TOXICITY IN CALIFORNIA WATERS

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TABLE OF CONTENTS TOC

TABLE OF CONTENTS TOC

Tables

Figures

TABLE OF CONTENTS TOC

TABLE OF CONTENTS TOC

EXECUTIVE SUMMARY E

Toxicity testing has been used to assess effluent and surface water quality in California since the mid-1980s. When combined with chemical analyses and other water quality measures, results of toxicity tests provide information regarding the capacity of water bodies to support aquatic life beneficial uses. This report summarizes the findings of monitoring conducted by the Surface Water Ambient Monitoring Program (SWAMP) and associated programs between 2001 and 2010. The report addresses a number of specific assessment questions related to surface water quality in California.

SWAMP monitoring has demonstrated that surface water toxicity occurs in all regions of the state, but that a greater number of toxic samples have been detected in the valleys and along the coast, the areas of greatest human activity. In monitoring conducted between 2001 and 2010, greater than 50% of collection sites have shown some degree of toxicity in fresh water and fresh water sediment samples and greater than 45% of the sites have shown some degree of toxicity in marine sediment samples. Agricultural and urban sites showed greater water and sediment toxicity than sites in less developed areas. While greater water toxicity has been observed in agricultural sites relative to urban sites, there has been no difference in sediment toxicity between urban and agricultural sites. The majority of the data presented in this report were obtained from monitoring studies designed to increase understanding of potential biological impacts from human activities. As such, site locations were generally targeted in lower watershed areas, such as tributary confluences or upstream and downstream of potential pollutant sources. Only a minority of the sites were selected probabilistically (i.e., at random). Therefore, these data only characterize the sites monitored, and cannot be used to make assumptions about unmonitored areas

Correlation analyses and toxicity identification evaluations (TIEs) were used to determine the likely causes of surface water toxicity. TIEs conducted in water samples suggest that toxicity to invertebrate test species (e.g., cladocera and amphipods) was most often caused by pesticides (e.g., diazinon and chlorpyrifos). More recent studies also show that pyrethroid pesticides play a role in water column toxicity to amphipods. There has been a limited number of TIEs conducted with fish and algae. Correlation analyses corroborate the TIE findings. Although there have been fewer TIE studies of the causes of sediment toxicity, those that have been conducted show that sediment toxicity is caused by pyrethroid and organophosphate pesticides. Fewer TIEs have been conducted to determine causes of toxicity in marine receiving waters.

The principal approach to determine whether observations of toxicity in laboratory toxicity tests are indicative of ecological impacts in receiving waters has been to conduct field bioassessments of macroinvertebrate communities. These studies have included “triad” assessments of chemistry, toxicity, and macroinvertebrate communities, the core components of SWAMP. Comprehensive studies in the Salinas and Santa Maria rivers on the central coast have demonstrated that pesticide toxicity in the lower reaches of these rivers is linked to impacts on resident aquatic insect communities. Similar studies of urban creeks in the central valley have also shown linkages between laboratory toxicity tests and impacts on urban stream macroinvertebrates. Additional research has suggested that habitat also plays an important role in the distribution of stream macroinvertebrates. One recommendation for future SWAMP monitoring is to conduct further investigations of the linkages between surface water toxicity and receiving system impacts on biological communities.

The value of utilizing the TST methodology is that it can be applied to all ambient and effluent toxicity monitoring improving consistency and comparability among all Water Board monitoring programs.

As part of State Water Board staff's development of a Policy for Toxicity Assessment and Control, a prescribed statistical methodology for determining whether a sample is toxic is being considered. The statistical approach uses the U.S. EPA's Test of Significant Toxicity (TST). The TST would be applicable to monitoring conducted by SWAMP so an assessment of SWAMP toxicity data was conducted comparing the results of toxicity tests using the TST and the traditional hypothesis testing approach. The analysis indicated there is little difference in the assessment of ambient toxicity regardless of which statistical method is applied to the data. The value of utilizing the TST methodology is that it can be applied to all ambient and effluent toxicity monitoring improving consistency and comparability among all Water Board monitoring programs.

Because evidence from SWAMP monitoring has demonstrated that pesticides are a primary cause of surface water toxicity in California, this report describes a number of management initiatives which together are designed to reduce pesticide loading. These include implementation of total maximum daily loads (TMDLs) in specific watersheds, and pesticide label changes recommended by the U.S. EPA and California Department of Pesticide Regulation to modify use of problem pesticides (e.g., organophosphate and pyrethroid pesticides, respectively). This report also describes state and regional programs to implement management practices to reduce runoff in urban and agricultural watersheds. These include incorporation of Low Impact Development and other techniques to curb pollution loading in urban stormwater runoff, installation of management practices such as vegetated treatment systems and enzyme additions to reduce pesticides in agriculture runoff, and implementation of setbacks and buffer zones to reduce loading in urban and agricultural watersheds. As the combination of management initiatives are implemented throughout California, continued monitoring will be necessary to document how well they reduce contamination and toxicity in State waters. SWAMP is uniquely positioned to document how management actions affect contamination and toxicity state-wide.

SECTION 1
INTRODUCTION

The California Water Quality Control Boards (Water Boards) began incorporating Whole Effluent Toxicity (WET) testing in wastewater discharge permits in the early 1980s. Since then toxicity has been used to assess ambient water quality in freshwater, estuarine, and marine habitats. When combined with chemical analyses and other water quality measures, results of toxicity tests provide information regarding the capacity of water bodies to support aquatic life beneficial uses.

The word "toxicity" is defined here as a statistically significant adverse impact on standard aquatic test organisms in laboratory exposures. A number of different species, including crustaceans, algae, fish, and mollusks, have been used, following widely accepted test protocols with strict quality assurance. Laboratory toxicity test organisms are surrogates for aquatic species found in the environment. Toxicity tests are especially useful in water quality monitoring because they can detect the effects of all chemicals (whether measured or not) as well as pollutant mixtures. The organisms used in these tests have been chosen because they are relatively sensitive to toxic chemicals.

The relationship between results of laboratory toxicity tests and the ecological health of aquatic ecosystems has been the subject of much debate in the scientific literature. The majority opinion of scientists involved in ecotoxicology have concluded that when appropriately applied, toxicity tests provide useful insights on the potential effects of anthropogenic contaminants on aquatic systems. Assessments of ecosystem health are most conclusive when they incorporate multiple measures, including toxicity tests, chemical analyses, bioassessments, and measures of bioaccumulation (e.g. see Grothe et al., 1996; Ingersoll et al. 1997). These are the core components of the State Water Board's Surface Water Ambient Monitoring Program (SWAMP).

This document is the first comprehensive, statewide assessment of ambient toxicity monitoring data in over a decade. The purpose of this document is to summarize the location and magnitude of toxicity observed since the beginning of the SWAMP monitoring in 2001, to make recommendations for improving ambient toxicity monitoring throughout the state, and to support recommendations for toxicity monitoring in the Water Boards' regulatory programs.

SECTION 2
ASSESSMENT QUESTIONS

This document presents a summary assessment of toxicity in California watersheds and coastal waters using data from SWAMP and partner programs. The following questions are addressed:

1. Where has toxicity been observed in California waters?
2. What is the magnitude of observed toxicity?
3. What chemicals have been implicated as causing toxicity?
4. What are the ecological implications of aquatic toxicity?
5. How do the results of toxicity measurements compare among waters draining urban, agricultural, and other land cover areas?
6. How do toxicity test results compare when different statistical methods are applied, particularly with respect to use of the EPA Test of Significant Toxicity?
7. What management initiatives have the potential to reduce toxicity associated with contaminants in surface water?

SECTION 3 CAVEATS

The following points should be kept in mind when considering the information presented here:

1. Most of the data presented here were obtained from monitoring studies designed to increase understanding of potential biological impacts from human activities. Site locations were generally targeted in lower watershed areas, such as tributary confluences or upstream and downstream of potential pollutant sources. Only a minority of the sites were selected probabilistically (i.e., at random). Therefore, these data only characterize the sites monitored, and cannot be used to make assumptions about unmonitored areas.
2. The word "toxicity" is defined here as a statistically significant adverse impact on standard aquatic test organisms in laboratory exposures. A number of different species, including crustaceans, algae, fish, and mollusks, have been used, following widely accepted test protocols with strict quality assurance. These results should not be extrapolated to assess impacts on human health.
3. These results may underestimate ambient toxicity because most samples were collected as "grabs" by filling a sample bottle or collecting sediment at one point in time. Toxic chemicals often flow downstream in pulses. Studies in which test organisms were caged in-stream often have detected toxicity when grab sample tests have not.
4. The suite of standard laboratory toxicity test organisms do not represent the full range of sensitivity of the resident community. That is to say that laboratory organisms represent a certain range of sensitivity to chemicals that may be either more or less sensitive than the organisms naturally living in the stream.
5. Acute toxicity tests might not reflect chronic toxicity. These programs assessed toxicity during a short duration of an organism's life cycle with the maximum duration of exposure of 10 days. These short duration exposures may underestimate chronic toxicity effects associated with more realistic instream exposures.
6. This assessment integrates data sets from a number of programs. This integration was made possible by the SWAMP quality assurance conventions and the SWAMP and California Environmental Data Exchange Network (CEDEN) data management system. There are, however, data from a number of other monitoring programs that have not yet been submitted to CEDEN and were not used in this analysis. Information on data sources is given in Tables 4 and 5.
7. The different programs often had different monitoring objectives, and there is large variation in the number of samples collected at each site and the number of sites surveyed in each Region. This limits the ability to make meaningful comparisons between different regions.

SECTION 4 FINDINGS

Information is presented below to answer the key assessment questions. Data sources and literature cited are presented at the end of the report.

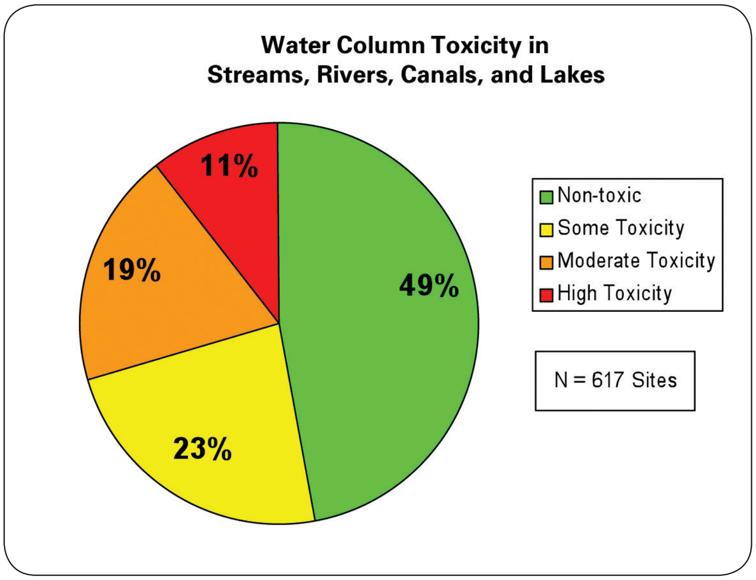
WHERE HAS TOXICITY BEEN OBSERVED IN CALIFORNIA WATERS?

The attached maps (Figures 1 – 6, at the end of this document) show locations of sites sampled for toxicity by SWAMP and partner programs, and the intensity of toxicity observed in the water and sediment samples collected at those sites. Sites are color coded using the categorization process described in Figure 11, which combines the results of all toxicity tests performed on samples collected at a site to quantify the degree of toxicity observed there. At sites where both water and sediment toxicity data were collected, two toxicity categories were calculated, to separately summarize the degree of toxicity in water and in sediment. Toxicity endpoints evaluated in this analysis included the mortality of fish and invertebrates and the density of cells in growing cultures of single-celled algae. To control for variation in test organism performance, every test result was expressed as a percentage of the survival or cell density observed in the laboratory control. Sublethal endpoints, such as inhibition of reproduction or growth, were not considered. Relative to the 303(d) impaired waterbody listing process, a site coded "green" would not be listed for toxicity. Sites coded "yellow" to "red" may be listed if the number of toxic samples met the criteria outlined in the State Water Board's Listing and Delisting Policy. This assessment used more recent data than was assessed for the most recent 303(d) impairment assessment so classifications identified here will not necessarily coincide with the impaired waters list.

Toxicity has been observed in all Regions. Streams in upper watersheds and mountainous areas tend to produce fewer toxic samples, while samples from downstream sites in the valleys and along the coasts tend to be more toxic. These lower watershed sites drain larger areas with greater levels of human activity. Consistent sediment toxicity has been observed in many bay and harbor sites. In most years since 1991, for example, annual surveys of San Francisco Bay have shown at least moderate sediment toxicity at a number of sites throughout the Bay.

WHAT IS THE MAGNITUDE OF OBSERVED TOXICITY?

Of the 617 sites monitored for water toxicity in this assessment, 327 (53%) had at least one sample in which toxicity to at least one test species was observed. Of these, 65 (10.5% of the total) were classified as high toxicity sites, meaning that the average result for the most sensitive species in all samples at the site was



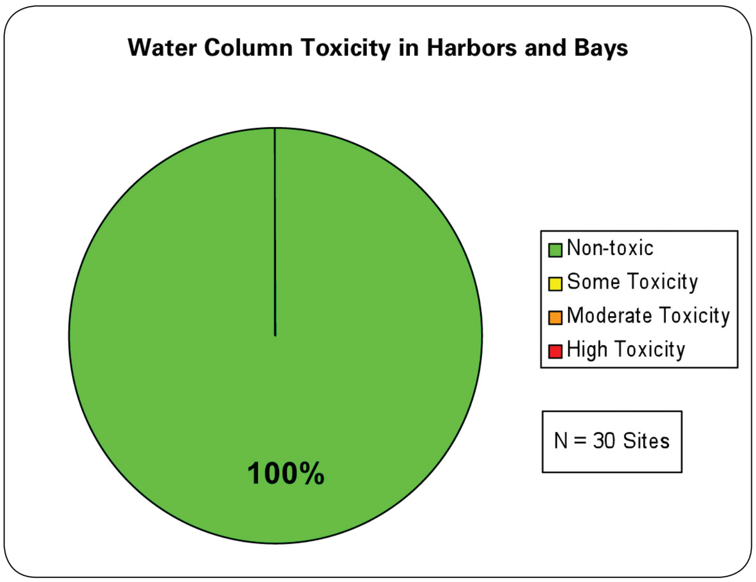


Figure 7. Magnitude of toxicity in water statewide, listed separately for freshwater sites and harbors/bays. Color coding is as shown in Figure 11.

more toxic than the high toxicity threshold for that species (Figure 7). In addition to freshwater sites, 30 samples collected in harbors and bays were evaluated for water toxicity to marine species. None of these samples were toxic.

Results of statewide water toxicity tests with the three standard EPA test species show that more samples were toxic to fish larvae (*Pimephales promelas*) and algae (*Selenastrum capricornutum*), than to water fleas (*Ceriodaphnia dubia*; Table 1). However, the magnitude of toxicity to fish was generally lower, with 5% of the sites demonstrating moderate to high toxicity. Results with water fleas showed that 23% of the sites demonstrated moderate to high toxicity. The magnitude of toxicity to algae was comparable to that observed with water fleas; approximately 19% of the sites demonstrated moderate to high toxicity to algae.

A total of 521 freshwater sites were monitored for sediment toxicity. Of these, 235 sediment samples (45.1%) were classified as demonstrating some toxicity (Figure 8), and 88 (16.9%) were classified as high toxicity. It should be noted that the majority of SWAMP freshwater sediment toxicity monitoring uses the 10 day toxicity test protocol with the amphipod *Hyalella azteca* which is conducted at a standard temperature of 23 °C. While this test is routinely used throughout the United States, recent data suggest this protocol likely underestimates sediment toxicity. Research by Ingersoll et al. (2005) show that the 28 day chronic growth and survival test with *H. azteca* is more sensitive than the 10 day test, and is a more conservative indicator of ecological impacts. In addition to underestimating chronic sediment toxicity, 10 day *H. azteca* tests conducted at 23 °C also underestimate pyrethroid pesticide toxicity because pyrethroids are more toxic at colder temperatures. Recent state-wide monitoring in California by Holmes et al. (2008) showed much greater sediment toxicity to amphipods in tests conducted at 15 °C vs. those conducted at 23 °C. Results showed this was due to pyrethroid pesticides. Results of a SWAMP database query of freshwater habitats sampled throughout California showed that the average surface water temperature was 15.3°C and ranged from ~0 to 37°C in the years 2001 – 2010

Table 1
Species-specific maximum levels of water toxicity observed at sites tested with *C. dubia*, *P. promelas* and *S. capricornutum* toxicity tests.

Species	Number of Sites	Maximum Toxicity Level Observed (%)			
		Non-toxic	Some Toxicity	Moderate Toxicity	High Toxicity
C. dubia	545	70.8	6.2	15.8	7.2
P. promelas	455	64.2	31.0	4.0	0.9
S. capricornutum	326	58.3	23.0	13.5	5.2

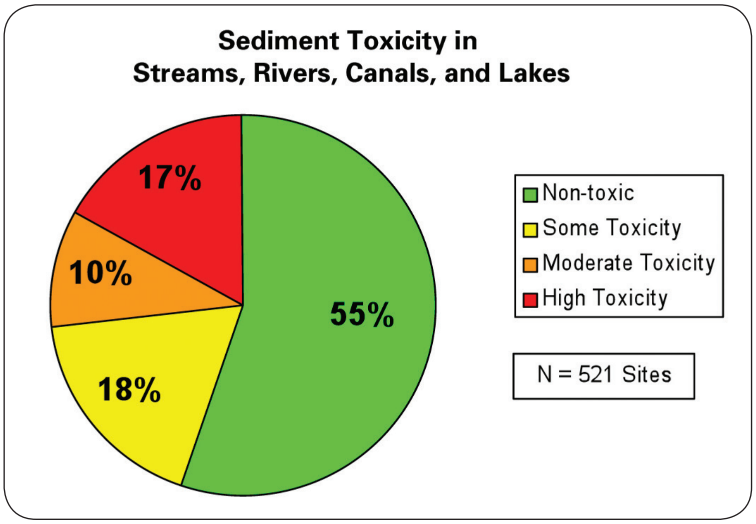
(n = 12, 279; personal communication Cassandra Lamerdin). The average temperatures ranged between 11.3 °C and 21.9 °C. Note that these reflect discreet daytime temperatures measured during routine SWAMP monitoring and do not represent continuous temperature readings. This demonstrates that ambient temperatures in California are sometimes far below the standard 23 °C *H. azteca* protocol temperature. Since photostable pyrethroids are present year-round, this suggests that surface water toxicity due to pyrethroids is likely to be an even greater problem in California watersheds than has previously been reported.

In addition to the freshwater stations, 171 harbor and bay sediment samples were monitored for toxicity, and 81 of these were non-toxic (47.4%). Of the remaining sites, 90 (52.6%) were classified as demonstrating some toxicity, and 24 (14%) were designated high toxicity (Figure 8).

HOW DO THE RESULTS OF TOXICITY MEASUREMENTS COMPARE AMONG WATERS DRAINING URBAN, AGRICULTURAL, AND OTHER LAND COVER AREAS?

Sites in agricultural and urban areas were significantly more toxic than sites in less developed areas (measured by minimum observed survival or algal growth (% of control performance), Figure 9a and 9b), and greater percentages of urban and agricultural sites were toxic (Figure 10a and 10b). Greater water toxicity was observed in samples from urban and agricultural sites relative to undeveloped ("other") sites (Figure 9a). The difference in water toxicity between undeveloped and urban areas was highly statistically significant ($p = 0.0007$); and the same is true for the difference between undeveloped and agricultural areas ($p < 0.0001$). Greater water toxicity was observed in agriculture sites relative to urban sites ($p < 0.0001$). As discussed below, evidence suggests that water toxicity in agriculture samples has largely been due to organophosphate pesticides (OPs). The greater water toxicity in agriculture sites may reflect this, since organophosphate pesticide use has declined in urban settings in recent years.

Greater sediment toxicity was observed in samples from urban and agricultural sites relative to undeveloped sites (Figure 9b), and toxic sediments were found at greater percentages of sites with urban or agricultural land uses (Figure 10b). The difference in sediment toxicity (as mean survival) between undeveloped and urban areas was highly statistically significant ($p < 0.0001$). The difference in sediment toxicity between undeveloped and



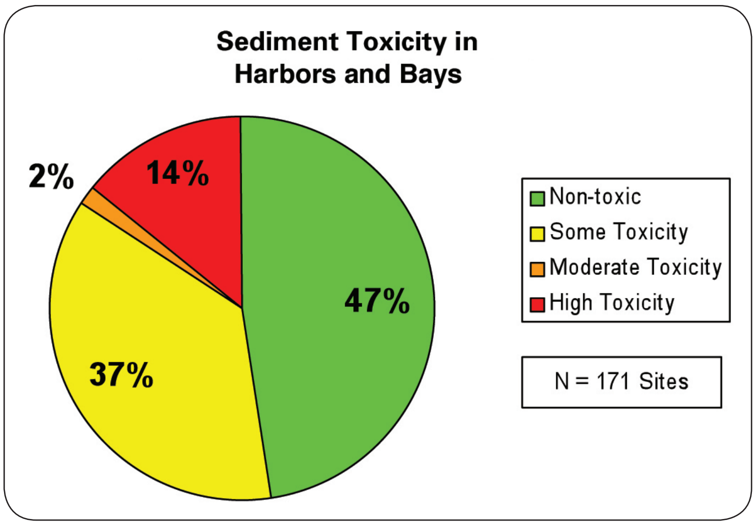


Figure 8. Magnitude of toxicity in sediment statewide, listed separately for freshwater sites and harbors/bays. Color coding is as shown in Figure 11.

agricultural areas was also statistically significant ($p < 0.0122$). There was no significant difference in sediment toxicity between agricultural and urban sites ($p = 0.0828$). As discussed below, evidence suggests that sediment toxicity in agriculture and urban samples has largely been due to pyrethroid pesticides. The lack of a significant difference in sediment toxicity between urban and agriculture sites may reflect that use of this class of pesticides is prevalent in both land use categories. It should be noted that classification of agricultural lands in this analysis included cultivated crops and hay and pasture lands, and that cultivated crops does not distinguish between different categories (e.g., row crop agriculture vs. orchards). Although pesticide use likely varies widely between these different agricultural land uses, these differences are not reflected in the current analysis

WHAT CHEMICALS HAVE BEEN IMPLICATED AS CAUSING TOXICITY?

There are thousands of pollutants that can cause biological impacts in waterways, and only about 140 are routinely measured. Ambient water and sediment samples often contain complex mixtures of many pollutants, often with additive effects. Toxicity tests are especially useful in pollution monitoring because they can detect the effects of all chemicals (whether measured or not) as well as pollutant mixtures. Two approaches are used to evaluate which chemicals are causing toxicity. Correlations analyses provide insights on statistical relationships between contaminants and effects, and these data can be used to build a weight of evidence for the cause of toxicity. Toxicity identification evaluations (TIEs) are laboratory procedures designed to provide direct experimental evidence of specific chemicals causing toxicity. Results of both approaches are provided below.

Causes of Water Toxicity

The vast majority of studies in California that have included TIE evidence have demonstrated that water toxicity is caused by pesticides. These studies primarily have been conducted with two test species, the cladoceran

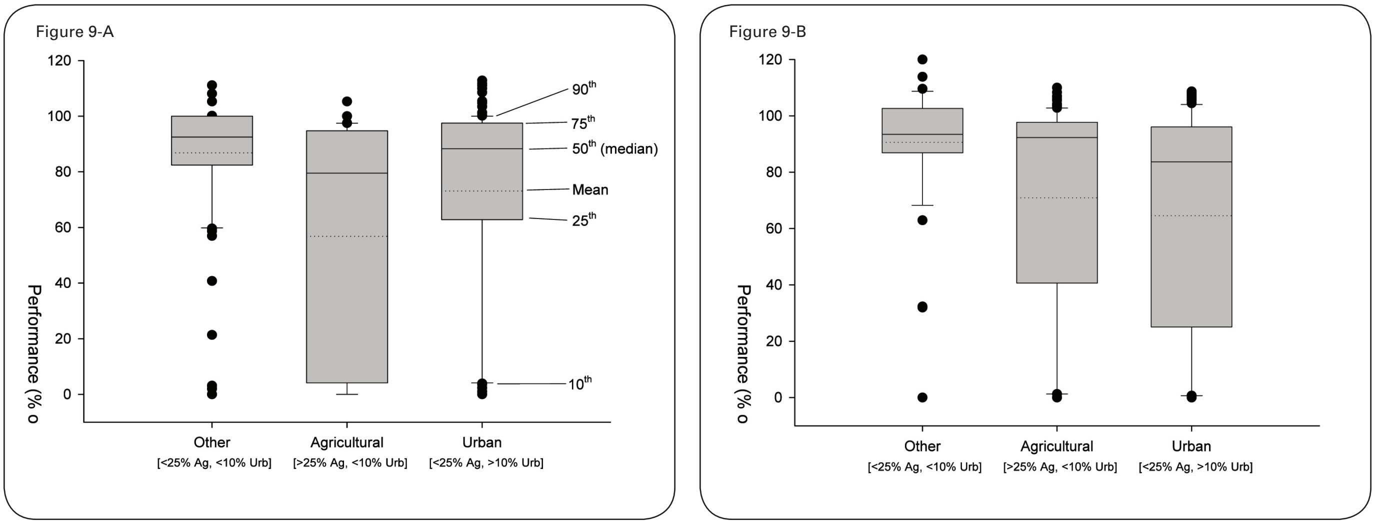


Figure 9. Toxicity distribution for samples collected from sites in urban, agricultural, and less developed areas. Lower values represent lower levels of survival, and indicate higher toxicity. Data are for the most sensitive test species at each site. Solid lines, from top to bottom, represent the 90th, 75th, 50th (median), 25th and 10th percentiles of the distribution. Dotted lines are the mean result. (A) Water column toxicity, (B) Sediment toxicity.

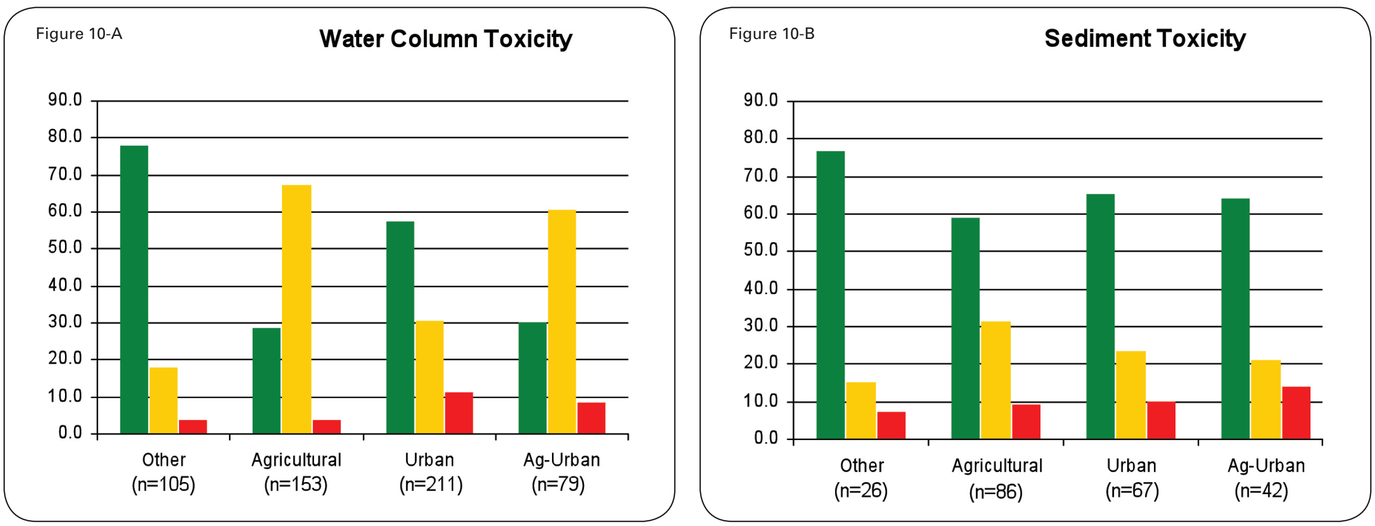


Figure 10. Numbers of sites (as a percentage of all sites in each land-cover category) where water and sediment were classified as nontoxic, moderately toxic, or highly toxic, using the coding system shown in Figure 11. "Some Toxicity" and "Moderate Toxicity" categories are combined here.

Ceriodaphnia dubia, and the amphipod *Hyalella azteca*. The first (published) ambient water TIE study conducted in the United States used water samples from the Colusa Basin Drain, in Sacramento County. This study identified methyl parathion, carbofuran, and malathion, pesticides associated with rice irrigation drainage, as the cause of toxicity to *C. dubia* (Norberg-King et al., 1991). Subsequent studies conducted by researchers at the UC Davis Aquatic Toxicology Laboratory (ATL) for the Central Valley Regional Water Quality Control Board used TIEs with *C. dubia* to document wide-spread toxicity from organophosphate pesticides in the Central Valley. The Central Valley TIE studies were conducted in agriculture drainage ditches, urban creeks, on main stem sections of the Sacramento

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and San Joaquin Rivers, and on tributaries to these rivers. This work is summarized in a review paper by de Vlaming et al. (2000) and demonstrated the principal pesticides responsible for ambient toxicity in this region were diazinon and chlorpyrifos. Toxicity was documented in both the dry season and rainy season and the majority of TIEs identified these two OP pesticides as the cause of toxicity (de Vlaming et al., 2000). Additional studies conducted in urban creeks in the San Francisco Bay area and at various sites in the San Joaquin River watershed showed that there was sufficient diazinon and chlorpyrifos in these watersheds to account for observed toxicity to *C. dubia*, but no TIEs were conducted using these samples (de Vlaming et al., 2000). TIE evidence from the Central Valley monitoring led to additional monitoring and TIE studies in other agricultural areas of California. TIE studies using samples from the Alamo River in the Imperial Valley demonstrated that toxicity to *C. dubia* was due to diazinon, chlorpyrifos and carbofuran (de Vlaming et al., 2000). TIE studies in the Calleguas Creek watershed on the Oxnard Plain demonstrated that toxicity to *C. dubia* was due to diazinon and chlorpyrifos. Toxicity to fish (*P. promelas*) in this watershed was due to ammonia (Anderson et al., 2002). TIE studies in the Central Coast region also have demonstrated that water toxicity to *C. dubia* is caused primarily by chlorpyrifos and diazinon. These include studies in the Salinas River and its tributaries (Anderson et al., 2003a; Anderson et al., 2003b; Hunt et al., 2003; Phillips et al., 2004), studies in the Pajaro River watershed (Hunt et al., 1999), and studies in the Santa Maria River and its tributaries (Anderson et al., 2006a; Anderson et al., 2010a; Phillips et al., 2010a).

In addition to TIE studies, the SWAMP statewide database evaluated for the current report was analyzed to determine statistical correlations between *C. dubia* survival and selected pesticide concentrations. Pesticide concentrations were divided by their corresponding median lethal concentrations (LC50s) for toxicity to *C. dubia* to provide toxic units (TUs) for each chemical. Based on the TIE evidence, TUs for pyrethroid pesticides and the OP pesticides diazinon and chlorpyrifos were calculated for all samples in which these pesticides were measured ($n = 466$). Pesticide TUs were summed and regression analysis was conducted. The results demonstrate a highly significant negative relationship between summed TUs and *C. dubia* survival ($p < 0.0001$).

Use of the OP pesticides diazinon and chlorpyrifos has decreased in California since the mid-1990s. Further declines have occurred in urban areas since U.S. EPA actions resulted in a phase out of these two pesticides for residential use in the early 2000s (Spurlock and Lee, 2008). While diazinon and chlorpyrifos are still used in agriculture, they have been replaced by pyrethroids for most urban applications. The majority of pyrethroids used in California (90%)

As pyrethroid pesticides have increased in use there is growing evidence of sediment toxicity to the amphipod H. azteca due to these pesticides

are those that are more photostable compounds, and these are the pyrethroids that have been identified as causing surface water toxicity. As pyrethroid pesticides have increased in use there is growing evidence of sediment toxicity to the amphipod *H. azteca* due to these pesticides (see following discussion). Because this species is particularly sensitive to pyrethroids and recent TIE studies have demonstrated water toxicity to *H. azteca* is due to pyrethroids, it is being incorporated into water toxicity monitoring programs as an indicator for this class of pesticides. These include studies in the New River in the Imperial Valley (Phillips et al., 2007), studies in various creeks in the Central Valley, and in the American and San Joaquin Rivers (Weston and Lydy, 2010a, b), and a study in the Sacramento-San Joaquin Delta in the northern San Francisco Estuary (Werner et al., 2010). Because *H. azteca* is considerably more sensitive to pyrethroids than *C. dubia*, there is increasing interest in using both species in agriculture water monitoring to account for mixtures of OP and pyrethroid pesticides. In addition, *H. azteca* is increasingly being considered as a replacement for *C. dubia* in urban stormwater monitoring because of the increasing use of pyrethroids in these settings.

No SWAMP studies to date have been specifically designed to determine whether changes in (urban) pesticide use patterns from OP to pyrethroid pesticides have resulted in changes in toxicity patterns. This would require continuous chemistry and toxicity monitoring at selected stations over the period before and after the residential restrictions in OP use, and no peer-reviewed studies were designed to capture this pattern. In addition, pyrethroid pesticides are toxic at very low concentrations. Analytical methods sufficient to quantify them at these low levels were not sufficiently developed in the late 1990s and early 2000s, particularly in ambient water samples. Several regional surface water monitoring studies provide data to illustrate how the change from OPs to pyrethroids in urban watersheds has resulted in changes in toxicity patterns. These include studies conducted in Bay Area urban creeks and in the San Francisco Estuary by the San Francisco Estuary Institute's Regional Monitoring Program (Anderson et al., 2003c), and a total maximum daily load (TMDL) study in the Chollas Creek watershed in San Diego (WestonSolutions, 2006, 2007). In addition, Johnson et al. (2010) modeled long-term trends showing reductions in surface water concentrations of diazinon and chlorpyrifos in urban and agriculture streams in the California Central Valley from the mid 1990s to 2005. Although no toxicity data was analyzed, diazinon concentrations in these Central Valley watersheds were shown to be below the chronic toxicity threshold after implementation of the California Central Valley TMDL for diazinon (Johnson et al., 2010). All three of these studies coincided with an increase in either sediment and/or water toxicity in San Francisco, Chollas Creek, and Central Valley watersheds due to pyrethroids.

While the preponderance of evidence shows that insecticides account for much of the water toxicity to freshwater invertebrate test species in California, a few (unpublished) TIE studies also have shown toxicity due to other contaminant classes. For example, testing with the green alga *Selenastrum capricornutum* has suggested herbicides inhibited algal growth in the Susan River (Fong et al., 2006) and in the Sacramento River Delta (AquaScience, 2002). In addition, TIE studies of stormwater toxicity in coastal receiving waters have shown toxicity due to zinc and copper to marine invertebrates and fish larvae (Phillips et al., 2003; Schiff et al., 2006).

TIE studies of stormwater toxicity in coastal receiving waters have shown toxicity due to zinc and copper to marine invertebrates and fish larvae

The SWAMP statewide database also was analyzed for correlations between algal growth (*S. capricornutum*) and selected chemicals in water, and between larval fish survival (*P. promelas*) and aqueous chemicals. There were no significant negative correlations between total herbicides in water and *S. capricornutum* growth ($p = 0.2528$; $n = 140$), or between algal growth and nitrates in water ($p = 0.4734$; $n = 169$). Weak negative correlations were determined for larval fish survival and copper ($p = 0.025$; $n = 151$) and between larval fish survival and zinc ($p = 0.021$; $n = 151$).

Causes of Sediment Toxicity

There are fewer examples of sediment TIEs conducted in California. This is due to the fact that there is a longer history of water toxicity monitoring in this state, and because sediment TIE methods have only recently been sufficiently developed for routine use. Sediment TIEs using *H. azteca* have been conducted in most regions of California where toxicity has been observed. As discussed above, the majority of these studies have demonstrated that sediment toxicity is due to pyrethroid pesticides. Other studies have shown sediment toxicity is due to the OP pesticide chlorpyrifos, or to mixtures of chlorpyrifos and pyrethroids. The majority of these studies have been conducted in the Central Valley and on the Central Coast. Sediment TIEs conducted in the Santa Maria River and its estuary demonstrated that toxicity to *H. azteca* was due to mixtures of chlorpyrifos and the pyrethroids cypermethrin and cyhalothrin (Anderson et al., 2006a; Anderson et al., 2010a; Phillips et al., 2010a). TIEs conducted in the Salinas River watershed have also reported sediment toxicity is due to the pyrethroids cyhalothrin and cypermethrin, and in some cases, the OP pesticide chlorpyrifos (Weston and Amweg, 2007; Anderson et al., 2008; Hunt et al., 2008). Studies conducted in the Central Valley have found the pyrethroid bifenthrin is largely responsible for toxicity to *H.*

Holmes et al. (2008) conducted statewide monitoring of sediment toxicity associated with pyrethroids in 30 urban creeks, and found pyrethroids pesticide concentrations were sufficient to account for the observed toxicity in the water bodies tested.

azteca, but toxicity also has been due to mixtures of bifenthrin, cyfluthrin, and cyhalothrin (Weston and Amweg, 2007; Anderson et al., 2008; Weston et al., 2008).

Holmes et al. (2008) conducted statewide monitoring of sediment toxicity associated with pyrethroids in 30 urban creeks, and found pyrethroids pesticide concentrations were sufficient to account for the observed toxicity in the water bodies tested. Four of these creeks were selected for TIE studies, and Phillips et al. (Phillips et al., 2010b) reported toxicity in the three southern California creeks was due to mixtures of bifenthrin, cyfluthrin and cypermethrin. Sediment toxicity in Marsh Creek, a tributary to the Sacramento/San Joaquin Delta, also was due to these same pyrethroids (Phillips et al., 2010b). There have also been numerous other studies that presented toxicity and chemical evidence of pyrethroid-associated sediment toxicity without TIEs. In addition to Holmes et al. (2008), these include studies in the Central Valley by Weston et al. (2004) and studies by Amweg et al. (2006) in Kirker Creek and other urban creeks in the eastern San Francisco Bay area.

In addition to TIE studies, the SWAMP statewide database also was analyzed to determine statistical correlations between *H. azteca* survival in sediments and selected pesticide concentrations. As discussed above, pesticide concentrations were divided by their corresponding LC50s for toxicity to *H. azteca* to provide Toxic Units (TUs) for each chemical. Based on the TIE evidence, TUs for pyrethroid pesticides and the OP pesticide chlorpyrifos were calculated for all samples in which these pesticides were measured (n = 185). For this analysis, the sediment pesticide concentrations were normalized to the total organic carbon content in the sediment because this has been shown to effect chemical bioavailability. Bioavailability refers to the capacity of chemicals to cross cell membranes and therefore cause toxicity. Studies have shown that the organic carbon content in sediments is a good predictor of organic chemical bioavailability and toxicity. Pesticide TUs were summed and the correlation was evaluated using the Spearman Rank Sum procedure. The results demonstrate a highly significant negative relationship between summed pyrethroid and chlorpyrifos TUs and *H. azteca* survival ($p = 0.0077$).

Table 2 summarizes dozens of studies in which TIEs have identified the causes of toxicity in ambient water and sediment samples from California, from 1991 to the present. With the exception of ammonia, all of these ambient TIEs implicated pesticides, primarily OPs and, more recently, pyrethroids. It is important to note that pesticides are implicated in streams draining residential and urban areas as well as agricultural land.

Table 2
Classes of chemicals and specific compounds shown to have caused toxicity in California. Numbers represent the numbers of water and sediment samples on which TIEs were conducted by the various studies.

Class	Compound	Water	Sediment
Ammonia	Ammonia	1	-
Carbamate Pesticide	Carbofuran	4	-
Organophosphate Pesticide	Chlorpyrifos	11	4
	Diazinon	13	-
	Ethyl Parathion	1	-
	Malathion	3	-
	Methyl Parathion	3	-
Pyrethroid Pesticide	Bifenthrin	4	8
	Cyfluthrin	3	3
	Cyhalothrin	2	7
	Cypermethrin	-	8
	Esfenvalerate	1	-
		-	1

Note that the likelihood of conducting a successful TIE increases with increasing magnitude of toxicity in a particular sample. As a result, more TIEs were conducted using *C dubia* because the magnitude of toxicity was greater with this species than to fathead minnows (*P. promelas*). This may skew the evaluation of what causes toxicity since for the most part there were few TIEs conducted on the fish.

WHAT ARE THE ECOLOGICAL IMPLICATIONS OF AQUATIC TOXICITY?

The principal approach to determine whether observations of toxicity in laboratory toxicity tests are indicative of ecological impacts is to conduct field bioassessments. Bioassessment is the characterization of environmental conditions through the use of biological organisms. Bioassessments may be conducted using a variety of different ecological communities ranging from periphyton to fish communities, but the most commonly used method

assesses impacts on macroinvertebrate communities. These studies are most informative when toxicity, chemistry and macroinvertebrate bioassessments are conducted at the same time and place, and these are referred to as "triad" studies because they involve three lines of evidence. Based on the findings of recent national workshops convened to evaluate indicators for water quality monitoring, programs that include chemistry, toxicity testing, bioassessments and bioaccumulation studies provide the most robust measures of environmental quality (Grothe et al., 1996; Ingersoll et al., 1997). These core indicators form the backbone of current SWAMP monitoring in California.

Because distributions of freshwater insects and other macroinvertebrates are influenced by habitat, water temperature, sedimentation, and numerous other non-chemical "stressors", the most comprehensive studies include characterizations of all factors which may influence their ecology. While there are a limited number of studies focused on California waters in the published literature, there is a large international body of evidence showing pesticides and other chemicals are linked to impacts on aquatic ecosystems, and these include microcosm and field studies. A review article by Schulz (2004) includes many of these papers.

The influence of habitat quality on macroinvertebrates was also assessed and it was concluded that habitat was a less important factor than pesticides

One of the most comprehensive series of studies linking water and sediment toxicity with impacts on resident macroinvertebrates in California was conducted in the Salinas River. In these studies, water and sediment toxicity was caused by diazinon and chlorpyrifos from agriculture runoff (Anderson et al., 2003a; Anderson et al., 2003b; Phillips et al., 2004). Bioassessments showed macroinvertebrate densities declined downstream of two agriculture drain water inputs, including densities of the resident amphipod *H. azteca* and of the mayfly genus Procloeon. A subsequent study demonstrated that Procloeon were sensitive to chlorpyrifos at concentrations found in the river, and that this species was not influenced by suspended particles (Anderson et al., 2006b). The influence of habitat quality on macroinvertebrates was also assessed and it was concluded that habitat was a less important factor than pesticides (Anderson et al., 2003b). A similar series of studies were conducted on the lower Santa Maria River. Toxicity studies and TIEs showed toxicity in the Santa Maria River was caused by mixtures of chlorpyrifos, diazinon, and pyrethroids (Anderson et al., 2006a; Phillips et al., 2006), and stations with the greatest contamination and toxicity also had the lowest macroinvertebrate densities. Because the amphipod *H. azteca* occurs naturally in freshwater systems throughout the central coast of California and is a standard toxicity test species, disappearance of these amphipods at the most toxic stations provides a key linkage between laboratory toxicity and field impacts. Subsequent studies in the Santa Maria River Estuary have demonstrated that water and sediment toxicity due to OP and pyrethroid pesticides extend into this estuary (Anderson et al., 2010a; Phillips et al., 2010a). Stations with the greatest contamination in this estuary also have the lowest macroinvertebrate densities, have lower numbers of amphipods, and higher numbers of pollution-tolerant species (Anderson et al., 2010a).

There is growing evidence that pyrethroids in coastal urban creeks may accumulate to toxic concentrations in nearshore marine systems. Holmes et al. (2008) found toxicity to *H. azteca* at a number of urban creeks in southern California including Switzer Creek in San Diego County, Peters Canyon Wash in the San Diego Creek watershed in Orange County, and Ballona Creek in Los Angeles County. Additional studies in San Diego Harbor (Anderson et al., 2005; Anderson et al., 2010b), Upper Newport Bay and the San Diego Creek watershed (Bay et al., 2005; Anderson et al., 2007; Ranasinghe et al., 2007; Phillips et al., 2010b), and the Ballona Creek Estuary (Bay et al., 2005; Ranasinghe et al., 2007; Lao et al., 2010) have shown persistent sediment toxicity, TIE evidence of toxicity due to pyrethroids, and degradation of marine infaunal communities in these receiving systems. Future monitoring should emphasize these important coastal habitats.

Additional triad investigations linking chemistry, toxicity, and macroinvertebrate bioassessments have been conducted in urban and agriculture-dominated creeks in the Central Valley. Weston et al. (2005) conducted bioassessments in Pleasant Grove Creek in the Roseville area of Sacramento. These authors found declines in densities of *H. azteca* in sections of the creeks that are toxic to this amphipod in laboratory tests. Synoptic chemistry showed several pyrethroids, including bifenthrin, exceeded toxicity thresholds to *H. azteca*. However, Hall et al. (2009) conducted additional assessments in Pleasant Grove Creek and Kirker Creek, in study areas that had previously demonstrated toxicity ascribed to pyrethroids. These authors concluded that physical habitat played a larger role in macroinvertebrate distributions than chemical contaminants. Hall et al. (2007) also used statistical analyses to investigate the relationships between farm level pesticide use and physical habitat metrics on benthic macroinvertebrate communities in Orestimba Creek. This creek has been the subject of previous Regional Water Board studies indicating water and sediment toxicity to pyrethroid and OP pesticides. Benthic macroinvertebrate densities and distributions varied widely in this system, and Hall et al. (2007) concluded that there were only weak statistical associations between benthic communities and pesticide applications and between benthos and physical habitat.

These authors found declines in densities of H. azteca in sections of the creeks that are toxic to this amphipod in laboratory tests.

HOW DO TOXICITY TEST RESULTS COMPARE WHEN DIFFERENT STATISTICAL METHODS ARE APPLIED, PARTICULARLY WITH RESPECT TO USE OF THE EPA TEST OF SIGNIFICANT TOXICITY (TST)?

State Water Board staff is developing, for Board consideration, a Policy for Toxicity Assessment and Control to establish numeric water quality objectives for toxicity. The proposed policy includes numeric water quality objectives for chronic and acute toxicity, a prescribed statistical methodology for determining whether a sample is toxic that is based on the U.S. EPA's Test of Significant Toxicity (TST), monitoring requirements for wastewater, and recommendations for monitoring stormwater and some non-point source discharges. The TST also would be applicable to monitoring conducted by SWAMP so this assessment was conducted using the TST.

In toxicity testing of ambient or stormwater samples, a single sample (e.g., ambient water) often is compared to a laboratory control. In these tests, the objective is to determine if a given sample of site water is toxic, as indicated by a significantly different organism response compared to the control using a t-test or similar statistic. To demonstrate the TST approach for ambient toxicity programs, U.S. EPA compiled SWAMP data from 409 chronic tests for *C. dubia* (crustacean) and 256 chronic tests for *P. promelas* (fish) to compare results of the two statistical approaches. The following data are from the U.S. EPA TST Technical Document (U.S. EPA, 2010).

Table 3 summarizes results of *C. dubia* tests analyzed with an $\alpha = 0.20$ for the TST test method. Although the majority (92%) of these comparisons resulted in the same decision using either the TST or the t-test approach, approximately 6 percent of the tests (24 tests) would have been declared not toxic using the traditional t-test approach when the TST would declare them toxic. In addition, 2 percent of the tests (7 tests) would have been declared toxic using the t-test approach when the TST would not indicate toxicity.

Table 3
Comparison of results of chronic *C. dubia* ambient toxicity tests using the TST approach and the traditional (t-test) analysis. For the TST approach $\alpha = 0.20$ and b value = 0.75. For the t-test approach $\alpha = 0.05$.

The two approaches agree 92% of the time.		EPA Test of Significant Toxicity	
		Toxic	Non-Toxic
Traditional (t-test)	Toxic	**20%**	2%
	Non-Toxic	6%	**72%**

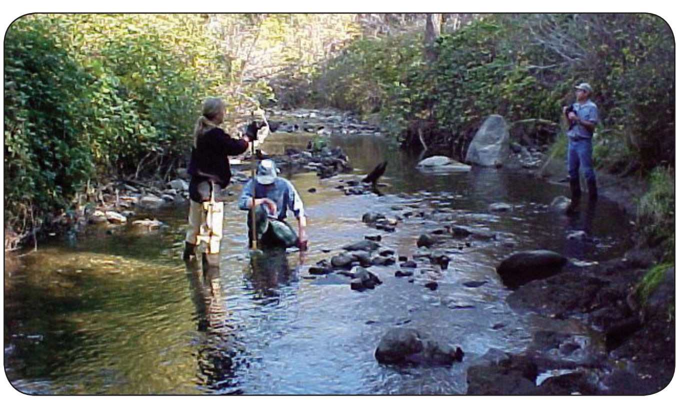
This analysis indicates there is little difference in the assessment of ambient toxicity regardless of which statistical method is applied to the data. The value of utilizing the TST methodology is that it can be applied to all ambient and effluent toxicity monitoring improving consistency and comparability among all Water Board monitoring programs.

WHAT MANAGEMENT INITIATIVES HAVE THE POTENTIAL TO REDUCE TOXICITY ASSOCIATED WITH CONTAMINANTS IN SURFACE WATERS?

Evidence suggests that pesticides are the most common cause of acute surface water toxicity in California, therefore initiatives that reduce agricultural and urban sources of pesticides should improve water quality. The primary regulatory mechanism for reducing pesticide loading in urban and agriculture watersheds in California is the development and implementation of a TMDL. A TMDL must be developed when a water body is listed as impaired per section 303(d) of the Federal Clean Water Act (see: http://www.swrcb.ca.gov/water_issues/programs/tmdl/, for TMDLs listed by region). There are several examples where the combination of U.S. EPA use changes and TMDLs for the pesticides diazinon and chlorpyrifos have resulted in significant reductions in loadings of these pesticides. These include use restrictions and TMDLs discussed above in the California Central Valley, and the Chollas Creek watershed. More recent pesticide TMDLs include those that have been, or are being implemented in the Calleguas Creek, Salinas River, and Santa Maria River Watersheds, and in the San Diego Creek watershed in Southern California.

Through implementation of TMDLs and label changes on use of specific agricultural pesticides, their occurrence in runoff should be reduced. For example, in July of 2006, the USEPA published the Re-registration Eligibility Decision (RED) for diazinon (http://www.epa.gov/oppsrrd1/reregistration/REDs/diazinon_red.pdf) which reduced the number and pounds of application of this pesticide for crops such as lettuce, strawberries, and broccoli. This RED requires that products sold after 2009 contain the new use guidance. A recent analysis of pesticide use data in Monterey County shows reduced use of diazinon after 2009, and increased use of malathion (personal communication, L. Harlan Central Coast Regional Water Quality Control Board). It is too soon to determine how these changes affect surface water quality.

U.S. EPA and the California Department of Pesticide Regulation (CDPR) have initiated reviews of pyrethroid pesticides registrations (Spurlock and Lee, 2008). CDPR currently is developing use restrictions for pyrethroid pesticides used by pest control businesses in urban settings (for public comment August, 2011; Personal Communication, John Sanders, California DPR). CDPR also plans on following urban restrictions with regulations to address agricultural use of pesticides affecting surface water quality in 2012 (Personal communication, John Sanders,



The primary regulatory mechanism for reducing pesticide loading in urban and agriculture watersheds in California is the development and implementation of a TMDL.

CDPR). The U.S. EPA also is requiring label changes for pyrethroid products to reduce their impact on surface water quality (http://www.regulations.gov/#!documentDetail;D=EPA-HQ-OPP-2008-0331-0021). In addition to restrictions on pounds of active ingredients applied per acre and number of applications per crop, use restrictions typically involve recommendations for vegetated buffer zones and setbacks to limit the potential for off-field transport of pesticides in spray drift, irrigation and stormwater runoff.

In addition to pesticide use restrictions, numerous state, regional and local programs are intended to reduce urban and agricultural contaminant runoff through implementation of various management practices. In urban settings these include incorporation of low impact development (LID) techniques to curb runoff and sequester contaminants, increased urban street sweeping to reduce total suspended solids in stormwater, and homeowner education programs to increase public awareness about water and biocide use. In agriculture settings management practices include use of integrated pest management techniques, efficient irrigation systems, setbacks, treatment wetlands and vegetated treatment systems to reduce pesticide and nutrient loading in runoff. Recent research also has demonstrated that incorporation of hydrolytic enzymes in integrated vegetated treatment systems work to detoxify organophosphate pesticides in agricultural runoff (Anderson et al., 2011). Similar enzymes systems currently are being developed for pyrethroid detoxification by the Australian government.

As the combination of management initiatives described above are implemented throughout California, continued monitoring will be necessary to document how well they reduce contamination and toxicity in State waters. For example, SWAMP's Stream Pollution Trends (SPoT) monitoring program is designed to detect long-term changes in contamination and sediment toxicity on a watershed scale. This program is uniquely positioned to document how management actions affect contamination and sediment toxicity state-wide. Coordination of SWAMP's regional and statewide monitoring programs with regional stormwater and agriculture cooperative monitoring programs and other state and federal agency monitoring will allow for more efficient use of resources and greater watershed coverage.

SECTION 5
CONCLUSIONS AND RECOMMENDATIONS

- Toxicity in California surface waters is widespread and evidence suggests it's largely due to pesticides.
- Increasing evidence of pyrethroid toxicity in water suggests the need for more water testing with the amphipod *Hyalella azteca*.
- Data from the SWAMP regional and statewide monitoring programs should be useful in detecting changes in toxicity patterns over larger spatial and temporal scales. There is a need for consistency in monitoring to capture emerging trends.
- The State Water Board should require replicate-level toxicity data be entered into or available for exchange through the California Environmental Data Exchange Network (CEDEN).
- The SWAMP regional and statewide monitoring needs to be better coordinated with other monitoring programs as appropriate (e.g., stormwater and other NPDES monitoring, Irrigated Lands Regulatory Program, CDPR, National Water-Quality Assessment Program (NAWQA)).
- More TIEs with fish and algae are needed to identify the cause of toxicity to these indicators.
- Linkage of toxicity, chemistry, and bioassessment monitoring programs would help strengthen the weight of evidence of ecological impacts from toxic pollutants.
- The SWAMP should consider implementing a statewide water column toxicity monitoring program to complement the existing statewide sediment toxicity monitoring program. Such a program would provide an unbiased, consistent assessment of water column toxicity throughout the state.

SECTION 6 DATA QUALITY OBJECTIVES FOR THIS ASSESSMENT

Comparability and data sources for this analysis: This analysis was able to leverage data collected by SWAMP Regional and Statewide monitoring programs, as well as by partner programs. This was possible because SWAMP established a systematic structure to document and evaluate data comparability. This structure gives data users the ability to quickly combine data from multiple sources to perform integrated assessments. The SWAMP Quality Assurance Program instituted standards for data quality and its verification while the SWAMP Data Management Program developed data formats, transfer protocols, and the California Environmental Data Exchange Network.

Statewide survey: Data were pooled from multiple sources to create the data set used in this statewide survey. The quality objective for data usability and comparability among data batches was defined as follows: data batches were usable for this analysis if toxicity test controls met test acceptability criteria as set by the test protocols. Other quality control and metadata information were not considered germane to the goals of this report. Data from multiple test protocols (indicator organisms) measured at multiple laboratories were integrated into a single data set for analysis. All toxicity test data were re-analyzed using the USEPA Test of Significant Toxicity to ensure that every toxicity test result conformed to a consistent statistical analysis (USEPA 2010, Denton et al 2011).

Threshold development: Thresholds for distinguishing between moderate toxicity and high toxicity were developed using data from multiple laboratories for all toxicity endpoints presented in this analysis. For this purpose, the quality objective for data usability and comparability among data batches was defined as follows: data batches were used only if classified as "SWAMP-Compliant." Data classified as "SWAMP-Compliant" has been verified to meet all measurement quality objectives and requirements as defined in the 2002 SWAMP Quality Assurance Management Plan or the 2008 SWAMP Quality Assurance Program Plan.

DATA SOURCES FOR THIS ASSESSMENT

The sources listed are for the data currently available in CEDEN. Many other studies by the State Board, the Regional Boards, and partner programs have been conducted but are not considered here. Many of those data sets will be entered into CEDEN as time and funding allow.

Table 4
Source programs, water toxicity test counts and test dates for toxicity data included in this report.

Region	Program	Test Count	Sample Date Range
1	No Water Toxicity Tests		
2	SWAMP	337	9/18/2001 - 2/16/2006
3	Salinas River Project	206	7/8/2002 - 9/22/2004
	Cooperative Monitoring Program (CMP)	1357	3/22/2005 - 3/31/2010
	SWAMP - Central Coast Ambient Monitoring Program (CCAMP)	449	12/3/2001 - 9/22/2009
	Other SWAMP	12	3/26/2009 - 9/22/2009
4	SWAMP	313	10/29/2001 - 6/11/2009
5	Irrigated Lands Regulatory Program	3178	3/26/2003 - 11/28/2007
	SWAMP San Joaquin River (SJR) Trends	838	1/28/2003 - 3/29/2007
	Other SWAMP	15	1/3/2003 - 3/15/2003
6	No Water Toxicity Tests		
7	SWAMP	150	5/6/2002 - 10/29/2008
8	SWAMP	4	5/24/2005 - 5/25/2005
9	SWAMP	228	3/12/2002 - 5/14/2009
Total Samples		7087	9/18/2001 - 3/31/2010

METHODS USED TO ASSESS HOW THE RESULTS OF TOXICITY MEASUREMENTS COMPARE AMONG WATERS DRAINING URBAN, AGRICULTURAL, AND OTHER LAND COVER AREAS

Land use was quantified around stream, canal and ditch sites at which samples were collected for testing in water column or sediment toxicity tests. Using ArcGIS, polygons were drawn to circumscribe the area within one kilometer of each site that was upstream of the site, in the same catchment, and within 500 meters of a waterway draining to the site. Land use was categorized according to the National Land Cover Database. All "developed" land types in the land cover database were collectively categorized as "urban". "Cultivated crops" and "hay/pasture" were categorized together as "agricultural". All other land types were categorized as "other" for the purposes of this analysis. Percentages of each land use type were quantified in the buffers surrounding the sample collection sites.

Urban land category represents sites with nearby upstream land use of greater than 10% urban areas and less than 25% agricultural areas.

Agricultural land category represents sites with nearby upstream land use of greater than 25% agricultural areas and less than 10% urban areas.

Results of toxicity bioassays were summarized by site as the minimum test organism performance seen for any test species in a given matrix over all toxicity tests performed with samples from a given site.

Table 5
Source programs, sediment toxicity test counts and test dates for toxicity data included in this report.

Region	Program	Test Count	Sample Date Range
1	Statewide Urban Pyrethroid Monitoring	6	11/14/2006 - 11/15/2006
	Stream Pollution Trends (SPoT)	9	10/14/2008 - 10/15/2008
2	Regional Monitoring Program for Water Quality	220	7/27/2004 - 8/29/2007
	Statewide Urban Pyrethroid Monitoring	16	12/28/2006 - 1/3/2007
	Stream Pollution Trends (SPoT)	10	6/17/2008 - 8/13/2008
	Other SWAMP	13	9/18/2001 - 6/19/2002
3	Salinas River Project	62	8/15/2002 - 9/22/2004
	CMP	195	5/14/2006 - 4/22/2009
	SWAMP - CCAMP	67	3/29/2004 - 5/3/2006
	Statewide Urban Pyrethroid Monitoring	10	1/6/2007 - 2/5/2007
	Stream Pollution Trends (SPoT)	11	5/22/2008 - 7/21/2008
4	Statewide Urban Pyrethroid Monitoring	12	1/3/2007 - 1/8/2007
	Stream Pollution Trends (SPoT)	7	5/19/2008 - 5/22/2008
	Other SWAMP	17	1/13/2003 - 6/7/2005
5	Irrigated Lands Regulatory Program	335	5/28/2002 - 9/25/2007
	Statewide Urban Pyrethroid Monitoring	12	11/14/2006 - 11/21/2006
	Stream Pollution Trends (SPoT)	31	4/28/2008 - 8/20/2008
	SWAMP Sediment Tox	61	10/9/2001 - 9/19/2005
	Other SWAMP	23	9/24/2004 - 11/7/2004
6	Statewide Urban Pyrethroid Monitoring	6	10/30/2006
	Stream Pollution Trends (SPoT)	9	9/17/2008 - 9/23/2008
7	Stream Pollution Trends (SPoT)	3	10/28/2008 - 10/29/2008
	Other SWAMP	85	5/6/2002 - 4/22/2008
8	Anaheim Bay / Huntington Harbor	59	8/7/2001 - 8/25/2001
	Lake Elsinore	60	5/1/2003 - 10/3/2003
	Statewide Urban Pyrethroid Monitoring	6	1/7/2007
	Stream Pollution Trends (SPoT)	5	5/20/2008 - 6/4/2008
9	Statewide Urban Pyrethroid Monitoring	12	1/7/2007 - 1/8/2007
	Stream Pollution Trends (SPoT)	7	5/21/2008 - 5/22/2008
	Other SWAMP	104	3/12/2002 - 4/11/2006

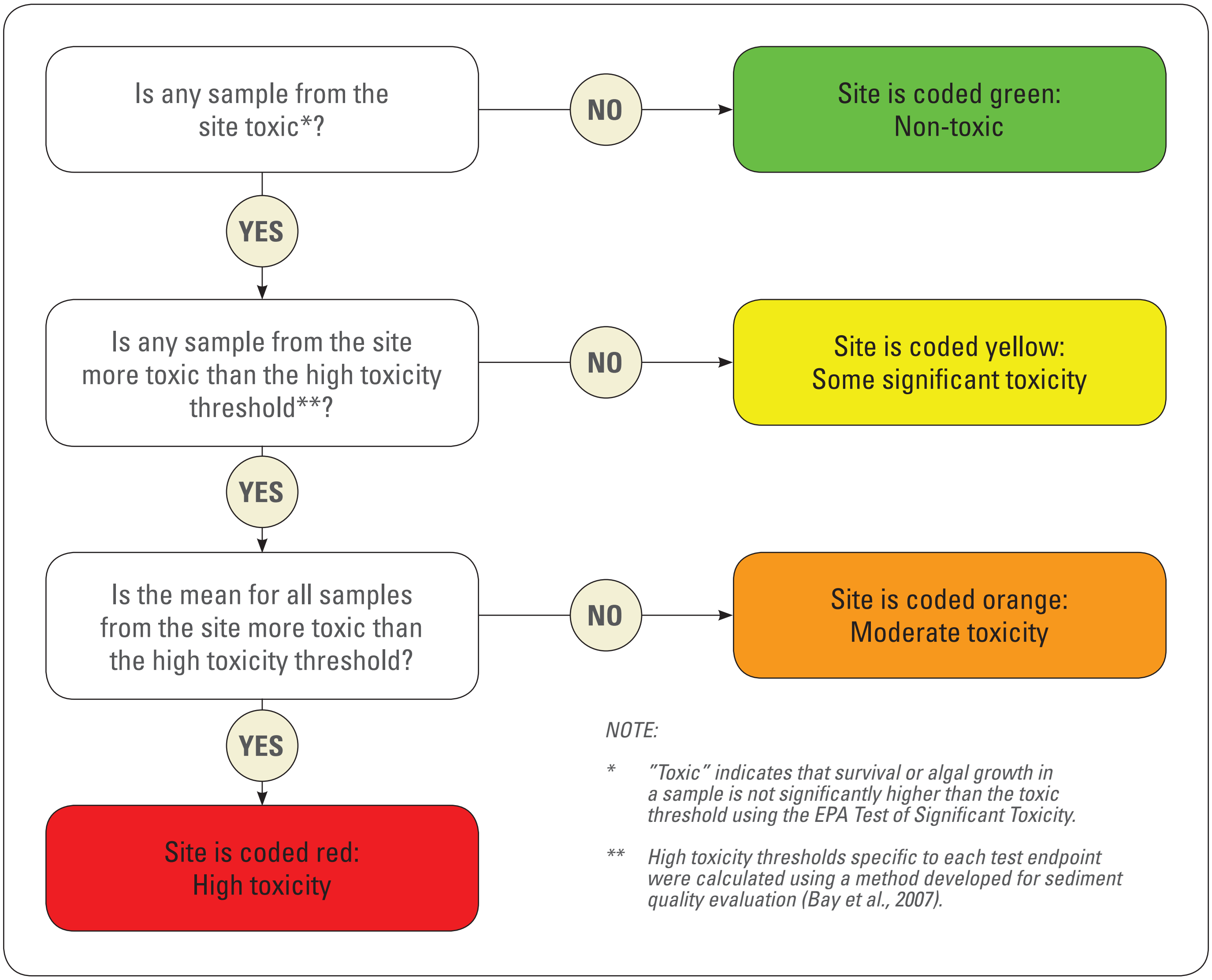


Figure 11. Site categorization process. The process used to characterize the magnitude of toxicity at each site was designed to take into consideration the widely varying number of samples and test endpoints (such as fish or crustacean survival) among sites. If any toxic samples were measured for a site, the site was categorized based on the most sensitive endpoint. This process considers both individual sample results and the mean results for sites with multiple samples. Relative to the impaired waterbody listing process, a site coded "green" would not be listed for toxicity. Sites coded "yellow" to "red" would be listed if the number of toxic samples met the criteria outlined in the State Water Board's Listing and De-listing Policy.

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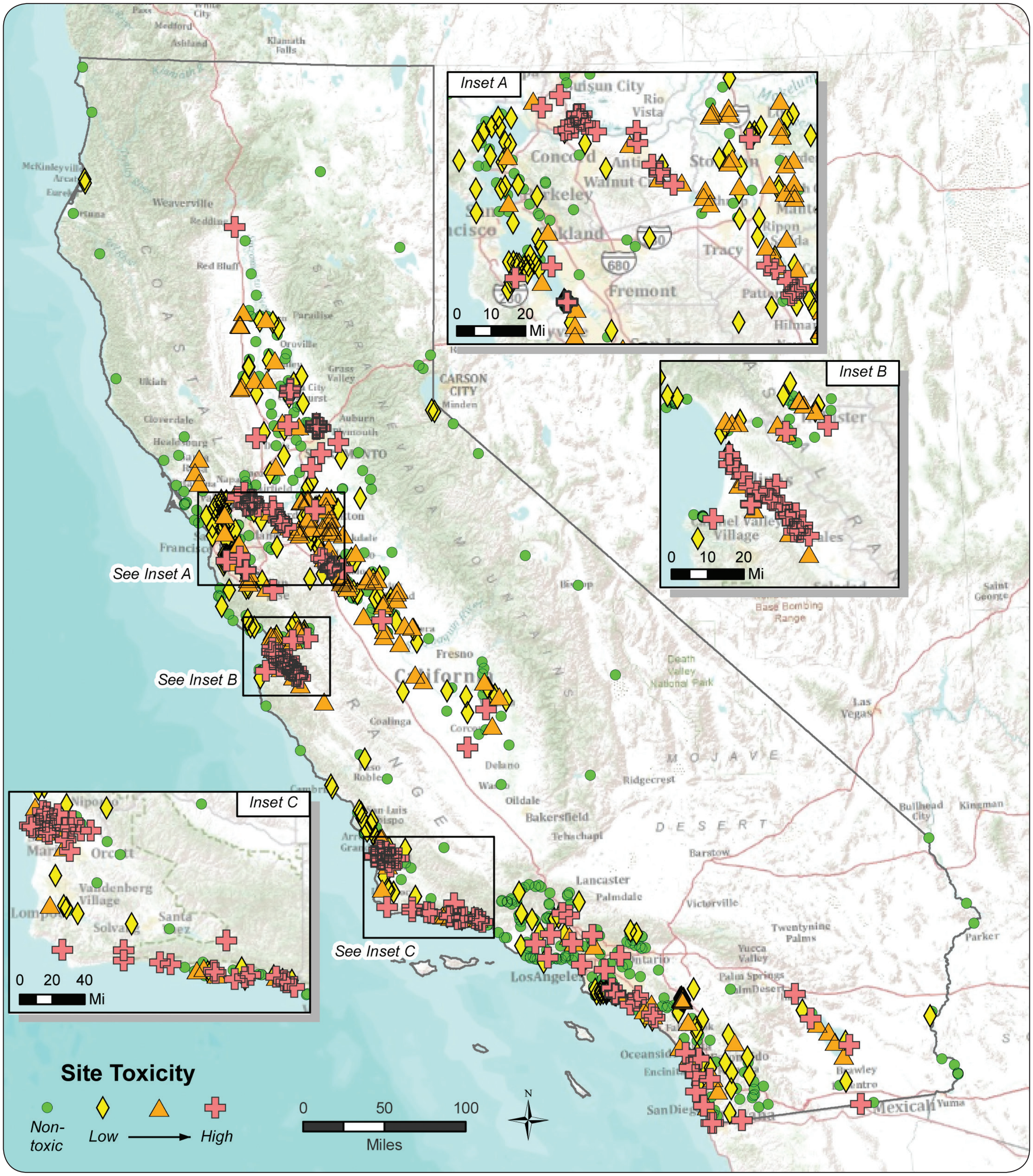


Figure 1. Magnitude of toxicity at all California sites assessed, based on the most sensitive species (test endpoint) in either water or sediment samples at each site.

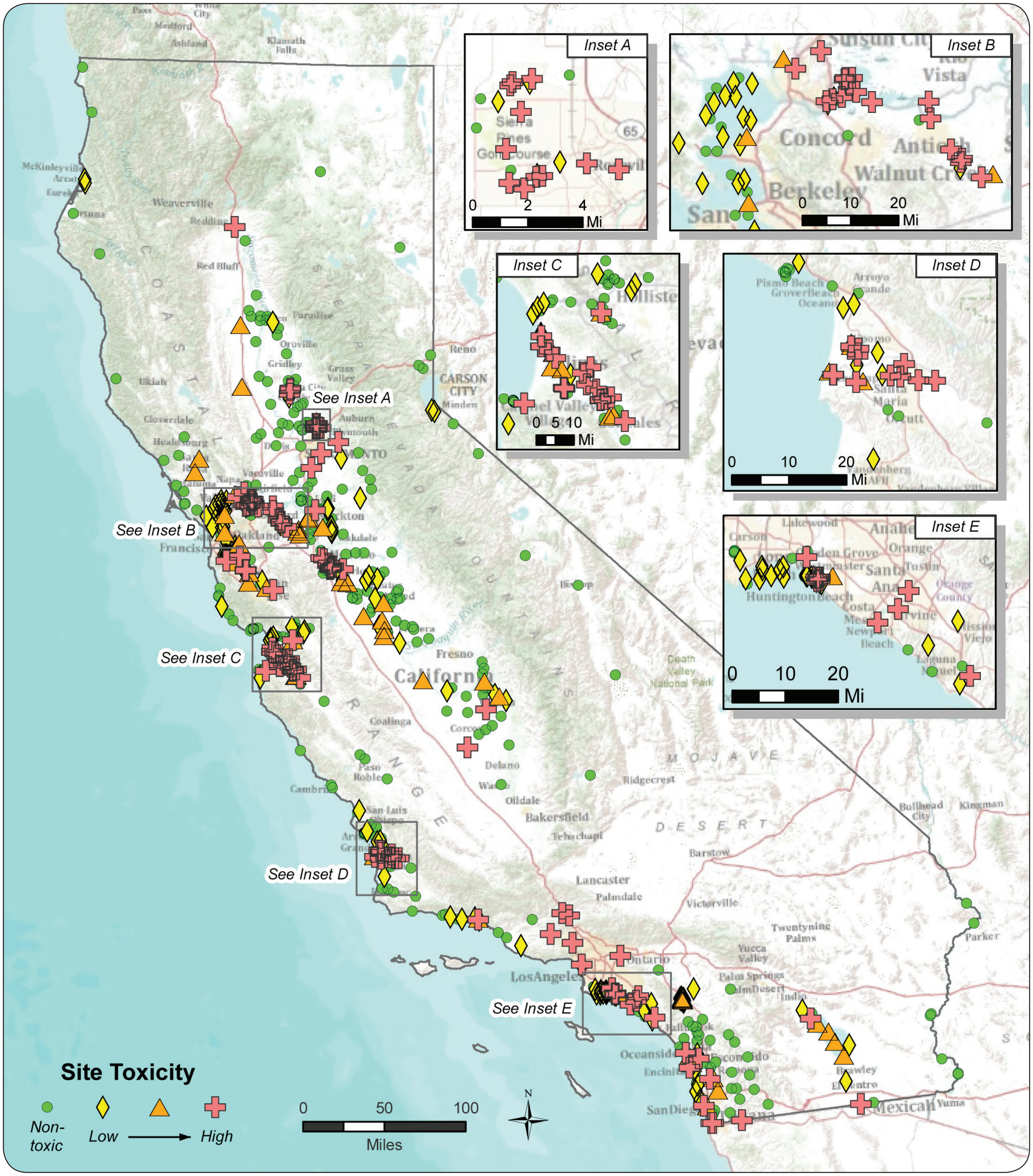


Figure 2. Magnitude of sediment toxicity at all California sites assessed, based on the most sensitive species (test endpoint) in sediment samples collected at each site.

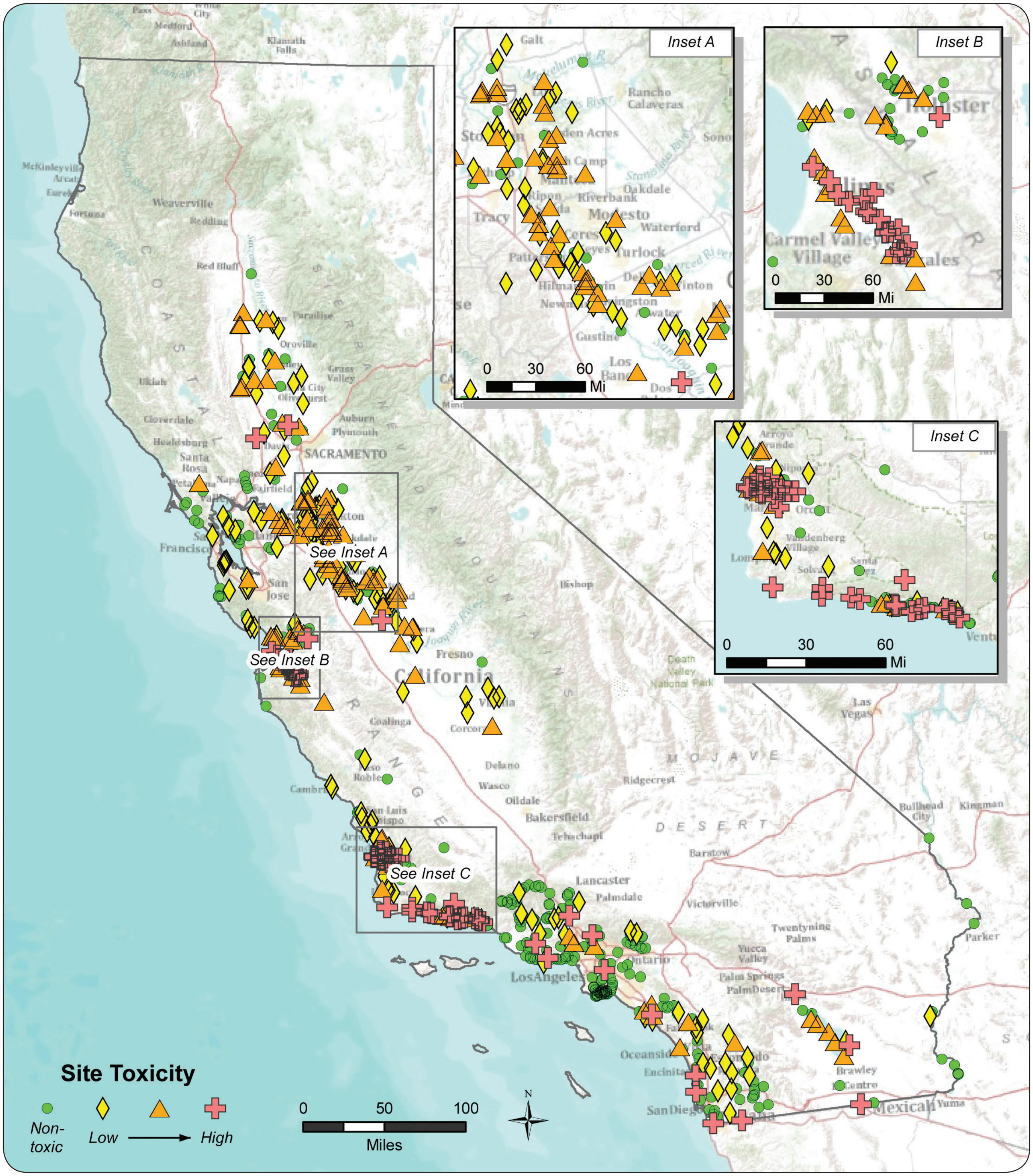


Figure 3. Magnitude of water column toxicity at all California sites assessed, based on the most sensitive species (test endpoint) in water samples collected at each site.

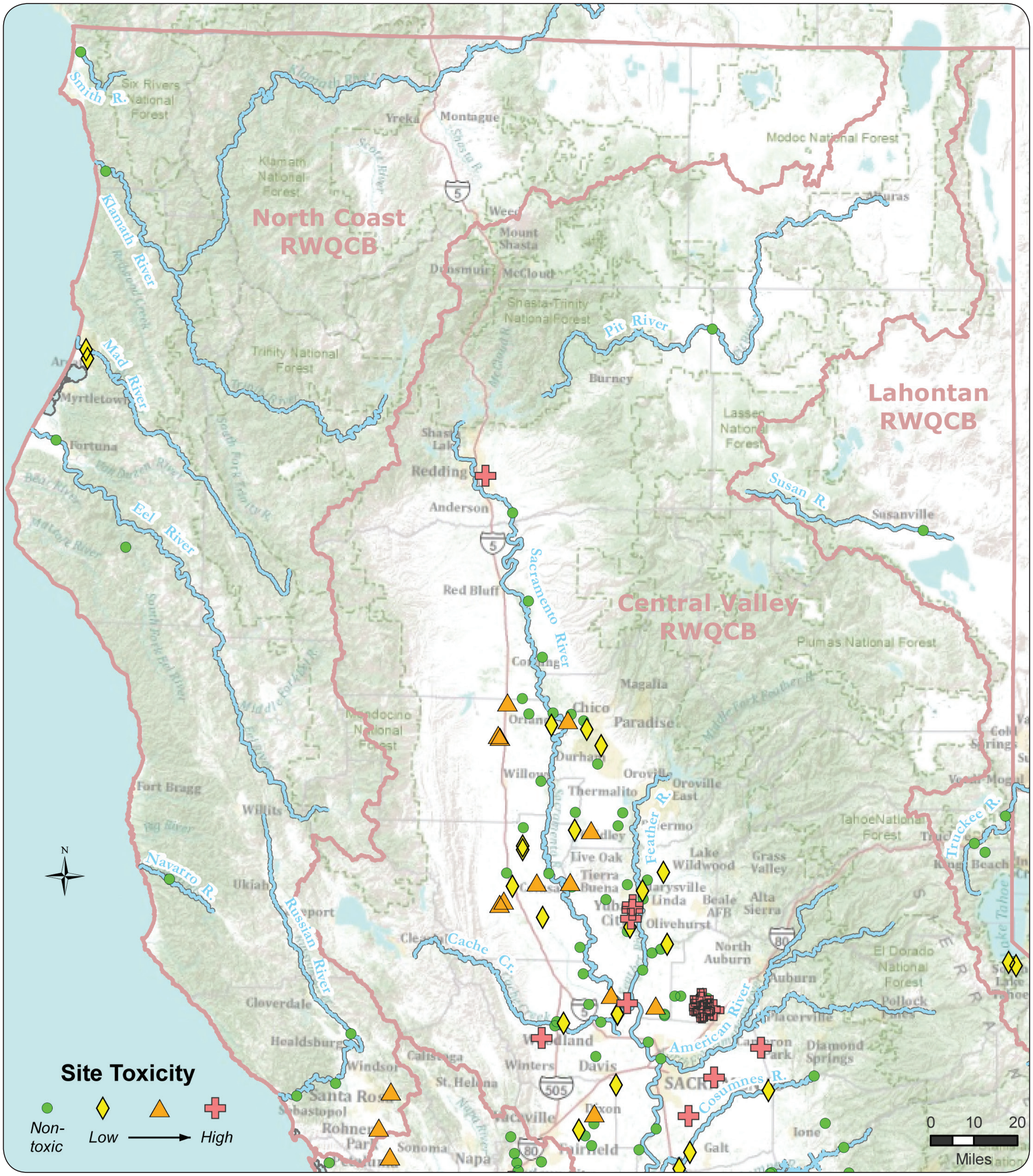


Figure 4. Magnitude of toxicity at sites in Northern California, based on the most sensitive species (test endpoint) in either water or sediment samples at each site.

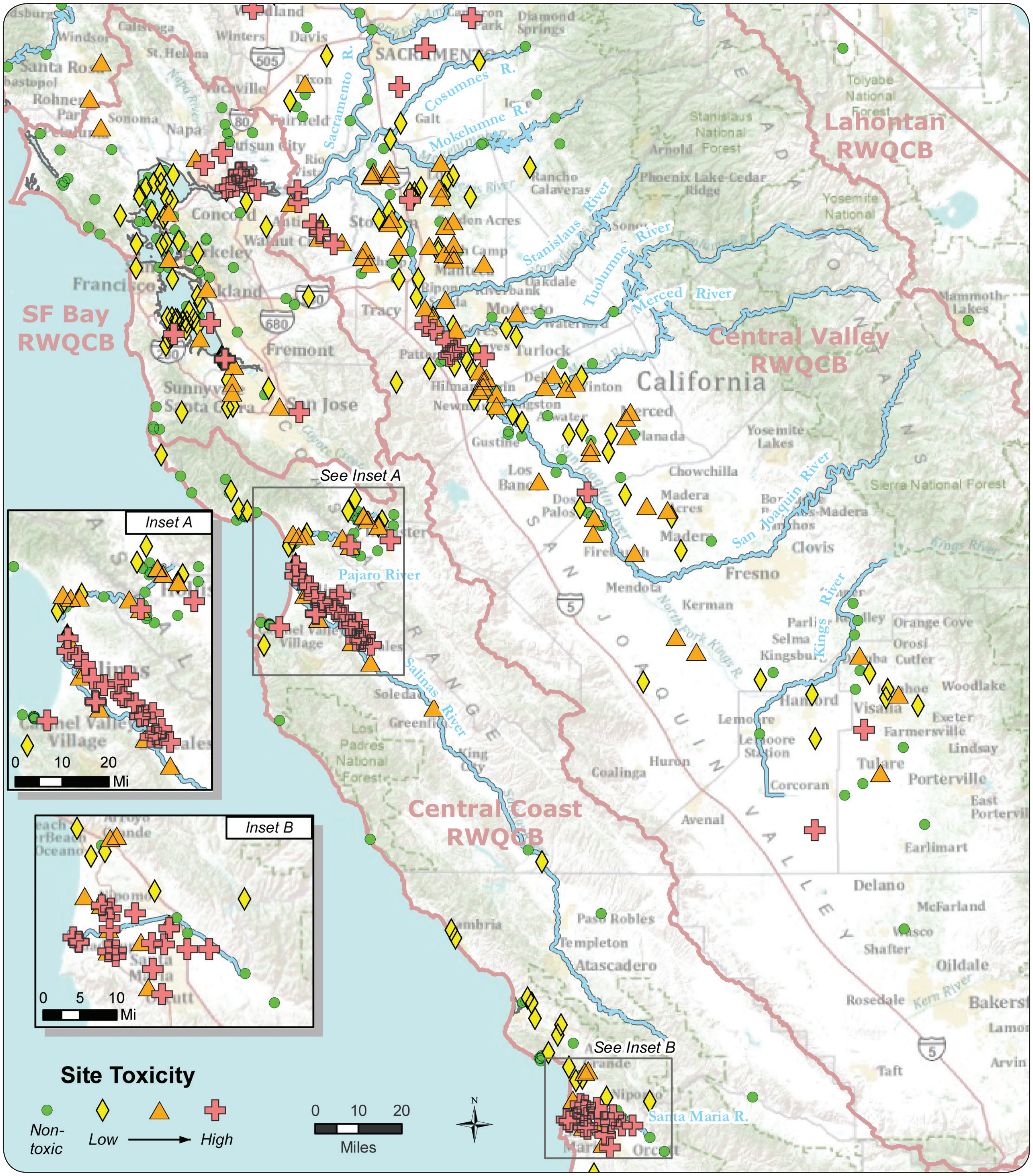
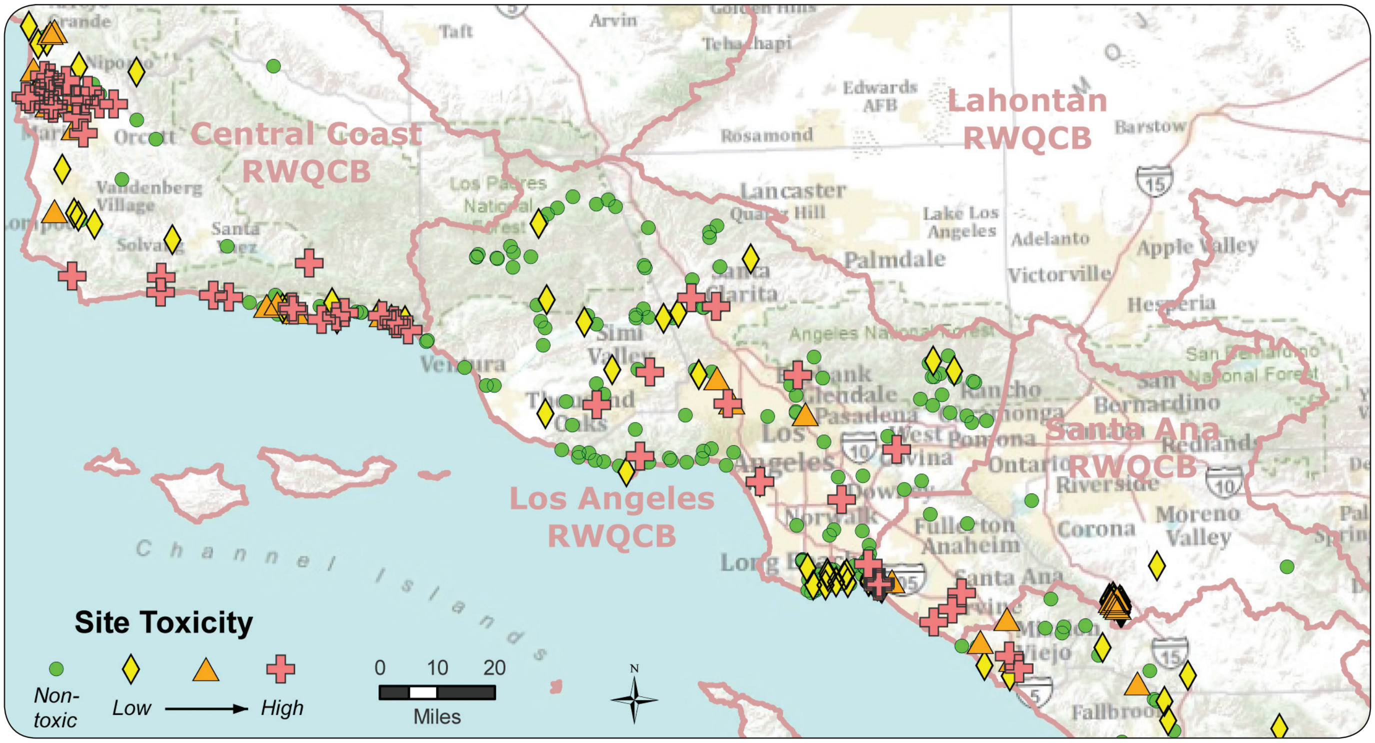


Figure 5. Magnitude of toxicity at sites in Central California, based on the most sensitive species (test endpoint) in either water or sediment samples at each site.



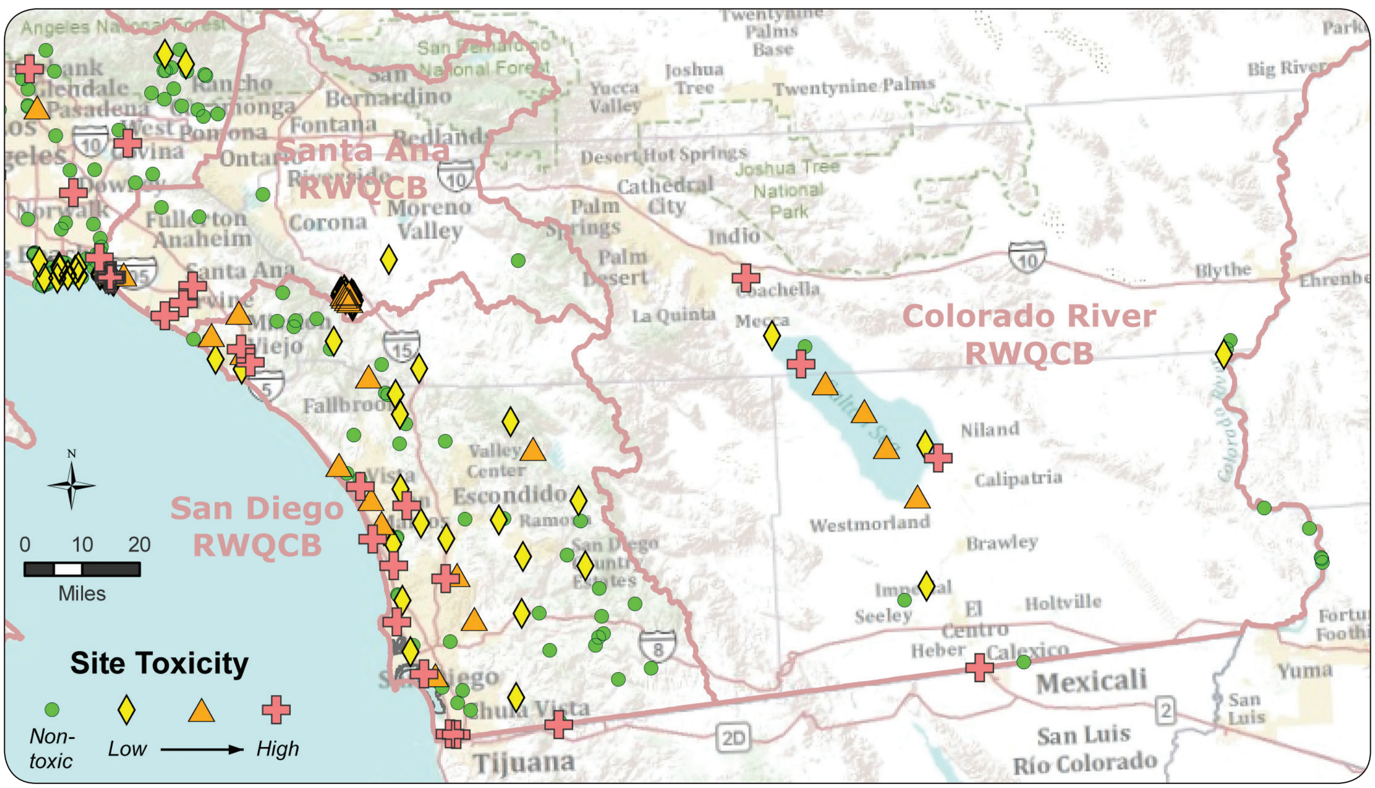


Figure 6. Magnitude of toxicity at sites in Southern California, based on the most sensitive species (test endpoint) in either water or sediment samples at each site.

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