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An Ecological Risk Assessment for Insecticides Used in Adult Mosquito Management

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ABSTRACT

West Nile virus (WNV) has been a concern for people across the United States since the disease was initially observed in the summer of 1999. Since 1999, WNV has caused the largest arboviral encephalitis epidemic in US history. Vector control management programs have been intensively implemented to control mosquitoes that carry WNV. Our deterministic ecological risk assessment focused on 6 common mosquito adulticides used in vector control, including 3 pyrethroids (d-phenothrin, resmethrin, and permethrin), pyrethrins, and 2 organophosphates (malathion and naled). Piperonyl butoxide, a synergist for the pyrethroids, was also assessed. Both aquatic and terrestrial nontarget organisms were considered for acute and chronic exposures to the adulticides. Tier I exposure estimates were derived from ISCST3 and AERMOD for deposition and air concentrations affecting terrestrial organisms and PRZM-EXAMS for standard pond concentrations affecting aquatic organisms. Nontargets exposed to adulticides included small mammals, birds, as well as aquatic vertebrates and invertebrates in a pond subject to receiving the chemical via drift and runoff. Risk quotients were obtained by comparing exposures to toxic endpoints. All risk quotients were low indicating that risks to ecological receptors most likely were small.

Keywords: Risk assessment Mosquito management Insecticides Synergists Nontarget receptors

Original Research

INTRODUCTION

West Nile virus (WNV) has been a concern for people across the United States since the disease was initially observed in the United States in the summer of 1999. Since 1999, WNV has caused the largest arboviral encephalitis epidemic in US history (Huhn et al. 2003). Despite the fact that the disease has resulted in thousands of morbidity cases and hundreds of deaths in humans, many people also are concerned about the risks associated with controlling the mosquitoes that vector WNV using a variety of insecticides (Peterson et al. 2006). This concern is related to the perception that ecological and human exposure to the insecticides will lead to risks that are more severe than WNV itself.

West Nile virus is vectored by several mosquito species, mostly from the *Culex* genus (Turrel et al. 2005). Mosquito management plans have been implemented in the United States and globally to control mosquito vectors of WNV and many other diseases. Common mosquito adulticides are pyrethroids such as d-phenothrin, resmethrin, and permethrin, as well as organophosphates such as malathion and naled. Natural pyrethrins are also available as mosquito control agents.

The ultra-low-volume (ULV) space sprays target adult mosquitoes as they are flying through the air. Adulticides used as space sprays become effective when they are released in the atmosphere as extremely small liquid droplets in ULV formulations. Droplet sizes range from 8 to 30 microns. This droplet size increases the surface area available to contact the target. The insecticide is absorbed through the insect cuticle and takes effect soon after contact. Each adulticide is a broad-spectrum insecticide that is toxic to many arthropods. Other insects in the spray zone may be deleteriously affected by the application of adulticides. Mosquito control programs often aggressively apply adulticides in the midst of a disease outbreak. Most are sprayed in the evening or early morning when female mosquitoes are seeking a blood meal and many other arthropods, particularly pollinators, are inactive.

For most mosquito insecticides, ecological incident reports (i.e., adverse effects) have been reported but have been typically associated with pest control programs in crops. Despite the much lower use rates and ULV delivery methods, it is plausible that adult mosquito management programs may pose similar ecological risks.

Risks from mosquito management plans to control WNV vectors can be quantified using a risk assessment framework. Risk assessment is a formal discipline that provides an objective evaluation of risk. The discipline has become widely used to make decisions about new or controversial technologies that may pose a risk to the public or the environment. Scientific data as well as societal considerations are made to describe the nature of the risk and communicate risk with decision makers. Within a risk assessment, assumptions and uncertainties are clearly presented. Ecological risk can be quantified as a function of hazard and exposure (NRC 1983). The approach uses a tiered modeling system that moves from deterministic models based on conservative assumptions erring on the side of environmental safety to refined probabilistic models using more realistic assumptions (SETAC 1994). Risk assessment follows a logical framework. The process proceeds in stepwise fashion, including 1) problem formulation, 2) hazard identification, 3) dose–response relationships, 4) exposure assessment, and 5) risk characterization. These steps allow for the comparison of an estimated environmental exposure with a reference dose derived from a toxic effect (NRC 1983).

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To our knowledge, ecological risk assessments for insecticides used in adult mosquito management have not been published in the scientific literature. Therefore, the objective of this study was to deterministically quantify the ecological risks to warm- and cold-water vertebrates and aquatic invertebrates that may be in the watershed where a mosquito control program is implemented as well as birds and mammals that are in the spray zone.

MATERIALS AND METHODS

Problem formulation

Mosquito control is important to effectively manage vector-borne illnesses such as WNV. Limiting mosquito habitat, using repellents, and targeting the insects using synthetic and natural chemicals are tactics for integrated mosquito management. In the case of pesticides, each active ingredient may pose an unacceptable risk to nontarget aquatic and terrestrial organisms where the chemicals are applied to control mosquitoes. Therefore, protecting the public and the environment from deleterious exposures to chemicals is a major concern when managers consider implementing a mosquito management program.

Applying adulticides via a ULV sprayer spreads small aerosol particles that target flying mosquitoes as they are moving through the air (CDC 2003). Our conservative (lower tiered) deterministic assessment considered acute and chronic exposure and risk of ULV insecticides to aquatic vertebrates and aquatic invertebrates after a truck-mounted sprayer treated an area where part of the spray drifted into the pond and a portion of the insecticide that deposited on the ground was transported to the pond as runoff. The assessment also estimated risk to terrestrial vertebrates from exposure to the insecticides from deposition on fur and feathers followed by grooming, inhalation for 24 h after the spray event, and ingestion of food items that the insecticides settled on.

Hazard identification

The ecological risk assessment focused on 6 common mosquito adulticides, including 3 broad-spectrum pyrethroids (d-phenothrin, resmethrin, and permethrin), pyrethrins, and 2 broad-spectrum organophosphates (malathion and naled). Piperonyl butoxide (PBO), a synergist for the pyrethroids and pyrethrins, also was assessed on its own.

Incident reports including nontarget receptor mortality have been recorded for each chemical. Most ecological incidents related to the application of the organophosphates have been related to crop pest control programs. Malathion has been listed as the probable cause of fish kills in 1990 in Massachusetts, USA, where the chemical was applied directly to a 700,000-acre estuarine environment for mosquito control (USEPA 2000). Permethrin incidents include a few terrestrial effects, including 3 cases related to bird kills. Twenty-one incidents have been related to aquatic wildlife with only a few classified as probable (USEPA 2006b). One aquatic incident has been recorded as attributable to the use of resmethrin in mosquito control, and 1 incident in 1992 is listed as probable that honeybees in the spray zone experienced high mortality from spray drift (USEPA 2005g).

Dose–response relationships and toxic endpoints

Toxicity endpoints were obtained for those organisms for which consistent data were available. These organisms included 3 birds, bobwhite quail (*Colinus virginianus*), mallard duck (*Anas platyrhynchos*), and ring-necked pheasant (*Phasianus colchicus*); 4 mammals common in the field, rat (*Rattus rattus*), shrew (*Blarina brevicauda*), vole (*Microtus pennsylvanicus*), and mouse (*Peromyscus maniculatus*); and 4 freshwater aquatic organisms, rainbow trout (*Oncorhynchus mykiss*), bluegill sunfish (*Lepomis macrochirus*), waterflea (*Daphnia magna*), and amphipods (*Gammarus* sp.).

Each of the insecticides is classified by the US Environmental Protection Agency (USEPA) as slightly to moderately toxic to birds with acute LC50 values (insecticide concentrations in food that cause mortality in half the test animal population) ranging from 2,538 to 21,117 mg/kg dry weight of food, with naled being the most toxic to ring-necked pheasants. Chronic no-observed-effect-concentrations (NOECs) were generally from 5-month reproductive studies for all the chemicals except malathion's toxicity to mallard duck, which was a growth and viability study, and resmethrin's toxicity, which considered reduced weight of adult males both for mallard duck and for bobwhite quail. NOECs ranged from 60 to 1,200 mg/kg dry weight of food; of the 5 insecticides for which chronic NOECs are available, resmethrin was the most toxic (Table 1).

Each mosquito adulticide is considered by the USEPA as slightly to moderately toxic to mammals. Most mammalian testing is done on rats and calculated for other small mammal species by using dose-related rat toxic endpoints and converting them to dietary endpoints for other small mammals by implementing the following (USEPA 1998):

$$TE_m = TE_r \times BW \div C \tag{1}$$

where TE_m is the estimated dietary toxic endpoint for other small mammals (mg/kg dry weight food), TE_r is the dose-related dietary endpoint for rats (mg/kg body weight), BW is the body weight of the small mammal (kg), and C is the food consumption in 1 d (kg). Consumption data for each of the small mammals evaluated in the risk assessment were obtained from the *USEPA Wildlife Exposure Factors Handbook* (USEPA 1993) or appropriate allometric equations. It was assumed that rats weighed 390 g and consumed 18 g/d, shrews weighed 15 g and consumed 8 g/d, mice weighed 21 g and consumed 4 g/d, and voles weighed 32 g and consumed 7 g/d. Acute LD50 values range from 443 mg/kg dry weight of naled in food for shrews to 73,500 mg/kg dry weight of pyrethrins in food for mice. Chronic NOECs collected from 5-month reproductive studies, except for malathion, for which the NOEC was based on cholinesterase reduction, ranged from 17 mg/kg dry weight of naled in food for mice to 530 mg/kg dry weight of d-phenothrin or PBO in food for shrews (Table 2).

Each adulticide is classified as practically nontoxic to highly toxic to fish and aquatic invertebrates with acute endpoints ranging from 0.000075 ppm (mg/L of water) of resmethrin to bluegill sunfish to 30,000 ppm of d-phenothrin to *D. magna*. Chronic life cycle NOECs ranged from 0.000039 ppm permethrin for bluegill sunfish to 1.31 ppm PBO for rainbow trout (Table 3).

Exposure assessment

Terrestrial exposures—In addition to standard dietary consumption, exposure routes exposure through inhalation and grooming were included in the exposure pathways. Adulticides were assumed to be sprayed a total of 10 times (days 1,

Table 1. Toxic endpoints of adulticides for birds. NA = not available

Chemical	Species	Acute LC50 (mg/kg dry wt)	Chronic NOEC (ppm)
Malathion	Bobwhite quail	3,497[a]	110[a]
	Mallard duck	5,000[a]	1,200[a]
	Pheasant	2,639[a]	NA
Naled	Bobwhite quail	2,117[b]	130[c]
	Mallard duck	2,724[b]	260[b]
	Pheasant	2,538[b]	NA
Permethrin	Bobwhite quail	5,200[d]	500[d]
	Mallard duck	10,000[d]	125[d]
	Pheasant	23,000[d]	NA
Resmethrin	Bobwhite quail	5,000[e]	60[e]
	Mallard duck	5,000[e]	60[e]
	Pheasant	NA	NA
PBO	Bobwhite quail	5,620[f]	300[f]
	Mallard duck	5,620[f]	300[f]
	Pheasant	NA	NA
d-Phenothrin	Bobwhite quail	5,000[c]	NA
	Mallard duck	5,620[c]	NA
	Pheasant	NA	NA
Pyrethrins	Bobwhite quail	5,620[g]	NA
	Mallard duck	5,000[c]	NA
	Pheasant	5,000[c]	NA

[a] (USEPA 2005d).
[b] (USEPA 1997c).
[c] (USEPA 1997a).
[d] (USEPA 2005a).
[e] (USEPA 2005g).
[f] (USEPA 2005h).
[g] (USEPA 2005f).

4, 14, 17, 27, 30, 33, 43, 46, and 56) at the maximum label application rates. Application rates were 39 g/ha for PBO, 4 g/ha for d-phenothrin, 8 g/ha for permethrin and resmethrin, 64 g/ha for malathion, 22.5 g/ha for naled, and 9 g/ha for pyrethrins. We used a total of 10 sprays to represent a reasonable worst-case application regime during an outbreak of WNV in humans (NYCDOH 2005; Peterson et al. 2006). AERMOD v. 1.0 (USEPA 1999) was used to predict the 7.62-m (25-foot) air concentration of each insecticide within 1- and 12-h time ranges for each adulticide and PBO at the ground level for mammals and at an altitude of 7 m for birds. AERMOD is an industrial source plume model developed to predict air concentrations of pollutants from industrial sources and has been adopted as a Tier I model for this project. The reasonable worst-case exposure scenario had the following assumptions: 1) Each chemical had a 24-h half-life in the environment except for naled, which was given an 18-h half-life; 2) the ULV applications had 3% of the emitted particles greater than the allowable particle size; 3) the insecticides were applied at the maximum application rate as stated on each label; 4) all the insecticides were susceptible to the same weather conditions using standardized weather data from Albany, New York, USA, in 1988; 5) all spray events occurred at 9:00 PM; and 6) each spray release was at 1.5 m.

Modeled receptors were set up on a Cartesian grid at 5 intervals of 2 at 7.62 m from each side of the spray emission area. The receptors were at ground level for mammals and 7 m aboveground for birds, as many bird species are likely to be exposed aboveground. Each receptor recorded the 1- and 12-h average air concentrations for each insecticide. An average was then taken of the readings from the 6 receptors at 7.62 m that were not at the edges of the spray zone. The following data were obtained using this network of receptors: the 1-h average concentration at 7.62 m and the 12-h average at 7.62 m.

The industrial source complex dispersion model (ISCST3; USEPA 1995) was used to model particle deposition at 7.62 m from the spray area at the 1-h average. ISCST3 is AERMOD's predecessor and was developed 1st for prediction of deposition and aerial concentrations of industrial pollutants. The same assumptions were used with this program as with AERMOD except that the default meteorological data

Table 2. Toxic endpoints of adulticides for mammals

	Species	Acute LC50 (mg/kg dry wt)	Chronic NOEC (ppm)
Malathion	Shrew	736	53
	Mouse	2,048	19
	Vole	1,857	21
	Rat	(LD50 390[a])	(NOEL 100[a])
Naled	Shrew	443	48
	Mouse	1,234	17
	Vole	1,119	19
	Rat	(LD50 235[b])	(NOEL 90[b])
Permethrin	Shrew	2,791	53
	Mouse	7,765	19
	Vole	7,043	21
	Rat	(LD50 1,479[c])	(NOEL 100[c])
Resmethrin	Shrew	8,753	265
	Mouse	24,355	95
	Vole	22,090	105
	Rat	(LD50 4,639[d])	(NOEL 500[d])
PBO	Shrew	8,623	530
	Mouse	23,992	190
	Vole	21,761	210
	Rat	(LD50 4,570[e])	(NOEL 1,000[e])
d-Phenothrin	Shrew	9,434	530
	Mouse	26,250	190
	Vole	23,810	210
	Rat	(LD50 5,000[f])	(NOEL 1,000[g])
Pyrethrins	Shrew	26,415	53
	Mouse	73,500	19
	Vole	66,666	21
	Rat	(LD50 14,000[g])	(NOEL 100[f])

[a] (USEPA 2005d).
[b] (USEPA 1997c).
[c] (USEPA 2005a).
[d] (USEPA 2005g).
[e] (USEPA 2005h).
[f] (USEPA 2005f).
[g] (WHO/FAO 1994).

were from Salem, Massachusetts, USA. The following assumptions were made in addition to those from AERMOD: 1) The ULV applications had 3% of the emitted particles greater than the allowable particle size as stated on the label, and 2) the particles were assigned a density in accordance with the specific gravity of each insecticide.

A similar Cartesian grid was used for ISCST3 that was used in AERMOD previously. Receptors were at 7.62 m from the spray source. The receptors were at ground level and 7 m in accordance with the grid used for AERMOD. All the same methods were used to calculate the average deposition at 7.62 m. AERMOD and ISCT3 did not consider deposition onto foliar surfaces or lateral impingement by vegetation. This serves as a reasonable worst-case scenario, as lateral impingement would limit deposition and inhalation exposure to birds.

The Kenaga nomogram was used to predict concentrations on food such as long grass, short rangegrass, fruits, seeds, and insects and broadleaf plants (Fletcher et al. 1994). The nomogram is a linear model that uses application rate to predict concentrations of the insecticide on different types of food. For this exercise, estimated environmental concentrations (EEC) were calculated for short rangegrass (high value) as well as fruits, seeds, and insects (low value) because these 2 food type groups were at the extremes of predicted concentrations.

Each spray event was followed through 90 d after the spray event for estimating chronic exposures to surrogate birds and mammals that would be likely to eat food in the spray area, using the following degradation model (USEPA 2004):

$$D = \sum_{j=1}^{n} Pe^{r_1 t} \tag{2}$$

where D is the sum of the deposition over one spray, P is the peak deposition after a spray event, r_1 is the rate of decay calculated by using each active ingredient's aerobic soil half-life, t is the time in hours, j is the spray day, and n is the decay period. The daily deposited concentration (DD) can be calculated by the following:

$$DD = (\sum_{j=1}^{n} Pe^{r}t + \sum_{j=4}^{n} Pe^{r}t + \ldots + \sum_{j=56}^{n} Pe^{r}t) \tag{3}$$

Total acute exposures for the terrestrial wildlife were assumed to be a summation of exposure to the animal through its diet, cleaning and preening of fur or feathers, and inhalation on the evening of the spray event. Chronic exposures were evaluated in a similar way, although the grooming and inhalation doses were assumed to correspond with 1-d pulse events each spray day. Deposition on the surface of the animal was assumed to cover the entire body at the concentrations calculated from ISCST3. Inhalation doses corresponded with the values predicted from AERMOD. Animals were assumed to breathe the peak average concentration for 2 h after the spray event and to breathe the 12-h average concentration for an additional 22 h to develop a reasonably conservative estimated inhalation exposure on the spray date. Additionally, all exposures were standardized for dietary ingestion, which is congruous with the assumptions specified in Equation 1. Equation 4 outlines the acute exposure calculation for terrestrial wildlife:

$$EE_A = (DD \times C + EEC_d \times SA + EEC_a \times IR) \div C \tag{4}$$

where EE_A is the estimated acute exposure, DD is the estimated environmental concentration on food (mg/kg dry weight) from Equation 3, C is the consumption rate of the animal (kg dry weight/day), EEC_d is the estimated environmental concentration at the given receptor height for deposition (mg/m^2), SA is the animal's total surface area (m^2) as estimated by allometric equations or outlined in USEPA (1993), EEC_a is the estimated aerial concentration of the adulticide (mg/m^3), and IR is the inhalation rate (m^3) (Table 4). Chronic exposures were assessed by Equation 5:

Table 3. Toxic endpoints of adulticides for aquatic organisms. NA = not available

Chemical	Species	Acute LC50 (ppm)	Chronic NOEC (ppm)
Malathion	*Daphnia magna*	0.00[a]	0.00059[a]
	Bluegill sunfish	0.02[a]	NA
	Rainbow trout	0.004[a]	0.002[a]
Naled	*D. magna*	0.0003[b]	4.5E-05[c]
	Bluegill sunfish	2.2[b]	NA
	Rainbow trout	0.16[b]	NA
Permethrin	*D. magna*	0.00011[d]	3.9E-05[e]
	Bluegill sunfish	0.00079[e]	0.0003[c]
	Rainbow trout	0.0084[c]	0.09[c]
Resmethrin	*D. magna*	0.00022[c]	NA
	Bluegill sunfish	0.000075[d]	NA
	Rainbow trout	0.00028[f]	NA
PBO	*D. magna*	0.51[g]	0.066[c]
	Bluegill sunfish	4[g]	NA
	Rainbow trout	1.8[g]	1.31[c]
d-Phenothrin	*D. magna*	30,000[d]	NA
	Bluegill sunfish	0.001[c]	NA
	Rainbow trout	0.0014[d]	1.1[c]
Pyrethrins	*D. magna*	0.0116[g]	9.6E-05[c]
	Bluegill sunfish	0.0187[d]	NA
	Rainbow trout	0.0032[g]	NA

[a] (USEPA 2005d).
[b] (USEPA 1997b).
[c] (USEPA 1997a).
[d] (USEPA 2006a).
[e] (USEPA 2005a).
[f] (USEPA 2005g).
[g] (USEPA 2005h).
[h] (USEPA 2005f).

$$EE_C = \frac{\sum_{n=1}^{90} EE_A}{90} \qquad (5)$$

where EE_C is the estimated chronic exposure and n is the date within the 90-d model. For this summation, EEC_d and EEC_a are zero within Equation 4 on nonspray days to mimic pulse events.

Aquatic exposures—The USEPA water quality software EXPRESS v. 1.00.00.012 (USEPA 2005c) was used to obtain EECs in the standard farm-pond, 2-m deep with a 1-ha

Table 4. Descriptive statistics for terrestrial animal exposures

	Food intake (kg/d)	Surface area (m^2)	Inhalation rate (m^3/d)
Bobwhite quail	0.0177	0.0331	0.1
Ring-neck pheasant	0.0582	0.1002	0.41
Mallard duck	0.0619	0.1132	0.45
Rat	0.018	0.0594	0.1002
Shrew	0.008	0.0054	0.026
Mouse	0.004	0.0089	0.024
Vole	0.0068	0.0139	0.043

Table 5. Input Parameters for PRZM-EXAMS. NA = not available

Chemical	Molecular weight (g/mole)	K_{oc}	Vapor pressure (mPa)	Solubility (ppm)	Aerobic soil half-life (d)	Foliar half-life (d)	Aerobic biolysis (d)	Anaerobic biolysis (d)	Aqueous photolysis (d)	Hydrolysis (d) pH 5	pH 7	pH 9
Malathion	330.4[a]	1,200[a]	0.45[a]	130[a]	1[a]	5.5[b]	3.5[b]	7.64[b]	42[b]	107	6.3	0.05[a]
Naled	381[a]	157[a]	26[a]	1.5[a]	4[a]	NA	1.5[c]	4.5[c]	4.4[c]	4	0.642	0.067[c]
Resmethrin	338.4[a]	100,000[a]	0.01[a]	0.0379[a]	197.5[d]	NA	36.5[d]	682[d]	0.033[d]	89	168	127[d]
d-Phenothrin	350.5[e]	56,000[e]	1.43E-7[e]	0.0097[e]	26[f]	NA	7.2[f]	61.9[f]	5[f]	301	495	120[e]
Pyrethrins	328.4[a]	100,000[a]	0.001[a]	0.001[a]	9.5[g]	NA	10.5[g]	86[g]	0.5[g]	Stable	Stable	17[g]
Permethrin	391.3[a]	39,300[a]	0.0029[a]	0.006[a]	30[a]	NA	38[h]	175[h]	30[a]	Stable	Stable	49.87[a]
PBO	338.5[i]	399[i]	5E-13[i]	14.3[i]	14[i]	NA	75[i]	181[i]	0.35[i]	Stable	Stable	Stable

[a] (USDA-ARS 2005).
[b] (USEPA 2000).
[c] (USEPA 1997c).
[d] (USEPA 2005g).
[e] (USNLM 2005).
[f] (WHO/FAO 1994).
[g] (USEPA 2005f).
[h] (USEPA 2005a).
[i] (USEPA 2005h).

surface area scenario through its interface with the Pesticide Root Zone Model v. 3.12.3 (PRZM; USEPA 2005e) and the Exposure Analysis Modeling System v. 2.98.04.06 (EXAMS; USEPA 2005b) for aquatic organism exposures. The input parameters used for all adulticides were 1) a 45-m buffer as the distance of the spray from the pond, as the chemicals are not to be sprayed into permanent water bodies; 2) spray drift into the pond was 1%, reflecting the default drift percentage from a high-boom, fine-particle ground sprayer; 3) applications were made on Florida turfgrass, a conservative, minimal vegetative cover that would be in a mosquito control area; 4) applications began on 1 July and ended 25 August with 3- and 10-d intervals between applications; and 5) applications were made at the maximum use rate listed on the label for mosquito control. PRZM-EXAMS input parameters for most of the mosquito insecticides were gathered from their respective USEPA Reregistration Eligibility Document or other sources (Table 5). Acute exposures for fish and *D. magna* were predicted as the peak estimated environmental concentration. Chronic exposures for fish were defined as the 60- or 90-d estimated average environmental concentration depending on the length of the study associated with the appropriate toxic endpoint. Chronic exposures for *D. magna* were defined as the 21-d estimated average environmental concentration.

None of the models used in this study specifically incorporates ULV applications. In ISCT3, a range of aerosol particle diameter values is incorporated into the model. PRZM-EXAMS was developed to mimic broad-spectrum crop pest control programs and only has a "very-fine" spray parameter. These models that do not use ULV spray application modeling are most likely very conservative because the particle size definitions would tend to concentrate the particle in an area of interest. Despite not having a "ULV" parameter, PRZM-EXAMS lets the modeler specify the amount of drift into a pond regardless of spray type.

Risk characterization

Ecological risks for the adulticides were assessed by dividing the EEC determined by PRZM-EXAMS for aquatic organisms or a combination of AERMOD, ISCST3, and the Kenaga nomogram for terrestrial organisms by the toxic endpoints for each organism to obtain a risk quotient (RQ). Exposure periods that corresponded to the period of each chronic endpoint study were used for aquatic organisms, and terrestrial organisms were assumed to be exposed for 90 consecutive d after the 1st spray date.

RESULTS

Acute risks to birds exposed to 1 spray application ranged from an RQ of <0.001 for permethrin to 0.02 for malathion (Table 6). Chronic RQs were calculated only for bobwhite quail and mallard duck because there were limited data on chronic toxicity endpoints for ring-necked pheasant. Chronic RQs ranged from 0.01 for permethrin, malathion, and naled to 0.2 for resmethrin. Chronic endpoints were not available for d-phenothrin or pyrethrins (Table 7). Ingestion from preening contributed most of the acute exposure to birds, ranging from 69% to 99% of total exposure. Dietary or preening ingestion contributed to most of the chronic exposure depending on the persistence of the chemical. If the chemical was deposited on fruits and seeds, the major contributing exposure route was preening, with a 61% to 97%

Table 6. Acute risk quotients for surrogate birds after one spray event. NA = not available

Chemical	Bobwhite quail	Ring-necked pheasant	Mallard duck
Malathion	≤0.02	≤0.02	≤0.01
Naled	≤0.01	≤0.009	≤0.008
Permethrin	≤0.002	<0.001	≤0.001
PBO	≤0.01	NA	≤0.009
Resmethrin	≤0.002	≤0.002	NA
d-Phenothrin	≤0.002	NA	≤0.002
Pyrethrins	≤0.001	≤0.001	≤0.001

contribution to total exposure. If it was applied to short rangegrass, the major contributor varied between dietary intake and preening.

Acute RQs were calculated for surrogate species that would represent the most common mammals in an area. Risk quotients ranged from <0.001 for permethrin, PBO, resmethrin, d-phenothrin, and pyrethrins to 0.07 for malathion (Table 8). Chronic RQs ranged from 0.003 for d-phenothrin to 0.5 for malathion (Table 9). Contributing exposure for mammals was similar to birds, with ingestion from grooming contributing the largest portion of the acute exposure in most cases, contributing 28% to 99% of total exposure. Grooming or diet contributed the largest portion of chronic exposure depending on the persistence of the chemical and the organism. Shrew exposure estimates were a particular anomaly because they consume almost half their body weight in food per day.

Daphnia magna and amphipods (scuds) were the only aquatic invertebrates evaluated in the assessment. Acute RQs for *D. magna* ranged from <0.001 for PBO to 0.04 for naled, malathion, and permethrin. Chronic risks for *D. magna* were <0.001 for PBO to 0.04 for permethrin. Available information for amphipods was limited. Acute RQs for scuds exposed to organophosphates ranged from 0.08 to 0.09 (Table 10).

Acute RQs for both warm- and cold-water fish ranged from <0.001 for PBO, permethrin, and naled to 0.02 for d-phenothrin. Chronic RQs for rainbow trout ranged from <0.001 for PBO and d-phenothrin to 0.004 for malathion (Table 10).

Table 8. Acute risk quotient for surrogate mammals after one spray event

Chemical	Shrew	Mouse	Vole
Malathion	≤0.07	≤0.04	≤0.04
Naled	≤0.03	≤0.03	≤0.006
Permethrin	≤0.001	<0.001	≤0.002
PBO	≤0.02	<0.001	≤0.01
Resmethrin	<0.001	<0.001	<0.001
d-Phenothrin	<0.001	<0.001	<0.001
Pyrethrins	<0.001	<0.001	<0.001

DISCUSSION

Levels of concern (LOCs) are regulatory decision tools used in the risk management process (USEPA 2006c). Managers compare the calculated RQ to the RQ LOC to determine if regulatory action is needed. The acute regulatory LOCs used by the USEPA are 0.5 for nonendangered birds and mammals, 0.2 for nonendangered terrestrial vertebrates exposed to restricted use insecticides, 0.1 for endangered birds and mammals, 0.5 for aquatic species, 0.1 for aquatic species exposed to restricted use insecticides, and 0.05 for endangered aquatic species. Chronic LOCs are 1.0 for all animals.

Only amphipods exposed to organophosphates exceeded USEPA RQ LOCs for any of the chemicals within any LOC category. Mammalian toxicity endpoints derived from studies done on rats did not account for interspecific variation within the mammalian surrogates considered. Although these extrapolations are common practice within USEPA risk assessment for insecticide reregistration, safety factors could be applied to the mammalian RQs to preserve reasonable worst-case scenario conservatism. Indeed, safety or uncertainty factors could be applied to any of the RQs in our assessment and then compared to RQ LOCs. We chose not to add safety factors because we believe our assessment is already highly conservative, and it also is beyond the scope of the assessment to make determinations as to the acceptability of risks.

The results from our risk assessment can be compared to a few other studies. Pearce and Balcom (2005) summarized a worst-case impact of pyrethroid insecticides sprayed around Long Island Sound (NY) that may have been responsible for a lobster kill within the fishery. Tier II modeling approaches

Table 7. Chronic risk quotients for surrogate birds after 10 spray events. NA = not available

Chemical	Bobwhite quail	Mallard duck
Malathion	≤0.08	≤0.01
Naled	≤0.03	≤0.01
Permethrin	≤0.01	≤0.06
PBO	≤0.08	≤0.08
Resmethrin	≤0.2	≤0.2
d-Phenothrin	NA	NA
Pyrethrins	NA	NA

Table 9. Chronic risk quotients for surrogate mammals after 10 spray events

Chemical	Rat	Shrew	Mouse	Vole
Malathion	≤0.06	≤0.5	≤0.5	≤0.5
Naled	≤0.04	≤0.09	≤0.3	≤0.3
Permethrin	≤0.06	≤0.1	≤0.3	≤0.3
PBO	≤0.02	≤0.03	≤0.09	≤0.09
Resmethrin	≤0.02	≤0.04	≤0.1	≤0.1
d-Phenothrin	≤0.003	≤0.005	≤0.07	≤0.01
Pyrethrins	≤0.03	≤0.06	≤0.2	≤0.1

Table 10. Acute and chronic risk quotients for surrogate aquatic receptors. NA = not available

Species	PBO	Pyrethrins	Permethrin	Resmethrin	Naled	Malathion	d-Phenothrin
Acute 1 spray event							
Daphnia magna	<0.001	0.001	0.04	0.01	0.04	0.04	NA
Bluegill sunfish	<0.001	0.001	0.005	0.04	<0.001	0.002	0.02
Rainbow trout	<0.001	0.006	<0.001	0.01	<0.001	0.01	0.01
Amphipod	NA	NA	0.002	NA	0.08	0.09	NA
Chronic 10 spray events							
D. magna	<0.001	0.04	0.04	0.01	0.04	0.03	NA
Rainbow trout	<0.001	NA	NA	0.002	NA	0.004	<0.001

concluded that insecticide concentrations in Long Island Sound of malathion ranged from 1 to 5 ppm; the highest estimated environmental concentrations were 0.099 ppm for d-phenothrin and 0.034 ppm for resmethrin (Landeck-Miller et al. 2005). Therefore, Pearce and Balcom (2005) concluded that the relative risk of the pesticides was negligible compared to other circumstances, such as persistent disease and warm water that were already affecting the lobster population.

Jensen et al. (1999) found no treatment effects on nontarget aquatic invertebrates downwind from ULV sprays of pyrethrins, malathion, and permethrin. Further, they found no impacts on species diversity within the seasonally impounded ponds and detected chemical concentrations that would be nontoxic to vertebrate receptors.

The New York City Department of Health (NYCDOH) conducted a geographically specific environmental impact statement (EIS) published online in 2005 in which risk outcomes from adulticiding were mostly consistent with our findings. Many of the chemicals evaluated in our risk assessment were evaluated by NYCDOH. Ecological risks for avian and mammalian receptors were below LOCs for all surrogates after a Tier II risk assessment. Potential nontarget impacts on terrestrial and aquatic invertebrates that exceeded an LOC were addressed by citing studies where evidence suggests that nontarget impacts on these communities would be negligible. NYCDOH (2005) used different predictive modeling resources to conduct their risk assessment. We used the PRZM-EXAMS model in our study and provided screening-level/Tier II aquatic concentration outcomes within a long-term scenario where consistent adulticiding was assumed to occur every summer. These predictive outcomes were of lower magnitude than the NYCDOH model but are considered robust by the USEPA (NYCDOH 2005).

An EIS similar to NYCDOH (2005) is currently being conducted by the Suffolk County Department of Public Works and Department of Health Services in Suffolk County, New York, USA (Suffolk-County 2006). The draft Suffolk County EIS is consistent with our results and predicts negligible acute and long-term ecological effects of adulticiding at both the individual and the community level. Terrestrial dietary RQs from the terrestrial screening-level assessment did not exceed 1.0 for any of the species examined. The screening level of the aquatic assessment identified risks that breached levels of concern in aquatic environments for malathion and permethrin synergized by PBO. A Tier II risk assessment conducted to mitigate potential risks reduced RQs below LOCs.

Final reregistration eligibility documents have been published by the USEPA for naled, malathion, and PBO. No regulatory issues were identified for mosquito control where the application rate parameter was 111 g/ha for naled. This estimate was made using an application rate 5 times of that used in our assessment (USEPA 1997c, 2000). Malathion exceeded regulatory LOCs for freshwater fish and invertebrates at the maximum application rate for mosquito control of 706 g/ha assuming 20% drift from a truck-mounted ULV application (USEPA 2000). The USEPA (2005h) found that PBO used in mosquito abatement did not breach acute or chronic LOCs for aquatic organisms in a scenario similar to our assessment, nor were any chronic LOCs breached for mammals when PBO was assumed to be applied at 560 g/ha, but breached some chronic risk to birds at the same application rate. The lowest application rate in the USEPA reregistration eligibility document was 14 times greater than the rate suggested for use in mosquito control. The application rates we used were gathered from insecticide labels and represent rates commonly used in mosquito control.

Many terrestrial arthropods will be exposed in the same way as flying or resting mosquitoes. Spray applicators generally apply mosquito insecticides when mosquitoes are active at dawn and dusk. Spray events that follow the mosquitoes' behavior are likely to limit exposure to honeybees and other terrestrial arthropods in the spray zone, but some mortality is likely in the sprayed areas. However, because of the nature of the applications and their frequency, it is unclear how arthropod communities will be affected in the field (Caron 1979). Therefore, more studies on terrestrial nontarget arthropods are warranted.

Coldburn and Langford (1970) found high bee mortality from applications of naled, malathion, and pyrethrum with PBO. These field tests did not use a ULV sprayer, but the study suggests that chemicals commonly used in mosquito control may lead to nontarget arthropod mortality. Caron (1979) exposed caged honeybees and hives to ULV sprays of malathion, naled, and pyrethrum. This study showed decreasing mortality as distance increased from the insecticide release. Pankiw and Jay (1992) found that honeybees in cages experienced significant mortality from malathion spray drift. Hester et al. (2001) observed significant bee mortality in hives that were exposed to malathion spray events both in open fields and in a forested environment. However, there was no effect on beehive weight or health of the colony over a season. Tietze et al. (1996) used sentinel crickets to measure spray deposition in a peridomestic environment. Cricket mortality varied between 12.5% and 48.7% depending on where crickets were put in residential yards in relationship to the spray zone.

Variability and uncertainty associated with the environmental modeling approaches used in our assessment exist. None of the models has a scenario for ULV spray applications, which limits their ability to predict EECs. A robust series of insecticide and aerosol studies that focus on environmental fate and the insecticide action in the environment would greatly benefit future modeling applications. The physicochemical input parameters for the environmental fate models also are uncertain. The USEPA uses conservative estimates in their reregistration eligibility documents on insecticide persistence depending on how many studies have been conducted on the degradation of each insecticide. Their input parameters are inconsistent with the properties listed in the USDA-ARS pesticide properties database and, in many cases, are multiples of these values. Carbone et al. (2002) found when variability was accounted for within PRZM-EXAMS, by changing input parameters; the model was robust for chemical registration purposes. The nature of the environmental fate models used in our study, also used by the USEPA, represent reasonable worst-case scenarios and most likely overpredict both acute and chronic exposures of nontarget organisms to insecticides.

Limited studies on downwind deposition and drift, specifically related to acute exposures, highlight the probable conservative outcomes from AERMOD and ISCST3. Moore et al. (1993) found that deposition of malathion on ground-level patches between 15.2 and 91.2 m ranged from 8.4×10^{-4} to 4.2×10^{-5} g/m^2. Tietze et al. (1996) measured an immediate-peak deposition of malathion in a peridomestic environment of 4.73×10^{-3} g/m^2. Average depositions in the same study were 3 times less, ranging from 2.99×10^{-4} to 8.8×10^{-4} g/m^2. Knepper et al. (1996) observed 0.1439 g/m^2 and 0.0922 g/m^2 concentrations of permethrin and malathion, respectively, in turfgrass after a ULV spray event in a suburban environment. Although Knepper et al. (1996) observed higher concentrations of malathion than Tietze et al. (1996) and Moore et al. (1993) immediately after spraying, Knepper et al. (1996) showed that malathion and permethrin had very low persistence in the environment with almost no trace of either after 36 h. The deposition values modeled in our assessment were 1.7 to 4.9 times greater than the averages from Tietze et al. (1996) and Moore et al. (1993) but less than Knepper et al. (1996). However, our chronic models used chemical half-lives much greater than those observed by Knepper et al. (1996).

We also conducted similar simulations using Agdrift v. 2.00.05 (Spray-Drift-Task-Force 2000) that resulted in much lower estimates of insecticide deposition at the ground level compared to ISCST3. Indeed, predicted ranges in the Tier I Agdrift model were 1% to 3% of the predicted surface deposition concentrations used in our assessment. Agdrift is currently the industry standard for predicting both on- and off-target spray drift deposition and is considered a conservative model.

Although not within the scope of the assessment presented here, including bootstrapping or Monte Carlo simulations that allow for uncertainty in the modeling process is an available technique until more studies are conducted on the degradation of these chemicals. Model variability can be varied within AERMOD and ISCST3 in the form of deposition and air concentration characteristics from topography, spray time, spray date, geographical location, and a variety of other factors. Crop type and spray date could be varied in PRZM-EXAMS. In some cases, different studies come to different conclusions about the persistence of chemicals given different substrates and varied environmental conditions. Variation in the inhalation, preening, and ingestion rates and food items in birds and mammals also add to the variation in RQ outcomes. Varying characteristics and entering them as distributions into stochastic models that develop a range of exposure predictions related to the variability of each input provides a range of risk values.

Synergized pyrethroids (containing PBO) have been shown to be more toxic to trout than technical grade pyrethroids (USEPA 1993; Paul et al. 2006). Based on our results, the RQs when PBO is added to the pyrethroids and pyrethrins would still be below LOCs for aquatic vertebrates. If toxicity was increased 10-fold for aquatic invertebrates, RQs would still be below LOCs for nonendangered species (Table 10). However, a large-scale management response to WNV where PBO is applied with pyrethrins or pyrethroids has been shown to synergize with more persistent pyrethroids in aquatic systems. Weston et al. (2006) found supplemental concentrations of PBO in creeks around Sacramento, California, USA, that increased residual pyrethroid efficacy, particularly bifenthrin, on nontarget amphipods. However, none of the 11 water samples collected in their study caused significant mortality to *Cerodaphnia dubia* (Weston et al. 2006).

Several receptors that may be sensitive to the insecticides evaluated in this assessment were not included because toxicity data were inconsistent both within each insecticide and between insecticides. Amphibians, benthic invertebrates, and estuarine species may be particularly important when assessing the risks of each insecticide in the environment.

Results from our conservative ecological risk assessment and the weight of scientific evidence suggest that risks to ecological receptors most likely are small from ULV insecticides applied within a mosquito management program. Because we used several conservative exposure assumptions, more realistic assessments most likely would result in RQ values lower than reported here. Further, environmental exposures from adulticide applications are not likely to add appreciably to background levels of the same active ingredients from agricultural and urban uses.

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