time period are any data from San Francisco Bay, so the lack of an urban signal in this period is partially due to incomplete sampling. The low concentrations of PCBs observed in many parts of the state away from urban centers indicate the weak influence of global atmospheric transport on PCBs in California fish.

In the 1988 – 1997 and 1998 – 2003 periods, concentrations away from urban centers were reduced relative to the earliest interval, suggesting a decline in sources in these areas. In 1988-1997, the largest cluster of locations with relatively high concentrations was in the Los Angeles/Orange County area. In 1998 – 2003, sampling in San Francisco Bay identified this water body as the broadest area in the state with relatively high PCB concentrations.

In general, these data suggest that PCB sources to California water bodies have diminished considerably over the past 25 years. Regions that were highly contaminated in the 1970s and early 1980s generally now have much lower concentrations. A prominent exception is San Francisco Bay, where concentrations remain elevated. Possible hypotheses for the unique behavior of San Francisco Bay include the erosional sediment regime in the Bay that is uncovering contaminated layers of buried sediment, the long residence time of sediment particles in the Bay, continuing inputs from local watersheds into the Bay, or perhaps a combination of two or more of these factors.

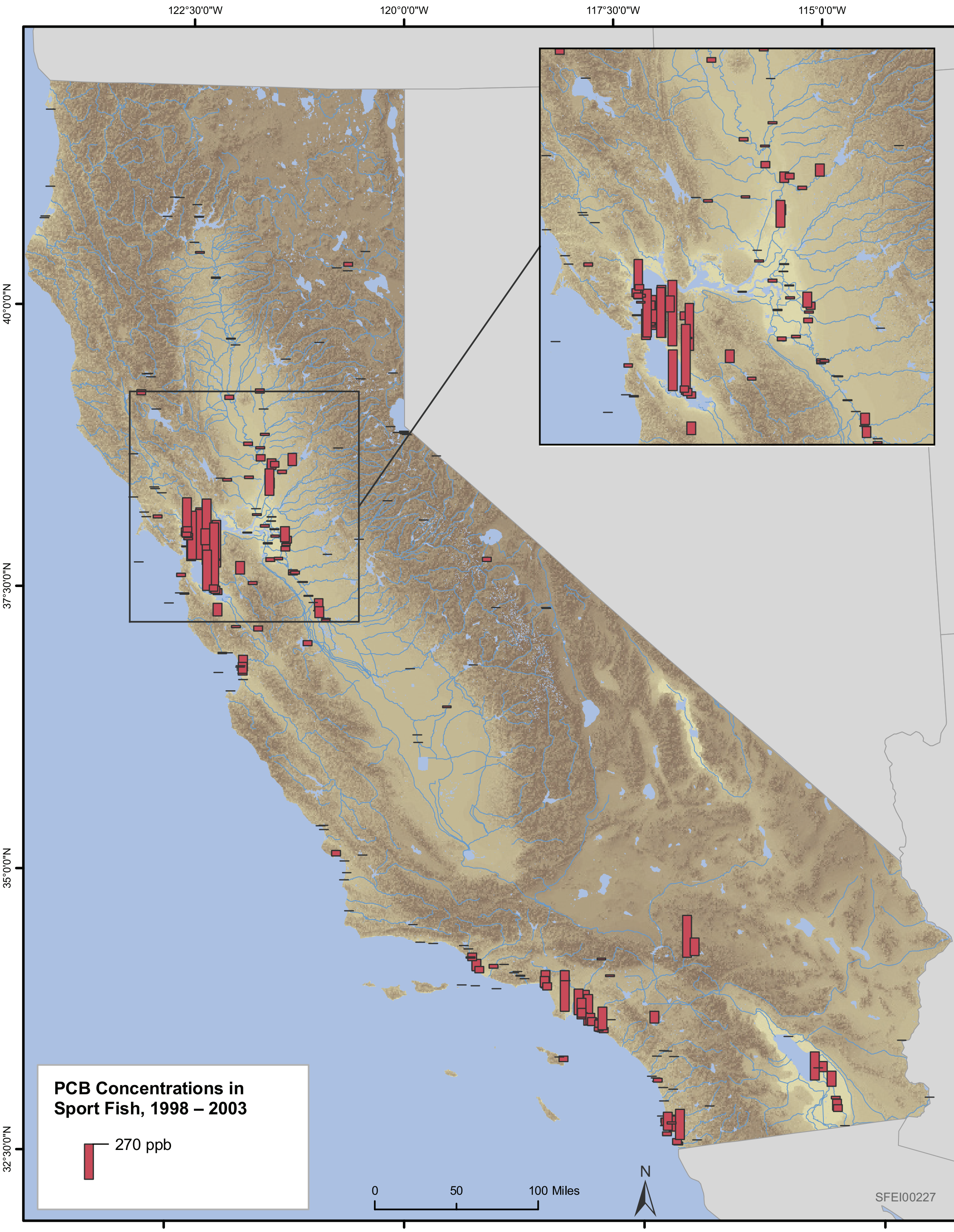


Figure 3.4.9. PCB concentrations (as sums of Aroclors or congeners, depending on the data source) in California sport fish, 1998 – 2003. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

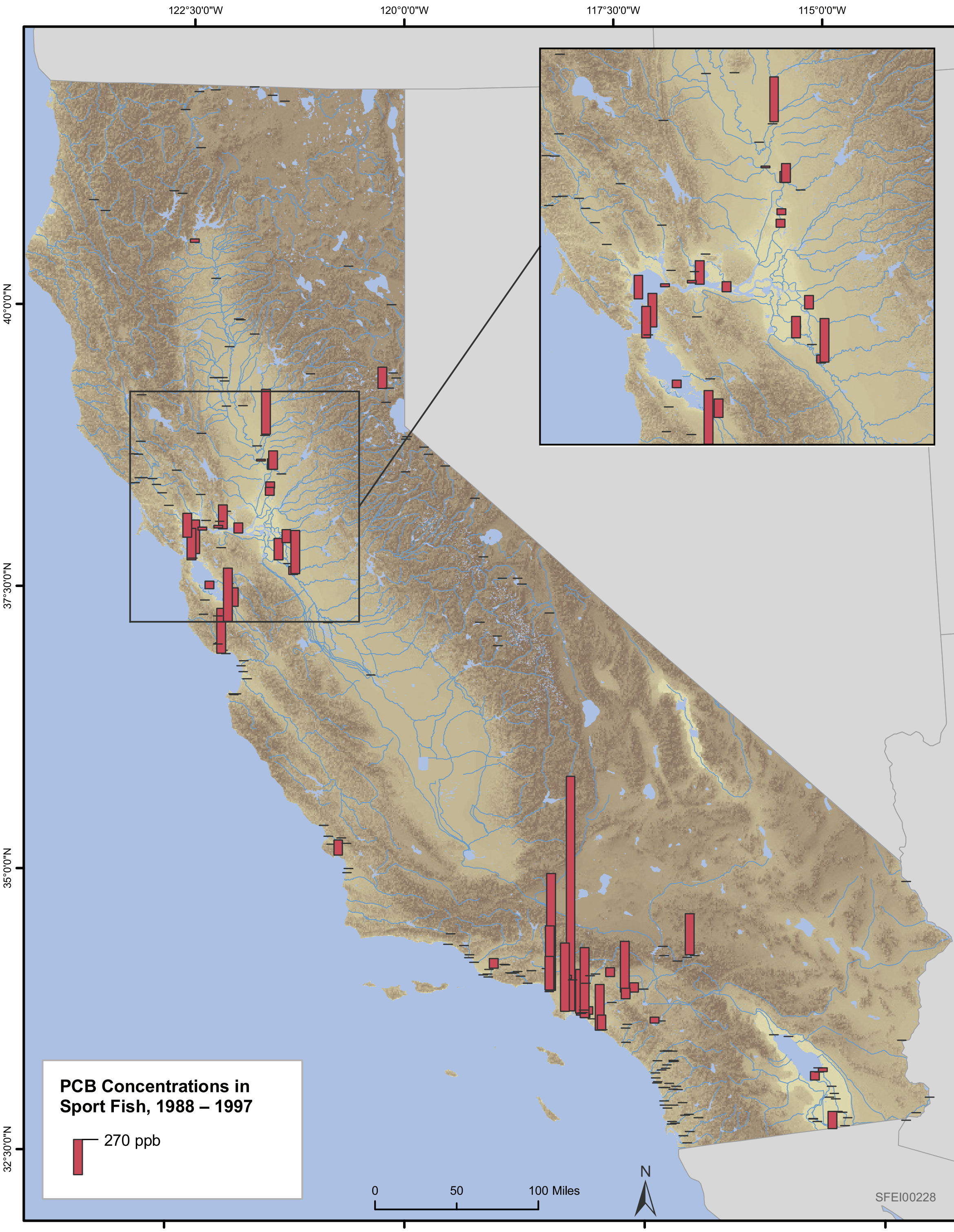


Figure 3.4.10. PCB concentrations (as sums of Aroclors or congeners, depending on the data source) in California sport fish, 1988 – 1997. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

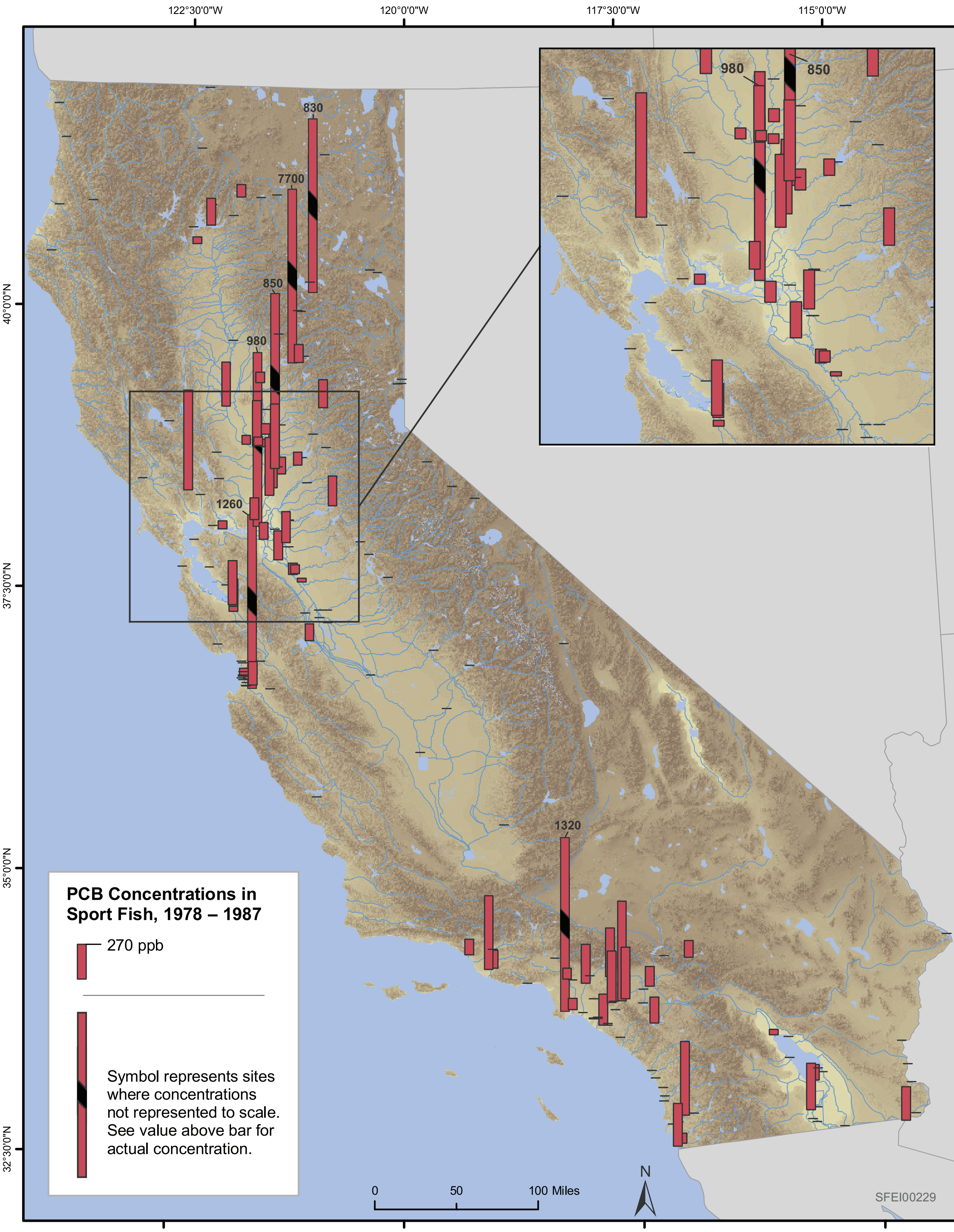


Figure 3.4.11. PCB concentrations (as sums of Aroclors or congeners, depending on the data source) in California sport fish, 1978 – 1987. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

### 3.4.3. Impact of PCBs on Aquatic Life in California

The limited information available on the effects of PCB bioaccumulation on aquatic life in California water bodies suggests that present concentrations may approach thresholds for concern. Considered against the backdrop of steadily declining concentrations across the state, this suggests that impacts were likely in the past when PCB concentrations were substantially higher. Currently, San Francisco Bay appears to be the ecosystem in California with the most severe and persistent PCB contamination – a detailed review of evidence for impacts on aquatic life in the Bay is therefore presented below.

**Statewide Assessment**

The impacts of pollutant bioaccumulation on the aquatic life beneficial use are best evaluated using a different set of indicators than the sport fish that are used to assess impacts on the fishing beneficial use. The best bioaccumulation indicators for aquatic life assessment are linked as directly as possible to exposure and effects in sensitive species. Since exposure and the potential for effects are often greatest in predators at the top of the food web, indices of exposure of piscivorous wildlife are commonly employed. In piscivorous birds, pollutant concentrations in eggs and in prey fish (analyzed as whole fish) are often measured to assess risks. In piscivorous marine mammals, risks are often assessed through analysis of blubber, blood, and prey fish. Since sport fish are typically larger than the prey fish consumed by wildlife and since only sport fish muscle is analyzed, sport fish data are not very well suited to assessment of wildlife risks.

Unfortunately, no sufficiently systematic monitoring has been conducted to support a statewide analysis of the impact of PCBs on aquatic life in California. Although the sport fish data are not appropriate for a rigorous assessment of aquatic life impacts, they can be used to provide a gross evaluation of potential risks. Birds are the most common fish predators in water bodies across the state, and therefore are the focus of the following discussion of risks to predators.

Studies of PCB movement through food chains have shown that eggs of piscivorous birds generally have total PCB concentrations that are 10 to 30 times higher than their prey. This ratio of concentration in predator and prey is known as a biomagnification factor (BMF). Glaser and Connolly (2002) compiled egg:prey ratios for several piscivorous species that ranged from 10 to 20, with variation among species and among different populations of the same species. For herring gulls in Lake Ontario, data from Braune and Norstrom (1989) indicated a ratio of 32 between eggs and prey (alewife). In order to use sport fish concentration data to evaluate risks to piscivorous birds, BMFs comparing sport fish to bird eggs are needed. These BMFs are not commonly reported in the literature. Data from San Pablo Bay (a sub-embayment of San Francisco Bay) indicate BMFs between cormorant eggs and sport fish ranging from 19 to 44 for seven sport fish species (Davis et al. 1999, 2004). Considering all of this information, a BMF of 30 seems appropriate as a protective ratio to use for converting sport fish concentrations to bird egg concentrations as part of a preliminary screening of potential risks to birds.

Ten locations in the recent sampling period (1998 – 2003) were found to have sport fish with concentrations above 270 ppb (or 0.27 ppm). Seven of these locations were within San Francisco Bay, where evaluations of bird eggs have concluded that PCB concentrations are near thresholds for effects (Davis et al. 2007). The remaining three locations were in Lake Chabot (Bay Area), Machado Lake (Los Angeles area), and Big Bear Lake (San Bernardino County). Using a BMF of 30, eggs of birds from these water bodies might be expected to have concentrations of 8 ppm or higher. A review by Hoffman et al. (1986) concluded that concentrations in the range of 8 to 25 ppm in eggs can lead to decreased hatching success for cormorants, terns, doves, and eagles. A study of cormorants (Yamashita et al. 1993) found an increased incidence of deformities beginning at 3.6 ppm. Water bodies assigned to the "very high" category for sport fish contamination therefore might also be expected to be at or slightly above the threshold for effects on piscivorous birds. Another 11 water bodies were assigned to the "high" category for sport fish (with PCB concentrations between 140 and 270 ppb) and would be predicted to have concentrations of approximately 4 to 8 ppm in bird eggs, which is still in a range where concerns exist for avian reproduction. Most (66%) of the locations sampled fell into the "low" category for sport fish (less than 30 ppb), which would correspond to egg concentrations of less than 0.9 ppm and low concern for risks to birds.

**San Francisco Bay**

Davis et al. (2007) and Thompson et al. (2007) recently reviewed the evidence for effects of PCB bioaccumulation on wildlife in San Francisco Bay. Several sources of information indicate that PCB concentrations in the Bay may be high enough to adversely affect wildlife, including rare and endangered species. Fish-eating species at the top of the food web generally face the greatest risks. Populations residing in PCB hotspots also face relatively high risks.

*Birds*

Studies of PCBs in eggs of the endangered California clapper rail, the endangered California Least Tern, and Double-crested Cormorants have found concentrations that are near the threshold for embryo mortality.

One study in the 1980s suggested that PCBs were adversely affecting Bay birds. Hoffman et al. (1986) found a negative correlation between PCB concentrations in eggs and embryo weights in Black-crowned Night Herons collected from Bair Island in 1983. PCB concentrations in these eggs ranged from 0.75 to 52 ppm wet weight. In the South Bay in 1982, three species, Caspian Tern (Sterna caspia), Forster's Tern (Sterna fosteri), and Snowy Egret (Egretta thula), showed organic contaminant concentrations similar to those of the night herons (Ohlendorf et al. 1988).

Several more recent studies of PCBs in Bay birds have found concentrations that were at or near the threshold for embryo mortality. Davis (1997) and Davis et al. (2004) studied Double-crested Cormorants as an indicator of PCB accumulation and effects in the open waters of San Pablo Bay. In samples collected in 1995, PCB concentrations in embryo yolk sacs from this colony were correlated with reduced egg mass, reduced embryo spleen mass, and induced cytochrome P450 in embryo livers (Davis 1997). The degree of cytochrome P450 induction in these embryos appeared to be just above the threshold for causing embryo mortality (Davis et al. 1997). Davis et al. (2004) measured PCB concentrations in freshly laid eggs. Concentrations observed in this study overlapped the lower end of the effects range for this species, with a maximum of 3800 ppb fresh wet weight observed in a composite sample from 2001. These studies indicated that PCB concentrations in the 1990s were still high enough to elicit measurable effects, but probably not high enough to have a significant impact on the viability of the Bay cormorant population.

Recent work on Caspian Terns (Sterna caspia), Forster's Terns (Sterna forsteri), and the endangered California Least Tern (Sterna antillarum browni) have found concentrations that approach thresholds for effects in these species (Adelsbach et al. 2003). Average PCB concentrations in eggs collected in 2001 from colonies distributed throughout the Bay were 1.6 ppm fresh wet weight (fww) in Caspian Terns, 2.0 ppm fww in Forster's Terns, and 2.7 ppm fww in Least Terns. The Least Terns forage in an area near one of the Bay's PCB hotspots, and probably represent a worst case scenario (high concentrations in the local habitat, high trophic level, threatened population) for possible PCB impacts on an avian population in the Bay.

Schwarzbach et al. (2001) examined organochlorines and eggshell thickness in California Clapper Rail eggs collected from South Bay marshes in 1992. PCBs, while elevated in one egg, were generally below effects thresholds, but the mean concentration observed in 1992 (1.30 ppm fww) had not declined from the mean concentration observed in 1986 (0.82 ppm fww). The authors concluded that PCBs in 1992 may still have been high enough in some rail eggs to produce embryotoxic effects.

*Seals*

PCB concentrations in Bay harbor seals (Phoca vitulina) are elevated in comparison to other parts of the world and a cause for concern for seal health. Risebrough et al. (1980) were the first to investigate the potential impacts of contaminants on Bay seals. PCB concentrations in some of the seals they analyzed were considerably elevated (up to 500 ug/g lipid in blubber) and comparable to concentrations that were later observed to cause reproductive problems in controlled feeding studies (Reijnders 1986).

In response to the slow recovery of the Bay harbor seal population, Kopec and co-workers (Kopec and Harvey 1995, Young et al. 1998) reexamined the potential influence of pollutants on this species. PCB concentrations (sum of congeners) in whole blood of 14 seals sampled in South Bay in 1991 – 1992 (averaging 50.5 ppb wet wt) were higher than the concentrations observed in the feeding studies of Reijnders (1986) and high relative to concentrations observed in harbor seals in other locations around the world. Data from this research suggested the possibility of contaminant-induced anemia, leukocytosis, and disruption of vitamin A metabolism in the Bay seal population.

To further explore the possibility of contaminant-induced health alterations in this population, Neale and co-workers (Neale 2004, Neale et al. 2005) measured blood concentrations of PCBs and other pollutants in Bay seals, examined relationships between pollutant exposure and several key natural blood parameters, and compared PCB concentrations in 2001 – 2002 with concentrations determined in Bay seals in the early 1990s. PCBs in harbor seal blood (defined as the sum of six congeners measured in both studies) declined significantly between the early 1990s and 2001–2002 (from 27 ppb wet to 18 ppb wet), but remained high enough that reproductive and immunological effects were considered possible. PCB concentrations in the Bay were higher than concentrations in Alaska and Monterey Bay. A positive association was found between leukocyte counts and PBDEs, PCBs, and DDE. The authors concluded from these studies that individual seals with high contaminant burdens could experience increased rates of infection and anemia.

Another recent study examined PCB exposure and health risks in harbor seals through modeling PCB movement through the Bay food web (Gobas and Arnot 2005). The authors concluded that there is a substantial probability that risk thresholds for seals are currently exceeded in the Bay. Based on current PCB concentrations in the sediments of the Bay, the probability that PCB concentrations exceed the threshold effects concentration in harbor seals was estimated to be 70 to 73% for male harbor seals and 56% for female harbor seals.

*Fish*

The most intensive study of PCB effects in Bay fish to date was performed in the 1980s (Spies and Rice 1988), and showed a negative correlation between PCB concentrations and survival of starry flounder embryos based on specimens collected in 1983 – 1985. No additional significant work was conducted on the possible effects of PCBs on Bay fish until the late 1990s, when a study by Ostrach and co-workers (SFEI 2005a) found developmental abnormalities in striped bass larvae that appeared to be associated with elevated concentrations of PCBs and other pollutants in eggs.

*Summary*

PCB concentrations in some Bay wildlife species appear to be above or near thresholds for effects. Given the long-term general trend of slow decline in PCBs in the Bay, concentrations should gradually fall below these thresholds. However, a major uncertainty with regard to PCB effects on wildlife is the extent to which PCBs combine with other stressors, such as other contaminants, diseases, or food shortage, to impair sensitive life-history processes such as reproduction, development, sexual differentiation, and growth. It is possible that the effects of PCBs on wildlife, in combination with other stressors, may be significantly greater than currently realized.

**Other Locations**

*Southern California Bight*

An extensive study was conducted in 1998 to examine concentrations of PCBs and other pollutants in fish that are potential wildlife prey in the Southern California Bight (Allen et al. 2002). This study focused on potential impacts on wildlife, and conducted whole-body analysis of fish in the sanddab guild. A total of 225 locations on the southern California shelf were sampled (Figure 3.4.12). The study found that PCB concentrations exceeded published risk guidelines for mammals in 8% of the area sampled and for birds in 5% of the area. The study also provided valuable recent data on the spatial distribution of PCBs in the region. Relatively high concentrations were observed in ports (median of 156 ppb wet weight for three

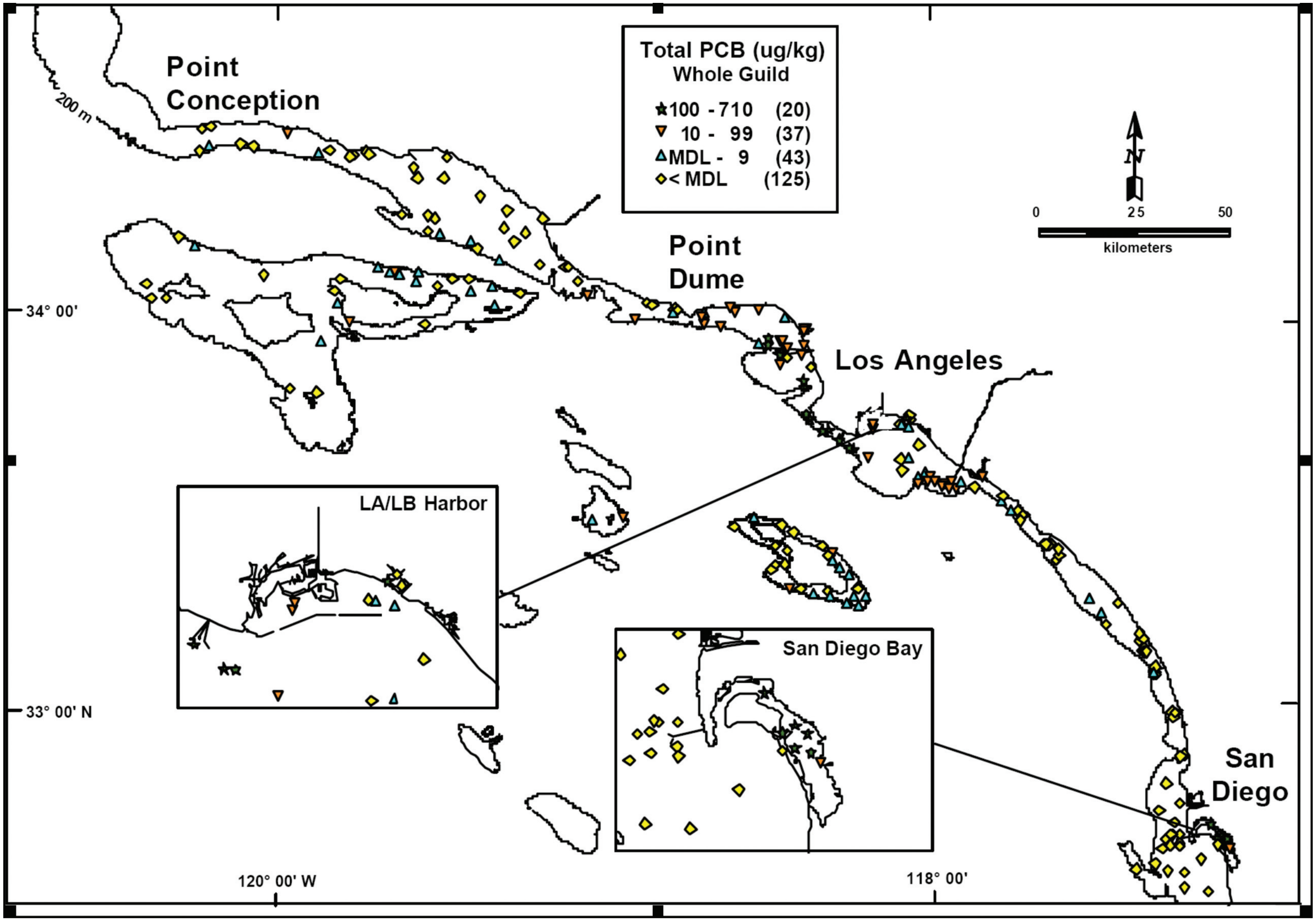


Figure 3.4.12. Distribution of total PCB concentrations in sanddab guild fishes on the southern California shelf at depths of 2-202 m, July-September 1998. From Allen et al. (2002).

samples), other parts of bays and harbors, and near outfalls of large POTWs on the continental shelf (median of 28 ppb for 32 samples). Similar to the studies in San Francisco Bay, this study indicated that PCB concentrations in the Bight in 1998 continued to pose some risk to wildlife predators.

*Central Coast*

Several studies of marine mammals along the Central Coast have found that PCBs are elevated in some individuals and may be high enough to have adverse impacts. Bacon et al. (1999) reported concentrations of PCBs in livers of stranded sea otters from the California coast from 1988 to 1991. Concentrations in these animals (n = 9) averaged 190 ppm PCBs on a wet weight basis and approximately 300 pg/g TEQ on a lipid weight basis. Concentrations in California were much higher than a location in southeast Alaska. Kannan et al. (2004) also measured PCBs in stranded sea otters collected from coastal locations between Half Moon Bay and Morro Bay from 1992 to 1996. PCB concentrations in the livers of some of these otters were higher than those observed by Bacon et al. (1999) and at or above a reported threshold for toxic effects in aquatic mammals (520 pg TEQ/g lipid). The average concentration for all of the otters analyzed was double the effects threshold. In related work, Nakata et al. (1998) provided additional data on PCBs in livers from stranded sea otters from the central California coast for 1992 to 1996. Though sample sizes were small, they found that Monterey Harbor had the highest concentrations, and that some of the otters they examined had concentrations above a threshold for effects in mink. Kajiwara et al. (2001) measured PCBs and other chemicals in California sea lions, elephant seals, and harbor seals stranded on the northern and central California coast from 1991 – 1997. Concentrations of PCBs in blubber or livers of some individuals of all three species were greater than estimated effects threshold concentrations (Kannan et al. 2000).

### 3.4.4. PCB Summary

The present impact of PCB bioaccumulation on fishing and aquatic life in California water bodies is moderately significant. In the most recent sport fish monitoring (from 1998 – 2003), 34% of the locations sampled had moderate, high, or very high PCB concentrations. The highest PCB concentrations are in a range where OEHHA discourages consumption for women of childbearing age and children 17 and younger (Klasing and Brodberg 2006). PCB concentrations in some areas also appear to be high enough to cause adverse impacts in wildlife. Concentrations are highest in water bodies near major urban centers, including the Bay Area, Sacramento, Los Angeles, and San Diego. PCB concentrations in San Francisco Bay are particularly high and appear to be unusually persistent. In general, PCB concentrations appear to be steadily declining across the state. The 1979 ban on PCB sale and production and other regulations relating to disposal of PCBs appear to have generally been effective at reducing the impact of PCBs in California water bodies. In some locations, however, particularly San Francisco Bay, recovery from PCB contamination appears likely to take many decades unless significant actions are taken to reduce continuing inputs. A PCB TMDL for San Francisco Bay is under development to identify appropriate management actions.

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## 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

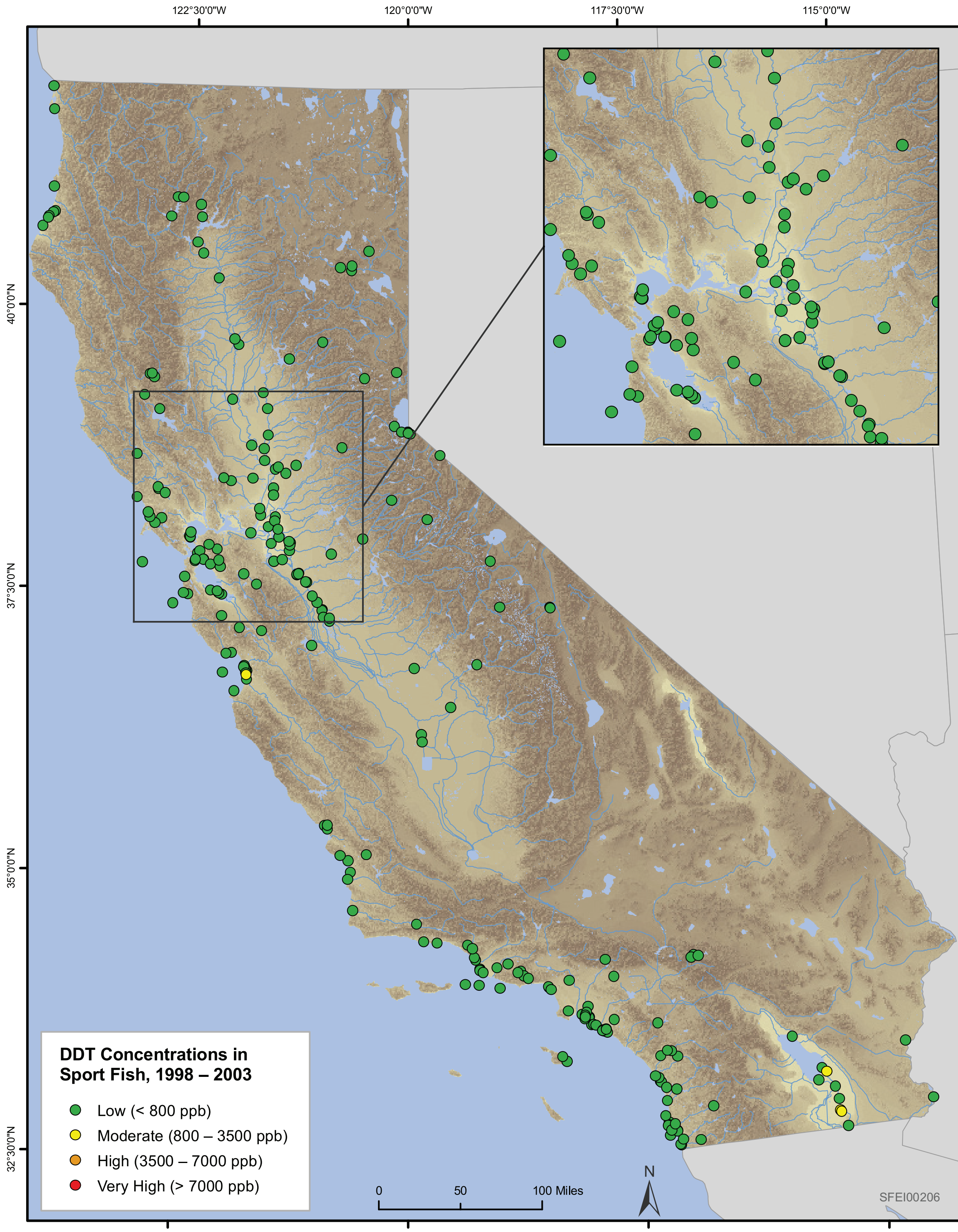


Figure 3.5.1. DDT concentrations in California sport fish, 1998 – 2003. Based on DDT measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

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-

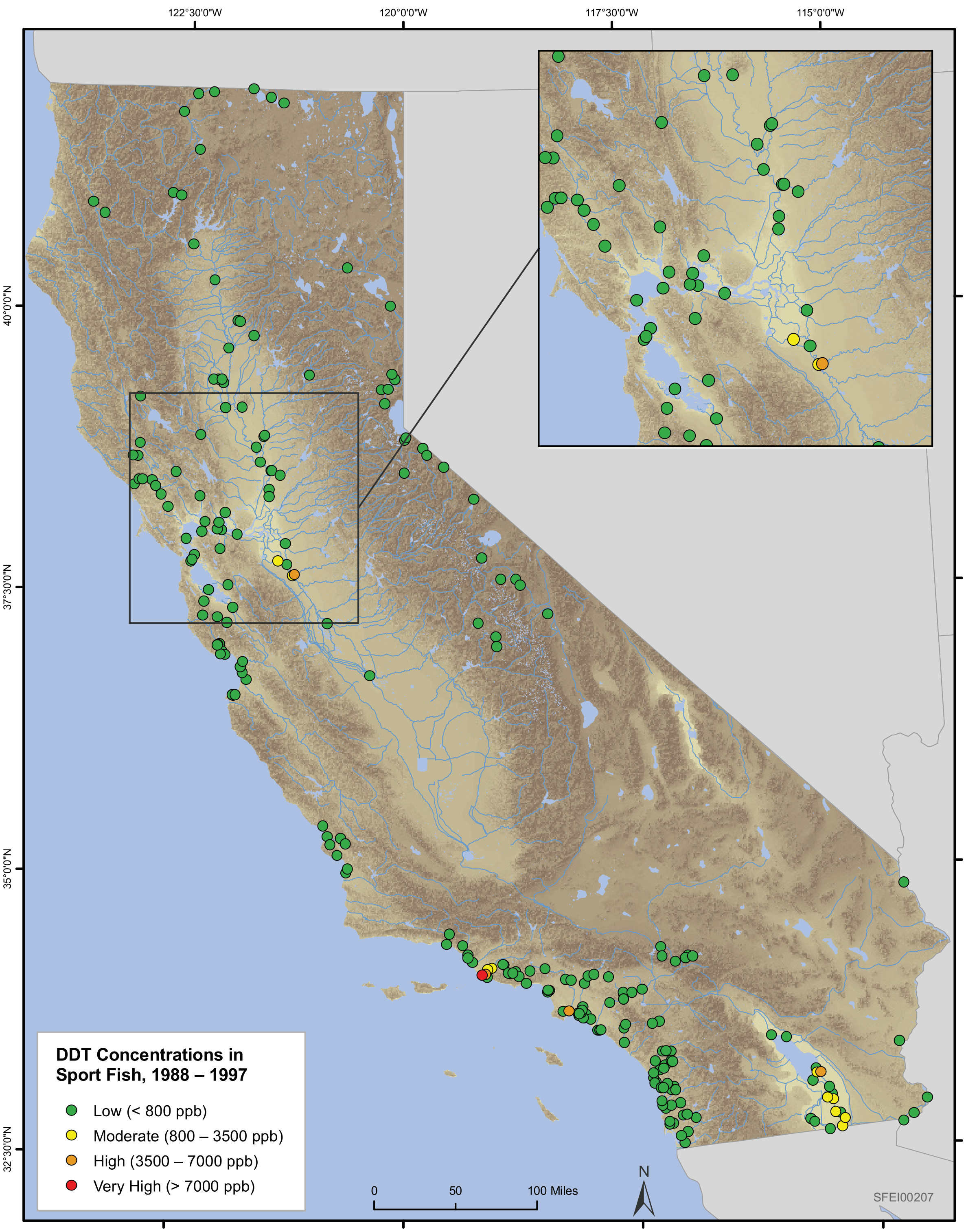


Figure 3.5.2. DDT concentrations in California sport fish, 1988 – 1997. Based on DDT measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

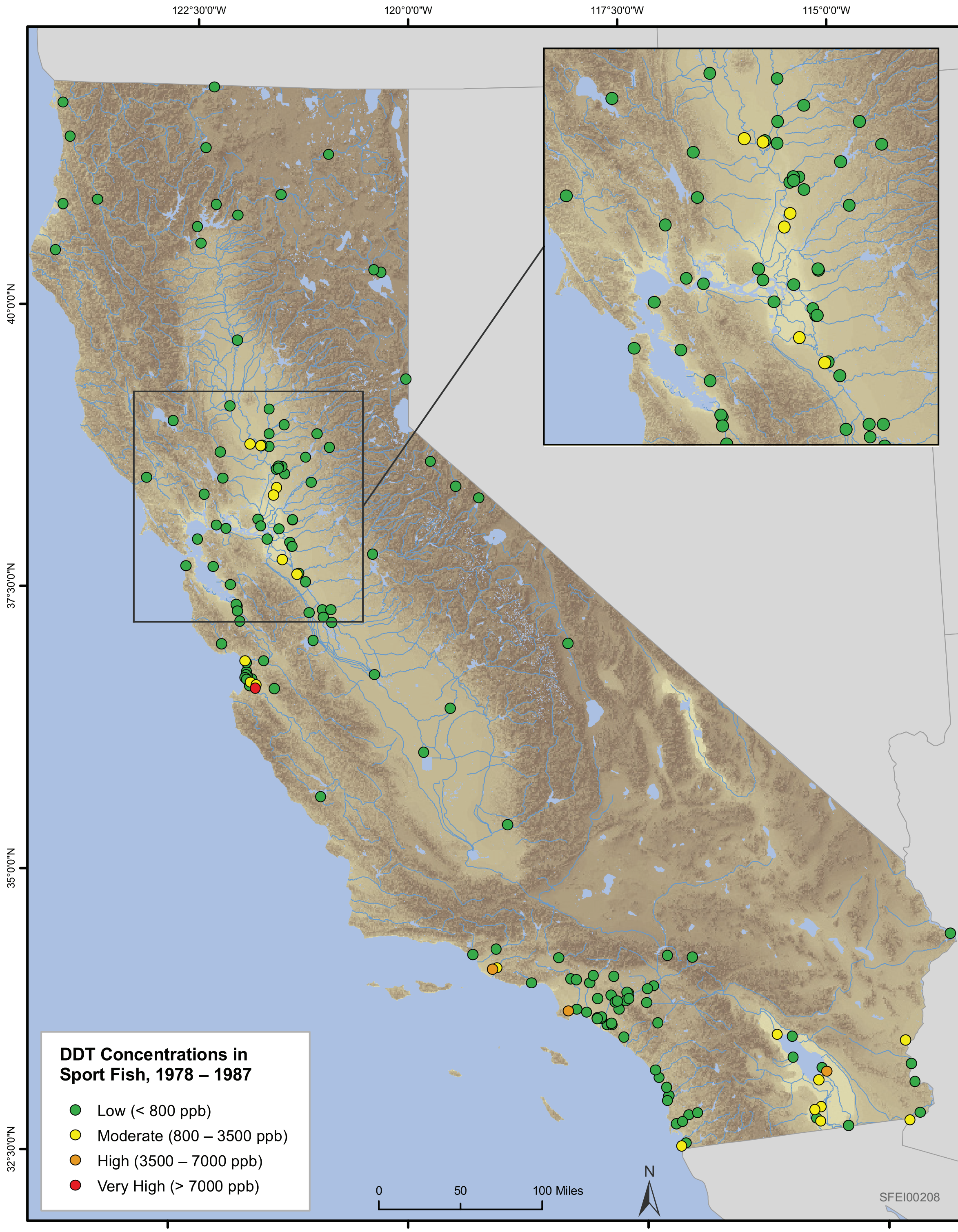


Figure 3.5.3. DDT concentrations in California sport fish, 1978 – 1987. Based on DDT measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

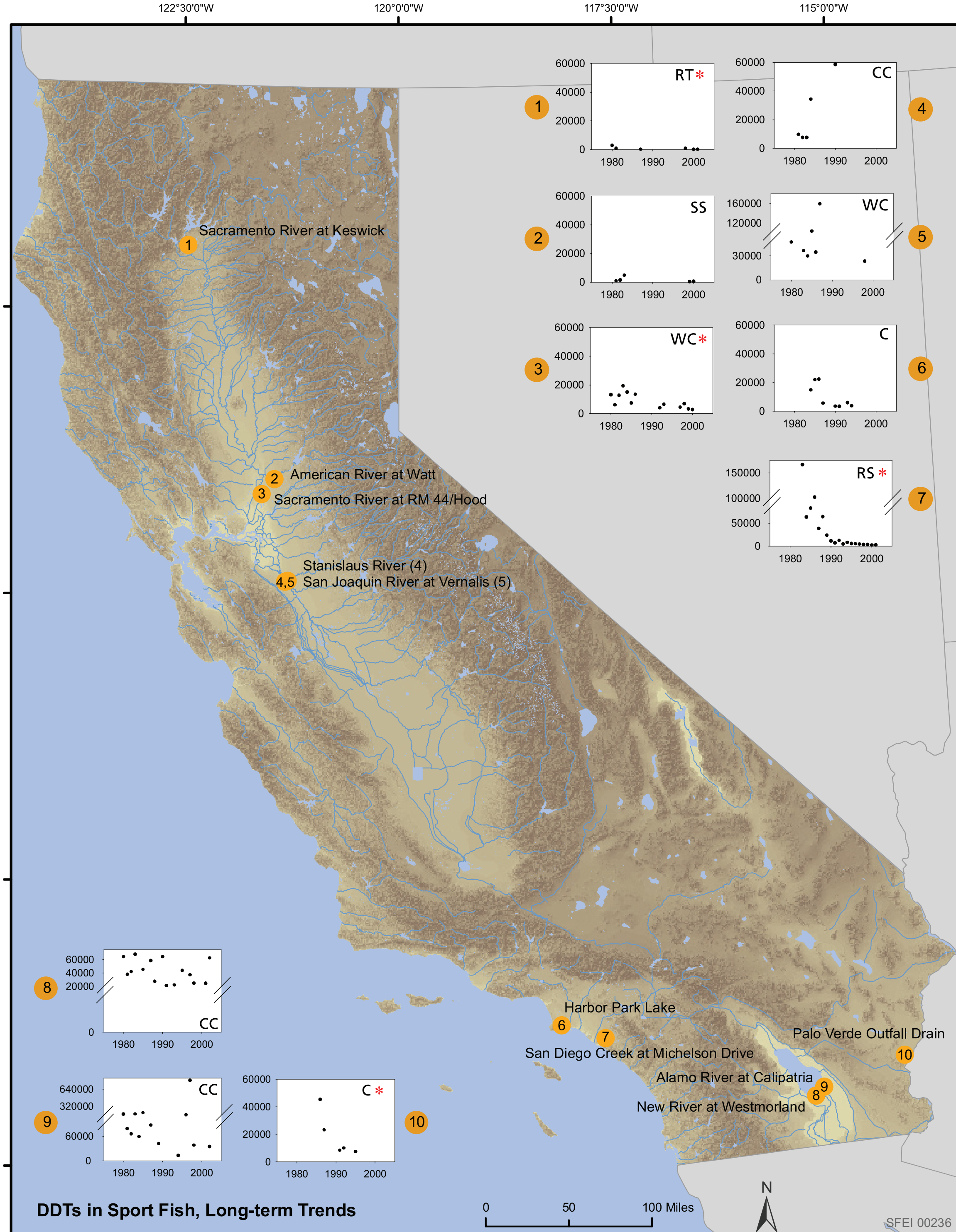


Figure 3.5.4. Long-term trends in DDT concentrations in California sport fish. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight. Species shown are rainbow trout (RT), 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

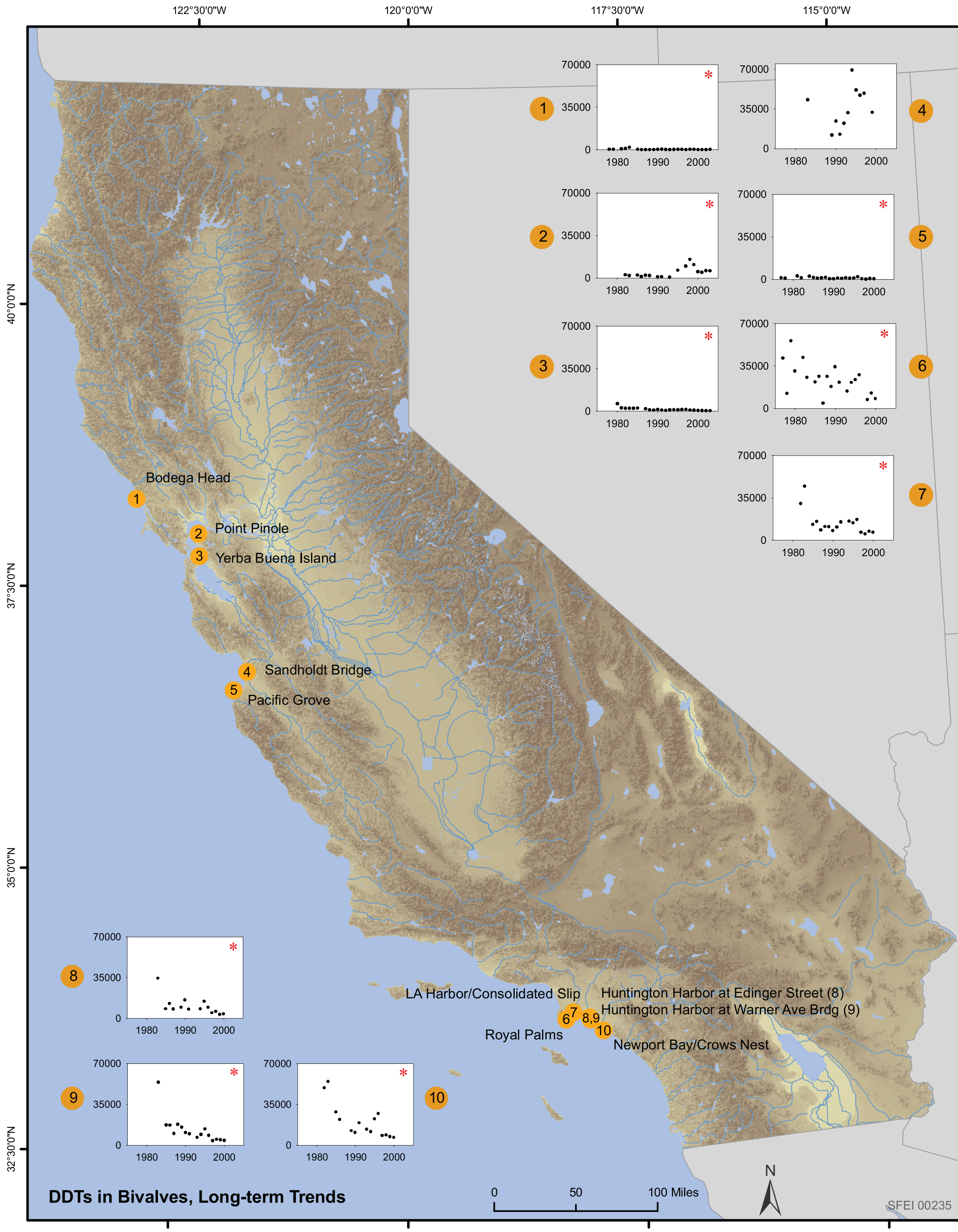


Figure 3.5.5. Long-term trends in DDT concentrations in California mussels measured by the State Mussel Watch Program. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight.

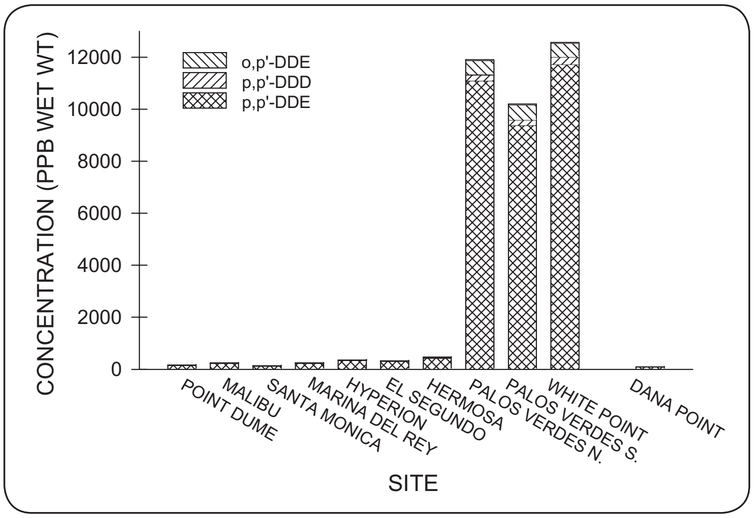


Figure 3.5.6. Mean concentrations of Total DDT in composites of white croaker collected from coastal Southern California in September 1990. From Allen & Cross (1992).

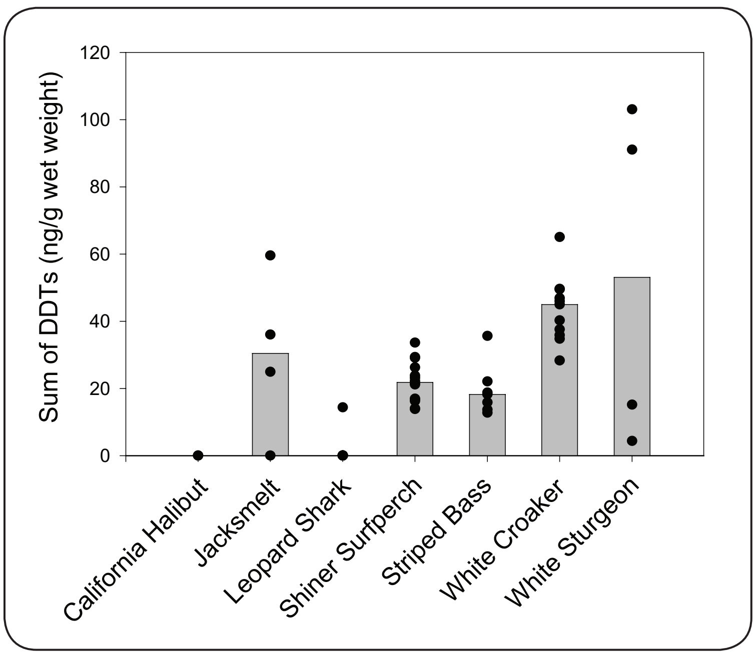


Figure 3.5.7. DDT concentrations (ppb) in San Francisco Bay fish, 2003. Points are concentrations in each composite sample analyzed. Bars indicate median concentrations. From RMP (2006).

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.

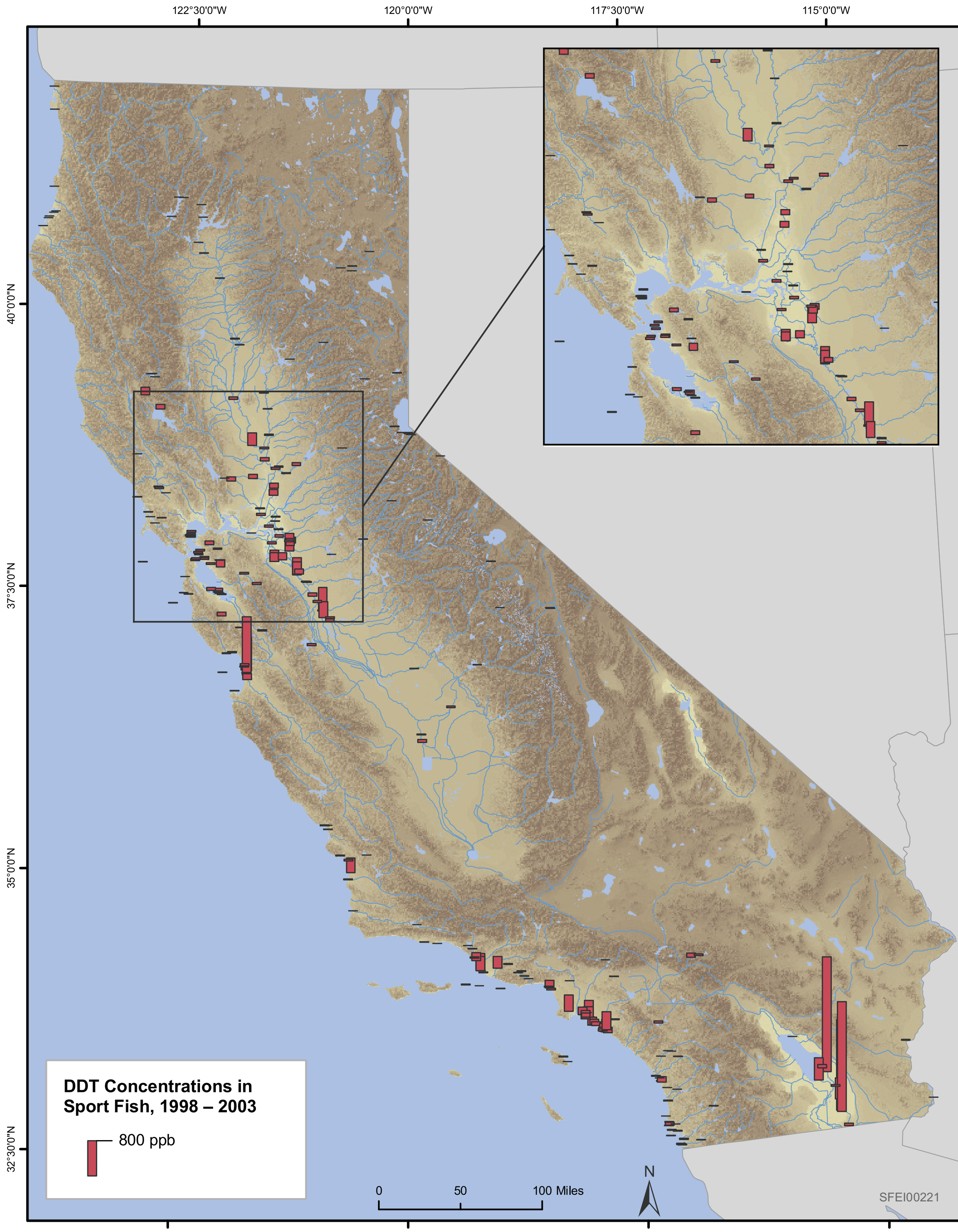


Figure 3.5.8. DDT concentrations in California sport fish, 1998 – 2003. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

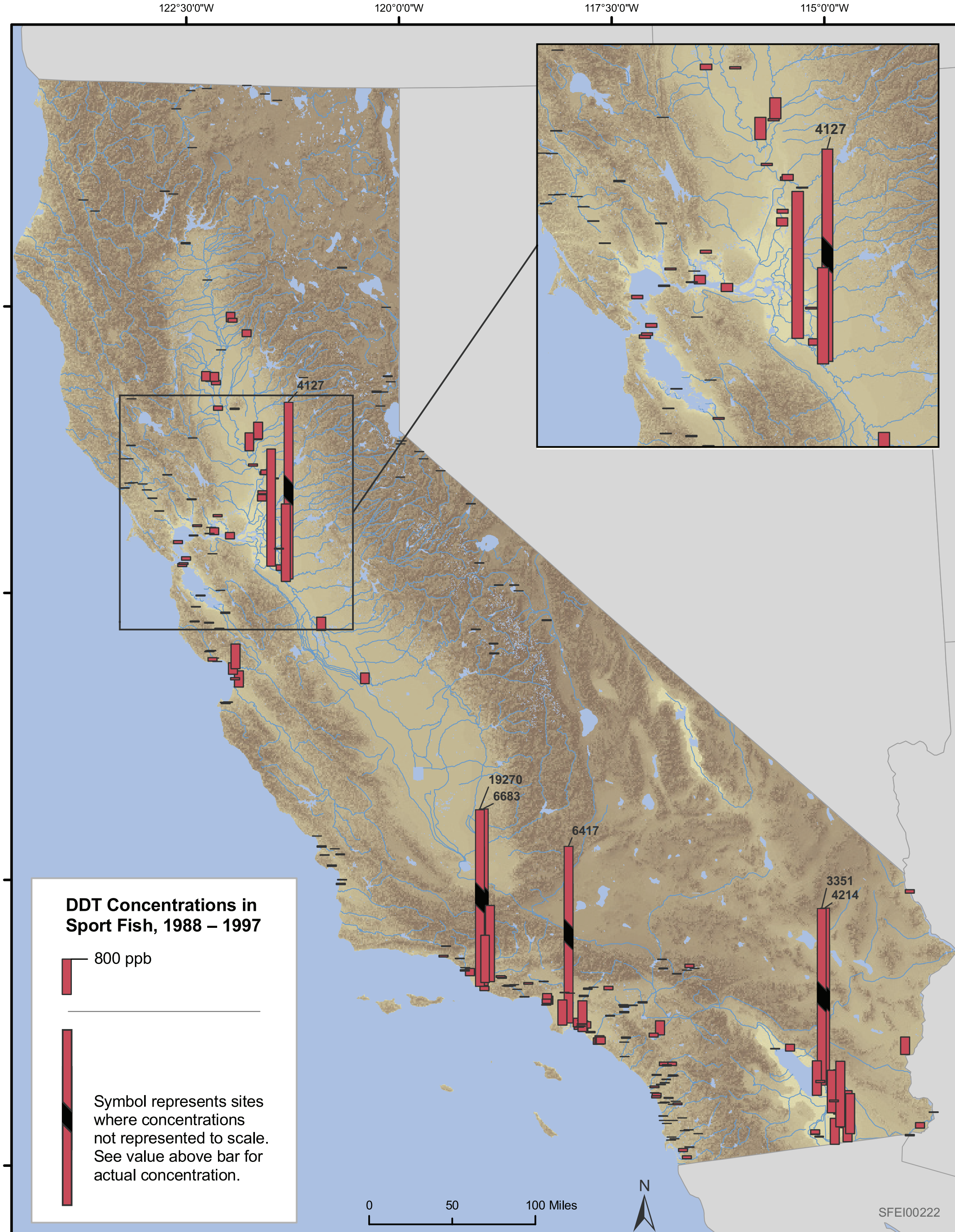


Figure 3.5.9. DDT concentrations in California sport fish, 1988 – 1997. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

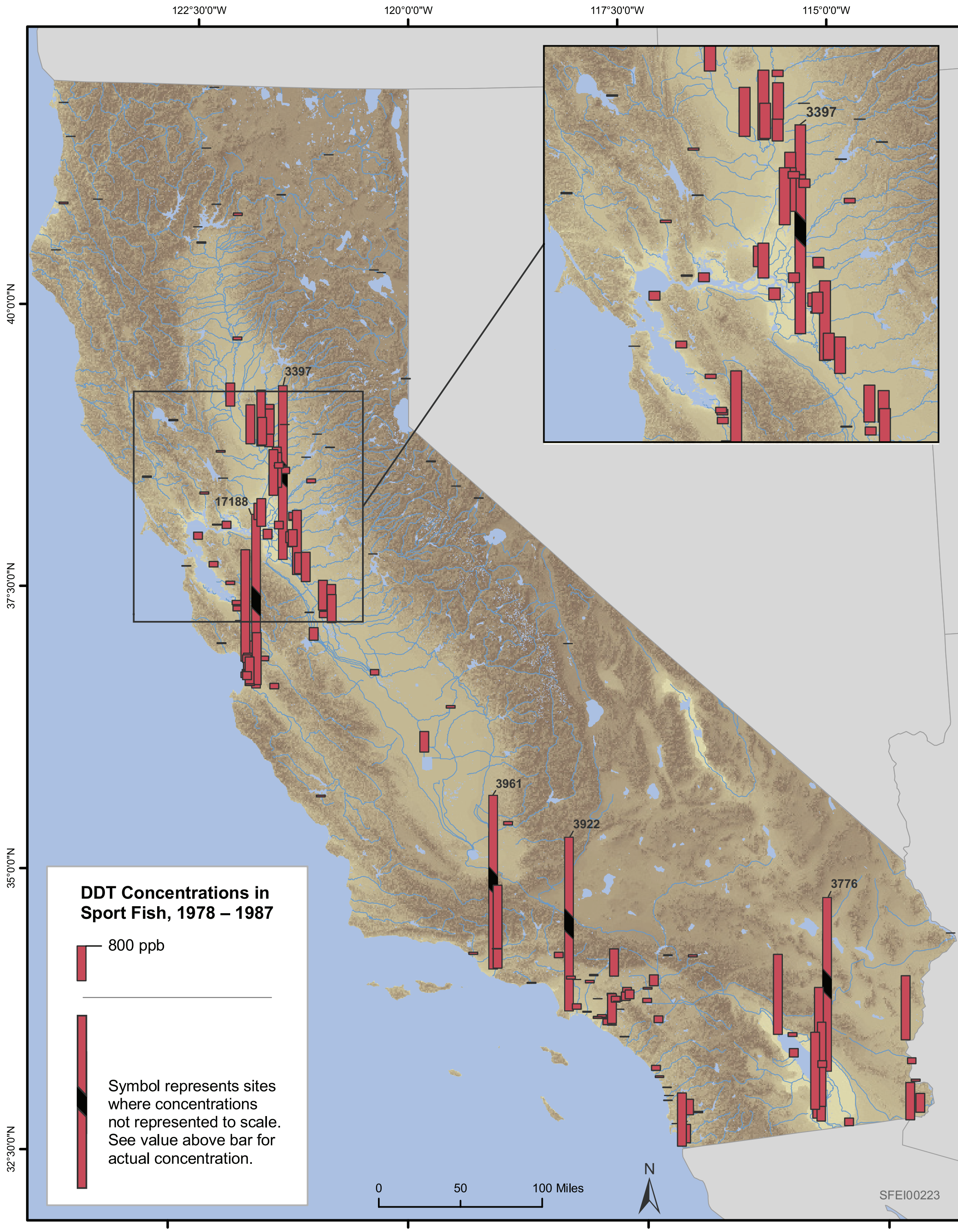


Figure 3.5.10. DDT concentrations in California sport fish, 1978 – 1987. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

### 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 ± 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

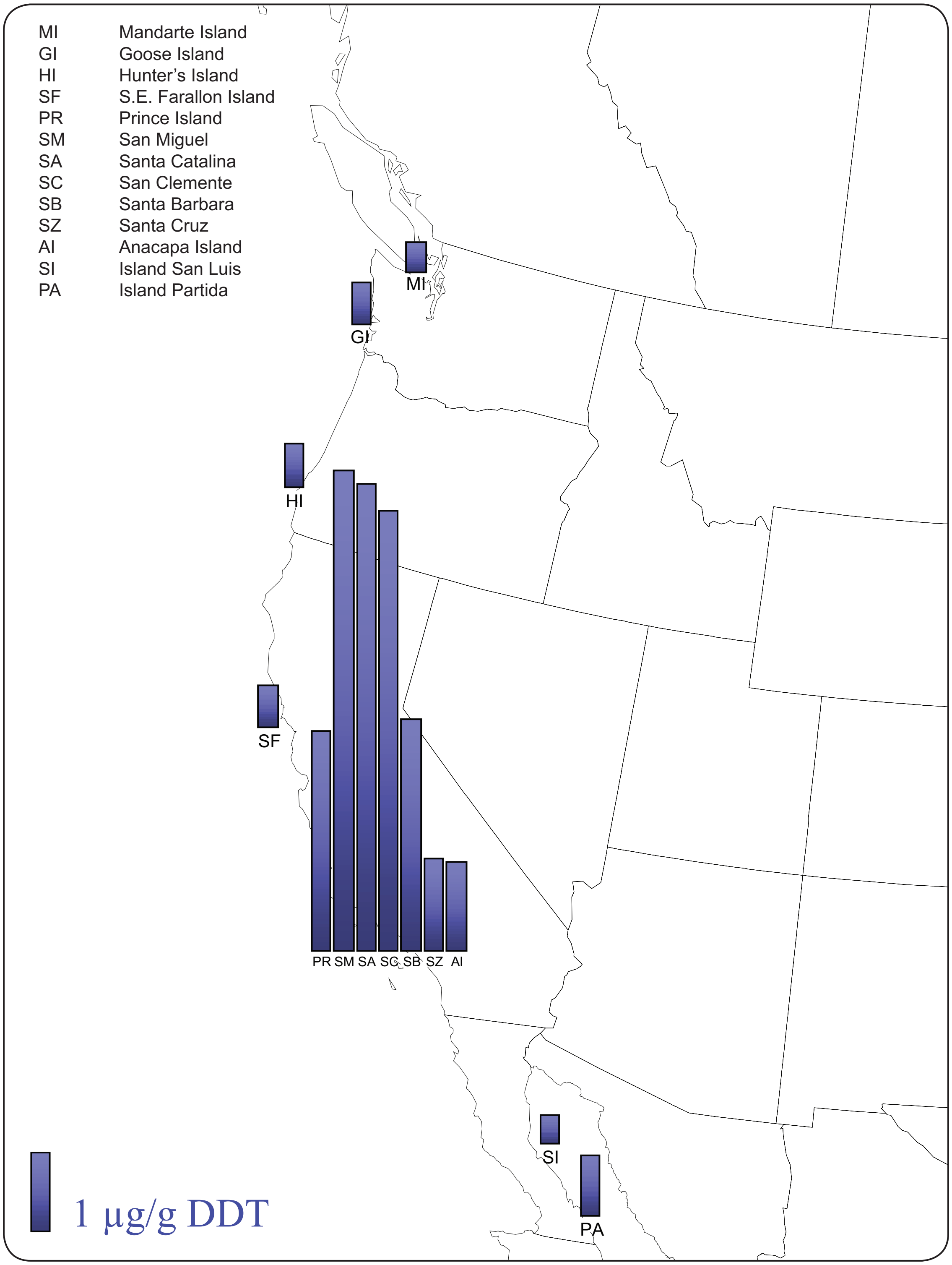


Figure 3.5.11. Total DDT in Western Gull eggs from island sampling locations along the Pacific Coast. From Fry (1995).

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.

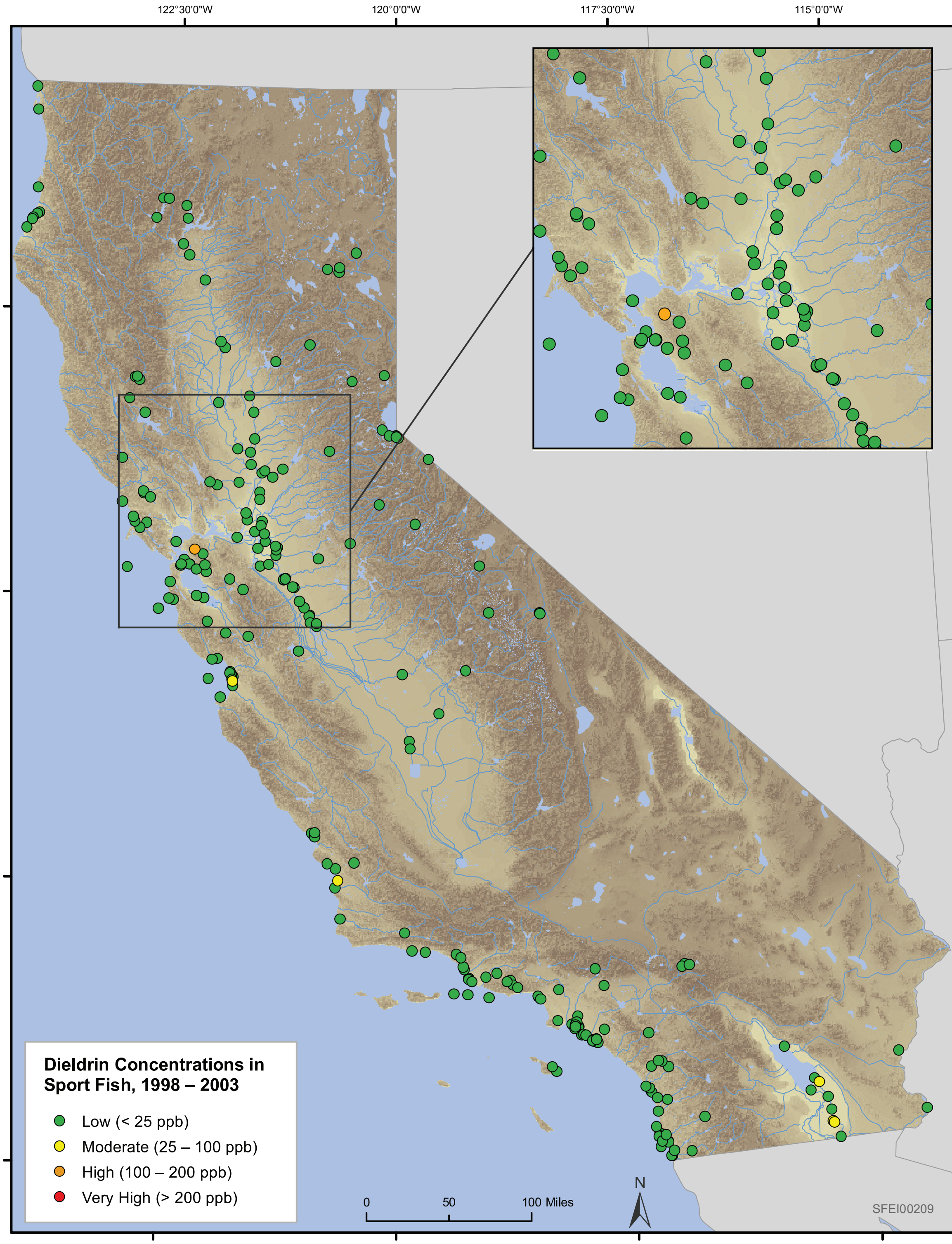


Figure 3.5.12. Dieldrin concentrations in California sport fish, 1998 – 2003. Based on dieldrin measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

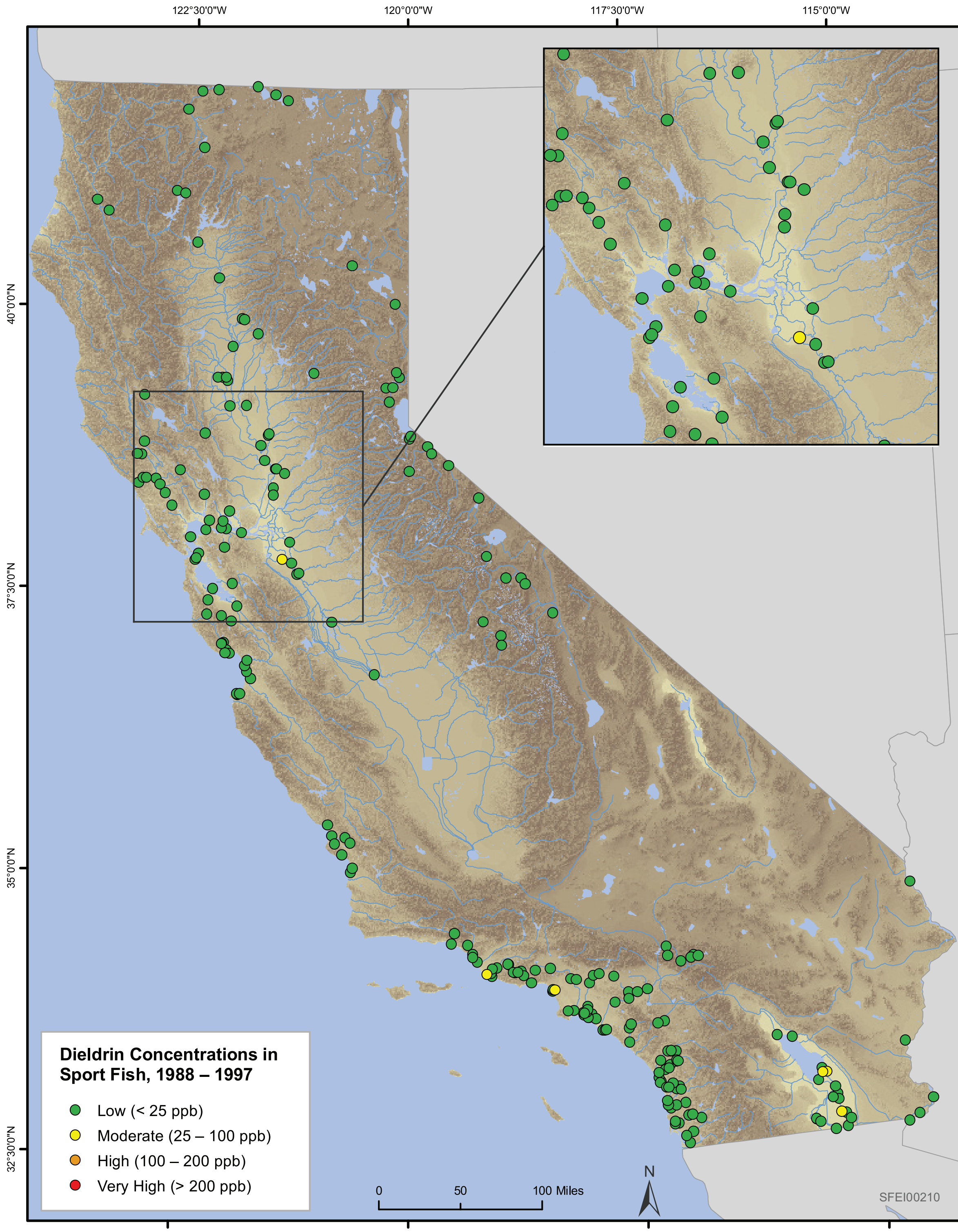


Figure 3.5.13. Dieldrin concentrations in California sport fish, 1988 – 1997. Based on dieldrin measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

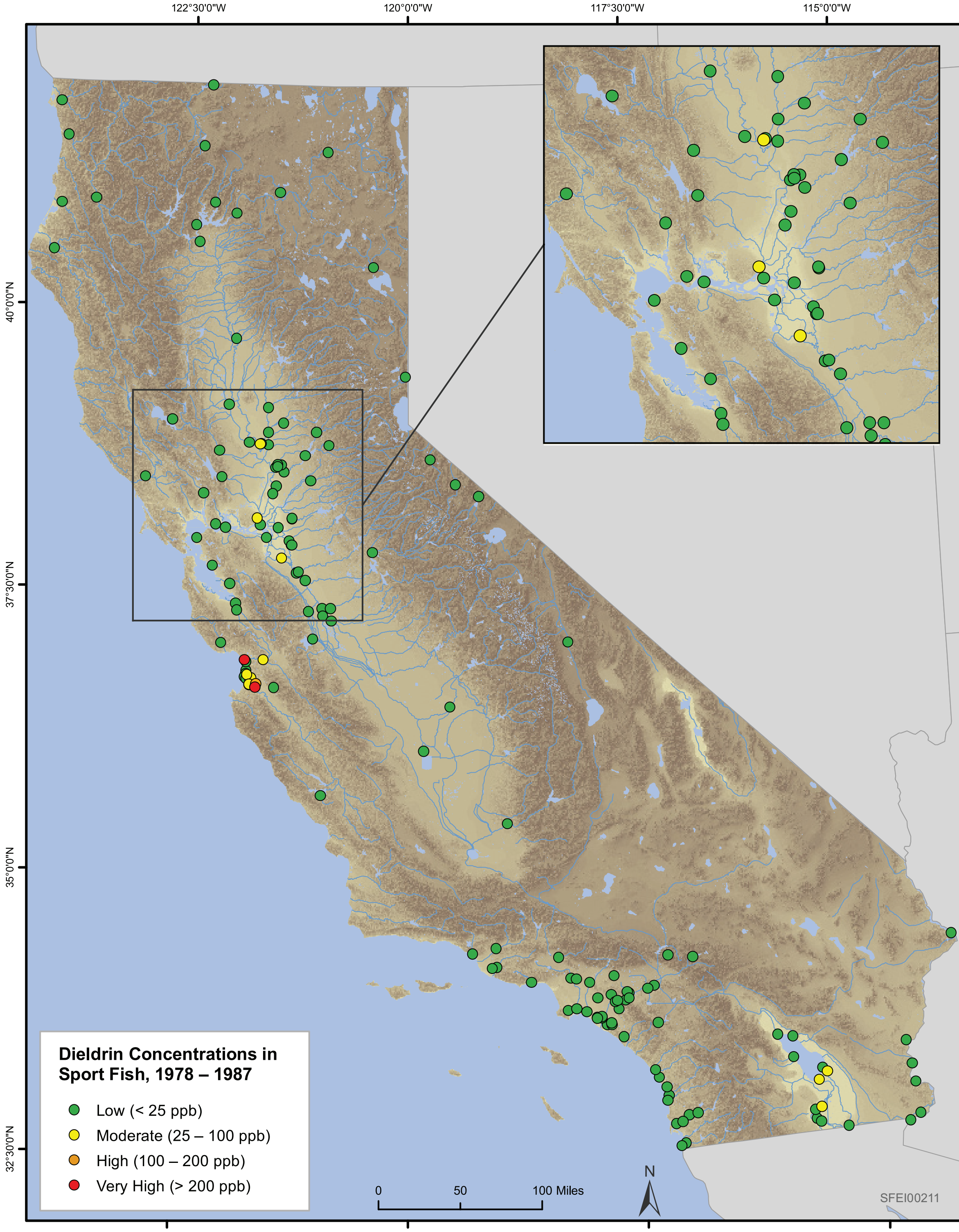


Figure 3.5.14. Dieldrin concentrations in California sport fish, 1978 – 1987. Based on dieldrin measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

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

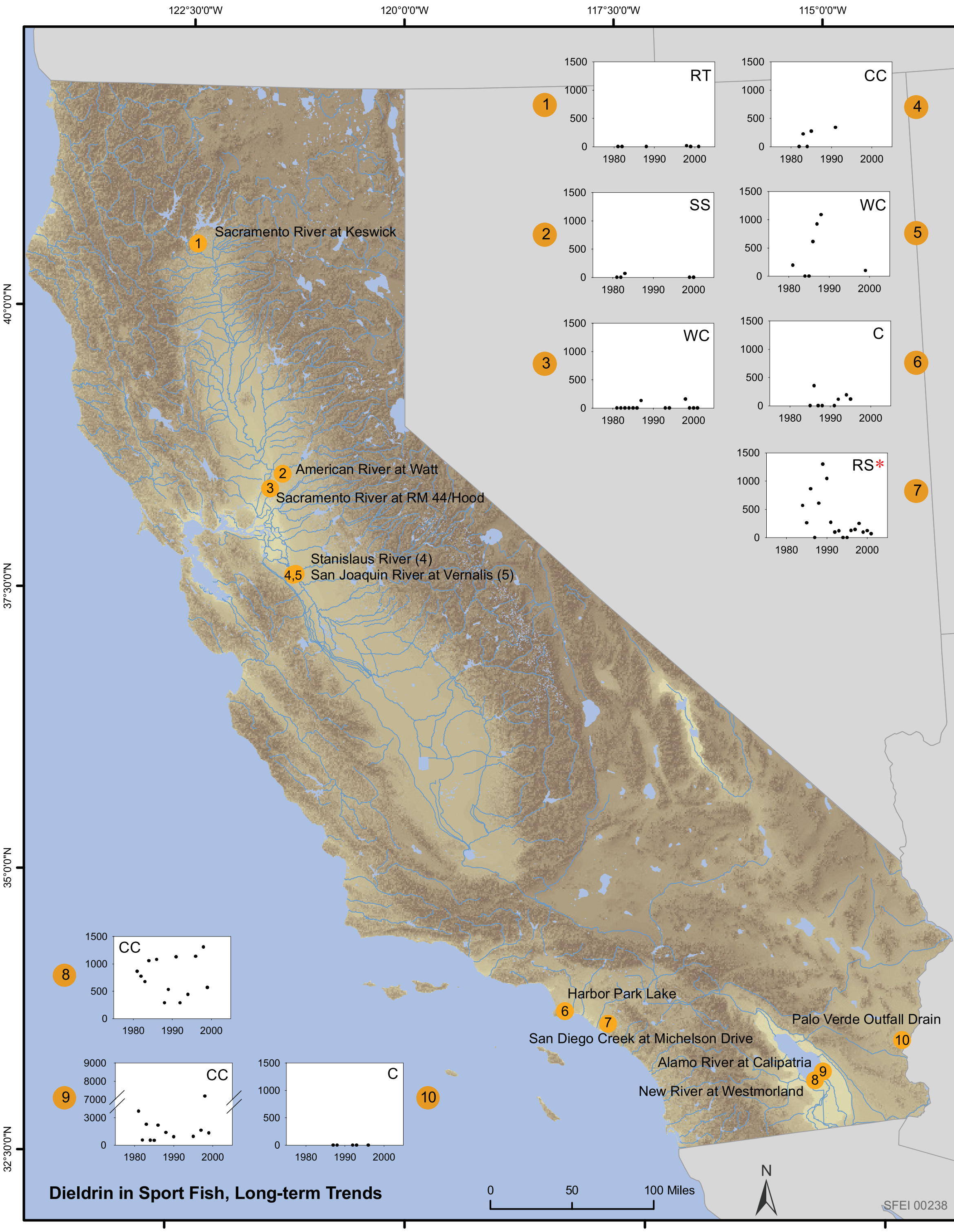


Figure 3.5.15. Long-term trends in dieldrin concentrations in California sport fish. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight. Species shown are rainbow trout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

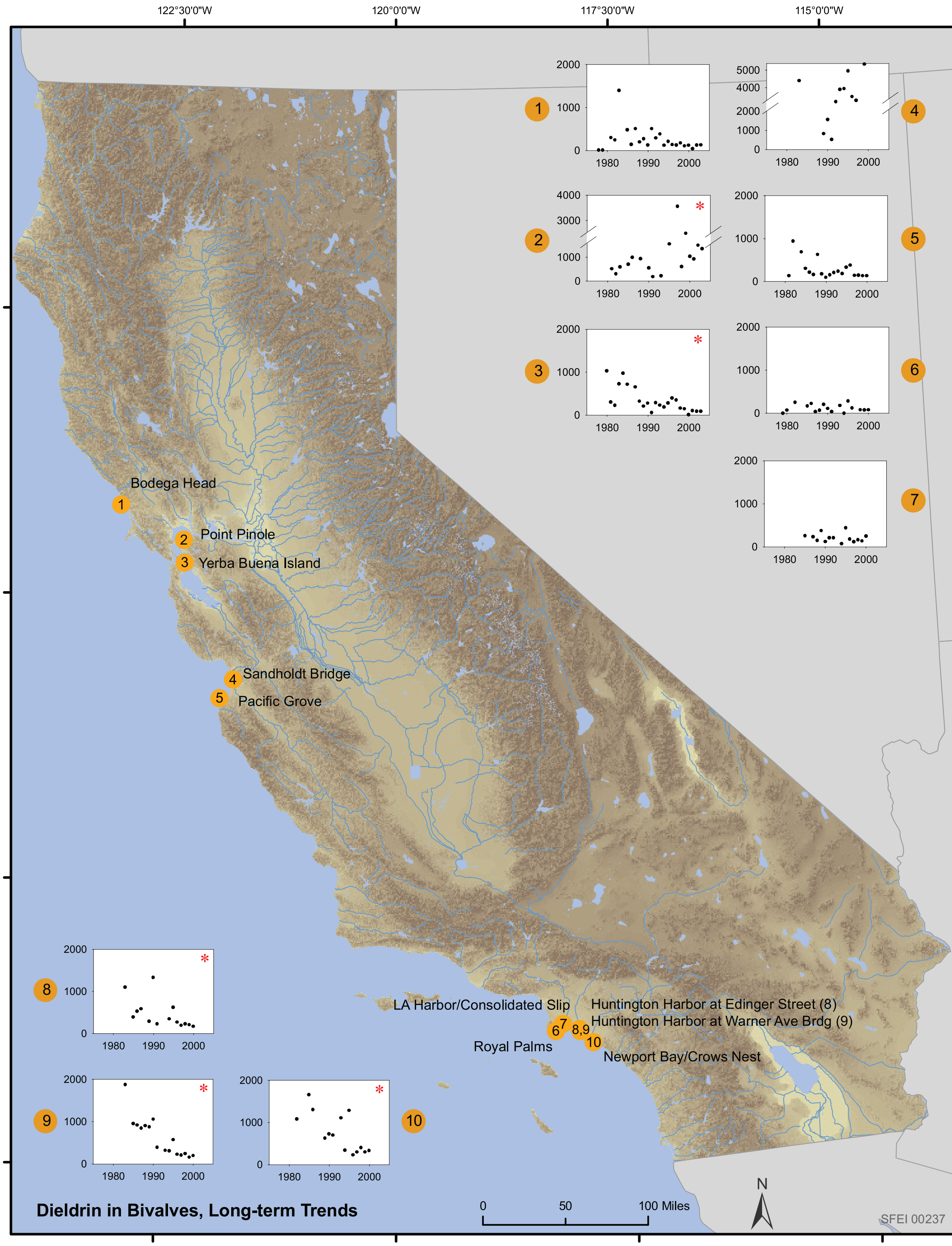


Figure 3.5.16. Long-term trends in dieldrin concentrations in California mussels measured by the State Mussel Watch Program. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid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

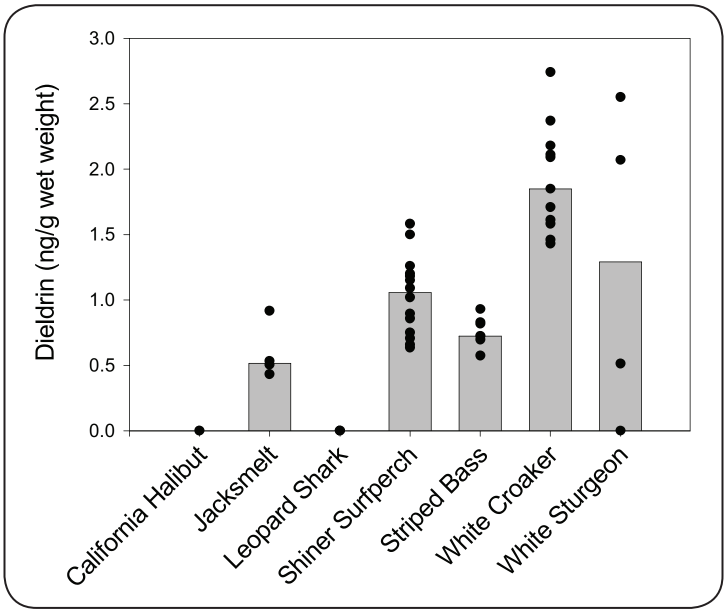


Figure 3.5.17. Dieldrin concentrations (ng/g) in Bay fish, 2003. Points are concentrations in each composite sample analyzed. Bars indicate median concentrations. From RMP (2006).

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.

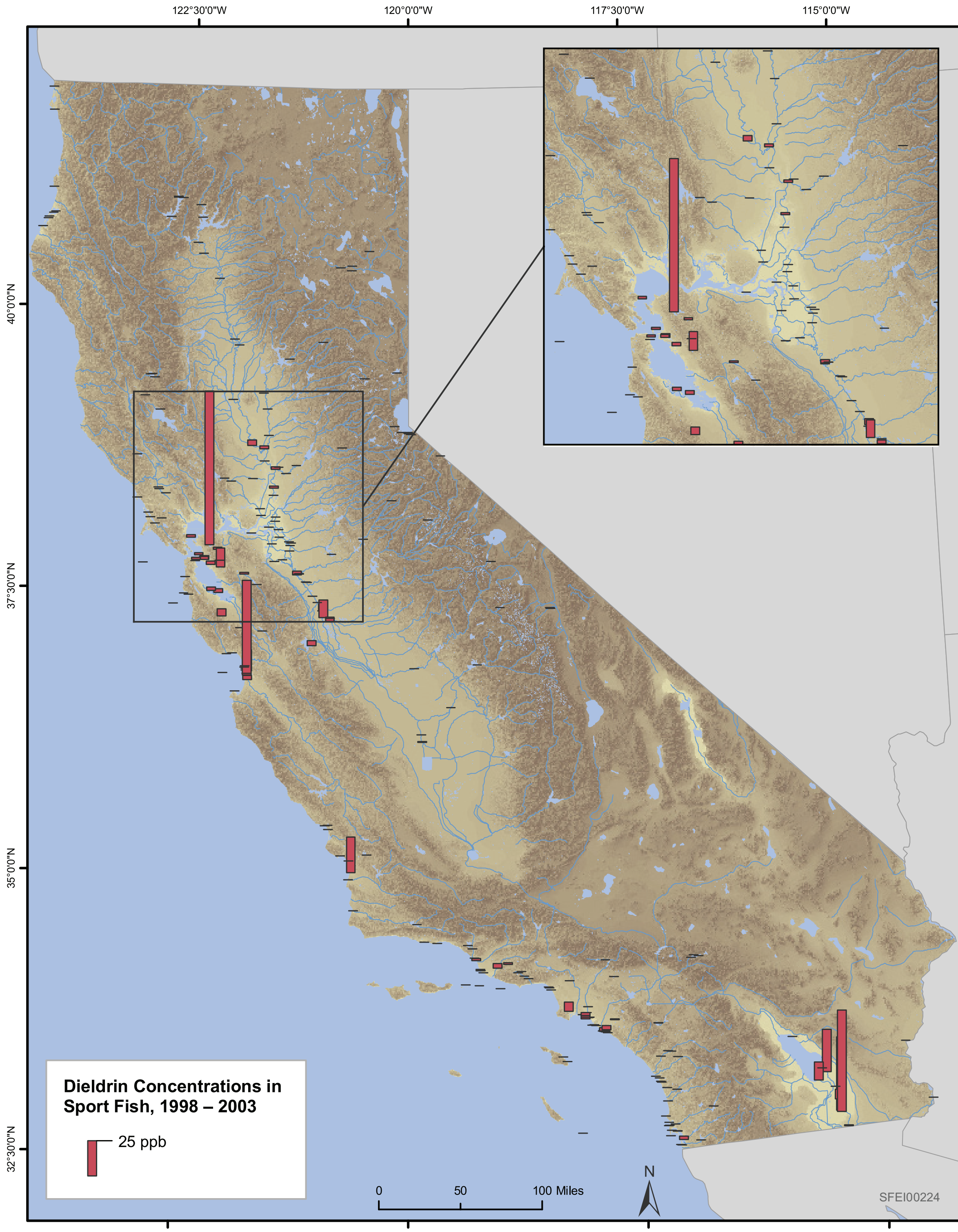


Figure 3.5.18. Dieldrin concentrations in California sport fish, 1998 – 2003. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

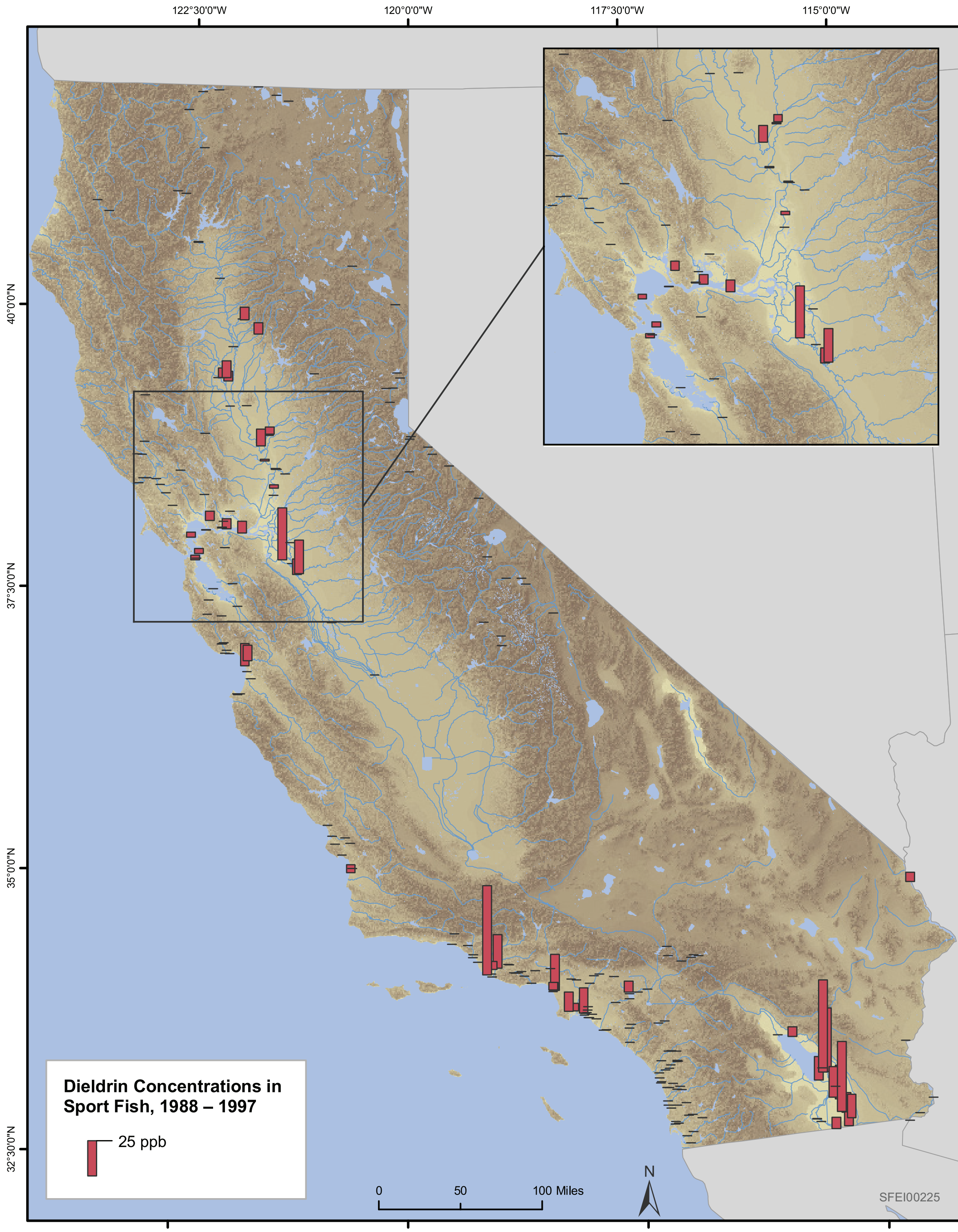


Figure 3.5.19. Dieldrin concentrations in California sport fish, 1988 – 1997. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

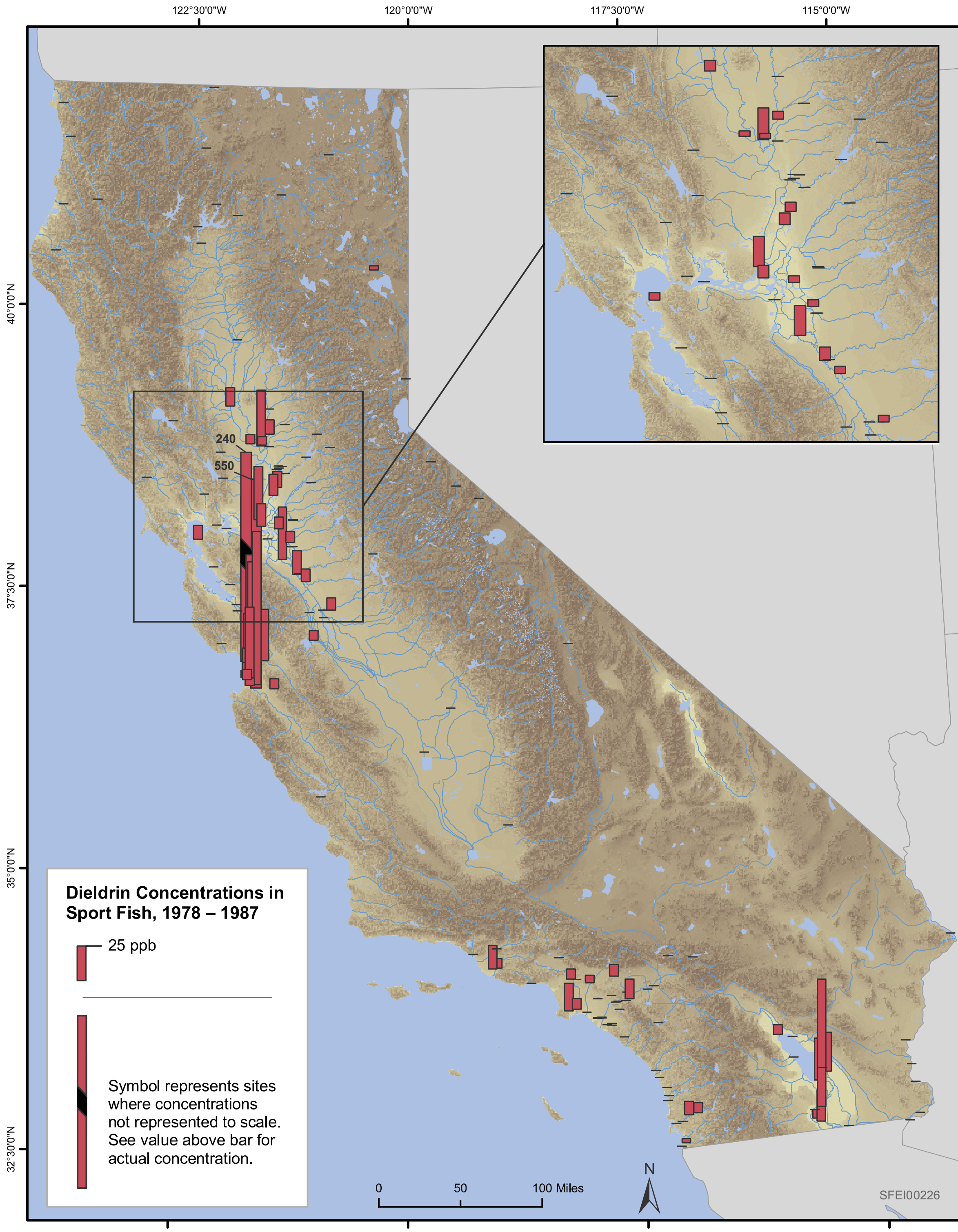


Figure 3.5.20. Dieldrin concentrations in California sport fish, 1978 – 1987. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

### 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.

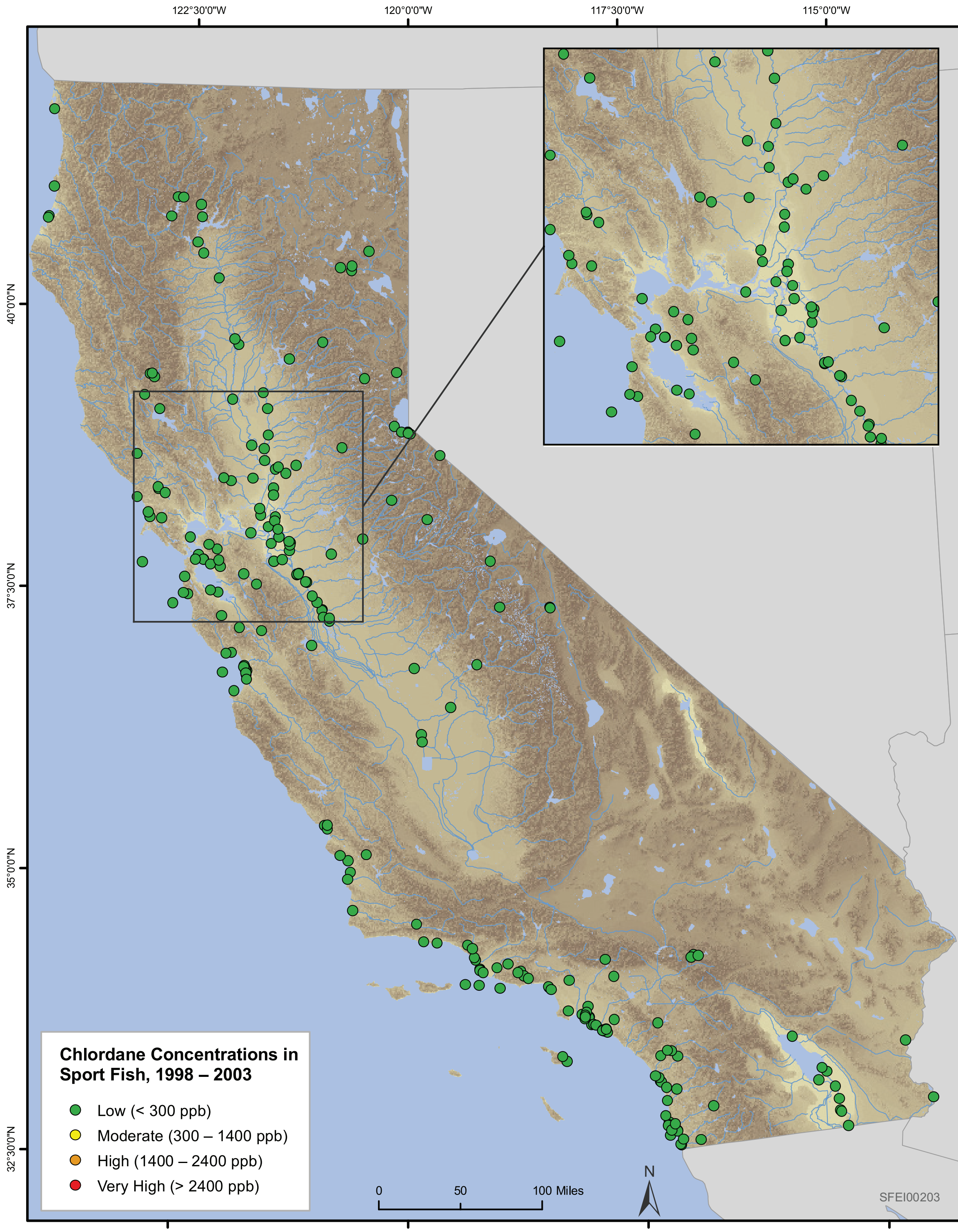


Figure 3.5.21. Chlordane concentrations in California sport fish, 1998 – 2003. Based on chlordane measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

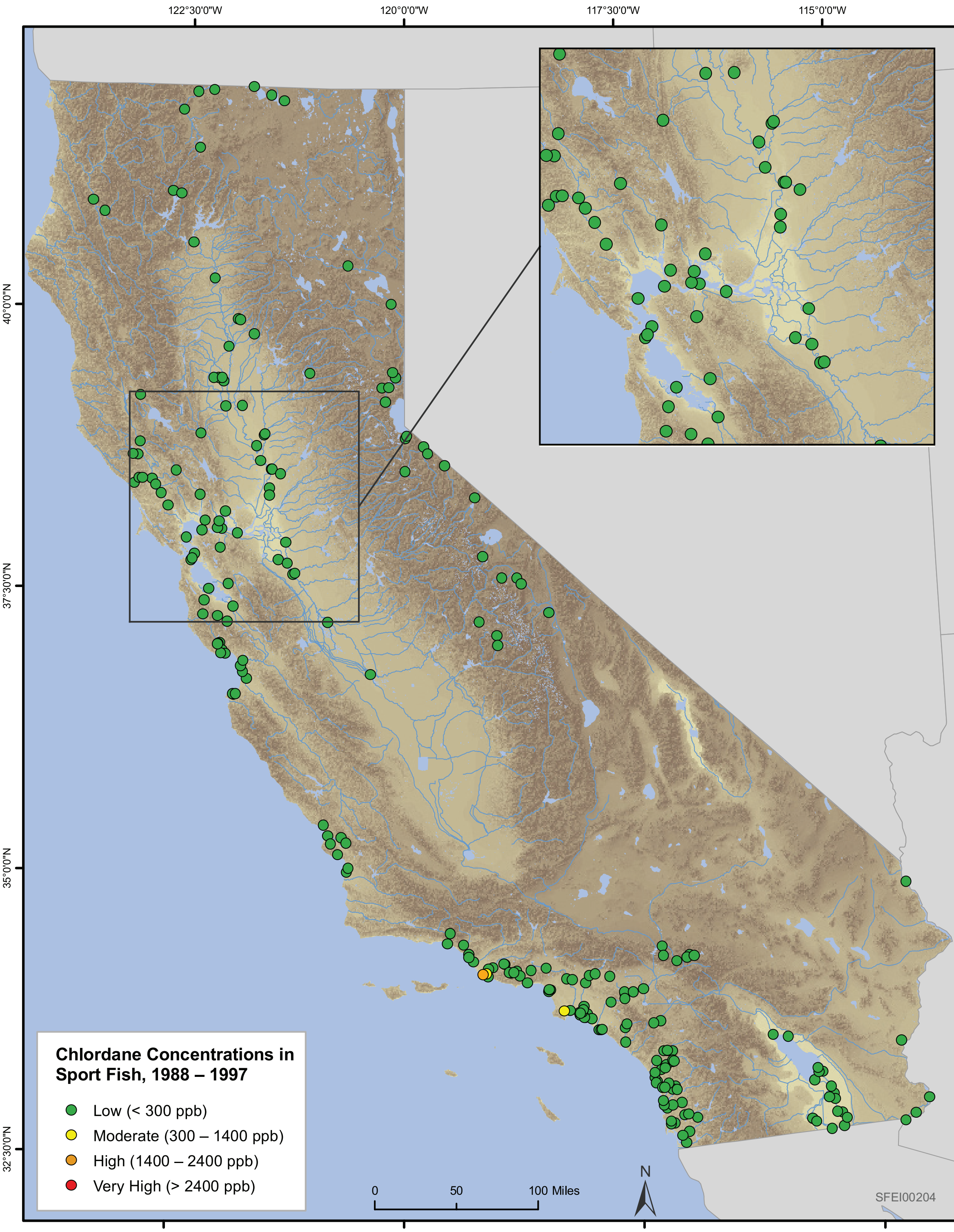


Figure 3.5.22. Chlordane concentrations in California sport fish, 1988 – 1997. Based on chlordane measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

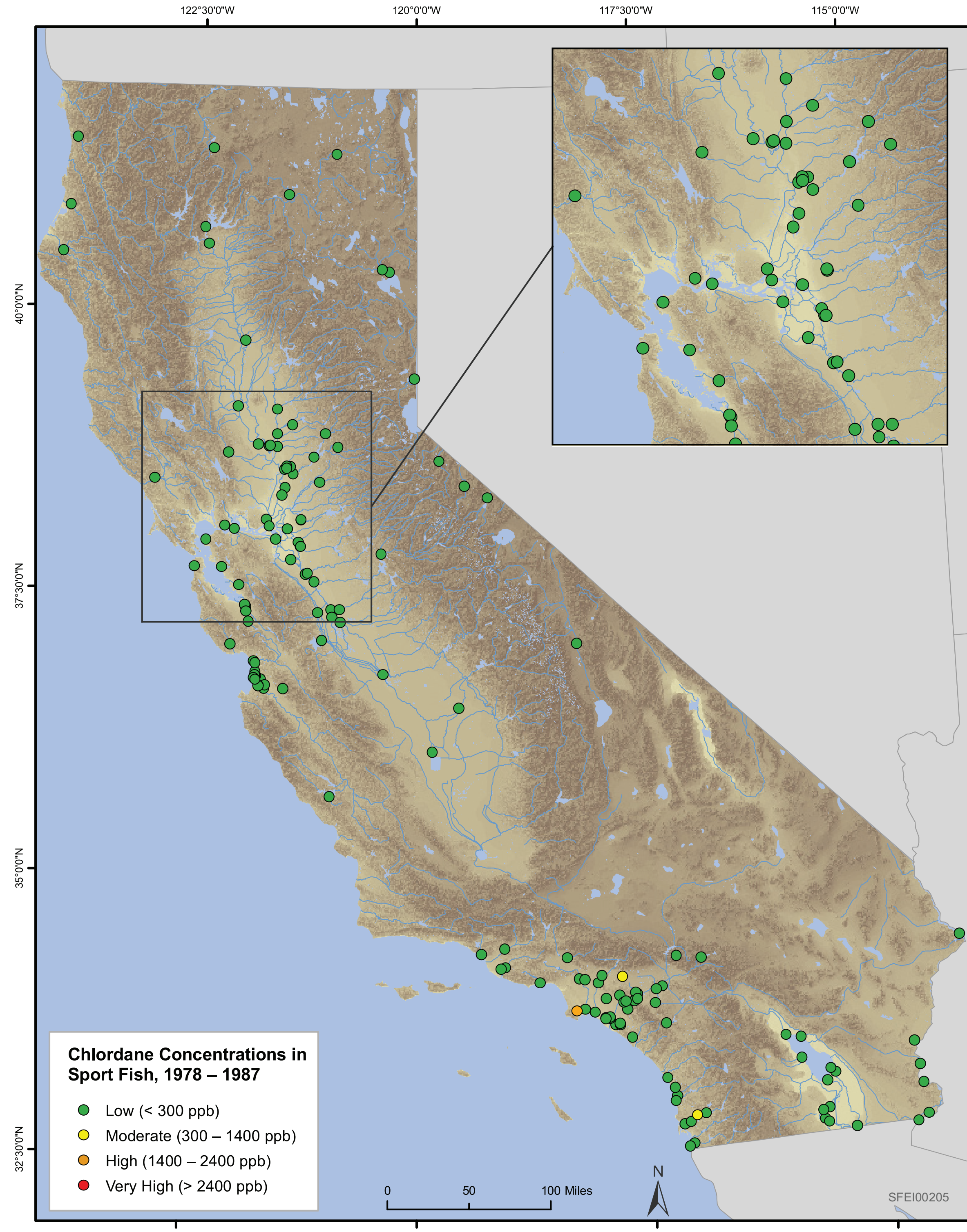


Figure 3.5.23. Chlordane concentrations in California sport fish, 1978 – 1987. Based on chlordane measurements (ppb wet wt) in muscle tissue from a variety of fish species. Dots represent sampling locations. Dot colors indicate the highest median concentration among species at each location.

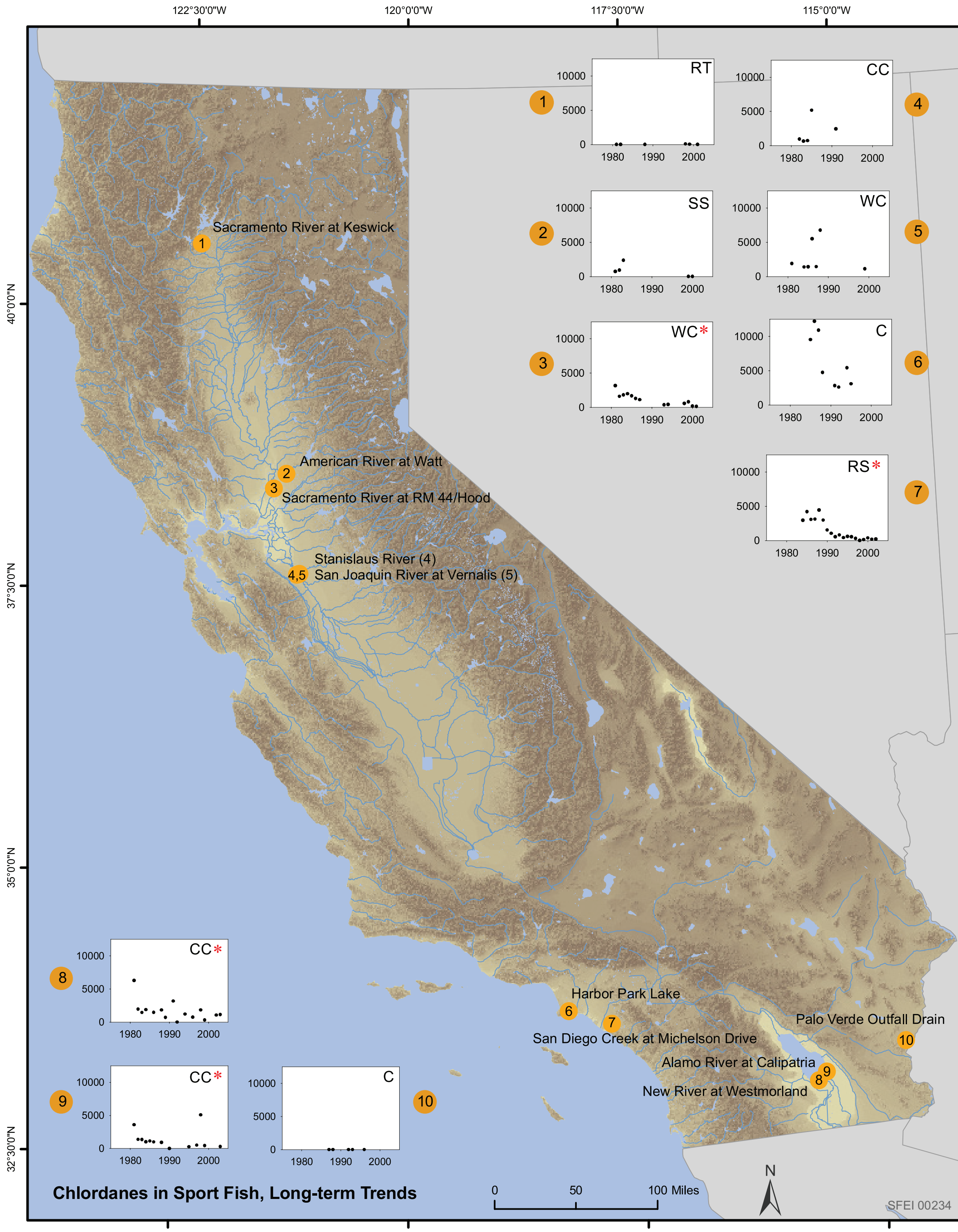


Figure 3.5.24. Long-term trends in chlordane concentrations in California sport fish. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid weight. Species shown are rainbow trout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

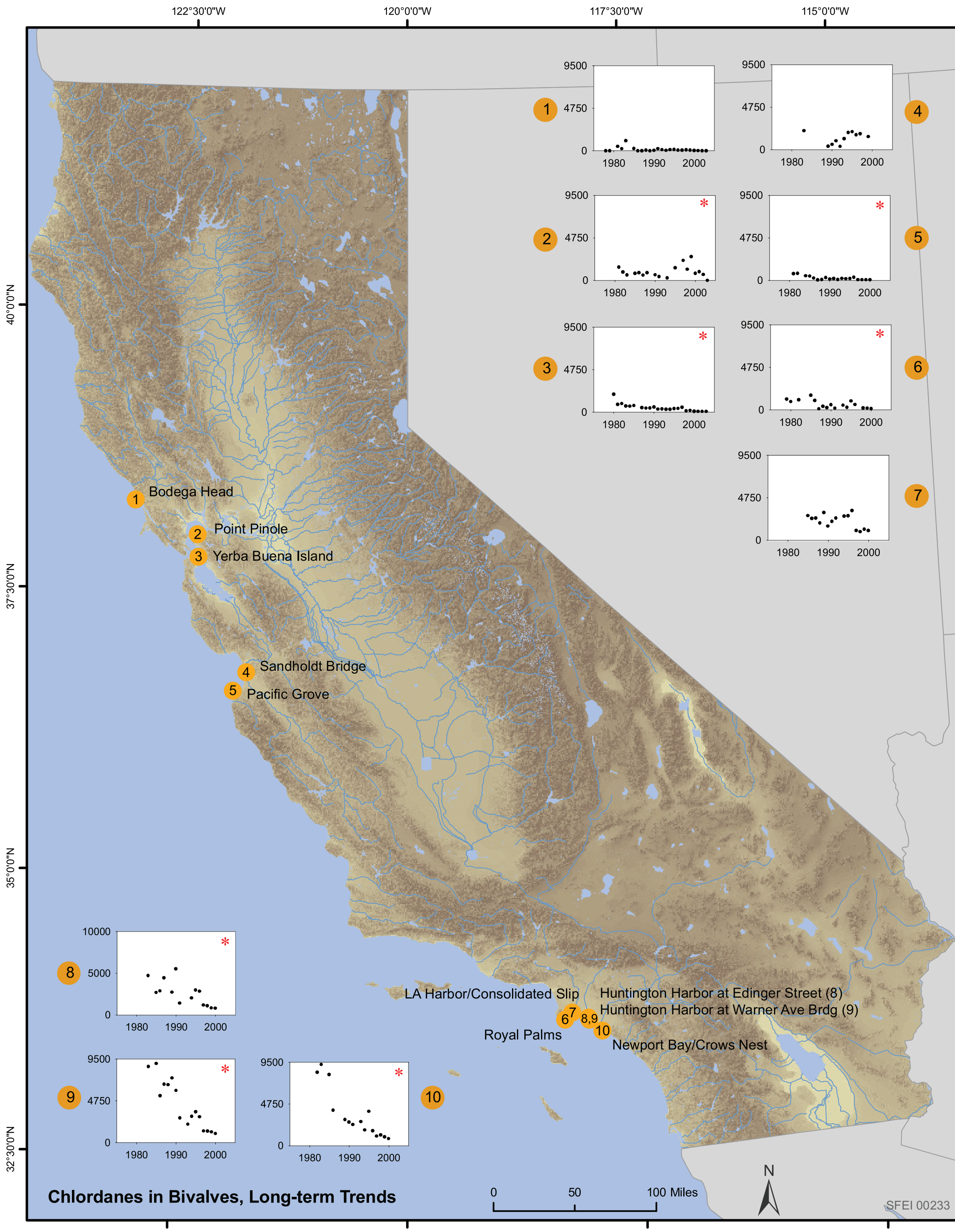


Figure 3.5.25. Long-term trends in chlordane concentrations in California mussels measured by the State Mussel Watch Program. Locations shown represent the best time series available for different parts of the state. The red asterisk indicates a significant trend. Concentrations are given in ppb lipid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

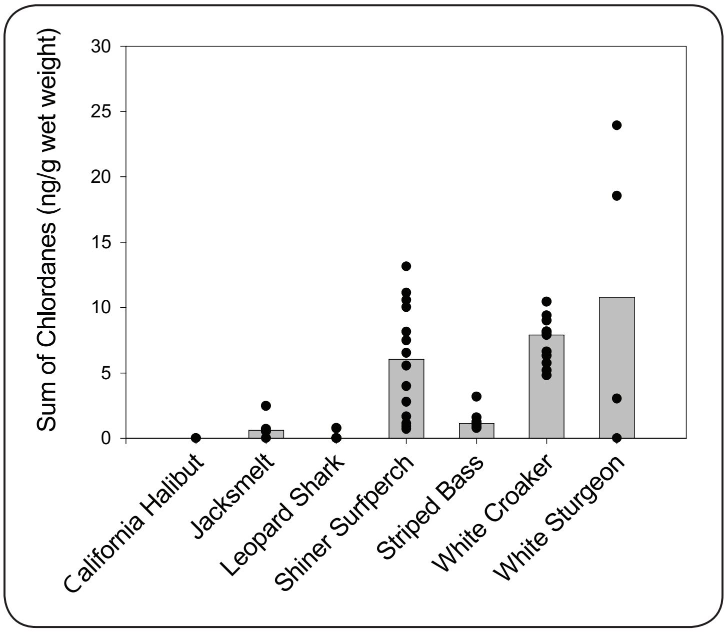


Figure 3.5.26. Chlordane concentrations (ng/g) in Bay fish, 2003. Points are concentrations in each composite sample analyzed. Bars indicate median concentrations. From RMP (2006).

## 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

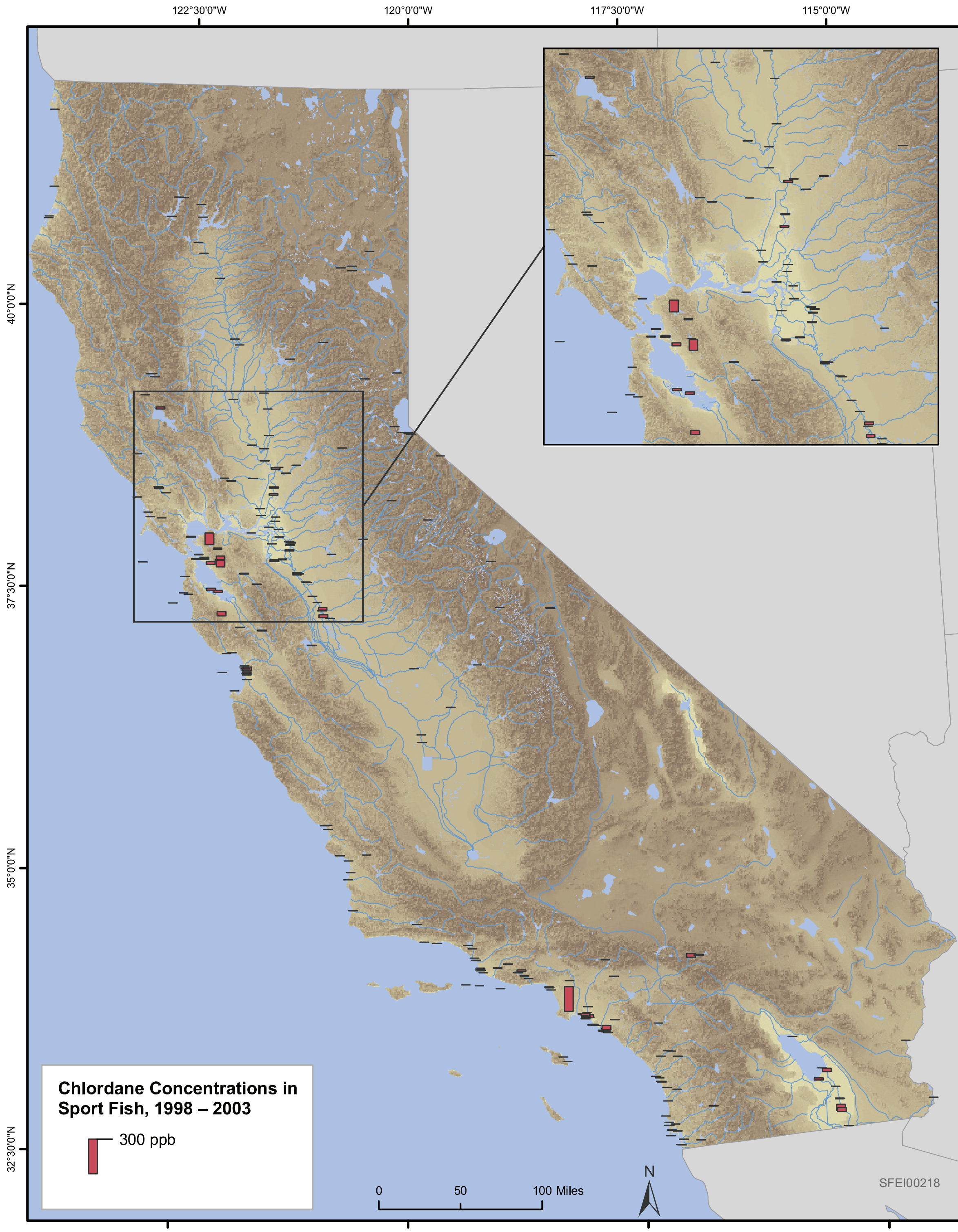


Figure 3.5.27. Chlordane concentrations in California sport fish, 1998 – 2003. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

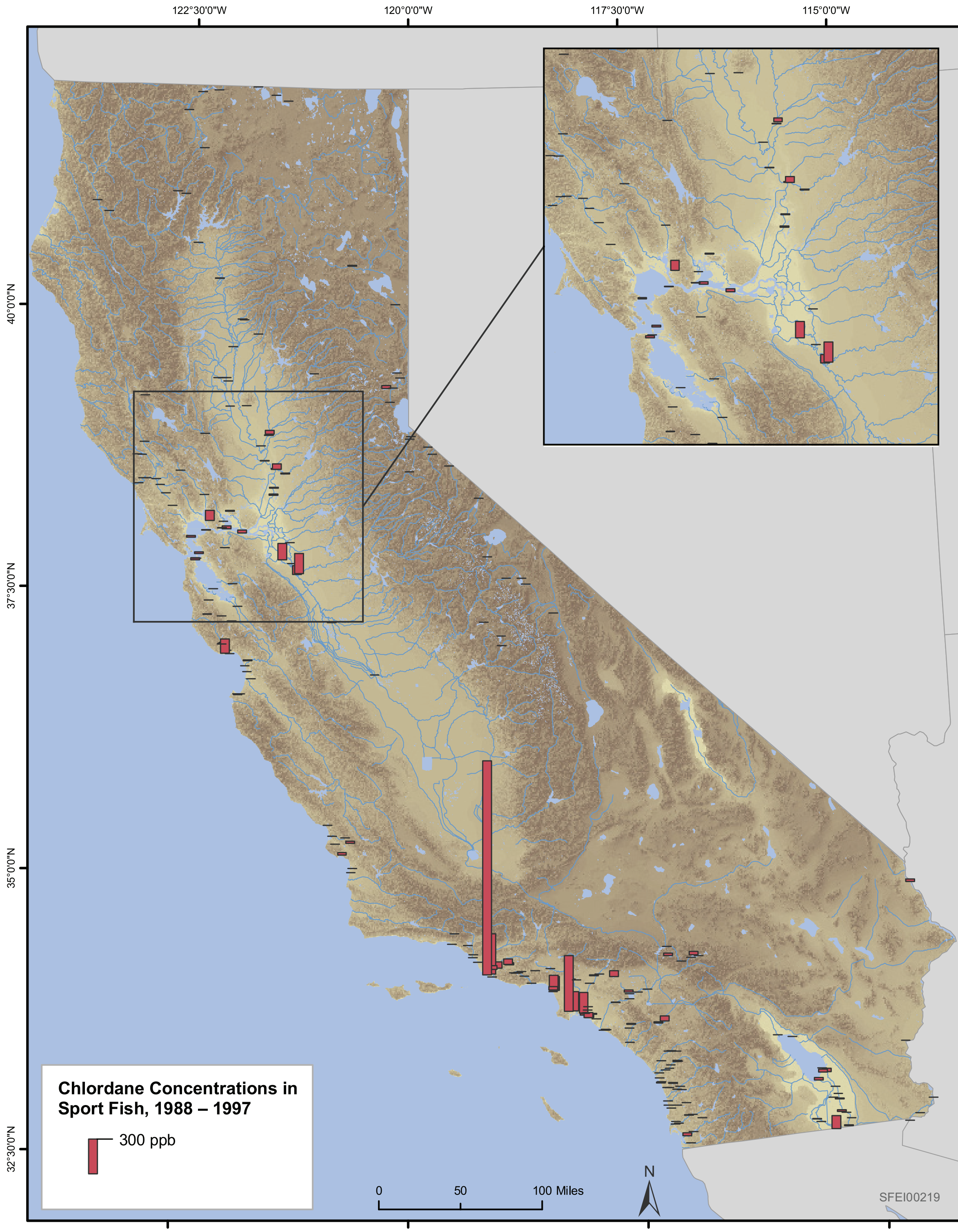


Figure 3.5.28. Chlordane concentrations in California sport fish, 1988 – 1997. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

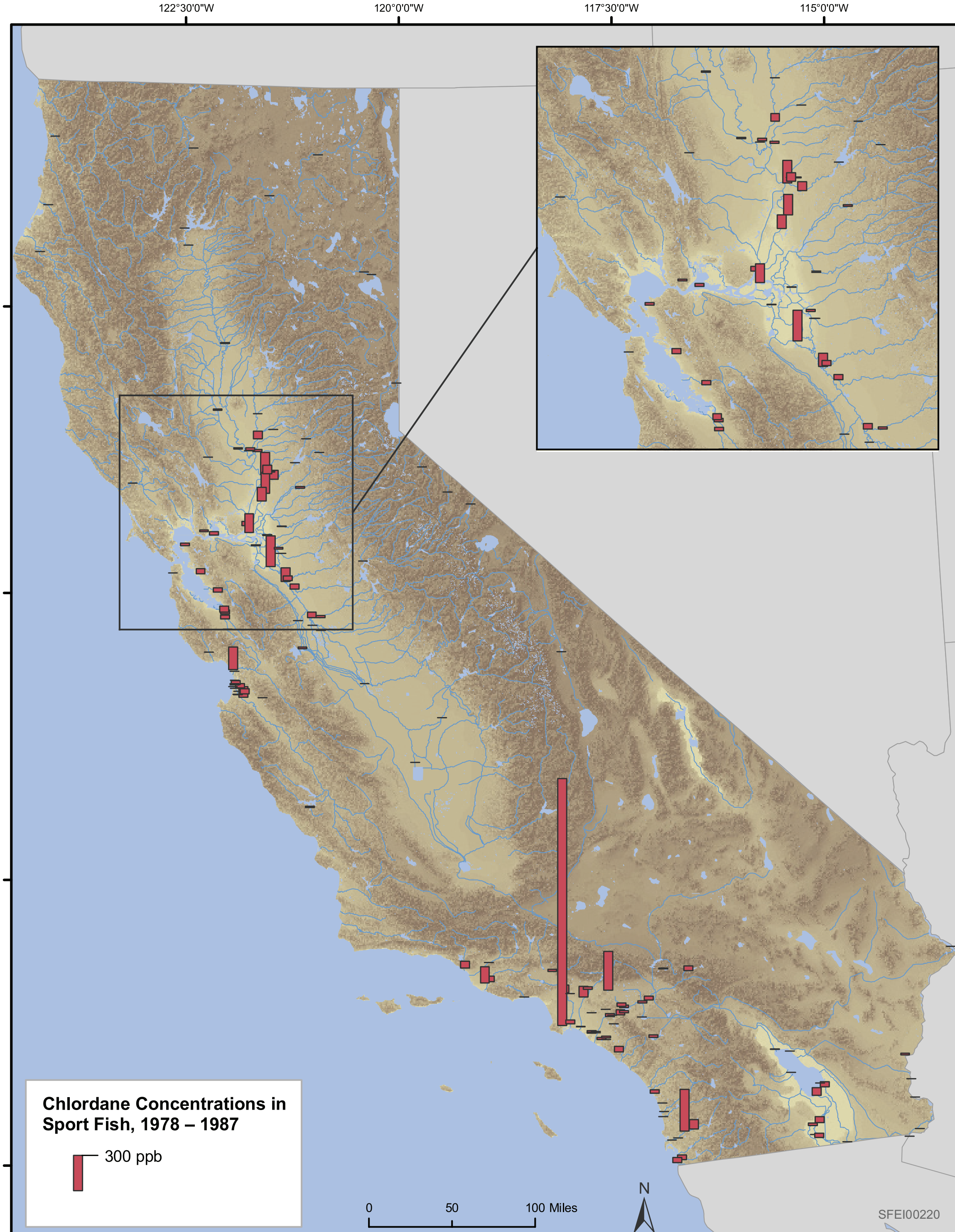


Figure 3.5.29. Chlordane concentrations in California sport fish, 1978 – 1987. Bars represent the highest median concentration (ppb wet wt) among species sampled at each location.

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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For more information, please contact:

Jay A. Davis
San Francisco Estuary Institute
7770 Pardee Lane
Oakland, California 94621
jay@sfei.org

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# APPENDIX 1 A

**Size limits (as total length) for fish included in mercury analysis.**

| Common Name | Scientific Name | Minimum Length (mm) | Maximum Length (mm) |
|---|---|---|---|
| American Shad | Alosa sapidissima | 294 | 392 |
| Arroyo Chub | Gila orcutti | 54 | 72 |
| Bairdiella | Bairdiella icistia | 229 | 305 |
| Barred Sand Bass | Paralabrax nebulifer | 153 | 204 |
| Barred Surfperch | Amphistichus argenteus | 147 | 196 |
| Bat Ray | Myliobatis californica | 493 | 657 |
| Bigscale Logperch | Percina macrolepida | 57 | 76 |
| Black Bullhead | Ameiurus melas | 184 | 245 |
| Black Crappie | Pomoxis nigromaculatus | 192 | 256 |
| Black Croaker | Cheilotrema saturnum | 215 | 287 |
| Black Perch | Embiotoca jacksoni | 190 | 253 |
| Black Rockfish | Sebastes melanops | 320 | 426 |
| Blue Rockfish | Sebastes mystinus | 229 | 305 |
| Bluegill | Lepomis macrochirus | 135 | 180 |
| Bocaccio | Sebastes paucispinis | 116 | 155 |
| Brook Trout | Salvelinus fontinalis | 135 | 180 |
| Brown Bullhead | Ameiurus nebulosus | 236 | 315 |
| Brown Rockfish | Sebastes auriculatus | 220 | 293 |
| Brown Smooth-hound Shark | Mustelus henlei | 621 | 828 |
| Brown Trout | Salmo trutta | 219 | 292 |
| Bullhead | Ameiurus sp. | 218 | 290 |
| California Corbina | Menticirrhus undulatus | 204 | 272 |
| California Halibut | Paralichthys californicus | 437 | 583 |
| California Killifish | Fundulus parvipinnis | 47 | 63 |
| California Sheephead | Semicossyphus pulchere | 262 | 349 |
| California Tonguefish | Symphurus atricauda | 110 | 147 |
| Channel Catfish | Ictalurus punctatus | 375 | 500 |
| Chilipepper Rockfish | Sebastes goodei | 296 | 394 |
| Chinook Salmon | Oncorhynchus tshawytscha | 356 | 475 |
| Chub Mackerel | Scomber japonicus | 106 | 141 |

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|---|---|---|---|
| Coast Cutthroat Trout | Oncorhynchus clarki clarki | 152 | 203 |
| Common Carp | Cyprinus carpio | 375 | 500 |
| Crappie | Pomoxis sp. | 179 | 239 |
| Diamond Turbot | Hypsopsetta guttulata | 191 | 255 |
| Eagle Lake Trout | Oncorhynchus mykiss aquilarum | 373 | 497 |
| English Sole | Pleuronectes vetulus | 98 | 131 |
| Fantail Sole | Xystreurys liolepis | 170 | 227 |
| Fathead Minnow | Pimephales promelas | 46 | 61 |
| Goldfish | Carassius auratus | 171 | 228 |
| Grass Carp | Ctenopharyngodon idella | 465 | 620 |
| Gray Smoothhound Shark | Mustelus californicus | 568 | 757 |
| Green Sturgeon | Acipenser medirostris | 861 | 1148 |
| Green Sunfish | Lepomis cyanellus | 98 | 131 |
| Greenstriped Rockfish | Sebastes elongatus | 195 | 260 |
| Halfmoon | Medialuna californiensis | 232 | 310 |
| Hardhead | Mylopharodon conocephalus | 238 | 317 |
| Hitch | Lavinia exilicauda | 182 | 243 |
| Jack Smelt | Atherinopsis californiensis | 240 | 300 |
| Kelp Bass | Paralabrax clathratus | 274 | 365 |
| Kelp Rockfish | Sebastes atrovirens | 232 | 309 |
| Klamath Smallscale Sucker | Catostomus rimiculus | 195 | 260 |
| Klamath Sucker | Castomidae snyderi | 308 | 411 |
| Kokanee | Oncorhynchus nerka | 295 | 393 |
| Lahontan Cutthroat Trout | Oncorhynchus clarki hen-shawi | 202 | 270 |
| Lake Trout | Salvelinus namaycush | 412 | 549 |
| Largemouth Bass | Micropterus salmoides | 305 | 405 |
| Leopard Shark | Triakis semifasciata | 915 | 1220 |
| Lingcod | Ophiodon elongatus | 586 | 781 |
| Mountain Whitefish | Prosopium williamsoni | 232 | 309 |
| Mozambique Tilapia | Tilapia mossambica | 140 | 186 |
| Opaleye | Girella nigricans | 168 | 224 |
| Orangemouth Corvina | Cynoscion xanthulus | 440 | 587 |
| Pacific Angel Shark | Squatina californica | 874 | 1165 |
| Pacific Hake | Merluccius productus | 410 | 547 |
| Pacific Herring | Clupea harengus | 140 | 186 |
| Pacific Sanddab | Citharichthys sordidus | 134 | 179 |
| Pacific Sardine | Sardinops sagax | 172 | 230 |

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|---|---|---|---|
| Pacific Staghorn Sculpin | Leptocottus armatus | 92 | 123 |
| Pile Surfperch | Rhacochilus vacca | 278 | 370 |
| Queenfish | Seriphus politus | 138 | 184 |
| Quillback Rockfish | Sebastes maliger | 279 | 372 |
| Rainbow Surfperch | Hypsurus caryi | 136 | 181 |
| Rainbow Trout | Oncorhynchus mykiss | 200 | 265 |
| Redbelly Tilapia | Tilapia zillii | 122 | 162 |
| Redear Sunfish | Lepomis microlophus | 141 | 188 |
| Redtail Surfperch | Amphistichus rhodoterus | 251 | 335 |
| Rosethorn Rockfish | Sebastes helvomaculatus | 232 | 309 |
| Round Stingray | Urolophus halleri | 293 | 391 |
| Sacramento Blackfish | Orthodon microlepidotus | 292 | 390 |
| Sacramento Perch | Archoplites interruptus | 97 | 129 |
| Sacramento Pikeminnow | Ptychocheilus grandis | 300 | 400 |
| Sacramento Sucker | Catostomus occidentalis | 355 | 470 |
| Sargo | Anisotremus davidsonii | 220 | 294 |
| Sculpin | Cottus sp. | 85 | 113 |
| Shiner Surfperch | Cymatogaster aggregata | 100 | 130 |
| Silver Surfperch | Hyperprosopon ellipticum | 178 | 238 |
| Smallmouth Bass | Micropterus dolomieu | 305 | 405 |
| Splittail | Pogonichthys macrolepido-tus | 326 | 434 |
| Spotfin Surfperch | Hyperprosopon anale | 108 | 144 |
| Spotted Bass | Micropterus punctulatus | 305 | 405 |
| Spotted Sand Bass | Paralabrax maculatofascia-tus | 266 | 355 |
| Spotted Scorpionfish | Scorpaena plumieri | 129 | 172 |
| Spotted Turbot | Pleuronichthys ritteri | 185 | 247 |
| Starry Flounder | Platichthys stellatus | 175 | 233 |
| Steelhead Rainbow Trout | Oncorhynchus mykiss gairdneri | 200 | 265 |
| Striped Bass | Morone saxatilis | 457 | 610 |
| Striped Mullet | Mugil cephalus | 294 | 392 |
| Tilapia | Tilapia leucosticta | 176 | 235 |
| Tilapia | Tilapia sp. | 176 | 235 |
| Top Smelt | Atherinops affinis | 167 | 223 |
| Tui Chub | Gila bicolor | 93 | 124 |
| Tule Perch | Hysterocarpus traski | 92 | 123 |
| Walleye Surfperch | Hyperprosopon argenteum | 157 | 209 |

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|---|---|---|---|
| White Bass | Morone chrysops | 263 | 351 |
| White Catfish | Ameiurus catus | 242 | 323 |
| White Crappie | Pomoxis annularis | 150 | 200 |
| White Croaker | Genyonemus lineatus | 215 | 290 |
| White Sturgeon | Acipenser transmontanus | 1190 | 1550 |
| White Surfperch | Phanerodon furcatus | 174 | 232 |
| Yellow Bullhead | Ameiurus natalis | 205 | 273 |
| Yellow Perch | Perca flavescens | 166 | 221 |
| Yellowfin Croaker | Umbrina roncador | 202 | 269 |