

Monitoring the Aquatic Toxicity of Mosquito Vector Control Spray Pesticides to Freshwater Receiving Waters

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ABSTRACT

Pesticides are applied to state and local waterways in California to control insects such as mosquitoes, which are known to serve as a vector for West Nile Virus infection of humans. The California State Water Resources Control Board adopted a National Pollutant Discharge Elimination System General Permit to address the discharge to waters of the United States of pesticides resulting from adult and larval mosquito control. Because pesticides used in spray activities have the potential to cause toxicity to nontarget organisms in receiving waters, the current study was designed to determine whether toxicity testing provides additional, useful environmental risk information beyond chemical analysis in monitoring spray pesticide applications. Monitoring included a combination of aquatic toxicity tests and chemical analyses of receiving waters from agricultural, urban, and wetland habitats. The active ingredients monitored included the organophosphate pesticides malathion and naled, the pyrethroid pesticides etofenprox, permethrin, and sumithrin, pyrethrins, and piperonyl butoxide (PBO). Approximately 15% of the postapplication water samples were significantly toxic. Toxicity of half of these samples was attributed to the naled breakdown product dichlorvos. Toxicity of 2 other water samples likely occurred when PBO synergized the effects of pyrethroid pesticides that were likely present in the receiving system. Four of 43 postapplication sediment samples were significantly more toxic than their corresponding pre-application samples, but none of the observed toxicity was attributed to the application events. These results indicate that many of the spray pesticides used for adult mosquito control do not pose significant acute toxicity risk to invertebrates in receiving systems. In the case of naled in water, analysis of only the active ingredient underestimated potential impacts to the receiving system, because toxicity was attributed to the breakdown product, dichlorvos. Toxicity testing can provide useful risk information about unidentified, unmeasured toxicants or mixtures of toxicants. In this case, toxicity testing provided information that could lead to the inclusion of dichlorvos monitoring as a permit requirement. *Integr Environ Assess Manag* 2014;10:449–455. © 2014 SETAC

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INTRODUCTION

West Nile Virus continues to pose a risk to people in California. In 2013, as of August, 73% of California counties had virus activity, and 28% of counties had confirmed human cases (www.westnile.ca.gov). Mosquitoes are the vectors for human infection by the virus. Habitat reduction and larval control are the initial means of managing mosquito populations, but ultra-low volume applications of pesticides are currently the most effective method to reduce adult populations and related virus activity (Bonds 2012; Mount 1998). Because spray activities often occur over or in close proximity to surface water where mosquitoes breed, the applied pesticides have the potential to cause toxicity to nontarget organisms in receiving systems.

A number of ecological risk assessments have been conducted to determine the potential impacts of spray activities

(Davis et al. 2007; Schleier and Peterson 2010a; 2013; Schleier et al. 2012). Some of these studies have assessed established models for pesticide loading that were designed to predict spray effects (Davis et al. 2007; Schleier et al. 2012), and others have examined in situ concentrations of spray pesticide deposition (Schleier and Peterson 2010b; Schleier and Peterson 2013). Schleier and Peterson (2013) state that only deterministic ecological risk assessments have been conducted and that most of the models are inappropriate for estimating environmental concentrations. Although these authors go on to accurately estimate environmental concentrations of active ingredients, and place these concentrations in the context of organism sensitivity, few studies have directly assessed the impacts to aquatic organisms immediately after actual spray events (Milam and others 2000; Weston et al. 2006; Schleier and Peterson 2010a).

In 2011, the California State Water Resources Control Board adopted a National Pollutant Discharge Elimination System (NPDES) General Permit to cover the discharge of pesticides to waters of the United States resulting from adult and larval mosquito control (NPDES No. CAG990004, WQO 2011-0002-DWQ, as amended). The permit requires the analysis of pre-application and postapplication water samples for active

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ingredients of the applied vector control products. As specified in the NPDES permit, the California State Water Resources Control Board, in cooperation with the Mosquito Vector Control Association of California (MVCAC), designed the current study to determine whether toxicity testing provided additional risk information in monitoring spray pesticide applications beyond that provided by the standard chemical analysis for residual active ingredient in the receiving water sample. In the context of the Clean Water Act, this study attempted to determine whether the pesticide applications were causing toxicity to nontarget aquatic organisms. This paper summarizes the results of chemistry and toxicity monitoring in water column and sediment samples conducted during the summers of 2011 and 2012.

METHODS

Mosquito control spray applications were conducted by individual vector control districts throughout the state. For this study, MVCAC coordinated the monitoring of spray applications conducted by member districts with University of California Davis researchers to develop sampling strategies, visit potential sampling areas, and select appropriate sampling sites before application events by assessing environmental setting and hydrology.

Among the pesticide active ingredients allowed for vector control under California's NPDES General Permit, a subset were prioritized for the toxicity study based on the United States Environmental Protection Agency (USEPA) Office of Pesticide Programs (OPP) aquatic life benchmarks. The aquatic life benchmarks (for freshwater species) are based on toxicity values reviewed by USEPA and used in the Agency's most recent risk assessments developed as part of the decision-making process for pesticide registration. The OPP considers studies required under the Federal Insecticide, Fungicide, and Rodenticide Act, as specified at 40 Code of Federal Regulations part 158, as well as a wide range of environmental laboratory and field studies to assess environmental risk. Each aquatic life benchmark is based on sensitive, scientifically acceptable toxicity endpoints for a given taxon from among all scientifically acceptable toxicity data available to USEPA. In the current study, the acreage treated and the use patterns of the active ingredients were also considered in prioritization of pesticides for monitoring. The active ingredients monitored in this study were the organophosphate pesticides malathion and naled, the pyrethroid pesticides etofenprox, permethrin, and sumithrin, pyrethrins, and the synergist piperonyl butoxide (PBO). These active ingredients were monitored during 15 spray events conducted from July to October 2011 and from May to September 2012.

Although the California NPDES General Permit only requires chemical analysis of water samples, combinations of water and sediment toxicity test methods were used in the current study because of differences in the physicochemical properties and short-term environmental fates of the different pesticides used in the applications. In addition to water testing, sediment toxicity and chemistry were monitored during applications of pyrethroids and pyrethrins, because these pesticides have a tendency to partition to sediments (Hladik and Kuivila 2012). All samples were collected before and after spray events in 3 environmental settings (agricultural, urban, and wetland), and up to 6 spray events were monitored for each active ingredient in each setting. The different environmental settings were targeted to determine whether local drain inputs affected pre-application and postapplication results. Because

spray events took place in the evenings when mosquitoes are active, the target collection period for pre-application water and sediment samples was from 4:00 to 8:00 PM on the day of the spray event. The target sampling period for the first postapplication water samples was from 6:00 to 10:00 AM the following morning, approximately 12 hours after the spray event. The target period for the final postapplication water samples was from 4:00 to 8:00 PM of the day after the spray event. Postapplication sediment samples were collected 4 to 7 days after the application, when it was presumed the active ingredients had partitioned to suspended sediments and settled. Water and sediment samples were collected according to the Standard Operating Procedure for Conducting Field Measurements and Field Collections of Water and Bed Sediment Samples in the California Surface Water Ambient Monitoring Program (SWAMP 2008).

Water toxicity for organophosphate pesticide applications was assessed using the 7-day survival and reproduction protocol for *Ceriodaphnia dubia* (USEPA 2002b). Water toxicity for pyrethrins and pyrethroid applications was assessed with the 96-hour survival protocol for *Hyaella azteca* (USEPA 2002a). Organisms were chosen for their sensitivity to the different chemical classes (Weston and Lydy 2010a; 2010b). Sediment toxicity for pyrethrin and pyrethroid applications was assessed using the 10-day survival and growth protocol for *H. azteca* (USEPA 2000). Although both amphipod growth and survival were measured, only the survival endpoint was considered in the final evaluation of the sediment toxicity data, because the growth data did not provide any meaningful response data beyond that provided by survival (data not shown). Water samples from both monitoring years and sediment samples from the first year were analyzed for a suite of pesticides that included the active ingredient applied during each event. For example, analysis of the active ingredient naled would also yield results for the naled breakdown product dichlorvos, and analysis of the active ingredient permethrin would yield results for other pyrethroids, such as bifenthrin. Not all year 2 sediment samples and 12-hour water samples from later events were analyzed for chemistry because of funding constraints, and only the toxic sediment samples from year 2 were frozen and archived for later analysis. Organophosphate pesticides in water were analyzed using USEPA Methods 8141 or 641 (USEPA 1993, 1994). Pyrethroids were analyzed using a gas chromatograph coupled to a tandem mass spectrometer (EPA 8270M; USEPA 1994). Pyrethrins were analyzed by a liquid chromatograph coupled to a tandem mass spectrometer (EPA 8321b M; USEPA 1994). The modifications for both methods include the use of a triple quadrupole mass spectrometer and the fact that pyrethroids and pyrethrins are not specifically listed in the method.

Samples analyzed for pyrethrins and pyrethroids were also analyzed for PBO (EPA 8270M; USEPA 1994). Some of the detections of sumithrin, permethrin, pyrethrins, and etofenprox were below the reporting limit for the laboratory, and therefore, estimated concentrations were evaluated.

Concentrations of detected active ingredients in the water column were evaluated relative to the permit trigger values (Table 1), which were derived by determining the pesticide's lowest median lethal concentration (LC50) available in the scientific literature and dividing it by a safety factor of 10 (CVRWQCB 2011). Analytical results were also compared with *C. dubia* and *H. azteca* LC50s when available. If organism-specific LC50s were not available, then analytical results were evaluated using LC50s from closely related

Table 1. Occurrence of toxicity, detection of active ingredients in water samples collected after pesticide applications, permit trigger values, and the number of samples exceeding the permit trigger

Active ingredient	Nr of sites tested	Nr of samples with significant toxicity		Nr of active ingredient detections		Permit trigger (ng/L)	Nr of samples exceeding permit trigger
		12-hour postapplication	24-hour postapplication	12-hour postapplication	24-hour postapplication		
Etofenprox	1	0	1	0	1	1.9	1
Malathion	2	0	0	1	2	100	2
Naled	9	8	3	2	0	14	2
Permethrin	12	0	2	2	4	30	0
Pyrethrin	12	1	1	4	2	140	0
Sumithrin	18	0	0	0	1	2.5	1

organisms. Statistically significant toxicity of all test results were determined using the Test of Significant Toxicity, following USEPA procedures (USEPA 2010).

RESULTS AND DISCUSSION

Quality assurance

All toxicity tests met test acceptability criteria, and all water quality parameters (pH, dissolved oxygen, temperature, conductivity, hardness, alkalinity; data not shown) were within acceptable ranges for the test organisms (USEPA 2000; 2002a; 2002b). All but 1 of the analytical chemistry laboratory control standards, surrogate spikes, matrix spikes, and matrix spike duplicates from samples analyzed during the first year of the project were within the acceptable range of 50% to 150%. During the second year of the project there were a number of surrogate recoveries for the water chemistry data that were not within acceptable limits. Based on other quality assurance samples, the data were deemed acceptable for analysis. See Supplemental Data for a complete list of active ingredient detections, detections of other pertinent chemicals, and a summary of quality assurance deviations.

Water toxicity and chemistry

Only 1 of the pre-application water samples was significantly toxic. This sample was moderately toxic to *H. azteca* before an application of sumithrin. No postapplication sumithrin samples were significantly toxic, but 16 of the 108 postapplication water samples were significantly toxic to either daphnids or amphipods after applications of etofenprox, naled, permethrin, or pyrethrin (Table 1). None of the applied active ingredients were detected in pre-application samples, but they were detected to varying degrees in postapplication samples, with roughly equal detection frequencies at both 12 and 24 hours after application. In some cases, a greater number of detections occurred at 24 hours than at 12 hours. This was likely caused by water movement in the various study areas, and the location of the sampling site.

Most postapplication water toxicity occurred after naled spray events (Table 1). Eight of nine 12-hour postapplication samples were toxic to *C. dubia*, and 3 of 9 24-hour postapplication samples were toxic. Naled was detected in only 2 of the samples, and at concentrations well below the LC50 for *Daphnia magna* of 360 ng/L (Frear and Boyd 1967),

but exceeding the permit trigger value. Naled rapidly breaks down to dichlorvos in water, and dichlorvos was detected in 13 of 18 postapplication samples. Nine of the dichlorvos detections were greater than the *C. dubia* LC50 of 130 ng/L (Ankley et al. 1991), and all samples were significantly toxic (Figure 1).

Dichlorvos is also a breakdown product of trichlorfon. Trichlorfon was not detected in any pre-application samples, but was detected in postapplication samples at 2 of 9 naled application sites. Both trichlorfon and dichlorvos are registered pesticides, and it is possible that these chemicals were applied separately from naled, but California pesticide use data shows that these pesticides are used mostly in urban areas.

No permit trigger value has been identified for dichlorvos, but a value could be calculated using the *C. dubia* LC50 and an appropriate safety factor, as the other permit trigger values were calculated. As an example, if the *C. dubia* LC50 were divided by 10, the resulting value of 13 ng/L would have been exceeded in every sample in which dichlorvos was detected. Dichlorvos likely caused the water toxicity associated with the naled spray applications, because toxicity testing results showed a high magnitude of toxicity in the absence of any other measured chemicals. This is an example in which the implementation of receiving water toxicity testing provided additional useful information to the spray event monitoring, beyond that provided through analysis of the spray pesticide active ingredient alone.

Complete mortality was observed in 12- and 24-hour postapplication pyrethrin samples analyzed at a single site. Pyrethrins were detected at 4 sites, but not at concentrations that would significantly contribute to toxicity, nor at concentrations that exceeded the permit trigger. Piperonyl butoxide was applied in combination with pyrethrins, and the concentrations of PBO were highest in the water samples that were toxic. The PBO added as part of the pyrethrin applications may have synergized concentrations of pyrethroids that were already present in the receiving system, as evidenced by the presence of pyrethroids in the corresponding sediment samples. Elevated concentrations of pyrethroids that are not used for mosquito control are often measured in urban sediments (Holmes et al. 2008). Although PBO does not generally cause toxicity directly, its use as a synergist can indirectly contribute to toxicity in systems in which pyrethroids are present (Amweg and others 2006; Weston and others 2006), and the concentrations of PBO detected in these samples well exceeded the concentration expected to synergize

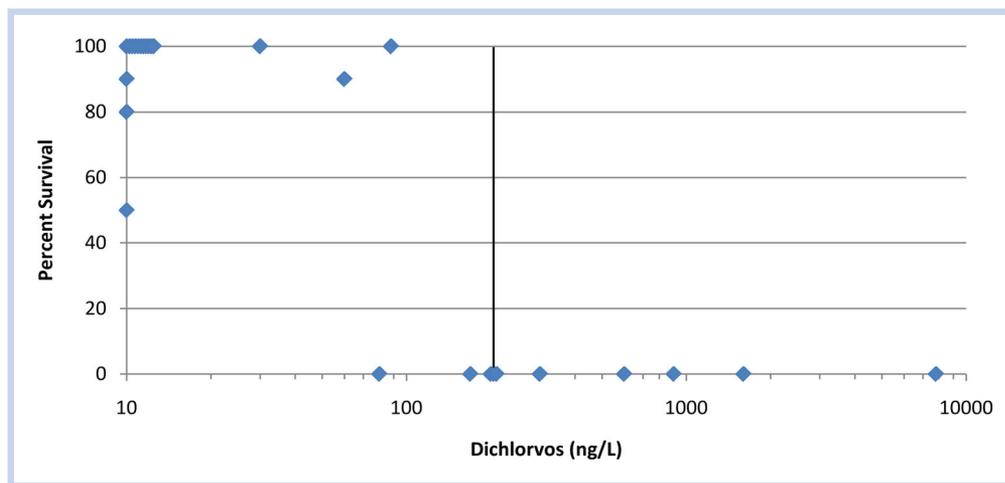


Figure 1. Concentrations of dichlorvos measured in water samples collected after applications of naled plotted against survival. Vertical line indicates dichlorvos LC50 for *C. dubia* of 130 ng/L (Ankley et al. 1991).

pyrethroids. The presence of pyrethroids in the water column samples could not be confirmed, because only the active ingredients applied (pyrethrins) were measured. Piperonyl butoxide was detected in 65 of 77 postapplication samples, as well as a number of pre-application samples. Individually, PBO did not exceed the permit trigger value of 49 $\mu\text{g/L}$, but an additional trigger value of 0.014 $\mu\text{g/L}$ is evaluated when PBO is applied with pyrethrins, because small concentrations of PBO greatly enhance the toxicity of pyrethroids, and the formulated products (i.e., pyrethrin and PBO) are generally more toxic than the active ingredient alone (USEPA 2006). This trigger was exceeded in every sample that was collected after pyrethrin applications. Concentrations of PBO in water at 24 hours postapplication were approximately 60% of the 12-hour postapplication concentrations.

Two of the 24-hour postapplication permethrin samples were significantly toxic. Permethrin was detected in 1 of these samples at concentrations well below the LC50, but the detection in the other sample was approximately equal to the *H. azteca* LC50 of 21 ng/L, (Anderson et al. 2006). None of the samples exceeded the permit trigger value. Permethrin was also detected in 4 additional postapplication water samples. Three of these samples contained permethrin at a concentration that was slightly higher than the *H. azteca* LC50, but none of these samples was significantly toxic, perhaps because of reduced bioavailability. Bifenthrin was detected in 1 of the 2 toxic water samples at a concentration high enough to cause the observed toxicity.

None of the postapplication sumithrin samples were toxic, and sumithrin was detected in only 1 postapplication water sample at a concentration exceeding the permit trigger value. Malathion was applied during only 1 event, and 2 sites were monitored. No water toxicity was observed in any of the samples. Three of the four postapplication samples contained detectable concentrations. These concentrations were well below the published LC50 for the *C. dubia* of 2120 ng/L (Ankley et al. 1991), but 2 concentrations exceeded the USEPA ambient water quality chronic criterion and permit limit (100 ng/L). Etofenprox was also applied during a single event, and only 1 site was monitored. No toxicity was observed in the pre-application sample, but significant toxicity was observed in the 24-hour water sample. This sample also contained a detectable concentration of the active ingredient (20 ng/L) at a concentration greater than the permit trigger value (1.9 ng/L, Table 1). No etofenprox LC50 has been published for *H. azteca*, but the detected concentration was well below the USEPA OPP benchmark of 400 ng/L.

Sediment toxicity and chemistry

Of the 24 sediment samples that were collected during permethrin applications, 7 were significantly toxic for survival (4 pre-application samples and 3 postapplication samples, Table 2). Six samples at 2 sites alternated between toxic and not toxic during 2 consecutive application events. For example, the pre-application sample was toxic at 1 site, the first postapplication sample was not toxic, but a second postapplication

Table 2. Occurrence of toxicity and detection of active ingredients in sediment samples collected during applications of pyrethroids and pyrethrins^a

Active ingredient	Nr of sites tested	Nr of samples with significantly greater postapplication toxicity	Nr of active ingredient detections	
			Pre-application	Post-application
Etofenprox	1	0	NA	NA
Permethrin	12	3	1*	1*
Pyrethrin	12	1	0	0
Sumithrin	18	0	2	5

^aOnly 1 pre-application and 1 postapplication sediment sample were analyzed for permethrin.

sample a week later was toxic. At another site that underwent an application of permethrin, both pre- and postapplication samples were toxic, and the postapplication sample had a significantly greater toxicity than the pre-application sample ($p < 0.05$). The observed toxicity in these samples was likely caused by concentrations of bifenthrin (97 ng/g), cyfluthrin (13.9 ng/g), and lambda cyhalothrin (8.19 ng/g), which contributed a total of 10 to 11 toxic units of pyrethroids. A toxic unit is equal to the sediment LC50 concentration of each individual chemical, and these units can be summed because pyrethroids toxicity is additive (Trimble and others 2009). The concentrations of permethrin in the pre- and postapplication samples did not differ and only accounted for approximately one-third of a toxic unit, indicating that the spray event did not contribute additional permethrin to the sample. Thus, toxicity observed in this spray event was not likely caused by the applied permethrin but was associated with pyrethroids from other sources, including the possibility of previous permethrin applications.

During the pyrethrin applications, 1 pre-application sample was significantly toxic, and no toxicity was observed in the postapplication sample. A single instance of significant postapplication toxicity was also noted. Pyrethrins were not detected in either case, but sediments from the first sample contained a toxic concentration of bifenthrin (42.2 ng/g). Sediment toxicity was observed in both pre- and postapplication samples from 2 sumithrin sites. Both postapplication samples had significantly greater toxicity than the pre-application sample. The sediments from this event were not analyzed for pyrethroids, but previous events at this site were analyzed for sediment concentrations of sumithrin, and this active ingredient was detected. The sites monitored during these events were exposed to multiple applications of sumithrin over 2 months, and possibly some of these detections were a product of accumulation after several applications.

Of the 8 pre-application sediment samples that were significantly toxic, 5 of the corresponding postapplication samples were not significantly toxic (Table 2). Two of the remaining corresponding postapplication samples were significantly toxic but did not produce statistically different results from the pre-application samples. Overall, 4 of the 43 total postapplication sediment samples were significantly more toxic than their corresponding pre-application sample. The toxicity of these samples was apparently not influenced by the spray application pesticides.

Results summarized by environmental setting

Up to 6 sites were to be sampled in each environmental setting for each active ingredient applied, but because of the nature of the vector control activities during the study period, full sets of samples were collected in only 7 of the combinations (Table 3). Urban settings were the most represented and had the most occurrences of toxicity, with 9 of 24 sites having at least 1 toxic postapplication sample. Sites that were monitored during naled applications accounted for two-thirds of the urban toxicity. Although urban settings have the most toxicity, the environmental setting did not appear to influence the toxicity results beyond the effects of naled applications, but additional monitoring of some environmental settings and active ingredients would provide more resolution.

Effects of multiple pesticide applications

During several events, multiple pesticide applications occurred within the event, or sites were treated multiple times during more

Table 3. Number of sites sampled for each combination of active ingredient and environmental setting^a

Active ingredient	Environmental setting		
	Agriculture	Urban	Wetland
Etofenprox	0	1 (1)	0
Malathion	1	0	1
Naled	1 (1)	6 (6)	2 (1)
Permethrin	6 (1)	5 (1)	1
Pyrethrin	0	6 (1)	6
Sumithrin	6	6	6

^aNumbers in parentheses indicate the number of events with a toxic sample.

than 1 event. For instance, some applications occurred for several nights, and the postapplication samples were not collected until after the final application, or sites were visited multiple times over several months, and postapplication samples were collected for each event. In some cases, different active ingredients were applied to minimize the possibility of mosquitoes developing tolerance. The first of these events consisted of 2 consecutive nights of pyrethrin applications. This was the only pyrethrin event in which the active ingredient was detected, or where toxicity occurred. The toxicity was assumed to not have been caused by pyrethrins, but more likely by PBO synergizing pyrethroids that were likely present in the water body, which was located in an urban area. Two other events consisted of 5 consecutive nights of pyrethrin and permethrin applications, respectively. Monitoring of these events did not result in any detection of water toxicity or active ingredients. Detected concentrations of PBO during these events were generally less than concentrations detected in other pyrethrin and permethrin application events. Sumithrin and naled were also applied to 2 sites on successive nights as part of a single spray event that included additional postapplication sampling after the application of sumithrin. No water toxicity was observed in these samples until the application of naled. Toxic concentrations of the naled breakdown product dichlorvos were measured in the toxic samples.

Sumithrin was applied to 2 sites during 3 events, and an additional 2 sites during 2 events. The first 2 events occurred approximately 2 weeks apart, and the third event occurred approximately 7 weeks later. No significant water or sediment toxicity was observed during these events. Sumithrin was detected in 1 postapplication water sample, and in several postapplication sediment samples. Sediment concentrations of sumithrin tended to be higher after successive applications.

Two sites underwent permethrin applications 1 week apart, and significant postapplication toxicity was observed in a 24-hour postapplication water sample after the second application. This sample contained permethrin at a concentration slightly less than the LC50. Two postapplication sediment samples were also significantly toxic, 1 after the first application and 1 after the second application. Permethrin concentrations were similar in pre- and postapplication samples and were below the toxicity threshold.

Two sites underwent applications of pyrethrin and permethrin approximately 1 month apart. No toxicity or active ingredients were observed in the water samples, and no significant toxicity was found in the sediment samples. Another 2 sites underwent applications of sumithrin and etofenprox

approximately 10 weeks apart. No toxicity or active ingredient was measured after the sumithrin application, but the 24-hour postapplication water sample was toxic after the etofenprox application. Etofenprox was detected in this sample, but at a concentration that was less than what would be considered toxic. Neither postapplication sediment sample had significantly greater toxicity than the pre-application sample.

Weston et al. (2006) suggested that multiple applications of pyrethrin and PBO in an urban environment could increase synergism of nonapplication pyrethroids in the receiving systems. This appears to have occurred after the 2-day urban application of pyrethrin described above but did not occur during similar applications of pyrethrin in wetland areas. Monitoring evidence suggests that many pyrethroids are more likely to be detected in water and sediment from urban receiving systems (Weston and Lydy 2010b), but vector applications are unlikely to be the source because of the types of pyrethroids detected (Weston et al. 2005). Overall, multiple applications within an event or among different events at the same site did not appear to increase the chances of toxicity or exceedance of permit trigger values in this study.

CONCLUSIONS

Of the 16 toxic postapplication water samples in this study, the toxicity of 9 samples was attributed to the naled breakdown product dichlorvos. Permethrin could have contributed to 1 of 2 toxic postapplication samples, but the toxicity of the other sample was likely caused by bifenthrin, which was not applied as part of vector spray pesticide activities. Two toxic water samples observed after the application of pyrethrin could have been caused by PBO synergizing ambient concentrations of pyrethroids that were previously present in the urban setting, but the concentrations of pyrethrins were well below toxic levels. Four water samples had toxicity that could not be attributed to the measured chemicals. Four of the 43 postapplication sediment samples were significantly more toxic than their corresponding pre-application sample. Two of these samples were collected as part of repeated applications and demonstrated a return to background toxicity. The measured active ingredients in the other sediment samples were not significantly higher in the postapplication samples, nor were they high enough to cause the observed toxicity. Although the cause of toxicity in some of these samples could not be directly attributed to the spray events, there is always the possibility that the application of pesticides contributed to the overall toxic load in the receiving systems, and therefore contributed to toxicity. This could not be determined from the results of the current study because of the limited number of chemicals analyzed.

This study was designed to determine whether toxicity testing provided additional useful information regarding the potential for impacts in the receiving systems beyond that provided by the analysis of the active ingredient alone. In the case of naled, analysis of the active ingredient alone underestimated the potential impacts to receiving waters. Other than malathion, naled is 1 of the only organophosphate pesticides available for adult mosquito control. Vector officials say naled is a necessary component of the suite of adulticides available, because its use helps prevent development of tolerance to other active ingredients such as pyrethrins and pyrethroids. Assuming its use will continue, risk to aquatic receiving systems may be mitigated by applying best management practices to naled applications. Based on discussions with vector officials, some management practices are currently being implemented,

including analysis of the appropriate application rate to maintain efficacy, and calibration of equipment to evaluate the optimum droplet size to promote longer aerosol resident time over spray sites. This could reduce the amount of naled used in applications, and the subsequent occurrence of dichlorvos in receiving systems, although it could also lead to the development of mosquito resistance to the product. The effectiveness of these practices should be further evaluated through chemical analysis and toxicity testing.

Toxicity testing also has the potential to capture the effects of spray applications to systems that may already be moderately contaminated but are not toxic until the spray application occurs. This is illustrated by the application of products containing PBO. If applied PBO is synergizing pyrethroids already present in the receiving system, then toxicity unrelated to the spray pesticide active ingredient could occur. This could be accounted for through toxicity testing and measurements of pyrethroid pesticides in the receiving systems before and after spray events. As spray products containing new active ingredients become available for application, toxicity testing can provide a valuable tool for receiving system impacts. This study demonstrated that monitoring for a single active ingredient does not provide a complete picture of potential impacts to receiving systems. Toxicity testing is a tool that integrates effects of the active ingredient and its degradates, formulation components, and any chemical stressors that may already be present in the receiving system.

Studies have predicted that increases in global and regional temperatures with the progress of climate change will impact medically important insects such as mosquitoes (Trumble and Butler 2009). In California, a predicted increase in temperatures has the potential to increase mosquito activities in areas where human populations are predicted to rise. These factors will make the role of mosquito control districts more critical and will require they have a full complement of larvicides and adulticides at their disposal. Vector control districts use a combination of organophosphate, pyrethroid, and pyrethrin pesticides to control adult mosquitoes in California. Multiple compounds with different modes of action are required to reduce the capacity of mosquitoes to develop tolerance to any 1 group of compounds. Use of these compounds will require best management practices and continued monitoring to reduce risk to aquatic systems, including the receiving water systems regulated under the NPDES General Permit.

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SUPPLEMENTAL DATA

Supplemental Table 1. Summary of active ingredient detections and detections of other pertinent chemicals.

Supplemental Table 2. Summary of active ingredient detections and detections of other pertinent chemicals in sediment samples.

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