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ASSESSING THE FATE AND EFFECTS OF AN INSECTICIDAL FORMULATION

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Abstract: A 3-yr study was conducted on a corn field in central Illinois, USA, to understand the fate and effects of an insecticidal formulation containing the active ingredients phostebupirim and cyfluthrin. The objectives were to determine the best tillage practice (conventional vs conservation tillage) in terms of grain yields and potential environmental risk, to assess insecticidal exposure using concentrations measured in soil and runoff water and sediments, to compare measured insecticidal concentrations with predicted concentrations from selected risk assessment exposure models, and to calculate toxicity benchmarks from laboratory bioassays performed on reference aquatic and terrestrial nontarget organisms, using individual active ingredients and the formulation. Corn grain yields were not significantly different based on tillage treatment. Similarly, field concentrations of insecticides were not significantly ($p > 0.05$) different in strip tillage versus conventional tillage, suggesting that neither of the tillage systems would enable greater environmental risk from the insecticidal formulation. Risk quotients were calculated from field concentrations and toxicity data to determine potential risk to nontarget species. The insecticidal formulation used at the recommended rate resulted in soil, sediment, and water concentrations that were potentially harmful to aquatic and terrestrial invertebrates, if exposure occurred, with risk quotients up to 34. *Environ Toxicol Chem* 2015;34:197–207. © 2014 SETAC

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INTRODUCTION

As of 2011, 75% of the total land area in Illinois, USA, was used for farming, and the state ranked second in the United States for corn production [1]. To ensure the long-term productivity of the land, and to reduce labor and equipment costs, conservation tillage and crop residue management have been widely utilized in Illinois, to the extent that conservation tillage is now more common than conventional tillage when all crops are considered [2]. Conservation tillage includes no-till and low-till practices that often are defined by the amount of residues left on the field after treatment. Strip tillage is a type of no-tillage system in which a shank and disk blades run parallel to the rows, thereby building a soil berm up to 25 cm wide; it is usually performed in the fall to aid soil drying and warming in the spring [2]. As of 2007, conventional tillage was still in use on approximately one-third of the total planted US acres [2]. In Illinois, this most common tillage system for corn production involves a chisel plow operation in fall, followed by a field cultivator before planting in the spring [2]. Conventional tillage eliminates most plant residues from the surface of the field and buries disease-bearing crop and weed residues, thereby reducing problems with weed competition and plant diseases [2]. As a result of the presence (or absence) of plant residues at the soil surface, tillage practices influence many soil parameters that play a major role in the plant cycle, such as soil temperature, moisture, compaction, and organic matter content [2].

Although the application of insecticides, excluding seed-coated insecticides, has decreased in corn in recent years in Illinois, 3 insecticides—phostebupirim, cyfluthrin, and tefluthrin—are currently the most commonly applied to corn, at totals of 70 tons, 4 tons, and 35 tons in 2010, respectively [1]. A formulation containing the active ingredients phostebupirim (2%; also called tebupirimphos), an organophosphate insecticide, and cyfluthrin (0.1%), a pyrethroid insecticide, is widely used across the state. This formulation is used to control corn rootworms, cutworms, and other soil insect pests in corn grown for seed and silage [3]. Although the organophosphate and pyrethroid insecticidal groups have been studied quite extensively, there is little research about the environmental fate of these 2 compounds individually or in a formulation. Field experiments are especially lacking, and only a few studies related to the formulation registration reviewed by the US Environmental Protection Agency (USEPA) are available [4–6].

Simulation models often are used to assess the exposure to the environment and/or population, especially in lower tiered regulatory risk assessments. The tiered risk assessment approach has been developed to avoid unnecessarily long and expensive pesticide studies. Using this method, tier I exposure models conservatively screen out molecules that do not require further investigation; for the other compounds, tier II exposure models are used to determine whether further study is needed, including higher-tier field monitoring and/or mesocosm tests. Lower-tier exposure models are based on the physical and chemical parameters of the compound, as well as several agronomic or soil parameters (e.g. crop, planting method, and soil texture). In the present study, we aimed to verify the accuracy of tier I and II models by comparing simulated data with actual measured field concentrations.

The environmental risk of this insecticidal formulation is extremely difficult to assess because few ecotoxicological data are available for cyfluthrin and phostebupirim, and no data were found in the literature for the mixture of the 2 compounds.

All Supplemental Data may be found in the online version of this article.
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Phostebupirim was shown to pose an acute risk above the USEPA's levels of concern for small birds, amphibians, reptiles, mammals, fish, and crustaceans [3]. Likewise, cyfluthrin is highly toxic to nontarget terrestrial invertebrates, as well as aquatic vertebrates and invertebrates [7]. Both insecticides are neurotoxins, although they operate with different modes of action. Phostebupirim is an acetylcholinesterase inhibitor that blocks the degradation of the neurotransmitter acetylcholine at nerve synapses, which causes hyperexcitation of the central nervous system [3]. Cyfluthrin is a type II synthetic pyrethroid that acts on nerve axons by inhibiting neurotransmitter delivery via inhibition of the calcium ion channels coupled with a stimulatory effect on the sodium ion channels, affecting both the peripheral and central nervous systems [7]. Because these 2 insecticides are neurotoxic, but are from different classes with different modes of action, the toxicity of the mixture is difficult to predict. A few studies have shown possible synergism or antagonism of the toxicity when nontarget organisms were simultaneously exposed to organophosphates and pyrethroids [8–10]. In addition to the 2 active ingredients present in the formulation, inert ingredients make up 97% of the product, which might potentially modify the toxicity [11].

The present study investigated the fate and transport of this insecticidal formulation in a continuous corn field in Illinois undergoing different tillage treatments. In an attempt to better understand the mechanisms governing the transport of the insecticides and to relate it to the conditions of the field, several soil chemical and physical parameters were monitored. Field concentrations of the 2 insecticides were then compared with model concentrations to examine their accuracy. Bioassays were also performed in the laboratory on reference nontarget organisms to assess the relative risks of the individual compounds and the formulation. The ultimate goal of the present study was to determine which tillage practice a farmer should adopt to minimize environmental impacts of insecticides without impacting profits on a long-term basis.

MATERIALS AND METHODS

Field study

The field portion of the project took place in Shelby County in Central Illinois. The 18-ha field site was managed under corn on corn (maize) from 2011 to 2013, and a commercially available insecticidal formulation was applied in-furrow at planting every year at a nominal rate of 8.2 kg/ha. The formulation was made of 2 active ingredients: 2% phostebupirim and 0.1% cyfluthrin. The composition of the remaining 97.9% inert ingredients was unknown by the authors. Total exchange capacity, pH, organic matter, and total organic carbon (TOC) contents were measured in the spring and fall of 2011 and fall 2012, each time before tillage, using 45 locations throughout the field from mineral soil collected from 0 cm to 15 cm of depth. In the top 15 cm, the silt loam soil (24% sand, 56% silt, 20% clay) had a total exchange capacity of 27 ± 6 meq/100 g, a pH of 6.0 ± 0.5, $3.7 \pm 0.4\%$ organic matter, and $2.1 \pm 0.3\%$ TOC, on average on the whole field for 2011 and 2012. Two tillage treatments were performed in alternate 18.3-m-wide bands every fall, containing 24 corn rows on a 72-cm wide row spacing. In the fall, conventional tillage was performed using a combination of disc gang and chisel plow shanks with twisted shovels, whereas the strip tillage used a single knife shank with disc coulters to berm the soil in a 25-cm wide strip. Tillage was performed each subsequent year by shifting the corn rows 36 cm to split the corn rows from the previous year. In the spring, a field cultivator was operated to a depth of 7.6 cm on the conventional tillage portions of the site to prepare the soil for planting. The entire field was planted with non-*Bt* corn in 2011, and *Bt* refuge-in-a-bag corn (contained 95% Bt and 5% non-Bt) was planted in the following years. Other traditional crop treatments were performed and were consistent across the field and included diammonium phosphate, or potash, applied before planting; herbicides (s-metolachlor, atrazine, mesotrione), applied immediately after planting; nitrogen stabilizer, anhydrous ammonia, and herbicides (simazine and 2,4-dichlorophenoxyacetic acid), applied following fall tillage.

Precipitation patterns were different for the 3 yr of the study [12]: 2011 was characterized by a wet spring with 11.2 cm and 15.8 cm of rain within 1 mo prior to and following planting, respectively, and had the driest summer, with 9.1 cm of rain over the subsequent 3 mo. Spring of 2012 was the driest season, with 3.2 cm and 11.5 cm of rain within 1 mo prior to and following planting, respectively; the following 2 mo of summer were dry as well, with 5.3 cm of total rainfall. A record drought occurred at the study area in the growing season of 2012. The year of 2013 had the wettest spring, with several flash floods, and 17.9 cm and 13.9 cm of rain within 1 mo prior to and following planting, respectively; the following month had a rainfall of 8.3 cm, and then late summer was dry, with 4.6 cm of total rainfall for the 2 following mo. Dry and warm conditions in 2012 allowed an early planting date of April 10, whereas corn was planted in late May in 2011 and 2013. Soil samples for insecticide analyses were collected monthly and consisted of surface soil samples collected from the top 2 cm of the mineral soil. Twenty different locations were sampled following a W pattern, with 10 locations from each tillage treatment. To answer an additional research question and reduce variability observed in previous years' data, care was taken to collect half of the samples in the crop row and half in the middle of the crop row within each tillage treatment between planting and tillage in 2013. In 2011 and 2012, samples were collected only within the crop row. Polyvinyl chloride (PVC) pipes (approximately $1.8\,m \times 0.4\,m$) were cut in half lengthwise along their axis to make overland flow collectors and were checked for absence of binding with hydrophobic compounds. The PVC troughs were installed in the field at 6 locations, 3 in each tillage treatment, to collect surface runoff samples. Runoff water and sediment samples were collected monthly following a rain event equal to or greater than 1.27 cm. As a result of equipment installation delays and a lack of significant rain events in summer 2011, runoff samples were collected only in 2012 and 2013.

Soil parameters and grain yield data

Infiltration rates of the soils were measured with a double-ring infiltrometer at 3 locations in each tillage treatment. The outer, buffer ring was 61 cm in diameter, and the center, measuring ring was 30 cm in diameter. Both rings were driven 15 cm into the soil. The measuring ring was centered within the buffer ring. Soil around the cylinder walls was slightly tamped to prevent short circuit flow or leakage [13]. Approximately 15 cm of water was ponded and maintained within both rings throughout the test using 3-L and 10-L Mariotte tubes; the 3-L Mariotte tube was connected to the measuring ring, and the 10-L Mariotte tube was connected to the outer ring. The level of water within the 3-L Mariotte tube was recorded in 5-min intervals for 1 h. The rate of fall of water level within the 3-L Mariotte tube was used to calculate the infiltration rate [13].

Saturated hydraulic conductivity was measured using a compact constant head permeameter at 3 locations in each tillage treatment. The device maintained a constant column of water

15 cm high within a hole 60 cm deep and 5 cm in diameter. The volume of water infiltrated was noted at 5-min intervals, and mean values were recorded after 3 consecutive readings of the same rate [14]. Soil bulk density samples were collected in 18 locations in each tillage treatment using a hammer-driven core sampler with a 5.4-cm diameter and a 5.9-cm height [15].

Harvest data were collected with a John Deere Greenstar™ 3 GPS unit, processed with Apex Farm Management software, and then analyzed using ArcGIS 10.0 (Esri 2011). Data points were collected approximately every 2.7 m throughout the field (>10 000 points). Sampling points were trimmed to include the 90th percentile of the harvest data to reduce the variation associated with low-density areas related to planting and field elevation. The trimmed data were then averaged to obtain mean field grain yield estimation for each tillage treatment.

Insecticide analyses

Soil and runoff water samples were collected using a trowel and an aluminum scoop, respectively. Care was taken with runoff samples to make sure high quantities of sediments were collected with the water. All samples were transferred into glass mason jars previously rinsed with pesticide-grade acetone (Fisher Scientific), transported to the laboratory in coolers, and stored in the dark at 4 °C. Soil samples were extracted within 4 wk of collection, and water samples were allowed to settle and then extracted within 2 wk of collection. Phostebupirim and cyfluthrin were extracted simultaneously in runoff and soil samples. Aqueous samples were extracted using liquid–liquid extraction, followed by solid-phase extraction (SPE) for cleanup. Solid samples, including soil, sediment, and earthworm tissue, were extracted by sample rotation in solvent, followed by SPE or gel permeation chromatography for cleanup. Detailed procedures are given in the Supplemental Data.

Following extraction, analytes were quantified on an Agilent Technologies 6850 gas chromatograph coupled with a 5975C inert XL electron impact/chemical ionization mass spectrometer (EI/CI–MS) detector. Cyfluthrin was present at very low concentrations, and negative chemical ionization (NCI) was needed to achieve the lower quantitation levels. Although phostebupirim was usually present at higher concentrations, the better sensitivity obtained in EI mode for this compound was necessary for some samples that were at detection limits; these samples were reinjected in EI mode for confirmation. The system was equipped with an HP-5MS Agilent column (30 m × 0.25 mm × 0.25 μm). The total run time was less than 25 min, starting with an oven temperature of 90 °C, which was increased to 275 °C at 15 °C/min, then to 285 °C at 2 °C/min, and finally to 300 °C at 10 °C/min, and then held at 300 °C for 6 min. The mass spectrometer detector was operated in the selected ion monitoring mode with a quadrupole temperature of 150 °C and source temperatures of 150 °C and 230 °C in NCI and EI modes, respectively. Each analyte was searched using a quantitation ion and 2 confirmation ions, which differed depending on the ionization mode. In the formulation, cyfluthrin existed as a mixture of 8 possible isomers, which produced 4 different peaks (for the 4 pairs of diastereoisomers) on the chromatogram (Supplemental Data, Figure S1). These peaks were not well resolved, especially the third and fourth peaks that coeluted; therefore, the 4 peaks were integrated together, and total cyfluthrin was reported.

Method detection limits (MDLs) were measured in spiked runoff water and soil from an adjacent field free of these insecticides (see Supplemental Data for MDL methods). In runoff water, MDLs were set at 1.9 ng/L in EI and 4.0 ng/L in NCI for phostebupirim, and 9.6 ng/L in EI and 3.2 ng/L in NCI for cyfluthrin. In soil, MDLs were 0.1 ng/g dry weight in EI and 1.7 ng/g dry weight in NCI for phostebupirim, and 1.5 ng/g dry weight in EI and 0.1 ng/g dry weight in NCI for cyfluthrin.

Bioassays

To assess the risk caused by the individual insecticides and the insecticidal formulation, toxicity bioassays were performed on nontarget organisms. Standards of phostebupirim (97.2% pure; Chem Service) and cyfluthrin (98% pure; Chem Service) were utilized to perform bioassays on individual insecticides. Because the commercial formulation investigated was a restricted use pesticide as a result of its toxicity to aquatic organisms, 3 aquatic species and only 1 terrestrial species were included—*Daphnia magna, Hyalella azteca, Pimephales promelas*, and *Eisenia fetida*. The bioassays followed protocols adapted from the Institutional Animal Care and Use Committee, the USEPA, and the Organisation for Economic Co-operation and Development guidelines [16–19]. All of the test species were exposed to the insecticides individually. *Daphnia magna* were exposed for 48 h to spiked water, *H. azteca* and *P. promelas* were exposed to spiked water for 96 h, and *E. fetida* were exposed to spiked soil for 14 d. Additional 10-d static sediment bioassays were conducted with *H. azteca* to assess the toxicity from sediment exposure. As recommended by the USEPA protocols, *H. azteca* were fed at the beginning of the bioassays and every 48 h with 0.2 mL/beaker of a yeast, Cerophyl, and trout chow (YCT) solution [17]. All other species were not fed during the tests, as proposed by the guidelines. For the water tests, 500 mL of moderately hard reconstituted water [20] were added to each beaker, with the exception of the *D. magna* bioassay, which used only 200 mL of moderately hard water in each beaker. Reference soil clean of any pesticide contamination was used in the soil and sediment bioassays and was collected 15 km south of Carbondale (IL, USA) and sieved to a particle size of ≤500 μm. This soil, which has been used as a reference soil previously [21], was classified as a silt loam (14% sand, 60% silt, 26% clay) after texture analysis (Bouyoucos hydrometer), contained 1.0% organic matter (combustion method), had a pH of 5.2, and a cation exchange capacity (standard method USEPA 9081) of 10.7 meq/100 g (Midwest Laboratories). The soil moisture content was 16% in the earthworm bioassay. Additional bioassays were conducted using the formulation granules dissolved in water or directly mixed into the soil to assess the toxicity of both insecticides in conjunction with their inert ingredients. For *H. azteca*, the formulation toxicity was assessed only for water exposure. For each species, preliminary range-finding tests were performed using 5 concentration levels, followed by definitive tests using 7 concentration levels. Concentrations were checked right before addition of the animals and at the end of each bioassay by liquid–liquid extractions or soil extractions following the field sample protocols. Bioassays were conducted in triplicate, and each replicate contained 10 organisms. All toxicity results reported in the present study had satisfactory survivorship in the negative and solvent controls (≥80% survival or no effect). Endpoints included lethality for all species, difficulty swimming and/or lack of or erratic movements for the aquatic invertebrates and fish, and growth for the earthworms.

Calculations

Economic return by tillage was calculated as the price obtained for the cost of tillage subtracted from corn grain income, per hectare. Tillage cost was not actually measured.

Instead, estimates of machinery cost using University of Illinois (IL, USA) indices were utilized: \$60.05/ha for conventional tillage (fall chisel plow + spring field cultivator), and \$41.76/ha for strip tillage [22]. The returns from corn grain yields were calculated from the prices received by farmers as of December 2011, 2012, and 2013 according to the US National Agricultural Statistics Service [23].

The times required for 50% (DT50) and 90% (DT90) of the initial insecticide concentrations to dissipate from the soil were calculated for each growing season using an exponential regression of the mean soil concentrations versus time in days, in the form of Equation 1

$$C = A_1 \times \exp(-xt_1) \quad (1)$$

where C represents the mean soil concentrations at a time x (in days). The initial insecticide concentration (A_1) and the dissipation rate constant (t_1) were calculated from the regression for each year and each compound.

Lethal concentrations to half of the population (LC50s) and effective concentrations to half of the population (EC50s) were calculated from measured concentrations of the media using SPSS (Ver 20.0; IBM) probit regression, or Trimmed Spearman–Karber nonparametric test (ToxCalc^{TM} 5.0; Tidepool Scientific) for the nonnormally distributed results. For the earthworms, the lowest concentration affecting worm growth (lowest-observed-effect concentration [LOEC]) and the highest concentration not affecting worm growth (no-observed-effect concentration [NOEC]) were calculated using SPSS Dunnett's analysis of variance (ANOVA) tests.

To assess the environmental risk, risk quotients (Q) were calculated as the ratios of insecticidal field concentrations to LC50s and EC50s obtained from our bioassays for each species and each insecticide, using Equations 2 and 3

$$Q_{\mathrm{EC50}} = \mathrm{HMFC/EC50} \quad (2)$$

$$Q_{\mathrm{LC50}} = \mathrm{HMFC/LC50} \quad (3)$$

where Q_{EC50} and Q_{LC50} are the sublethal and lethal risk quotients, respectively, and HMFC is the highest mean field concentration for 1 sampling event.

Statistics regarding comparisons of soil parameters, tillage treatments, and within rows versus in-between rows were performed using independent t tests and/or ANOVA after checking for normality and transformation of the data to a normal distribution when necessary (SPSS). Each statistical test was conducted for $\alpha = 0.1$.

Models

Tier I and II exposure models from the USEPA that are used in regulatory risk assessments—that is, FIRST, GENEEC2, and EXPRESS Ver 1.03.02 (EXAMS–PRZM Exposure Simulation Shell for pesticide aquatic exposure assessment)—were evaluated. Based on standard scenarios, the tier I FIRST model provides estimation of peak day (acute) and annual (chronic) concentrations in a watershed reservoir, whereas the tier I GENEEC2 model was designed to predict peak, 4-d, 21-d, 60-d, and 90-d concentrations in a standard farm pond. Tier II EXAMS and PRZM are more complex models that provide refined estimation of pesticide concentrations in surface waters and include limnetic and benthic dissolved exposure concentrations at different time periods, and 1-m deep leachate concentrations, and are based on 30 yr of meteorological data. The input data used for the models were from the USEPA registration reviews for phostebupirim [3] and cyfluthrin [7].

RESULTS

Agroeconomics

Insufficient rainfall and extreme heat in summer 2011 and in 2012 greatly affected corn productivity, with average grain yields on the whole field being higher in 2013 at 11 800 ± 2197 kg/ha, compared with 7280 kg/ha ± 1695 and 3140 ± 1632 kg/ha in 2011 and 2012, respectively. Corn grain yields at the field study site in 2011 and 2012 were lower than the average yields of Shelby County, Illinois, which were estimated at 9164 kg/ha in 2011 and 4519 kg/ha in 2012; the opposite was true for 2013, with an estimated 11 173 kg/ha average yield in Shelby County. Over the course of the 3-yr project, no significant differences were observed regarding the influence of tillage treatments on grain yields (Table 1). Although conventional tillage (fall chisel plow followed by spring field cultivator) cost an additional \$18.29/ha compared with strip tillage [22], conventional tillage provided a higher economic return over tillage cost in 2011 and 2012 (Table 1). In 2013, a higher economic return was obtained for strip tillage. However, the differences in economic return between tillage treatments were not significant (overlap of standard deviations) and were equal to \$11, \$185, and \$8, in 2011, 2012, and 2013, respectively.

Soil

Soil organic matter and TOC percentages increased significantly ($p = 0.067$ and $p = 0.0001$, respectively) on the whole field between spring 2011 and fall 2012, both before respective tillage (Table 1). During this period, organic matter content increased from 3.6% to 3.8%, and TOC content increased from 2.0% to 2.3%, representing a 6% and 15% increase, respectively. The pH and total exchange capacity changed significantly during this period: the pH increased from 5.8 to 6.1 (5% increase), and the total exchange capacity dropped from 28.8 meq/100 g to 25.0 meq/100 g (15% decrease). None of the soil parameters, including organic matter/TOC, pH, and total exchange capacity, were significantly different based on the tillage treatment ($p > 0.1$). Saturated hydraulic conductivity and infiltration rates were measured after tillage in spring 2012 and were both higher in conventional tillage (1.73 ± 0.54 cm/h in conventional tillage vs 0.55 ± 0.55 cm/h in strip tillage for the saturated hydraulic conductivity, and 3.3 ± 1.0 cm/min in conventional tillage versus 2.9 ± 0.8 cm/min in strip tillage for the infiltration rates); however, only the saturated hydraulic conductivity was significantly different based on tillage treatment ($p = 0.057$). Bulk density was measured in fall 2011 and 2012, and spring 2012 and 2013 and was significantly higher in strip tillage than in conventional tillage ($p < 0.05$) only in the fall of 2011 and 2012. The bulk density values decreased over time, starting in spring 2011 at 1.48 g/cm^3 and 1.42 g/cm^3 for strip tillage and conventional tillage, respectively, and dropping to 1.34 g/cm^3 and 1.27 g/cm^3 by spring 2013.

Neither phostebupirim nor cyfluthrin were applied to the field before the project started, and this finding was confirmed in the preapplication samples of 2011 where all samples collected were below detection limits. Following planting in 2011, both insecticides were detected in almost all of the soil samples, although at detection limits during the winter, between tillage and planting. The highest concentrations measured over the 3 growing seasons were 5121 ng/g dry weight of phostebupirim

Table 1. Agroeconomic and soil factors as influenced by conventional- and strip-tillage systems in 2011, 2012, and 2013

		Tillage	
Agroeconomic or soil factor	Year	Conventional	Strip
Treatment cost ($/ha)	2011–2013	60.05	41.76
Corn grain yield (kg/ha)	2011	7407 ± 1632	7281 ± 1758
	2012	3578 ± 1632	2825 ± 1506
	2013	11926 ± 2071	11864 ± 2260
Corn grain benefits ($/kg)	2011	0.231	0.231
	2012	0.270	0.270
	2013	0.174	0.174
Economic return over tillage cost ($/ha)	2011	1651 ± 377	1640 ± 406
	2012	906 ± 441	721 ± 407
	2013	2015 ± 360	2023 ± 393
Organic Matter (%)	Spring 2011	3.65 ± 0.43	3.60 ± 0.42
	Fall 2011	3.82 ± 0.46	3.69 ± 0.42
	Fall 2012	3.81 ± 0.41	3.76 ± 0.43
Organic Carbon (%)	Spring 2011	2.02 ± 0.22	1.99 ± 0.26
	Fall 2011	2.10 ± 0.32	2.03 ± 0.29
	Fall 2012	2.35 ± 0.29	2.33 ± 0.33
Bulk density (g/cm^3)	Fall 2011	1.42 ± 0.09	1.48 ± 0.08
	Spring 2012	1.36 ± 0.10	1.41 ± 0.09
	Fall 2012	1.24 ± 0.12	1.38 ± 0.13
	Spring 2013	1.27 ± 0.13	1.34 ± 0.16

and 328 ng/g dry weight of cyfluthrin, both 9 d after planting in 2011. Soil concentrations followed the same pattern each year, with the highest concentrations observed a few days after planting, and these concentrations decreased over time until they reached detection limits after fall tillage (Figure 1). There was no significant difference in insecticide concentrations between the 2 tillage treatments for any of the sampling events. However, concentrations in samples collected within corn rows in 2013 were significantly higher ($p < 0.05$) than the ones collected in-between rows until they both reached low levels, approximately 3 mo and 1 mo after planting for phostebupirim and cyfluthrin, respectively. There was therefore little horizontal movement of the insecticides from the application site (within rows) toward the space between rows.

Using the mean concentrations of the 10 soil samples per tillage treatment, DT50s were calculated for each year, insecticide, and tillage treatment (Table 2). In 2011, both insecticides and both tillage treatments gave similar DT50s, between 29 d and 37 d. In 2012, the dissipation times were longer for phostebupirim, especially in conventional tillage. In 2013, DT50s were similar to those of 2011, but the same trend toward longer DT50s in conventional tillage was observed as in 2012.

Runoff water and sediments

The highest runoff water concentrations were 2315 ng/L for phostebupirim and 4 ng/L for cyfluthrin, both 7 d after planting in 2013, in a sample collected in strip tillage. The same sample also had one of the highest phostebupirim concentrations detected in runoff sediments, at 192 ng/g dry weight. The highest measured cyfluthrin sediment concentration was 307 ng/g dry weight, and the corresponding sample was also collected 7 d after planting in 2013 in strip tillage. On day 20 after planting, the phostebupirim concentrations in runoff water and sediments were 13 times higher in 2013 than in 2012 (Figure 2). Cyfluthrin was not detected in runoff water in 2012, but the average concentrations observed in sediments were 178 times higher in 2013 than in 2012 at 20 d following insecticide application.

Soil–water partitioning

An estimation of the soil–water partitioning coefficient (K_d) was made using the ratio of the concentrations of each insecticide in the sediments to the concentrations in (unfiltered)

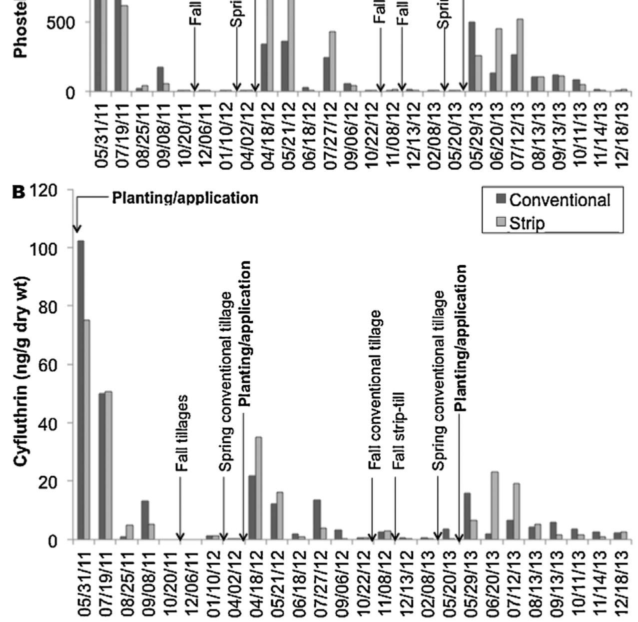

Figure 1. Soil concentrations (ng/g dry wt) of (**A**) phostebupirim and (**B**) cyfluthrin. For clarity, error bars are not represented. $n = 10$ for each treatment.

Table 2. Soil dissipation half-lives[a]

	2011			2012			2013		
	Whole field	Conventional	Strip	Whole field	Conventional	Strip	Whole field	Conventional	Strip
Phostebupirim	30 (0.96)	30 (0.96)	29 (0.96)	63 (0.85)	67 (0.81)	52 (0.96)	31 (0.93)	38 (0.91)	34 (0.95)
Cyfluthrin	34 (0.93)	32 (0.94)	37 (0.88)	25 (0.94)	30 (0.92)	22 (0.96)	40 (0.89)	37 (0.93)	30 (0.89)

[a]Dissipation half-lives (DT50s) are in days. R^2 values in parentheses.

water. The log K_d values were calculated to be 2.3 ± 0.3 and 4.4 ± 0.5 for phostebupirim and cyfluthrin, respectively. The mean TOC content for the sediments was $3.2\% \pm 0.4\%$. Using this TOC value, the logarithms of sediment organic carbon–water partitioning coefficients (K_{OC}) were calculated to be 3.5 ± 0.2 and 5.9 ± 0.5 for phostebupirim and cyfluthrin, respectively. The log K_{OC} value estimated for phostebupirim in the present study is close to the values of 3.0 to 3.4 reported in the registration studies, whereas our cyfluthrin value was slightly higher compared with the values of 4.9 to 5.3 reported in the registration studies [3,7].

Models

The maximum average phostebupirim water concentrations measured in the field in 2013 (1070 ng/L) were close to the acute concentrations estimated using the FIRST model (1220 ng/L).

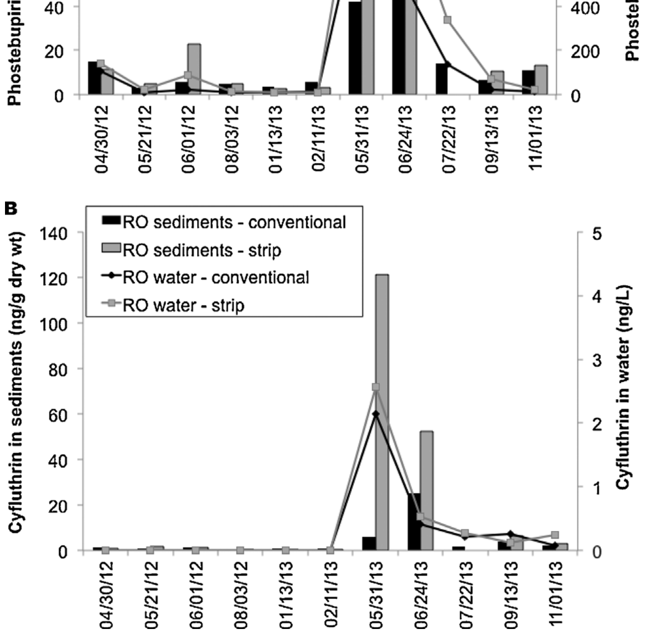

Figure 2. (**A**) Phostebupirim and (**B**) cyfluthrin concentrations in runoff (RO) water and sediments in 2012 and 2013; no runoff samples were collected in 2011. For clarity, error bars are not represented. $n = 3$ for each treatment.

For cyfluthrin, maximum average field water concentrations were approximately 10 times lower than the estimated peak value (28.25 ng/L). Therefore, the FIRST model appears to be fairly accurate at modeling the highest concentrations detected in the field and is more likely to overestimate these values for some compounds, especially for a dry year, such as 2012. When compared with the average concentrations measured in the field in 2013, the annual average (chronic) concentrations from the same model were slightly underestimated for phostebupirim and slightly overestimated for cyfluthrin, with field annual mean values of 170 ng/L (vs 142 ng/L estimated) and 0.29 ng/L (vs 0.61 ng/L estimated) for phostebupirim and cyfluthrin, respectively.

The second model evaluated was GENEEC2, which estimated maximum concentrations at different intervals of time. The model predicted maximum values at 4 d and 21 d post application of 1260 ng/L and 1120 ng/L for phostebupirim, respectively, and 7.56 ng/L and 2.43 ng/L for cyfluthrin, respectively. In the field, the average values for 7 d and 31 d post application in 2013 were 1070 ng/L and 700 ng/L for phostebupirim, respectively, and 2.4 ng/L and 0.5 ng/L for cyfluthrin, respectively. The model values were slightly overestimated for phostebupirim and cyfluthrin, but in the same range. Overall, both of these tier I models seemed to be accurate and to reflect a worst-case scenario of the field concentrations, which may be much lower in the absence of precipitation during the growing season.

The tier II EXPRESS model predicts pesticide concentrations in surface waters and estimated highest surface water concentrations at 188 ng/L for phostebupirim and 0.35 ng/L for cyfluthrin. Annual estimated concentrations were 25 ng/L for phostebupirim and 0.009 ng/L for cyfluthrin. These concentrations were far below the measured field runoff concentrations and may be more reflective of surface water concentrations. No adjacent stream water was monitored during the present field study and thus these data could not be compared. Another result obtained from this model is the unlikeliness of leaching of both insecticides with leachate concentrations estimated at 0.

Bioassays

Short-term toxicity bioassays were conducted on nontarget aquatic and terrestrial species to assess the risk of the individual insecticides and the insecticidal formulation. The EC50 and LC50 values were measured, and the results are given in Table 3. For earthworms, growth was assessed as a sublethal effect, and the highest concentration not affecting growth (NOEC) and the lowest concentration affecting growth (LOEC) are presented in Table 3 instead of EC50s.

Insecticide granules dissolved in water affected aquatic and terrestrial invertebrates as expected based on the sum of individual toxicities; in other words, additive toxicity was noted with no effect found attributable to the inactive ingredients. For fish, the formulation showed potentiation or synergism, with

Table 3. Sub-lethal and lethal concentrations (EC50 and LC50, respectively) to half of the test organisms of individual insecticides to reference non-target organisms

	Matrix	Units	Phostebupirim[a]		Cyfluthrin[a]	
			EC50	LC50	EC50	LC50
Hyalella azteca	Water	ng/L	498 (450–549)	555 (505–610)	0.8 (0.7–0.9)	1.3 (1.0–1.8)
H. azteca	Sediment	ng/g dry weight	2.6 (2.0–3.4)	4.9 (3.6–6.7)	7.5 (5.7–9.8)	10.0 (8.3–11.9)
Daphnia magna	Water	ng/L	51 (29–64)	100 (68–136)	30 (25–35)	35 (30–42)
Pimephales promelas	Water	μg/L	357 (296–429)	808 (562–1375)	0.27 (0.16–0.50)	1.21 (1.01–1.46)
Eisenia fetida[b]	Soil	μg/g dry weight	LOEC[c] = 0.38	1.19 (0.71–2.59)	NOEC = 0.384	>0.384

[a]95% confidence intervals in parentheses.
[b]For earthworms, the highest concentration not affecting growth (NOEC) and the lowest concentration affecting growth (LOEC) are presented instead of EC50s, and were calculated at $p \leq 0.05$.
[c]$30 \pm 1\%$ loss of body mass over 14 days vs $14 \pm 3\%$ in the controls.

toxicities 7 times higher than expected based on the sum of individual insecticide toxicities. Additional testing showed that the increased toxicity to fish was mainly the result of potentiation of cyfluthrin toxicity because of phostebupirim, even at nontoxic levels of phostebupirim, and that toxicity attributable to the inert ingredients was unlikely (unpublished data).

Comparison with literature values was particularly difficult for phostebupirim, because the only available benchmark value in common with the species used in the present study was a 48-h EC50 of 78 ng/L for *D. magna*, which was in the same range as the EC50 of the current study (51 ng/L) [3,24]. More data were available for cyfluthrin, and the benchmark values found in the ECOTOX database were rather close to the values found in the present study: *D. magna* 48-h EC50 of 25 ng/L (vs 30 ng/L in the present study) and *P. promelas* 4-d LC50 of 1.08 μg/L (vs 1.21 μg/L in the present study) [24]. *Daphnia magna*'s LC50s from the database were higher than in the current study (0.035 μg/L) and ranged from 0.17 μg/L to 0.62 μg/L. For *H. azteca*, only aqueous 10-d LC50s were available in the database, but they were in the same range (1.7–5.7 ng/L) as the ones obtained in the 4-d test (1.3 ng/L). For *E. fetida*, only one study using a commercial insecticide containing β-cyfluthrin was found with a LC50 of 5.5 $\mu g/g_{soil}$, which is 14 times higher than the NOEC of the present study (0.384 μg/g dry wt) [25].

DISCUSSION

Strip tillage and conventional tillage corn provided equivalent grain yields over the 3 yr of the study, with a trend toward higher yields in conventional tillage in very dry conditions, as seen in 2012. Hendrix et al. obtained similar results, with corn grain yields being greater in conventional tillage in 2001, which was the driest year of their 2-yr study [26]. They suggested that tillage before planting in conventional tillage may have released soil moisture from below the soil surface, and then increased the corn population, and thus the grain yields. Economic return was equivalent for the 2 tillage treatments over the course of the 3-yr study but seemed to be advantageous in conventional tillage in the case of severe drought. The higher benefits in 2012 in conventional tillage were attributable to yields approximately 750 kg/ha higher than in strip tillage, and a higher than usual value of corn given the very low grain yields obtained across the US Midwest.

The bulk density in the surface soil horizons decreased over time in both tillage treatments, but the difference between conventional tillage and strip tillage was significant only in the fall. Lower soil compaction was noted in conventional tillage, reflecting only a short-term effect on soil compaction as a result of tillage at this point of the study. Previous research reported higher bulk density in no-tillage soils, compared with conventionally tilled soils, or no difference in bulk density based on tillage treatment [26]; therefore, changes in compaction may occur in the field as part of the present study, but will require a longer period of time (>3 yr) to see the effects. The conventional tillage had higher mean water infiltration rates than strip tillage, although the differences were not statistically significant, possibly because of the high degree of variability in soil infiltration rates across an agricultural field. It is likely that a greater number of infiltration measurements would have resulted in a statistical difference between the 2 tillage treatments, but the scope of the study did not allow for further measurement. In addition, sealing of the mineral soil surface was expected to be greater in strip-tilled compared with conventionally tilled soils over the time span of the study, which should result in less infiltration and more runoff. Saturated hydraulic conductivity was statistically higher in conventional tillage soils. These findings suggest possibly easier transport of the water from the surface into the soil and higher percolation of the water within the soil. Even though this potentially lower water infiltration rate in strip tillage soils was unlikely to have modified the transport of the 2 insecticides downward into the soil (because they both have poor potential for leaching), it may have resulted in an increase in runoff, causing a greater amount of soil-bound insecticides in runoff, thus increasing the insecticide concentrations in runoff water and sediments. In addition, the higher insecticide concentrations in runoff water and sediments under strip tillage may also be attributed to the higher surface soil concentrations in these areas, with erosion of this more contaminated soil leading to more contaminated runoff media. These results are in contrast to a study on a smaller field site in more controlled conditions, which showed increased runoff water and sediment volumes in chisel-plowed plots compared with strip tillage, following a 60-min simulated rainfall [27]. However, the same authors also found a correlation between runoff volumes and losses of the organophosphate insecticide terbufos, which were higher in areas subject to higher runoff [27].

The reasons for the trend toward higher soil concentrations of both insecticides in strip tillage are unclear. Both compounds are moderately persistent under aerobic soil degradation [3,7], but they may be susceptible to biodegradation by microbes in the soil. We hypothesized that differences might have existed in either the type of organic matter or the composition of the microbial communities based on tillage type. Organic matter content of a soil is important in terms of the bioavailability of the compounds, because cyfluthrin and phostebupirim are both hydrophobic and will bind tightly to soil organic matter, thereby making the pesticides unavailable for biodegradation by the microbes. However, organic matter was not found to be different based on tillage treatment. In terms of microbial communities, 29 microbial taxa were identified throughout the field, 1 mo after planting in 2012, with no taxonomic differences between the 2 tillage treatments. Further analyses are being conducted to determine whether differences existed at the species level, and it is possible that biodegradation occurred because of an enhancement of certain microbial biochemical activities rather than shifts in microbial populations (A. Fakhoury, College of Agricultural Sciences, Southern Illinois University, Carbondale, IL, USA, personal communication). Therefore, different biodegradation processes might explain the differences between the tillage treatments. The primary routes of dissipation of phostebupirim may be aqueous and soil photolysis, hydrolysis, and volatilization [3]. Another more likely possibility is that plant residues present in the strip tillage soils may lower soil temperatures, prevent volatilization of the insecticides, and block sunlight, thereby preventing phostebupirim dissipation and/or degradation. The last assumption may also be true for cyfluthrin, whose primary routes of dissipation are aqueous and soil photolysis [7].

The DT50s of the insecticides in surface soil were similar for each tillage treatment in 2011 and in the same range of DT50s obtained in 2013. In 2012, DT50s were longer for phostebupirim in both tillage treatments but were slightly lower for cyfluthrin, especially in the strip tillage soils. Spring 2012 was very dry before and during planting; thus, less upward transport of the insecticides to the surface likely occurred, reducing dissipation of residues by runoff transport and aqueous abiotic processes. The dissipation rate of phostebupirim, which is less hydrophobic and more sensitive to hydrolysis than cyfluthrin [3,7], may have been slowed down because of the lack of water in the soil. In addition, soil photolysis may have increased the rate of degradation of cyfluthrin (half-life [$t_{1/2}$] = 5.6 d), while not affecting phostebupirim, which has a much lower soil photolysis rate ($t_{1/2}$ = 106 d) [3,7]. The main degradation process for phostebupirim was therefore most likely microbial degradation, which would help explain the longer dissipation rates, because its aerobic soil metabolism half-lives can be up to 1849 d [3]. When it did rain, more transport of the insecticides may have occurred via runoff in strip tillage soils, which likely had lower infiltration rates because of surface sealing and saturated hydraulic conductivity, thereby increasing dissipation of the insecticides compared with conventional tillage soils. This resulted in shorter DT50s in strip tillage; despite the faster decrease in soil concentrations in strip tillage, the soil concentrations were still higher than in conventional tillage, mostly because of the greater starting concentrations in the strip tillage treatment. This could indicate better movement of phostebupirim in soil in strip tillage, first to reach the soil surface after the in-furrow application, and then to dissipate off site via runoff.

Therefore, the fate and transport of both insecticides were strongly related to rainfall, the water infiltration capacity of the soil, and thereby the runoff amount. In the absence of (or with low) precipitation, as in 2012, biodegradation may be the major process impacting the fate of these compounds, slowing the rate of degradation of both, but especially of phostebupirim, which is more resistant to aerobic soil metabolism [3,7]. An evaluation of the 90% dissipation times over the 3 yr of the study showed that both insecticides were not very persistent in soil, because their DT90s were less than 1 yr. In 2012, however, the average DT90 for phostebupirim was 209 d, which coincided with the tillage period, meaning that more than 10% of the initial insecticide residues may have persisted longer in the surface soil if no tillage had occurred. This result was specific to 2012, because in 2011 and 2013, phostebupirim DT90s ranged from 97 d to 126 d, which was before harvest and tillage. For cyfluthrin, DT50s ranged from 75 d to 132 d throughout the 3 yr of the study, and background levels were achieved before tillage. Soil disturbance may thus be an important factor affecting the dissipation of some compounds under certain weather conditions and should be taken into account when exposure to insecticides is evaluated for an agricultural purpose.

The overall degradation of cyfluthrin was not consistent for the 4 pairs of isomers, because a different distribution of cyfluthrin isomers was found over time. The extent of the difference in isomer relative distribution was dependent on the year. The 3 resolved peaks obtained for cyfluthrin by gas chromatography–mass spectrometry using our analytical conditions corresponded to the groups of isomers I, III, and II + IV, which are *cis*, *trans*, and *cis* + *trans* isomers, respectively [28,29]. Over time, the proportion of the second peak, cyfluthrin *trans*-isomer III, decreased from 29% of the sum of the 3 peak areas right after planting to approximately 12% after 76 d in 2013. In 2012 and 2011, the proportion of the second peak went from 31% and 18% right after planting to 20% and 17% after 100 d and 86 d, respectively. The ratio of the second peak right after planting to 2 mo to 3 mo later was therefore close to 1 in 2011, 1.6 in 2012, and 2.5 in 2013. Chromatograms of cyfluthrin immediately after planting and 5 mo later are given in Supplemental Data, Figure S1 to show the preferential metabolism of *trans* isomers. Li et al. [30] showed that, for β-cyfluthrin, the *trans*-isomer was degraded faster than the corresponding *cis*-isomer, and abiotic degradation processes, such as hydrolysis and photolysis, were not likely to depend on stereoisomerism [31]. Therefore, microbial degradation in soil was likely a pathway of cyfluthrin degradation, becoming more important every year. Unfortunately, the samples in the present study were not injected on a chiral column, and thus the 8 optical isomers could not be separated. Because we showed that the isomer distribution was different over time, and because only 2 of the 8 isomers are toxic [29], it would be of interest to conduct a better assessment of ecological risk to determine which isomers actually underwent biodegradation and which remained in the field longer. This is especially true because the enantiomer-selective degradation seemed to depend on soil and sediment characteristics, including the pH of the soil [30,31].

Environmental relevance

Because of the modes of action of both compounds, aquatic invertebrates have been found to be more susceptible to the insecticides than fish in laboratory bioassays. In water, *D. magna*, a common indicator species for freshwater systems, was the most sensitive species for phostebupirim, whereas the aquatic amphipod *H. azteca* and the fish *P. promelas* were less sensitive. For cyfluthrin, *H. azteca* was the most sensitive species, and *D. magna* and *P. promelas* were less sensitive. For both compounds, the fish toxicity laboratory benchmarks were several orders of magnitude higher compared with the aquatic invertebrate benchmarks. To determine potential risk to nontarget species, laboratory bioassays were compared with field insecticide concentrations to calculate risk quotients based on environmental exposure.

Table 4. Risk quotients (Q) calculated for sub-lethal and lethal endpoints for the reference species

		Phostebupirim			Cyfluthrin		
	Matrix	EC50/LC50 (μg/L or μg/g dry wt)	HMFC (μg/L or μg/g dry wt)	Q_{EC50}/Q_{LC50}	EC50/LC50 (μg/L or μg/g dry wt)	HMFC (μg/L or μg/g dry wt)	Q_{EC50}/Q_{LC50}
Hyalella azteca	Water	0.498/0.555	1.069	2.1/1.9[a]	0.0008/0.0013	0.0024	3.0/1.8[a]
H. azteca	Sediments	0.0026/0.0049	0.089	34.0/18.2[a]	0.0075/0.0100	0.064	8.5/6.4[a]
Daphnia magna	Water	0.051/0.100	1.069	21.0/10.7[a]	0.030/0.035	0.0024	0.08/0.07
Pimephales promelas	Water	357/808	1.069	0.003/0.001	0.27/1.21	0.0024	0.009/0.002
Eisenia fetida	Soil	0.38[b]/1.19	1.716	4.5/1.4[a]	>0.384	0.091	<NOEC

[a]Risk quotient > 1.
[b]Lowest-observed-effect concentration.
EC50 = sub-lethal and lethal concentration affecting half of the test organisms; LC50 = lethal concentration affecting half of the test organisms; HMFC = highest mean field concentration; NOEC = no-observed-effect concentration.

Risk quotients were calculated as the ratios of insecticidal field concentrations to LC50s and EC50s obtained from the bioassays conducted in the present study for each species and insecticide (Table 4). The acute risk for sublethal and lethal effects in the aquatic invertebrates and *E. fetida* for the 3 matrices (soil, runoff water, and sediments) was higher than the 0.1 level of concern for aquatic animal acute restricted use exposure for at least 1 of the individual insecticidal compounds. Risk quotients greater than 0.1 indicate the potential for acute risk as a result of exposure [32]. For the invertebrates tested, the toxicity of the formulation was equivalent to the sum of the toxicity of each individual compound; therefore, the risk quotient for the formulation is equal to the sum of the individual quotients. The sum of the risk quotients calculated for the formulation was similar in value to those obtained for phostebupirim alone for *D. magna* and *E. fetida*; however, 2 to 5 times the risk was observed for *H. azteca* exposed to water and sediments, respectively. For *P. promelas*, individual risk quotients were well below 0.1, even after addition of the risk quotient of both compounds. However, bioassays performed using the formulation and the mixture of individual active compounds showed that the formulation was 7 times more toxic than expected if the toxicity were additive. With the appropriate multiplication factor applied to the sum of the risk quotients, they remained below 0.1; therefore, the formulation should not present an acute risk to freshwater fish.

The tillage treatment did not significantly affect the risk potential for any of the species investigated in the present study, even though slightly higher concentrations in strip tillage may present slightly higher risk. Risk quotients were calculated using peak mean insecticide concentrations following planting to assess the maximum potential acute risk posed to nontarget species. A steady decline in concentrations was observed through the remainder of the year in soil and water matrices, indicating a lesser degree of risk for these time periods. Maximum concentrations observed in the field following planting suggest a potential risk to aquatic and terrestrial nontarget invertebrate species.

In addition, earthworm body residues were measured after the 14-d *E. fetida* bioassays, and both phostebupirim and cyfluthrin were detected in significant amounts, up to 240 μg/g dry weight and 0.135 μg/g dry weight for phostebupirim and cyfluthrin, respectively. The logarithm of worm body residue concentrations followed a linear relationship with the logarithm of soil concentrations, for both phostebupirim and cyfluthrin, with respective regression coefficients of $R^2 = 0.97$ and $R^2 = 0.99$ from soil implemented with the formulation. To better understand the potential for contaminant trophic transfer, the worms were not depurated prior to extraction. At soil concentrations of the formulation equivalent to the ones applied to the field, worm residues were 2866 ng/g dry weight and 18 ng/g dry weight for phostebupirim and cyfluthrin, respectively. These concentrations are well below toxicity benchmarks for birds, with a chronic NOEC around 9 μg/g diet of phostebupirim and reproductive effects observed at 250 μg/g diet of cyfluthrin for birds [3,7]. However, the worm residue concentrations showed a potential for bioaccumulation, especially for phostebupirim, and whole body residues in worms were on average 2690% and 20% of the soil residues (concentrations of worm and soil in ng/g dry wt) for phostebupirim and cyfluthrin, respectively. In this test, worms were exposed to soil for only 14 d, and the linearity of the relationship between body residue and concentrations may suggest potential higher bioaccumulation over longer exposures, as would be the case in field conditions.

CONCLUSION

The present field study showed that the insecticidal formulation of phostebupirim and cyfluthrin following labeled application methods could potentially cause short-term risk to aquatic and terrestrial nontarget organisms, especially to invertebrates. It should be noted, however, that the exposure as measured in the present study did not reflect the bioavailability of the compounds, especially in water, because the total runoff water concentrations were measured without differentiation of particulate, organic matter–bound, or freely dissolved concentrations, whereas the bioassays were performed in water free of particulates and organic matter. Therefore, the actual environmental risk was likely overestimated in the present study, especially for species exposed through diffusion processes, and for cyfluthrin, which is more hydrophobic and more likely to associate with organic matter. Furthermore, although *E. fetida* were shown in the present study to be potentially at risk following application of the formulation, additional studies of other soil nontarget species are necessary to form conclusions on the effects of this formulation on soil invertebrates.

Currently, the degradation rate determinations from terrestrial field dissipation studies are not used in risk assessments conducted by the USEPA during the registration process, although these data provide the best indication of the persistence of parent compounds and metabolites under actual use conditions [33]. The present study showed that exposure models may accurately predict runoff water concentrations under average weather conditions, but soil dissipation rates may be longer and aqueous concentrations may be lower during dryer years. Under dryer than normal weather conditions, therefore, using only estimates of exposure calculated based on laboratory data may overestimate the acute risk to nontarget organisms and may underestimate the chronic risk.

At this stage of the project, conventional and strip tillage provided similar economic returns in term of tillage cost versus benefits from corn grain yields, except in 2012, which had a very particularly dry spring and exceptionally low grain yields all over the Midwest; these conditions increased the price paid to farmers for corn grain and therefore the economic return in conventional tillage because of higher grain yields. Tillage treatments did not significantly affect yields throughout the 3 growing seasons, but this may change after a longer period. Indeed, Ismail et al. [34] observed a change in corn grain yields based on tillage treatments after more than a decade, with conventional tillage outyielding no-tillage during the first 12 yr [34]. Concentrations of, and thus exposure to, insecticides were higher, yet not statistically significantly higher, in strip tillage. Tillage treatments are expected to affect insecticide fate over a time span longer than 3 yr. Indeed, changes in soil parameters take years to occur, and it is these soil parameters (e.g., organic matter content and infiltration rates) that will likely affect insecticide fate and transport through retention, degradation, and bioavailability [35,36]. Therefore, a longer term study is necessary to determine how tillage treatments affect the environmental risk. The current project is ongoing, and longer term results are expected in a few years.

SUPPLEMENTAL DATA

Figures S1–S2. (92 KB DOC).

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