Open-File Report 95-156

COLLECTION OF SHORT PAPERS ON THE BEAVER CREEK WATERSHED STUDY IN WEST TENNESSEE, 1989-94

U.S. DEPARTMENT OF THE INTERIOR
MARCH 3, 1849

Prepared by the
U.S. GEOLOGICAL SURVEY

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TRANSPORT AND DEGRADATION OF ALDICARB IN THE SOIL PROFILE: A COMPARISON OF CONVENTIONAL TILLAGE AND NON-TILLAGE

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ABSTRACT: Paired cotton fields (conventionally tilled and non-tilled) in the Beaver Creek watershed, West Tennessee, were selected to (1) develop a sampling strategy that accurately characterizes the spatial and temporal distribution of aldicarb and its metabolites in the soil profile, (2) evaluate the transport and degradation of aldicarb and its metabolites in the soil profile, and (3) compare the behavior of aldicarb and its metabolites in conventionally tilled and non-tilled cotton fields. The observed concentration distributions support sampling in multiple positions within transects perpendicular to the orientation of the rows. Multiple transects were proven necessary to achieve field representation for a given transect position. This sampling strategy provided suitable data sets for calculating representative and statistically defensible field values. Base-catalyzed and surface-catalyzed hydrolysis appeared to be the most important mechanisms for the disappearance of aldicarb sulfoxide in the soil profile in both tillage systems. Degradation rate constants were 0.046 day^{-1} for the conventionally tilled field and 0.040 day^{-1} for the non-tilled field. The half-lives of aldicarb sulfoxide were 15 days and 16 days for the conventionally tilled and the non-tilled fields, respectively. Horizontal transport was negligible, and vertical transport was minimal for both tillage systems.
KEY TERMS: transport; degradation; soils; aldicarb; aldicarb sulfoxide; aldicarb sulfone.

INTRODUCTION

Non-tillage has been effectively used to reduce soil erosion. However, researchers have suggested that non-tillage can enhance chemical transport and increase the potential for ground-water contamination (Dick and others, 1989; Hall and others, 1989; Isensee and others, 1990). Resource-management agencies in the Mississippi embayment, where the soils are highly susceptible to erosion, have recommended non-tillage as a best management practice (BMP). The risk of ground-water contamination associated with the implementation of non-tillage in this region needs to be addressed before this BMP is widely adopted.

Field studies comparing pesticide transport and degradation in conventionally tilled and non-tilled soils in this region are needed. An important precursor for such studies is the development of sampling strategies that systematically address the spatial and temporal distribution of pesticides within the soil profile. An insufficient characterization of the pesticide distribution can bias the evaluation of pesticide transport and degradation, and subsequently can bias the comparison of conventional tillage and non-tillage results.

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In 1993 the U.S. Geological Survey, in cooperation with the Tennessee Department of Agriculture, initiated a study was initiated to (1) develop a sampling strategy that accurately characterizes the spatial and temporal distribution of aldicarb and its metabolites in the soil profile, (2) evaluate the transport and degradation of aldicarb and its metabolites in the soil profile, and (3) compare the behavior of aldicarb and its metabolites in conventionally tilled and non-tilled cotton fields. This study is part of a comprehensive research and monitoring program to evaluate agricultural non-point source pollution and BMP's in the Beaver Creek watershed, West Tennessee. This report discusses results for the 1993 growing season.

ALDICARB DEGRADATION

Aldicarb is a widely used insecticide and nematicide for cotton crops in the Beaver Creek watershed. Granular aldicarb is typically integrated into the soil during planting. The 0.3 meter (m) wide application band is centered along the row tops. Application rates range from 0.6 to 0.9 kilogram per hectare (kg/ha) of the active ingredient. Upon contact with soil moisture, aldicarb is released. Plant uptake is generally less than 5 percent of that applied (Smith and Parrish, 1993). Transport in surface runoff is less than 10 percent of that applied.

Because of its high water solubility (5,730 milligrams per liter), aldicarb is considered highly mobile (Seiber, 1987). However, aldicarb mobilization can be retarded by adsorption into organic phases in the soil profile. Rao and others (1986) reported that in sandy soils, aldicarb sorption is controlled by the organic carbon content. The molecular structure of aldicarb implies that the pesticide has potential for binding ionically to clays. However, the sorption mechanism and sorption coefficient are unknown.

Laboratory experiments conducted by Lightfoot and others (1987) show that microbial oxidation is the most important degradation mechanism for aldicarb in soils. Half-lives of 1 to 2 days were reported in these experiments. The first product of aldicarb oxidation is aldicarb sulfoxide, which can further oxidize to form aldicarb sulfone. Aldicarb, aldicarb sulfoxide, and aldicarb sulfone concurrently hydrolyze to form non-carbamate oximes and nitriles.

Hydrolysis is the most important degradation mechanism for aldicarb sulfoxide and aldicarb sulfone in soils (Lightfoot and others, 1987; Hansen and Speigel, 1983; Lemeley and Zhong, 1984). Data presented by Lightfoot and others (1987) show that the hydrolysis rates for aldicarb sulfoxide and aldicarb sulfone follow first-order kinetics and are pH and temperature dependent. Lightfoot and others (1987) also reported half-lives for these metabolites in experimental soil microcosms ranging from 10 days to greater than 2,000 days, depending upon the pH and temperature.

Aldicarb and its metabolites are relatively toxic (Fukuto, 1987). Their primary mechanism of action is to inactivate the acetylcholinesterase enzyme (Fukuto, 1987; Metcalf, 1971). Rat oral LD_{50} values are 1 milligram per kilogram (mg/kg) for aldicarb and aldicarb sulfoxide, and 24 mg/kg for aldicarb sulfone (Lightfoot and others, 1987). Non-carbamate metabolites of aldicarb are of little environmental concern because their toxicities are relatively low and they continue to degrade to form aldehydes, acids, and alcohols.

PROTOCOLS AND METHODS

Two adjacent cotton fields in the Beaver Creek watershed were selected for this study (See Figure 1). The fields are located in Shelby County, near the Collierville-Arlington Road. These fields are of similar size, slope, and soil type. The south field was cultivated by conventional tillage (the top 0.3 meter was disked before planting). The north field was cultivated by non-tillage (crop residues from the previous growing season were not disturbed during planting). The nominal rate of application was 0.84 kg of aldicarb (active ingredient) per hectare. Agricultural limestone was added to these fields as recommended by the University of Tennessee, Agricultural Extension Service (John C. Wilson, farmer, oral commun., 1993).

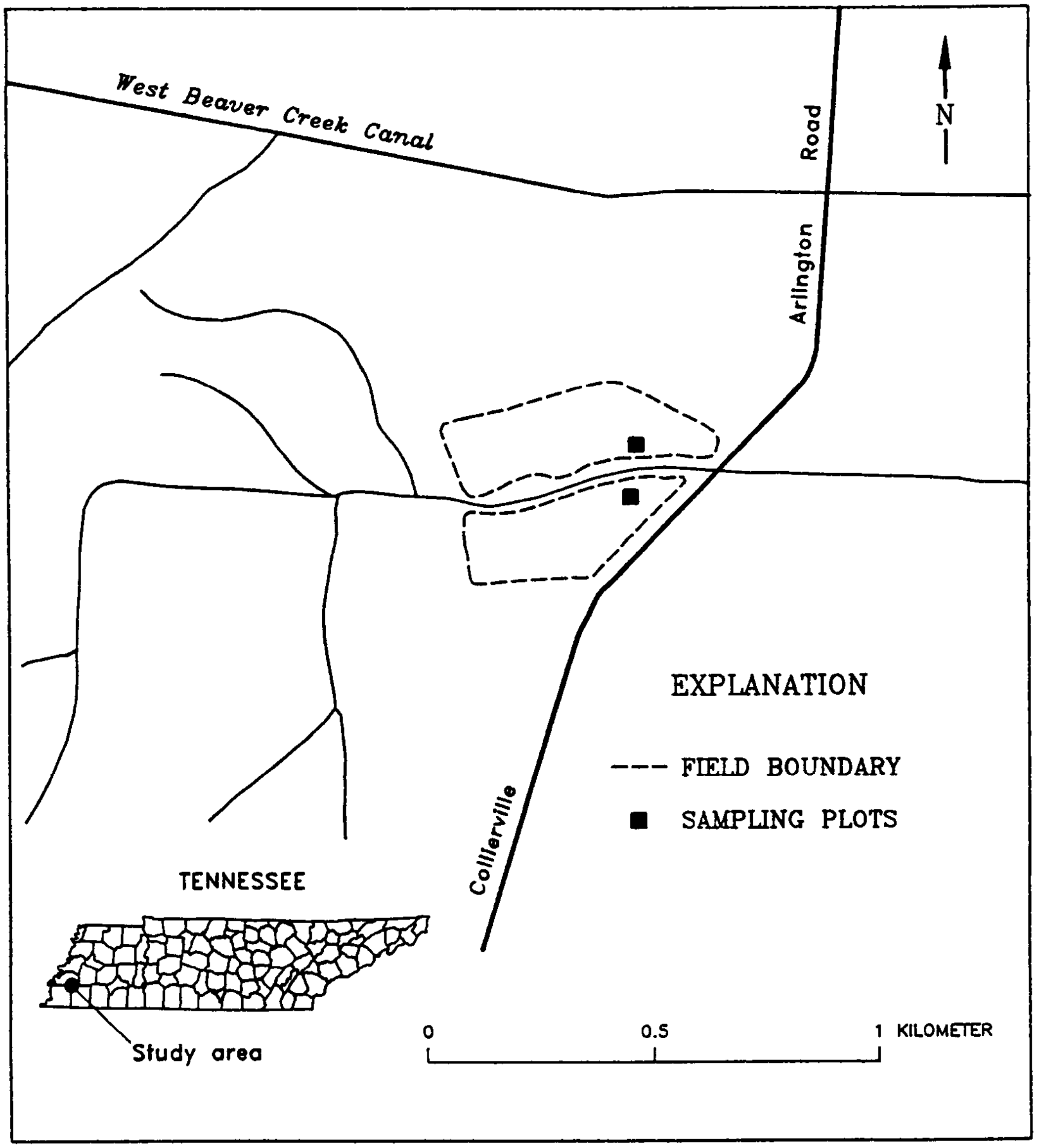

Figure 1. Location of study area and sampling plots.

Soils in these fields were classified in 1966 by the U.S. Department of Agriculture, Soil Conservation Service, as Falaya silt loam. Soil pH ranges from 5.1 to 5.5. However, these soils are usually limed to bring the pH to about 6.5. The water table is often within a foot of the land surface in winter and spring, and declines several feet below the surface in the summer and fall (U.S. Department of Agriculture, Soil Conservation Service, 1989).

Sampling Protocols

A 30- by 30-m sampling plot was delineated in each field. All sampling in this study was conducted within the plot boundaries. Prior to pesticide application, soil samples were taken from these plots to verify the absence of aldicarb and its metabolites. Aldicarb was applied to both fields on May 18, 1993. Soil samples were collected from the conventionally tilled field at 9, 22, 52, 84, 115, and 149 days after the application of aldicarb and from the non-tilled field at 3, 21, 43, 70, 113, and 148 days after the application date. For each sampling date, three transects were randomly established from row top to adjacent row top (see Figure 2). Five equidistant coring positions were sampled in each transect. Soil cores were taken at each position in 0.15-m intervals from the land surface to a maximum depth of 0.90 m. Soil cores were collected using a 0.054-m diameter stainless-steel auger. The auger was cleaned with deionized water between sampling depths to avoid contamination from one depth interval to the next. For the same reason, the top 3 centimeters of each soil core (except for the uppermost interval) were discarded.

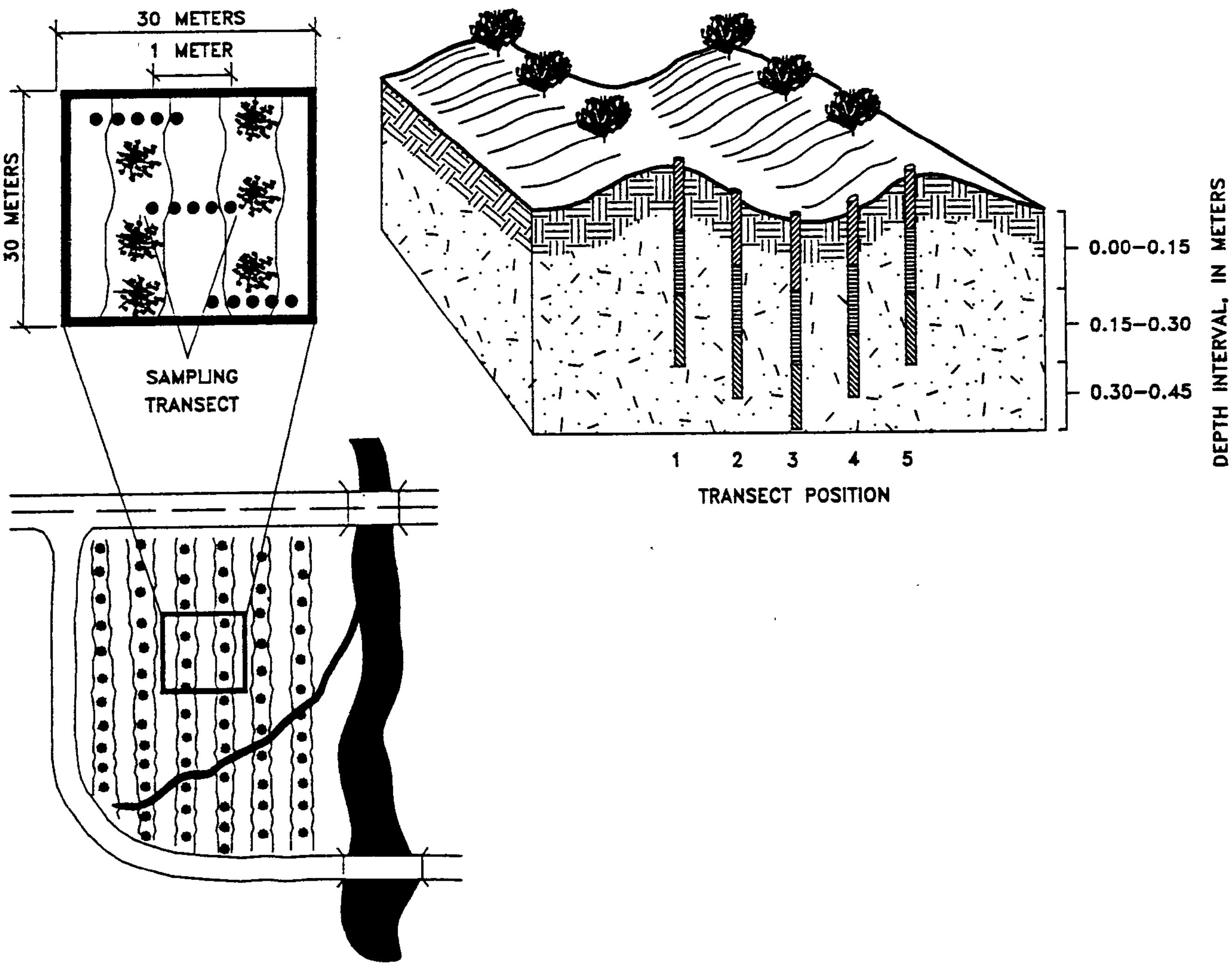

Figure 2. Schematic diagram of soil-sampling strategy.

Each soil core (0.15 m long) was transferred to a large sealable plastic bag and homogenized. Half of each soil sample was transferred to a second plastic bag, packed with ice, and sent by overnight carrier to The Institute of Wildlife and Environmental Toxicology (TIWET), Clemson University, for analysis of aldicarb, aldicarb sulfoxide, and aldicarb sulfone.

Analytical Methods

The samples were analyzed using TIWET standard operational procedure #401-53-01. To summarize the procedure, aldicarb residues for each sample were isolated by shaking together 0.050 kg of homogenized soil and 0.100 kg deionized water in a sealed polyethylene bottle for 1 hour, centrifuging the slurry, and extracting the supernatant with methylene chloride. The methylene chloride fractions were then rotary evaporated to dryness. Aldicarb residues for each sample were resuspended in 2.0 mL of acetonitrile and analyzed with a Hewlett-Packard 1090 Liquid Chromatograph (any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government) with a fluorescence detector.

Ten percent of the samples were run in triplicate. Analyses were accepted only if the coefficients of variation were less than 5 percent. Laboratory blanks representing 5 percent of the total number of samples were also analyzed.

RESULTS AND DISCUSSION

Most of the results and discussion presented in this paper are centered on the transport and degradation of aldicarb sulfoxide. Aldicarb was not detected after the first sampling date in each field, and the occurrence and distribution of aldicarb sulfone was similar to that of aldicarb sulfoxide.

Evaluation of Sampling Protocols

In general, the occurrence of aldicarb sulfoxide was limited to the row tops (transect positions 1 and 5) where the pesticide was applied (see Figure 2). Aldicarb sulfoxide concentrations in the slopes (positions 2 and 4) and furrows (position 3) were low or below the detection limit of 1.5 micrograms per kilogram. This distribution was observed in all transects, at all depth intervals, for all sampling dates, and in both fields (see Figure 3).

For a given sampling date, the variances in the aldicarb sulfoxide concentrations at the row-top position were relatively high among and within transects (see Table 1). Therefore, a larger number of samples are needed to obtain representative concentrations at this position than at the slope and furrow positions where concentration variances were smaller.

The aldicarb sulfoxide concentration distribution supports sampling in multiple positions within transects perpendicular to the orientation of the rows. Because the concentrations for a given transect position varied widely among transects, multiple transects proved necessary to achieve field representation for that particular position. Considerable spatial variability in the pesticide residue concentrations in soil samples has been reported in many field studies (Smith and others, 1987; Jones and others, 1988; Hornsby and others, 1990). Concentration medians were assumed to provide representative and statistically defensible field values for a given transect position. Medians were selected over other measurements of central tendency because they are less biased by extreme values and, therefore, better suited to describe data sets with large variances. Concentration medians were used to characterize the spatial and temporal distribution of aldicarb sulfoxide in the two fields.

Transport

Aldicarb sulfoxide concentration distributions were used to determine the vertical and horizontal transport in the soil profile. Because aldicarb was applied at the row top position at about 0.07 m below land surface, occurrence of aldicarb or its metabolites in other transect positions or depths would constitute horizontal or vertical transport. The non-occurrence of aldicarb sulfoxide in the slope and furrow positions of both fields indicates that horizontal transport is negligible. The absence of horizontal mobilization for aldicarb and its metabolites was also observed by Smith and Parrish (1993) in peanut fields in sandy loam soils.

Aldicarb sulfoxide concentrations peaked in the uppermost interval, 0 to 0.15 m, throughout the sampling period (See Figure 4 and Table 1). Concentrations in the uppermost interval of the row-top position accounted for about 85 percent of the total aldicarb sulfoxide in the soil profile. No occurrence of aldicarb sulfoxide was

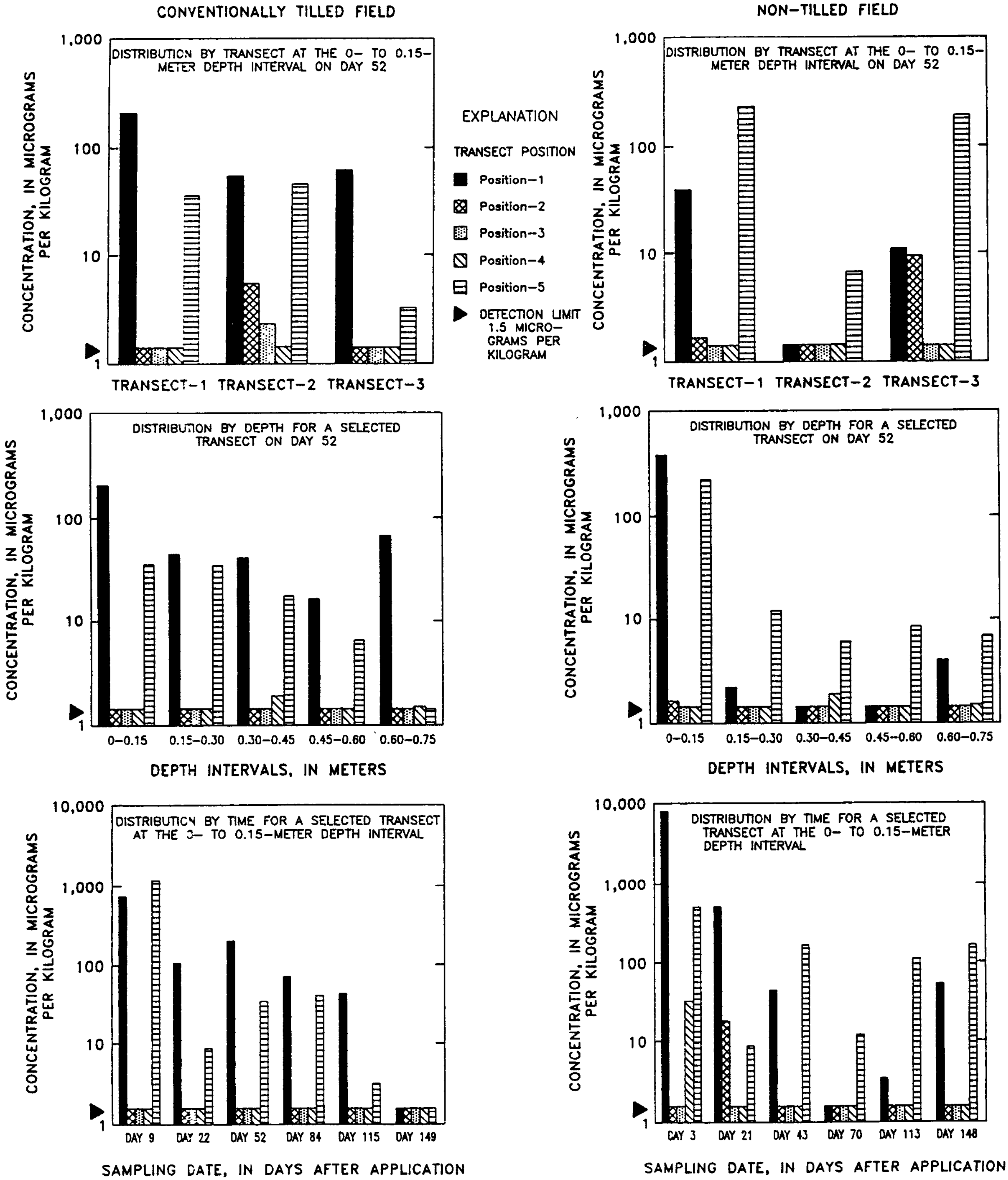

Figure 3. Typical aldicarb sulfoxide distribution for the conventionally tilled and non-tilled fields.

Table 1.--Descriptive statistics for aldicarb sulfoxide in the conventionally tilled and non-tilled fields

[m, meters; n, number of observations; µg/kg, micrograms per kilogram; (µg/kg)², micrograms per kilogram squared; <, less than; --, no data are available]

Conventionally tilled	Day 9		Day 22		Day 52		Day 84		Day 115		Day 149	
	Median (µg/kg)	Variance (µg/kg)²	Median (µg/kg)	Variance (µg/kg)²	Median (µg/kg)	Variance (µg/kg)²	Median (µg/kg)	Variance (µg/kg)²	Median (µg/kg)	Variance (µg/kg)²	Median (µg/kg)	Variance (µg/kg)²
0 to 0.15 m												
Row top, n=6	1,005	87,616	9.1	1,901	50.5	5,213	16.5	759	3.9	320	<1.5	0.0
Slope, n=6	<1.5	5.4	<1.5	0.0	<1.5	2.8	<1.5	0.0	<1.5	0.0	<1.5	0.0
Furrow, n=3	6.6	23.1	<1.5	1,030	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
0.15 to 0.30 m												
Row top, n=6	25.5	3,003	2.2	610	2.8	1,089	2.4	2.3	<1.5	0.5	<1.5	0.0
Slope, n=6	<1.5	7.5	<1.5	0.4	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
Furrow, n=3	3.4	2.2	<1.5	0.0	<1.5	0.3	<1.5	0.0	<1.5	0.0	<1.5	0.0
0.30 to 0.45 m												
Row top, n=6	79.0	5,432	2.0	1,544	30.0	1,183	2.0	0.7	<1.5	0.1	<1.5	0.0
Slope, n=6	<1.5	64.3	<1.5	0.0	<1.5	0.5	<1.5	0.0	<1.5	0.0	<1.5	0.0
Furrow, n=3	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
0.45 to 0.60 m												
Row top, n=6	--	--	--	--	<1.5	396	<1.5	0.0	<1.5	0.0	<1.5	0.0
Slope, n=6	--	--	--	--	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
Furrow, n=3	--	--	--	--	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
0.60 to 0.75 m												
Row top, n=6	--	--	--	--	7.9	681	<1.5	0.0	<1.5	0.0	<1.5	0.0
Slope, n=6	--	--	--	--	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
Furrow, n=3	--	--	--	--	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
0.75 to 0.90 m												
Row top, n=6	--	--	--	--	--	--	--	--	<1.5	0.0	<1.5	0.0
Slope, n=6	--	--	--	--	--	--	--	--	<1.5	7.0	<1.5	0.0
Furrow, n=3	--	--	--	--	--	--	--	--	<1.5	0.0	<1.5	0.0

Table 1.--Descriptive statistics for aldicarb sulfoxide in the conventionally tilled and non-tilled fields--Continued

Non-tilled	Day 3		Day 21		Day 43		Day 70		Day 113		Day 148	
	Median (μg/kg)	Variance (μg/kg)²	Median (μg/kg)	Variance (μg/kg)²	Median (μg/kg)	Variance (μg/kg)²	Median (μg/kg)	Variance (μg/kg)²	Median (μg/kg)	Variance (μg/kg)²	Median (μg/kg)	Variance (μg/kg)²
0 to 0.15 m												
Row top, n=6	260	67,600	90.5	4,816	9.7	2,272	24.5	10,201	2.0	180	30.0	615
Slope, n=6	4.7	22.6	<1.5	19.5	<1.5	1.1	<1.5	9.5	<1.5	0.0	<1.5	0.0
Furrow, n=3	<1.5	0.0	<1.5	3.2	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
0.15 to 0.30 m												
Row top, n=6	15.0	449	1.6	2.2	2.5	166	1.8	18.1	<1.5	9.1	3.6	8.5
Slope, n=6	<1.5	0.1	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
Furrow, n=3	<1.5	6.8	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
0.30 to 0.45 m												
Row top, n=6	5.1	32.9	<1.5	172	<1.5	106	<1.5	5.3	<1.5	1.5	<1.5	1.3
Slope, n=6	<1.5	0.0	<1.5	15.4	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
Furrow, n=3	<1.5	0.7	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
0.45 to 0.60 m												
Row top, n=6	--	--	--	--	<1.5	9.4	<1.5	237	<1.5	0.0	<1.5	0.1
Slope, n=6	--	--	--	--	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
Furrow, n=3	--	--	--	--	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
0.60 to 0.75 m												
Row top, n=6	--	--	--	--	<1.5	0.0	1.8	4.8	<1.5	0.0	<1.5	0.0
Slope, n=6	--	--	--	--	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
Furrow, n=3	--	--	--	--	<1.5	0.0	<1.5	0.0	<1.5	0.0	<1.5	0.0
0.75 to 0.90 m												
Row top, n=6	--	--	--	--	--	--	<1.5	0.6	<1.5	0.0	<1.5	0.0
Slope, n=6	--	--	--	--	--	--	<1.5	0.0	<1.5	0.0	<1.5	0.0
Furrow, n=3	--	--	--	--	--	--	<1.5	0.0	<1.5	0.0	<1.5	0.0

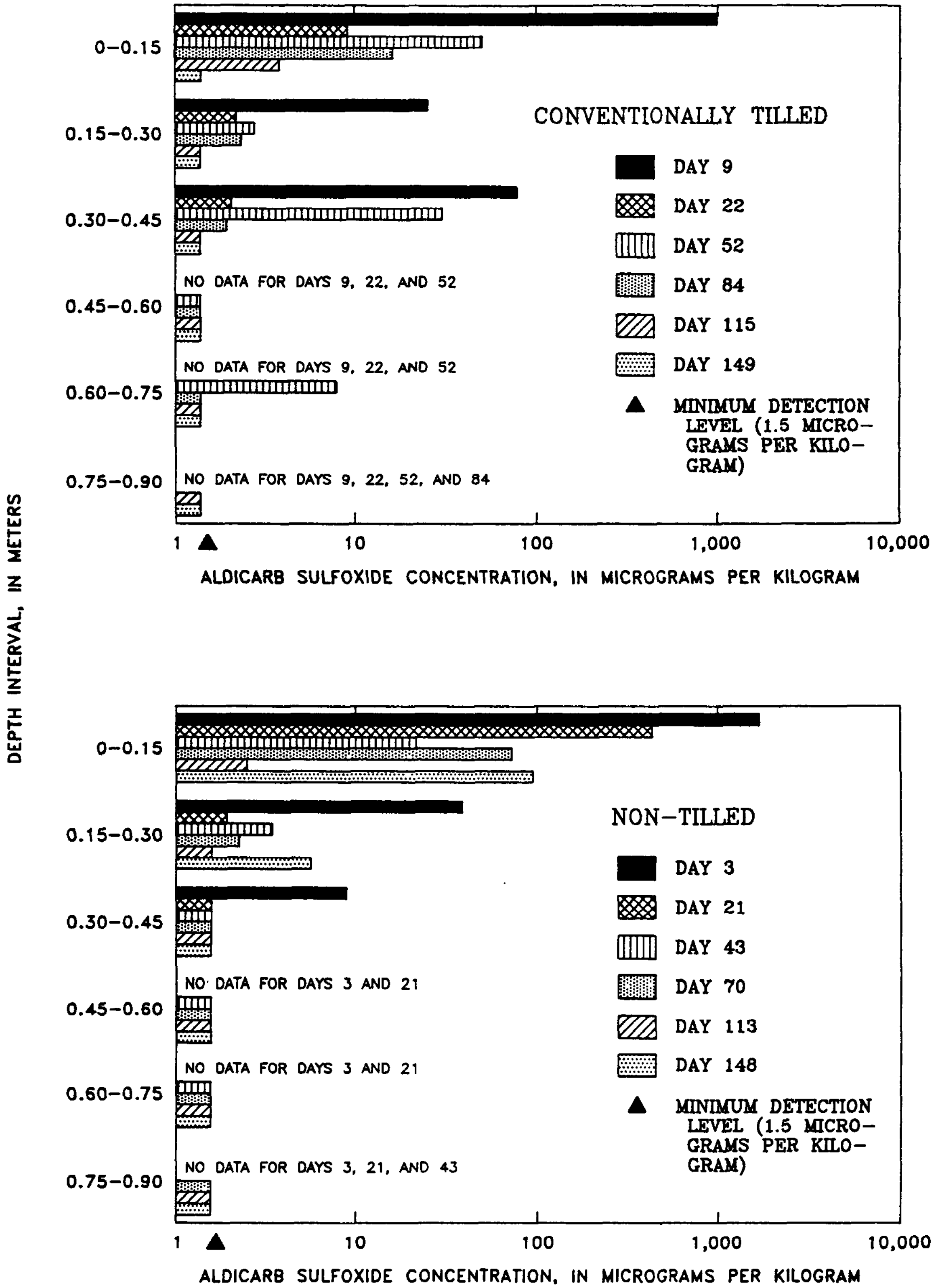

Figure 4. Median aldicarb sulfoxide for row-top position.

observed below the 0.60- to 0.75-m depth interval in either field. This indicates that the vertical transport in the row-top position was minimal. However, it was not determined if aldicarb sulfoxide had migrated below the 0.30- to 0.45-m depth interval early in the growing season. Soil samples below this depth interval were not collected until day 53 and day 43 in the conventionally tilled and the non-tilled fields, respectively.

Degradation

The median aldicarb sulfoxide concentrations for the 0 to 0.15-m depth interval were observed to decrease exponentially with time (see Figure 5). Correlation coefficients for the natural logs of the median aldicarb sulfoxide concentrations and time since application were -0.80 and -0.61 for the conventionally tilled and the non-tilled fields, respectively (See Figure 5). An analysis of the residuals showed that the data points for day 22 in the conventionally tilled field and for day 148 in the non-tilled field are outliers, therefore, respective adjusted correlation coefficients were -0.99 and -0.92. These correlation coefficients demonstrate that the disappearance of aldicarb sulfoxide at the 0 to 0.15-m depth interval follows a first-order kinetics reaction.

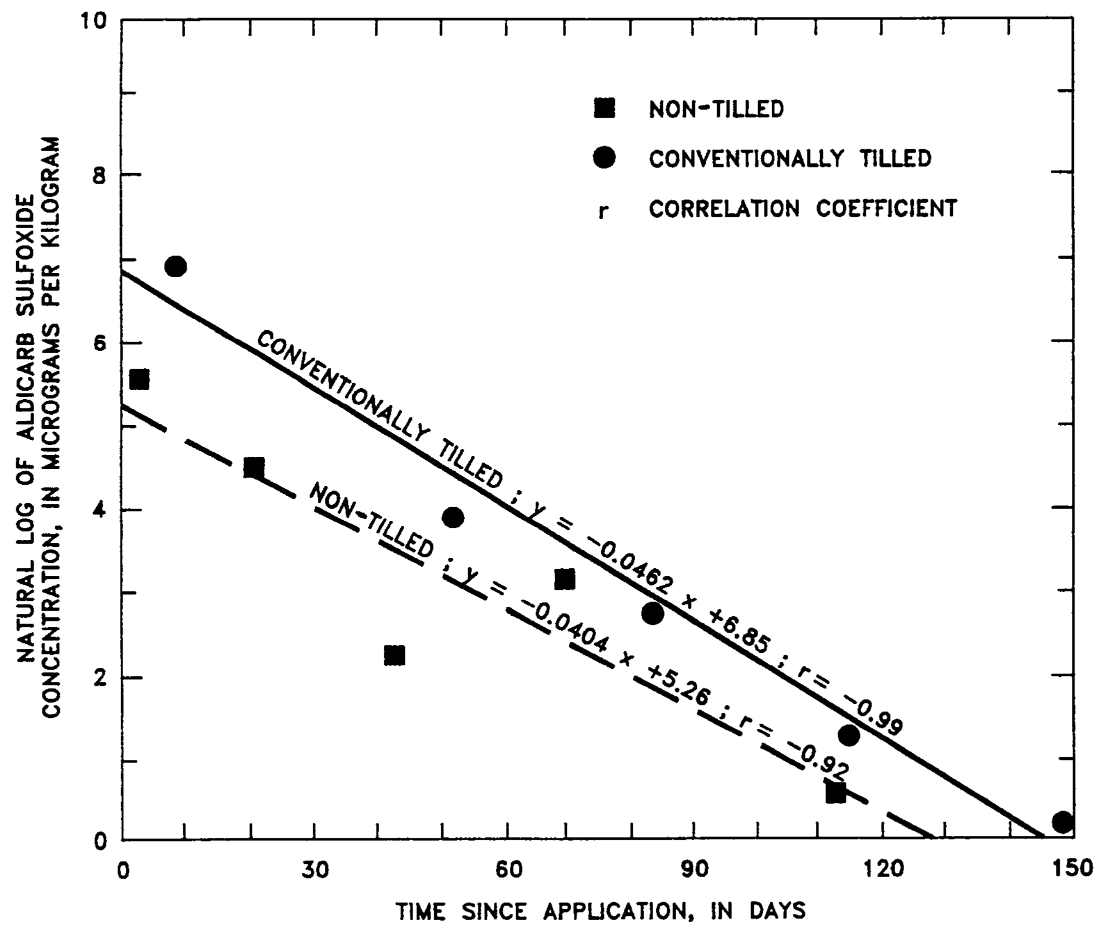

Figure 5. Natural log of the median aldicarb sulfoxide concentration for soil samples at the 0- to 0.15-meter depth interval within the band of application as a function of time since application.

The degradation rate constant, k, is the negative of the slope of the line generated by the correlation of the natural log of aldicarb sulfoxide concentration and the time since application (see Figure 5) (Castellan, 1983). Degradation rate constants were 0.046 day^{-1} for the conventionally tilled field and 0.040 day^{-1} for the non-tilled field. The half-lives of aldicarb sulfoxide calculated from these rate constants were 15 and 16 days for the conventionally tilled and the non-tilled fields, respectively.

These half-lives are about seven times shorter than the half-lives computed from the base-catalyzed distilled water hydrolysis equation presented by Lightfoot and others (1987) for a pH of 6.5 and a temperature of 303 kelvin. Surface-catalyzed hydrolysis can explain the shorter half-lives computed from our field data. Bank and Tyrrell (1984), Bromilow and others (1986), and Lightfoot and others (1987) have shown that distilled water hydrolysis rates are considerably lower than those measured where a solid phase is present.

Smith and Parrish (1993) reported half-lives of 13 to 20 days for total aldicarb residues in a sandy loam soil. In a study conducted in sandy loam soils in the Western United States, half-lives of 15 to 65 days for total aldicarb residues were reported (Jones, 1987). Hornsby and others (1990) showed that the half-life of aldicarb metabolites taken collectively averaged about 69 days for a sandy soil in Florida. In general, the half-lives reported in these studies were somewhat higher than those computed in this study. However, the effects of surface-catalyzed hydrolysis on aldicarb sulfoxide degradation are somewhat greater in the clay-rich Falaya silt loam soils of the sampling plots in this study.

CONCLUSIONS

The aldicarb sulfoxide concentration distributions support sampling in multiple positions within transects perpendicular to the orientation of the rows. Multiple transects were proven necessary to achieve field representation for a given transect position. This sampling strategy provided suitable data sets for calculating representative and statistically defensible field values.

Base-catalyzed and surface-catalyzed hydrolysis appeared to be the most important mechanisms for the disappearance of aldicarb sulfoxide in the soil profile in both tillage systems. Degradation rate constants were 0.046 day^{-1} for the conventionally tilled field and 0.040 day^{-1} for the non-tilled field. The half-lives of aldicarb sulfoxide were 15 and 16 days for the conventionally tilled and the non-tilled field, respectively. Horizontal transport was negligible, and vertical transport was minimal for both tillage systems.

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RELATION BETWEEN NITRATE CONCENTRATIONS AND POTENTIAL NITRATE SOURCES FOR WATER-TABLE AQUIFERS IN THE BEAVER CREEK WATERSHED, WEST TENNESSEE: A STATISTICAL ANALYSIS

Shannon Williams[1] and Angel Roman-Mas[2]

ABSTRACT: Non-parametric statistical methods were used to examine the relation between nitrate concentrations and potential sources of nitrate for the water-table aquifers in the Beaver Creek watershed. Water-quality data collected in 1992 from 81 domestic wells were used for these analyses. The data showed that nitrate concentrations (measured as nitrate) exceeded 13 milligrams per liter in 16 percent of the water samples analyzed. The Spearman Rank correlation analysis indicated that nitrate concentrations are well-depth dependent. The Kruskal-Wallis test indicated that the variance in nitrate concentration can be explained in terms of selected potential nitrate-source categories. The Student-Newman-Keuls multiple comparison of the means test showed that: (1) the mean nitrate concentration for wells in the deeper than 45-meter category was lower and statistically different than the mean concentrations for the other categories, (2) the mean nitrate concentrations for categories that include septic tanks were higher and statistically different than the mean concentration for the crop category, and (3) the mean nitrate concentration for the confined animals category was higher and statistically different than the mean concentration for the crop category.
KEY TERMS: Water-table aquifers; nitrate; rural areas; statistical analyses.

INTRODUCTION

The increasing incidence of ground-water contamination has resulted in a need for: (1) ground-water quality and land-use data and (2) analytical tools to establish relations between anthropogenic activities and ground-water quality. Row-crop agriculture has been recognized as a potential non-point source of ground-water contamination. Water draining cropland can contribute large amounts of nitrate and other contaminants to receiving aquifers, thus impairing their suitability for designated uses. However, row crops are not grown isolated from other nitrate sources, such as septic tanks and facilities with confined animals. Accordingly, analytical tools must be able to differentiate the various nutrient sources.

Parametric and non-parametric statistical analyses are commonly used to relate anthropogenic activities to ground-water quality conditions. Although statistical analyses can provide significant insight, investigators should exercise caution in the selection of the proper statistical methods. Objectives of the evaluation and the statistical characteristics of the water-quality data and other relevant

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variables, such as well depth and distance-to-source, also should be considered when selecting appropriate statistical methods. In performing the statistical analyses, it is assumed that each well or spring sampled and the associated water-quality data can be categorized according to potential sources of a contaminant. The categorization process should be consistent with the physiographic and hydrogeologic setting of the area.

Agriculture is vital to the economy of the Beaver Creek watershed in West Tennessee. About 70 percent of the watershed is under crop production that consists mainly of soybean, cotton, and corn. As in many rural areas in the United States, residents in the Beaver Creek watershed rely on domestic wells as their primary source of water. The residents also share a growing concern that agricultural activities could be affecting the quality of the ground water in the watershed.

In the summer of 1992, a water-quality reconnaissance of the water-table aquifers in Shelby, Tipton, Fayette, and Haywood Counties, Tennessee, was conducted by the U.S. Geological Survey, in cooperation with the Tennessee Department of Agriculture, and the University of Tennessee Agricultural Extension Service. The reconnaissance included the collection of water samples from 81 domestic wells completed in the water-table aquifers in the Beaver Creek watershed (Figure 1) (Fielder and others, 1994).

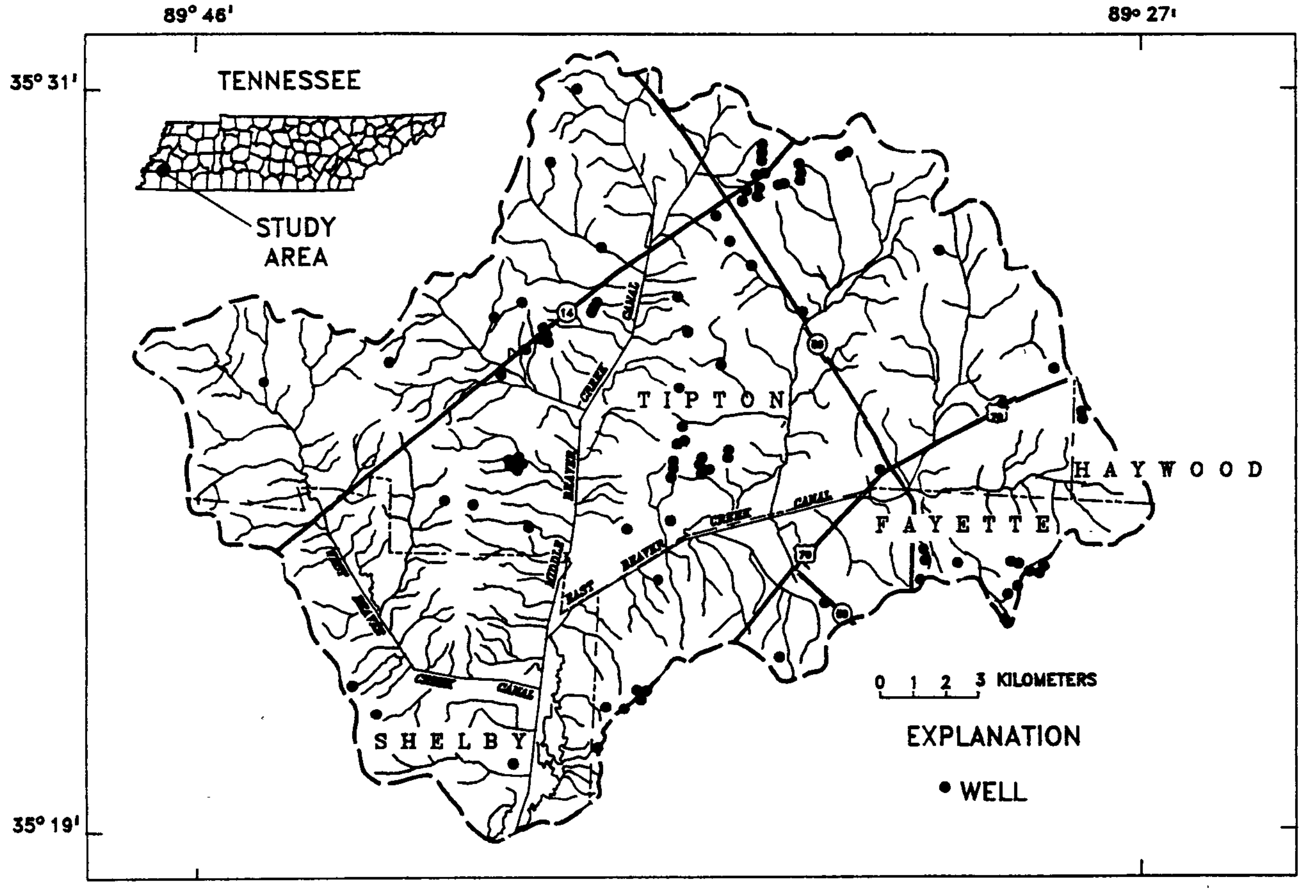

Figure 1. Location of Beaver Creek study area and sampled wells.

Nitrate concentrations (measured as nitrate) exceeding 13 mg/L (milligrams per liter) were observed in 16 percent of the water samples for wells in the Beaver Creek watershed. Concentrations above 13 mg/L can indicate influence of anthropogenic activities (Madison and Brunett, 1985). Water samples from two of the wells exceeded the primary drinking water standard for nitrate (measured as nitrate) of 45 mg/L adopted by the Tennessee Department of Environment and Conservation (1993). Ingestion of water with high nitrate content has been found to cause methemoglobinemia in infants (Klaassen and Rozman, 1991). Statistical analyses are used in this paper to relate nitrate concentrations to potential nitrate sources in the Beaver Creek watershed.

HYDROGEOLOGY

Alluvium and fluvial deposits of Quaternary and Tertiary(?) ages make up the shallow aquifers in the Beaver Creek watershed (Parks and Carmichael, 1990a, 1990b; Parks, 1990). Alluvium, consisting of sand, gravel, silt, and clay, is present beneath the bottomland areas. Fluvial deposits, consisting of sand, gravel, minor clay, and ferruginous sandstone, are present beneath the loess in upland and valley slope areas. Locally, the alluvium and fluvial deposits are underlain by the Cockfield Formation, Cook Mountain Formation, or the Memphis Sand (Parks and Carmichael, 1990a, 1990b). The Cockfield Formation and Memphis Sand consist of sand, silt, clay, and lignite and make up the Cockfield and Memphis aquifers. The Cook Mountain Formation consists primarily of clay and is the confining unit overlying the Memphis aquifer. Hydrogeologic information for the deeper wells sampled during this investigation was insufficient to determine whether or not some of these wells are screened in the Cockfield or Memphis aquifers.

METHODS

Protocols for well selection and for sample collection and analysis have been described by Fielder and others (1994). A detailed land-use inventory was conducted for each well included in the sampling. Land use near each well site was described and distances and topographic gradients between wells and potential nitrate sources were recorded. Well construction data were provided by well owners. Well records from the Tennessee Department of Environment and Conservation (TDEC) were used to verify information obtained from well owners.

The selection of statistical methods used in this paper follows the discussion by Harris and others (1987) and Montgomery and others (1987). All statistical analyses were performed using CoStat statistical software (CoHort, 1990) (Any use of trade, product, or firm names in this article is for descriptive purposes only and does not imply endorsement by the U.S. Government).

RESULTS AND DISCUSSION

Well Depth and Distance-to-Source

Researchers have shown that the nitrate concentration in ground water is a function of well depth and distance-to-source (Madison and Brunett, 1985; Murphy, 1992). Correlation analyses can be conducted to assess the concentration/well depth and the concentration/distance-to-source dependencies. The well depth and distance-to-source at which the water quality of the well is no longer affected by a particular source can be determined from these analyses. These outer limits should be integrated into the classification process. If nitrate concentrations are dependent on well depth and distance-to-source, then depths and distances among potential nitrate-source categories cannot be statistically different for a comparison among categories to be valid.

Well depth and nitrate concentration data were not normally distributed; they were skewed to the right (positive skewness). Accordingly, a non-parametric correlation test (Spearman Rank) was used. Nitrate concentrations were observed to decrease with depth and the Spearman Rank coefficient was -0.55 (Figure 2). This analysis indicates that nitrate concentrations are well-depth dependent; therefore, well depth must be considered when evaluating the relation between potential nitrate sources and nitrate concentrations.

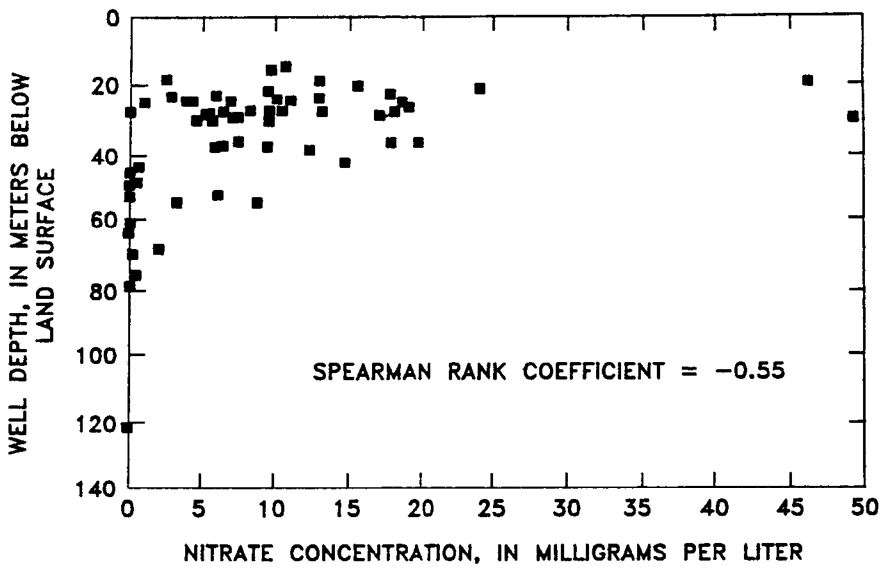

Figure 2. Relation between nitrate-as-nitrate concentration and depth of sampled wells.

The y-intercept resulting from the correlation between well depth and nitrate concentration (45 m) was used to segregate the data into two categories (less than 45 m and greater than 45 m in depth). The segregated data were subjected to a series of statistical analyses to prove or disprove the null hypothesis: the variance in nitrate concentration between the two well-depth categories is random. Nitrate concentrations for both well-depth categories also showed positive skewness. Therefore, the Mann-Whitney test (non-parametric) was utilized to test the null hypothesis. The test indicated that the variation in the nitrate concentration has a low probability (10^{-6}) of being due to random variation (Figure 3). Therefore, the variance in the nitrate concentration can be explained in terms of the selected well-depth categories. These analyses indicate that nitrate concentration for wells deeper than 45 m are not affected by anthropogenic activities.

Distance-to-source (nearest source) and nitrate-concentration data were also positively skewed. Because these data were not normally distributed, the Spearman Rank non-parametric correlation test was used. A significant relation between the two variables was not observed; the Spearman Rank coefficient was -0.21 (Figure 4).

Nitrate-Source Categories

In order to apply statistical analyses to relate the nitrate concentrations to potential nitrate sources, each well sampled was classified into one potential nitrate-source category. Each category included one or more nitrate sources. Only those sources that were relevant to the individual well were considered. The

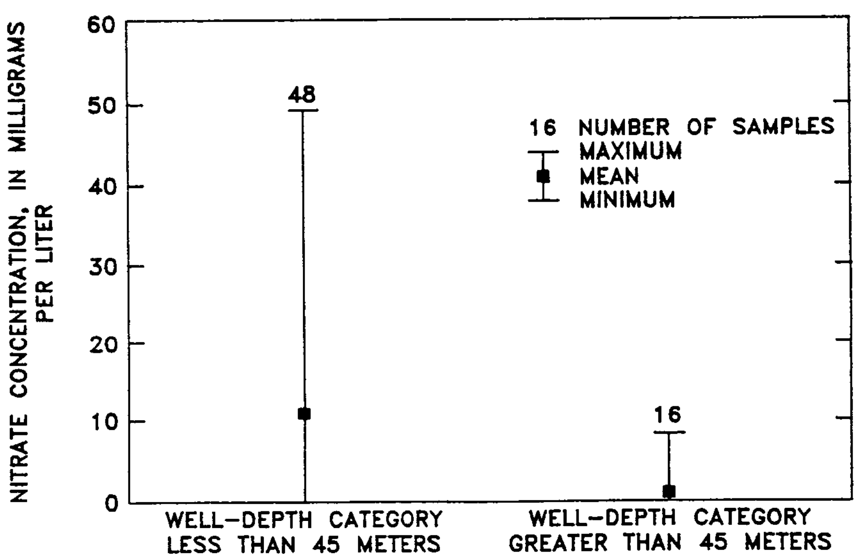

Figure 3. Summary of nitrate-as-nitrate concentration data by well-depth category.

hydrogeologic characteristics of the aquifer and well hydraulics were used for determining which were the potential contributing sources. However, the available hydrogeologic and hydraulic data were somewhat limited.

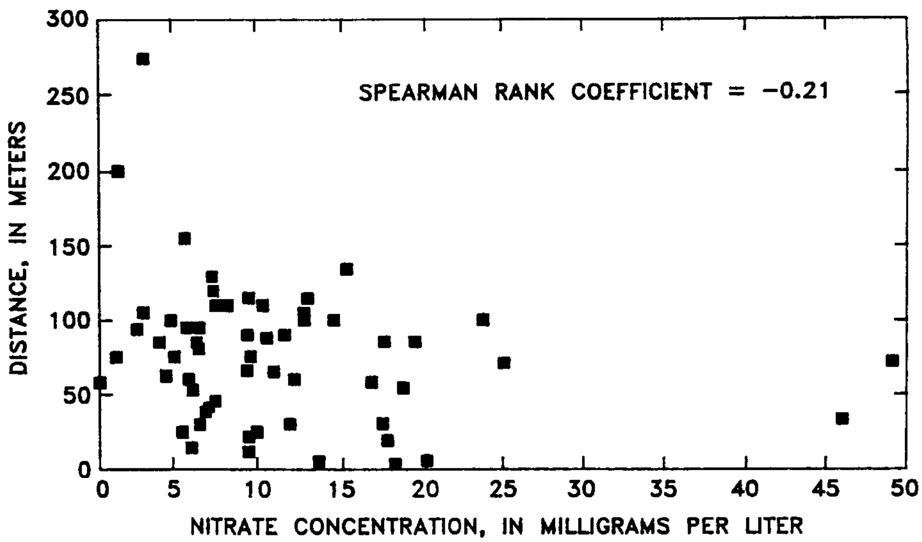

Figure 4. Relation between nitrate-as-nitrate concentration and distance to nearest potential nitrate source for sampled wells.

Distances and topographic gradients were used to classify the sampled wells into potential nitrate-source categories. The direction of ground-water flow in the shallow aquifers was assumed to generally correspond to topographic gradients. Accordingly, all potential nitrate sources upgradient were considered in the classification. The sampled wells are low-yield domestic wells that are pumped intermittently to holding tanks. Pumping periods are usually less than 5 minutes and pumping rates are less than 38 liters per minute. The well radius of influence is assumed to be relatively small. Therefore, potential nitrate sources more than 15 m downgradient were not considered in the classification.

Potential nitrate sources considered in the classification were: (1) cropland (fertilizer application), (2) septic tank systems, and (3) confined animal facilities. These sources resulted in six combinations of categories (Table 1). The wells deeper than 45 m were considered as an individual category. Nine of the wells sampled did not fall into any of the selected categories and were not used for further analyses.

TABLE 1. Classification of wells by potential nitrate sources

Potential Nitrate Sources	No. of Wells
Septic tanks only (ST)	12
Septic tanks and crops (ST/CR)	26
Septic tanks and confined animals (ST/CA)	9
Crops only (CR)	3
Confined animals only (CA)	3
Crops and confined animals (CR/CA)	3
Wells more than 45 meters deep (TD)	16
Wells that did not fit into any of the above categories	9
TOTAL	81

Statistical Analyses

In order to evaluate the relative contribution of potential nitrate sources to nitrate concentrations, the variance in the nitrate concentrations should be explained in terms of the selected potential nitrate-source categories. The Kruskal-Wallis test (non-parametric) was used for this evaluation. Skewness tests indicated that nitrate concentration data for several of the categories were not normally distributed; therefore, parametric tests could not be used. The Kruskal-Wallis test determines only the probability at which the variance for a particular variable (nitrate concentration) is due to random variation (null hypothesis). If the null hypothesis is rejected, the selected data treatments (potential nitrate-source categories) can be a plausible explanation for the observed variance. Results of the Kruskal-Wallis test indicated that the variance in the nitrate concentration has a low probability (10^{-6}) of being due to random variation. Therefore, the variance in nitrate concentration is explained in terms of the selected potential nitrate-source categories.

Because more than two categories were considered, the analysis of the variances was followed by a multiple comparison of the means. The Student-Newman-Keuls multiple comparison test (non-parametric) was used to rank the nitrate-concentration means and to determine which means were statistically different at a 0.05 level of significance (Figure 5). The tests showed that: (1) the mean concentration for wells classified as deeper than 45 m was lower and statistically different than the means for the other six categories, (2) the means for categories that include septic tanks were higher and statistically different than the mean for the crop category, and (3) the mean for the confined animals category was higher and statistically different than the mean for the crop category.

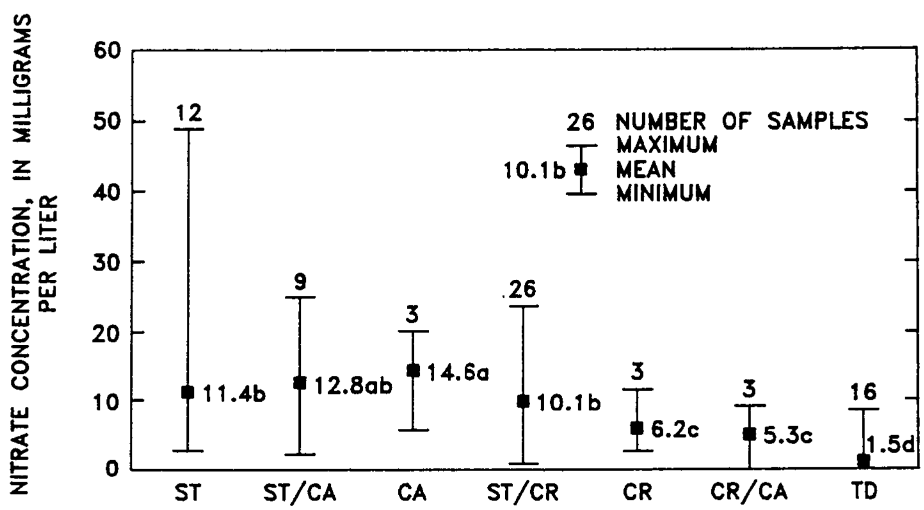

Figure 5. Summary of nitrate-as-nitrate concentration data for selected potential nitrate-source categories (letter following mean value represents populations that are not statistically different at the 0.05 probability as determined by the Student-Newman-Keuls multiple com-parison test. ST, septic tank; CA, confined animals; CR, crops; TD, wells deeper than 45 meters).

The test of the variance and the multiple comparison of the means test assumed that the nitrate data were stationary in time and space. These assumptions were supported by: (1) the relatively short time period (July-August) in which the samples were collected, (2) the uniformity in the climatic conditions during the sampling period (dry season), (3) the absence of on-going agricultural activities that could affect nitrate concentrations (planting season ends in early June), and (4) the relatively low hydraulic conductivity of these aquifers. For these reasons, the data were assumed to be independent and serial correlations were not conducted.

Because nitrate concentrations were determined to be dependent on well depth, the Kruskal-Wallis test was used to compare well-depth distribution among the various potential nitrate-source categories. The results showed that the variation in well depth among the various categories can be explained by random variation ($p = 0.35$) (Figure 6). Therefore, well depths do not bias the comparison among the nitrate-source categories.

CONCLUSIONS

Results from the Spearman Rank correlation test indicated that nitrate concentrations are well-depth dependent. Therefore, well depth had to be considered when evaluating nitrate source-concentration relations. The results of the Mann-Whitney test supported the assumption that nitrate concentrations for wells deeper than 45 m are not affected by anthropogenic activities.

The Kruskal-Wallis test indicated that the variance in nitrate concentration can be explained in terms of the selected potential nitrate-source categories. The Student-Newman-Keuls multiple comparison of the means test showed that: (1) the mean nitrate concentration for wells in the deeper than 45 m category was lower and statistically different than the means for the other six categories, (2) the mean nitrate concentrations for categories that include septic tanks were higher and statistically different than the mean for the crop category, and (3) the mean nitrate concentration for the confined animals category was higher and statistically different than the mean for the crop category.

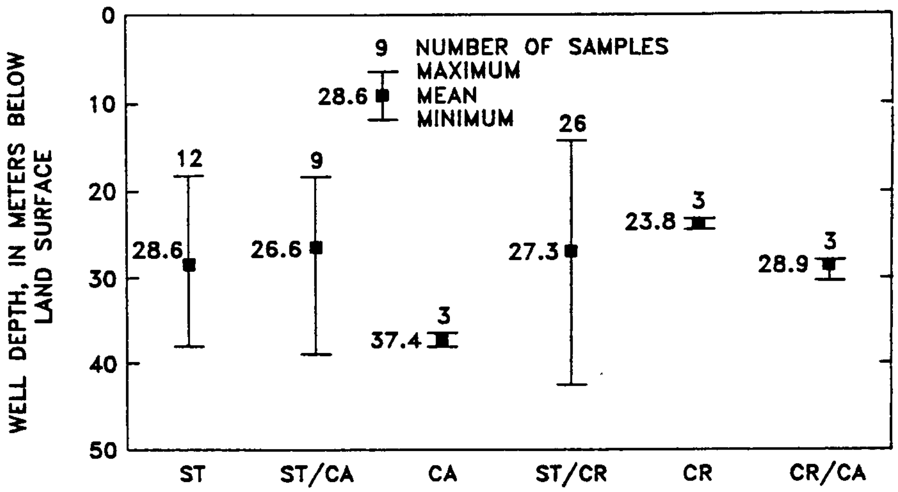

Figure 6. Relation between well depth and selected potential nitrate-source categories (ST, septic tank; CA, confined animals; CR, crops).

The results of these analyses are limited by the relatively small sample sizes for some of the potential nitrate-source categories. However, the number of wells per unit area for a given category is relatively large when compared to similar investigations reported in the literature. Furthermore, the distribution of sampled wells among the various categories reflects the occurrence of potential nitrate sources with respect to well location throughout the watershed.

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FATE AND TRANSPORT OF NITROGEN IN AN AGRICULTURAL WATERSHED

A. Roman-Mas[1], H.H. Cochrane[2], J.A. Smink[3], and S.J. Klaine[3]

The fate and transport of nitrogen in an agricultural watershed in West Tennessee was studied at the field and watershed levels. Runoff and water-quality data for four ditches draining row crop fields in the Beaver Creek watershed were used to quantify the amount of nitrogen exported from cropland. Drainage areas for these fields ranged from 7 to 200 hectares, with about 75 to 95 percent of the drainage areas under row crop cultivation. Runoff and water-quality data for a 3rd-order and a 5th-order stream site downstream of the ditches in the Middle Beaver Creek Canal were used to evaluate potential changes in nitrogen speciation and load in the watershed streams. Drainage areas for these sites were 1,200 and 12,000 hectares, respectively, with about 70 percent of the areas under row crop cultivation.

Data from about 100 rainfall-runoff events were used for this study. Discharge records were developed from modified U.S. Geological Survey operational procedures. Discrete water samples were collected at intervals ranging from 5 to 30 minutes and analyzed for total and dissolved nitrogen species.

Data for the four ditches show that nitrogen in runoff from row crop fields was dominated by the suspended-organic fraction. Crop residue in different stages of decomposition accounted for about 80 percent of the total nitrogen export. Nitrite plus nitrate accounted for about 20 percent of the total nitrogen export from these fields. Ammonia was detected in trace concentrations only. Some seasonal patterns were evident. Late winter and early spring rainfall-runoff events accounted for 75 percent of the suspended-organic nitrogen exported annually. Late spring rainfall-runoff events accounted for 80 percent of the nitrite plus nitrate exported annually. Growing-season rainfall-runoff events accounted for less than 10 percent of the annual nitrite plus nitrate export.

Data for the 3rd- and 5th- order stream sites show that no significant changes in the nitrogen chemical speciation or the load per unit area occurred downstream from the row crop fields. Data for these sites show that channelized streams in this watershed are efficient at conveying runoff as well as dissolved and particulate nitrogen in runoff. However, these streams do not change the nitrogen content of cropland runoff.

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SUSPENDED-SEDIMENT YIELD AND CHANNEL EVOLUTION IN A SMALL AGRICULTURAL CATCHMENT

Robert W. Stogner, Sr.[1] and Timothy H. Diehl[2]

Research on suspended-sediment yield from agricultural land and channel evolution in permanent field ditches is being conducted in a small catchment in the Beaver Creek watershed of West Tennessee. This catchment has an area of about 11 hectares, 90 percent of which has been planted in cotton annually since 1989. The soils are Grenada, Loring, Memphis, and Adler silt loams.

Surface-water discharge was measured from July 9, 1991 to October 15, 1993, except for short periods of equipment failure. Rainfall during this period was typical of average conditions. Total runoff volume during 95 measured storms was 61,000 cubic meters. The median elapsed time from initial hydrograph rise to hydrograph peak for 20 selected storms was 15 minutes.

Suspended-sediment concentration was measured during 43 storms, typically at a sampling interval of 5 minutes. The total runoff volume for these storms was 30,000 cubic meters, about 49 percent of the total runoff volume during the study period. The measured total suspended-sediment yield from these storms was 170 metric tons (mt). Suspended-sediment yield from storms during which sediment concentration was not measured was estimated to be about 200 mt using the regression of storm load on storm volume. The 90-percent confidence interval for this estimate, based on the standard error of the regression, is from 100 to 460 mt. The estimated total suspended-sediment yield from all 95 storms was 370 mt, with a 90-percent confidence interval from 250 to 640 mt.

Evolution of the permanent ditch draining the cotton field was studied. The ditch is about 22 m long, about 1 m wide at the bank tops, and from 0.6 to 0.9 m deep. Its bed slope is about 1 percent. During the study period, the thalweg of the ditch degraded an average of 0.11 m, and a net volume of 5 cubic meters of bed and bank material was eroded, contributing an estimated 7 mt of suspended sediment. Erosion in the ditch contributed between 1 and 3 percent of the total suspended-sediment yield from the catchment.

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MECHANISTIC EVALUATION OF PESTICIDE TEMPORAL PATTERNS FOR A FIRST-ORDER STREAM

Angel Roman-Mas[1], Carol P. Weisskopf[2], and Stephen J. Klaine[2]

Processes and factors affecting the temporal patterns in the concentration of aldicarb and its metabolites were evaluated for a first-order stream in the Beaver Creek watershed, West Tennessee. Mathematical algorithms proposed to explain solute transport mechanics for soil-applied chemicals and, in particular, the mass transfer of a chemical from the soil solution into runoff were examined.

Two general patterns in the concentration distribution of aldicarb and its metabolites were observed during rainfall-runoff events. In the early stages of the event, the natural log of the concentration increased linearly with time and inversely in proportion to the rainfall-runoff ratio. The duration of this initial pattern was similar to the basin response (lag) time. The pattern ended when the concentration peaked. After peak concentration, a second pattern was observed. The natural log of the concentration decreased exponentially with time and was proportional to the rainfall-runoff ratio. These patterns in the concentration of aldicarb and its metabolites are consistent with the rate-limited transfer algorithm used in physically based models and are inconsistent with the instantaneous-equilibrium algorithm used in lumped-parameter models.

Peak concentrations for aldicarb and its first metabolite, aldicarb sulfoxide, for rainfall-runoff events decreased exponentially with elapsed time between application and an event. The rate at which the peak concentrations of aldicarb and aldicarb sulfoxide in runoff decreased with time was similar to the degradation rates in the soil profile. The peak concentrations of aldicarb sulfone, the last metabolite in the oxidation of aldicarb, increased exponentially with time until day 18 and decreased exponentially with time after that. These patterns were consistent with those observed in soils. Multiple regression analyses indicated that peak concentration in runoff for a given rainfall-runoff event is controlled primarily by the concentrations in the soil solution.

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RESEARCH AND MONITORING NEEDS FOR THE IMPLEMENTATION OF THE USDA WATER QUALITY HUA PROGRAM: THE BEAVER CREEK WATERSHED PROJECT

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Abstract: The hydrologic unit area (HUA) program was initiated because of increasing concern for water resources in rural United States. The HUA program is a major element of the President's Water Quality Initiative within the U.S. Department of Agriculture. Previous experiences with rural water-quality programs have demonstrated the need for intensive monitoring programs to identify the sources and magnitudes of agricultural non-point source pollutants and to document water-quality improvements. Additionally, many questions remain about the effects of best management practices on the transport and fate of potential contaminants, including sediment, nutrients and pesticides. HUA projects should be designed to address these issues and concerns. They should also serve as demonstration projects to increase awareness in water-quality issues within the agricultural community.
KEY TERMS: agriculture; nonpoint source pollution; water quality.

INTRODUCTION

Concern for water-quality issues in rural United States has increased in the past decade. Nonpoint source pollution and watershed protection have become areas of special attention (Knopman and Smith, 1993). As a result of this concern, the President's Water Quality Initiative has initiated the hydrologic unit area (HUA) program within the U.S. Department of Agriculture (USDA). The HUA program targeted watersheds at locations throughout the United States where water quality was considered impaired as a result of agricultural non-point source pollution. The primary objective of the HUA program was to improve water quality in these watersheds through the implementation of best management practices (BMPs).

The Soil Conservation Service (SCS), Agricultural Extension Service (AES) and Agricultural Stabilization and Conservation Service (ASCS) are USDA agencies that each play a major role in the HUA program. The SCS, AES and ASCS have the responsibility to provide technical, educational and financial assistance, respectively, to farmers. Landowners can choose among many

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alternative resource management options with different production, economic and environmental consequences. The USDA representatives must be able to offer landowners the most accurate information available to assist them in making the best possible decisions for each unique situation. This can only be accomplished with a reliable understanding of the targeted watershed.

Experiences from the Rural Clean Water Program and agriculture nonpoint source studies have demonstrated the need for intensive monitoring programs to achieve reliable information about the watershed (Richards and Baker, 1993). Successful monitoring will help identify the sources and magnitudes of agricultural non-point source pollution (NPS). Continued water-quality monitoring was also deemed important to document improvements in water quality resulting from changes in land-management practices. Additionally, many questions remain about the effects of BMPs on the transport and fate of potential contaminants, including sediment, nutrients and pesticides (Cooper, 1993).

Another objective of the HUA program is to demonstrate the effectiveness of voluntary programs at mitigating water quality in the targeted watersheds. This feedback is vital to farmers, policy makers and the general public. Demonstrating improved water quality as a result of BMPs encourages other farmers to participate and become involved. The HUA projects should demonstrate economic feasibility, and improve environmental sensitivity and public acceptability. These are key determinants in the decision-making process of contemporary agriculture.

Successful implementation of many HUA projects has been hindered by the absence of data and modeling capabilities that (1) accurately define the nature and extent of the impacts of agricultural activities on water quality, and (2) support the selection and implementation of BMPs that best fit conditions at the field and at the watershed level. The intent of this paper is to focus on several major needs USDA agencies have in order to better assist landowners in managing their natural resources. Examples will be drawn from the Beaver Creek HUA in Southwestern Tennessee to highlight successes and future needs.

Beaver Creek HUA Project

The Beaver Creek HUA project in Southwestern Tennessee was initiated in 1990 as a result of farmers' concern about NPS pollution. Reasons for the concern ranged from statements in the national media attributing significant water-quality impairments to modern agriculture, to an assessment level evaluation conducted by state agencies in Tennessee. The water-quality degradation in the Beaver Creek Watershed (BCW) was attributed to agricultural activities (Thompson, 1993). A comprehensive water-quality monitoring program was initiated in addition to implementing BMPs in the Beaver Creek HUA to achieve the objectives of the HUA program.

RESEARCH NEEDS

Research activity that would improve the ability to manage the natural resource base is needed. The goals of this research effort should include: (1) improving our ability to determine the nature and extent to which agricultural activities impact water quality and threaten environmental integrity, (2) improving our ability to differentiate agricultural NPS pollution from other sources of pollution and from natural conditions, (3) developing and evaluating monitoring and sampling strategies that accurately represent water quality, (4) understanding the processes and factors that control the transport and fate of agricultural pollutants, (5) assessing the validity of present criteria used to set standards for water quality, (6) calibrating and validating water-quality models.

The effects of agricultural BMPs such as conservation tillage, nutrient management and pesticide management have been studied throughout the country (Sharpley and others, 1988; Jayachandron and others, 1993). Inconsistent results have been obtained by virtue of a number of site specific factors influencing the processes of soil erosion and nutrient pesticide movement (Betson and Marius, 1969; Griffin, 1992)). More research is needed to quantify site specific variables which determine the effectiveness of BMPs used alone and in various combinations.

The use of the simulation model, CREAMS, in Pennsylvania, resulted in substantial variance among sites and practices with regards to effectiveness of BMPs and nutrient management programs (NMPs) in reducing off-site sediment and nutrient loads (Hamlett and Epp, 1994). It was concluded that "typically there is not a single BMP or NMP that is adequate for controlling pollutant losses from agricultural lands. Therefore, to control sediment and nutrient losses from agricultural areas, control programs should emphasize the use of improved NMPs with the more traditional BMPs", (Hamlett and Epp, 1994). It was also concluded that field conditions such as slope, crop rotation, and the predominant transport pathways should be considered when selecting BMPs and NMPs.

A study using the paired watershed approach in the limestone soils of northern Alabama was conducted by Soileau and others (1994). Analyses of the data collected over a 6-year period suggested a strong effect of tillage on runoff water. Specifically, a greater volume of runoff water occurred with the use of conservation tillage than conventional tillage. However, soil losses for both tillage systems were below soil tolerance (T), but significantly lower for conservation tillage when soil loss calculations were determined using the erosion index. Also, the presence of a grass filter strip together with crop residue cover in the winter and spring also helped minimize sediment loss from the watershed, especially with conventional tillage.

Soileau and others (1994) found that tillage operations appeared to have no significant effect on nitrogen (N) losses, either as Kjeldahl nitrogen or as NO_3-N. Phosphorus (P) runoff tended to be greater with conservation tillage than with conventional tillage. This contradicts studies done by Sharpley and others (1991) who found a reduction in nutrient (N and P) transport from a sorghum field located in the southern plains with conservation tillage. The dissimilar results could be explained in part by the unique site variables. Most of the annual P runoff from the Alabama site occurred with the filtered water component instead of the sediment-bound fraction because this site had a relatively large volume of runoff water and small sediment loss. Another explanation could be the use of different sampling and analytical techniques.

Again, these different research projects demonstrate the variability in results, possibly due to sampling or research strategies, geographical location, soil types, climate or even the duration of the project. They also demonstrate the need for additional research utilizing consistent sampling methodology so that legitimate comparisons between data sets become possible.

An important factor that must be considered is the financial feasibility of the BMP to the farmer. As concluded by Hamlett and Epps (1994), various approaches may prove effective in the field, but they may negatively influence the farm enterprise or may not be financially feasible in a real-world situation. Therefore, economic analyses of all practices and programs are needed.

USDA agency personnel need to understand how the implementation of no-till and combinations of other BMPs behave with respect to the fate of nutrients and pesticides. This includes the migration of agri-chemicals through different types of soils. The agent must be able to integrate that knowledge with the myriad combinations of crops, vegetative cover, soil types, field slopes, climate and economic feasibility. Ultimately a system of conservation and nutrient practices which will provide beneficial effects to the environment must be achieved.

Additional information is needed on the types of vegetation best suited for erosion control and nutrient and/or pesticide uptake and deposition under the varying climatic and geographic conditions. An enormous amount of research has been conducted on these topics, but the results need to be integrated to provide the best recommendations for land-users.

The success of BMPs is dependant upon the types of vegetation associated with these practices. Promoting certain types of vegetation as winter cover crops, conservation cropping systems or in filter strips and vegetated waterways has proven to be extremely successful at reducing the transport of soil, nutrients and pesticides that have an affinity to clay. For example, it has been shown that grasses and legumes were effective for erosion control purposes in row crops. It has also been demonstrated that vegetative barriers (known as stiff grass hedges) can reduce sediment yield and decrease runoff velocity (Dabney and others, 1993).

The introduction of legumes into a conservation cropping system has proven to be useful both as erosion control and adding nitrogen to the soil for the following seasons crop. It has been demonstrated that some winter cover crops, such as rye, can scavenge nitrogen from previous applications and prevent its movement through the soil profile into ground water. However, some of these same cover crops may actually promote NPS pollution into adjacent water systems. An example of this condition has been shown with P leaching from cotton foliage, wheat straw (Soileau and others, 1994), and the canopy of sorghum and soybeans (Sharpley, 1981). The P leached from the crops was a major portion of the total P runoff in these studies. This P leached from plant tissue, may partly explain why cotton plus rye in the conservation tillage cropping system had more total P runoff loss than cotton only in the conventionally tilled system in the northern Alabama project (Soileau and others, 1994).

Recognizing the sources of nutrient loading is a critical factor in developing natural resource plans. All of the pollution sources potentially affecting soil, water, air, plants, animals, and humans must be identified as plans are developed to alleviate problems affecting designated use impairments. If potential nonpoint pollution sources are not accounted for, the restoration plan may never achieve its intended goals. Thus, we must continue to develop our understanding of both the natural and human influences on the environment.

Monitoring Needs

Optimized sampling strategies to assess agricultural NPS pollution and BMPs must be considered as the foundation for determining baseline conditions, sources of contaminants, and the effectiveness of decisions made and actions taken. Meeting these needs requires an intensive monitoring program at both the field, sub-watershed, and large watershed level (Richards and Baker, 1993). Analyses and subsequent conclusions are only as reliable as the methodology that is utilized. Therefore, it is imperative that quality assurance and quality controls are employed as part of the monitoring strategy.

There are additional concerns when monitoring nonpoint pollution parameters in large watersheds. These concerns include site selection and multiple nonpoint pollution sources. Proper selection of sampling sites is imperative when sampling at the watershed level. One should be aware of hydrologic factors, such as backwater at the mouth of a watershed, that could influence the chemical and biological parameters being assessed. Caution must be taken, as these conditions generally yield higher concentrations of the contaminants being evaluated owing to the retardation of water flow.

Another concern that a successful monitoring program must address is that of multiple nonpoint pollution sources. It is extremely difficult, if not impossible, to differentiate sources of NPS pollution at the watershed level. A thorough reconnaissance of the watershed must be done to catalog the possible sources of pollutants. Specific sites should be monitored to determine the NPS contribution of an area to the watershed. Additional sampling should be done downstream to determine if the NPS pollution has been removed from the system (by assimilation and breakdown) or has migrated downstream. Without characterizing this process, it is possible to incorrectly identify and target a source as the primary contributor.

Many of the desired water-quality goals were developed for low-flow conditions. However, NPS pollutant loading occurs primarily during rain events. These elevated levels of NPS pollutants last for relatively short periods of time. This implies that chemical monitoring must be intensive during the rain event to accurately characterize the NPS pollutant load, especially in small basins (Richardson and Baker, 1993). Temporal and spatial considerations are critical to a successful chemical monitoring program. A great deal of financial and personnel resources are required to accurately characterize the NPS pollution in a watershed.

Many agencies are turning to biological monitoring to avoid the problems associated with a chemical monitoring program. However, there are concerns with the use of biological monitoring to draw conclusions regarding NPS pollutants. For example, the biological database is presently too small to establish scientifically defensible limits for specific geophysical regions (Griffin and others, 1992). Preliminary research must be done to develop an acceptable baseline data set. This is essential to factor out the inherent diversity of the watershed system, as well as seasonal and annual fluctuations. Acceptable ranges must be established to account for natural variability.

Many sampling protocols are available for biological monitoring (Byl and Roman-Mas, 1994). However, these protocols were developed for specific hydrologic and geographic conditions that may or may not apply to the watershed under investigation. For example, the use of the sensitive taxa (Ephemeroptera, Trichoptera, and Plecoptera) to evaluate water quality may not be applicable to watersheds that do not supply the correct habitat for these organisms. It is entirely possible that these organisms never inhabited the watershed in question, and therefore, accurate comparisons cannot be made.

Despite the many needs of monitoring programs, accurate biological and chemical assessments are necessary in order to properly characterize the contaminants and their sources. Otherwise, application of conservation management systems, or BMPs, to restore or enhance the impaired water quality will be ineffective.

The Beaver Creek HUA Program

The Beaver Creek HUA project has provided a vehicle to conduct intensive monitoring and research in real farm situations. It has presented the opportunity to improve methodologies for surface and ground-water monitoring. Experience gained from several years of pre-BMP monitoring has allowed the Beaver Creek program to develop a considerable data base to use in water-quality models. The intensive studies conducted in the Beaver Creek HUA have also improved our understanding of in-stream assimilation processes. This has led to the development and testing of innovative new BMPs, such as a constructed wetland to mitigate runoff from a field of row crops.

The U.S. Geological Survey, in cooperation with USDA, local and state agencies, is conducting a comprehensive assessment of the effect of agricultural activities on water quality and the effectiveness of BMPs in the Beaver Creek Watershed. The main goal of this project is to develop tools that can be used by federal and state agencies in their resource management programs. A holistic watershed approach has been adopted for this program. Intensive water sampling has been done during rain-events to get chemical data and characterize the transport of NPS pollution in the watershed. The chemical monitoring has examined both surface and ground water. Biological samples have also been gathered in connection with the chemical and geomorphology studies. Some of the results of these monitoring activities can be found in these proceedings.

SUMMARY

Effectively addressing NPS water-quality problems from agricultural sources depends upon (1) careful identification of the NPS problems and their source, (2) implementation of effective and cost efficient practices to address the problems identified, and (3) the capability to demonstrate the benefits of resource management decisions on water quality. Research is needed to aid in identifying NPS sources, to sort out the sources of the pollutants identified, and to demonstrate the effectiveness of BMPs in management systems. These results are crucial to provide resource management agencies with the best possible information.

Monitoring is needed to (1) document baseline conditions, (2) track progress and identify possible modifications or refocusing of efforts, and (3) document water-quality improvements. Demonstrating improvement is essential to justify the resources invested both to the landowners and, perhaps more importantly, to the tax-paying public demanding improved water quality and effective agency programs.

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