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**ADSORPTION, DISSIPATION AND MOVEMENT OF FLUOMETURON AND
NORFLURAZON IN THREE SOUTHEASTERN SOILS**

A Dissertation

Presented for the

Doctor of Philosophy

Degree

The University of Tennessee, Knoxville

W.T. Willian

December, 1995

AD-VET-MED.

Thesis

95b

.W545

DEDICATION

To my parents, Mr. and Mrs. Kermit Willian, whose encouragement and prayers made this effort possible, I dedicate this dissertation.

ACKNOWLEDGMENTS

The author wishes to express his sincere appreciation to the following persons:

To Dr. Thomas C. Mueller, his major professor, for his guidance in conducting this research, for his advice and assistance in preparing this dissertation, for his friendship and encouragement throughout the course of this study, and most of all for his patience and helpfulness.

To Dr. Robert M. Hayes for much helpful advice, guidance, and assistance in the planning and implementation of this research, and for his friendship throughout the course of my graduate studies.

To Drs. William A. Krueger, Donald D. Tyler, and Fred D. Tompkins for serving on my graduate committee, for their advice and suggestions in preparing this dissertation, and for their friendship and encouragement.

To Drs. Charles E. Snipes and David C. Bridges (and their respective staffs) for providing labor, resources, and expertise in plot establishment and soil sampling for this research.

To Dr. G. Neil Rhodes, Jr. for friendship, encouragement, and for making me laugh.

To Dr. John E. Foss and the entire Department of Plant and Soil Science for providing me the opportunity and resources for furthering my education.

To the personnel at the West Tennessee Experiment Station, Delta Branch Experiment Station, and Georgia Experiment Station at Griffin for their invaluable assistance in field sampling for this project.

To Patricia Brawley for her assistance, friendship, and conscientiousness.

To Ernest Merriweather for his assistance in plot maintenance and sampling.

To Dr. Tom Ammons, Janice Branson, and the entire Soil Genesis and Classification laboratory personnel for their helpful assistance in soil characterization.

To my fellow Weed Science graduate students (past and present), Buster Kirksey, Blake Brown, Kent Gallaher, Bob Montgomery, Angela Harrison, and Ann Moore, who provided me with countless advice, encouragement and friendship.

To Dave O'Dell, Terry Kelley, Neal Denton, Danny Peek, Bill Halfman and Bob and Debbie Paris, for their fellowship, friendship, and encouragement during the past few years.

To the undergraduate assistants, Cheryl Ashburn, Monica Stephens, Michelle Howard, Neal Ashburn, Rex Robinson, Andy Price, Carrie Stiles, Kip Barnett, Dale Norman, Elizabeth Theobald, and Brandon Smith, for their invaluable assistance in soil preparation, and for being a pleasure to work with.

To the secretaries, Dawn Brown, Martha Cox, Debbie Bowles, Marilyn Dorn, Betty Gorman, Laura Webster, Jenifer Huey, Janet Privetera, and Dotty Casella, who aided me in countless ways.

To my brothers, Tim and Bud, my sister-in-law, Lisa, and my nephews, Michael and Andrew, for their love and support.

To the Millers: Bob, Karen, Laura, and Al for their love, prayers, support, and encouragement. Thanks also for providing a place to get away from it all, for many hot meals, and for letting me sleep on the most comfortable couch in the world!

To my parents, Mr. and Mrs. Kermit Willian, for their endless encouragement,
support and love.

To God for providing me the strength, perseverance and patience to complete this
research.

ABSTRACT

Studies were conducted in 1992 and 1993 to evaluate the adsorption, dissipation, and movement of fluometuron and norflurazon in three soils representative of the cotton growing regions of the southeastern United States. Field experiments were established in Jackson, Tennessee; Stoneville, Mississippi; and Griffin, Georgia on areas which had no fluometuron or norflurazon exposure for three years prior to plot establishment. The studies utilized a randomized complete block design and treatments were replicated four times in tilled areas.

Fluometuron and norflurazon were applied preemergence (PRE) at 1.7 kg ai ha⁻¹ broadcast. Soil samples were obtained in one of two ways: surface (0 to 8 cm) soil samples were collected from each plot prior to and immediately following herbicide application, and at approximately 14, 28, 56, and 112 days after treatment (DAT) using hand-held soil probes. These samples were utilized to examine surface fluometuron and norflurazon dissipation. Two full-length depth cores from the 0- to 120-cm soil zone were collected at approximately 28, 56, and 112 DAT. Samples obtained from the 0 to 120 cm depth were divided into 0- to 8-, 8- to 15-, 15- to 30-, 30- to 45-, 45- to 60-, 60- to 90-, and 90- to 120-cm soil zones to examine fluometuron and norflurazon distribution within the soil profile.

Additional studies were conducted in 1993 to examine fluometuron adsorption and dissipation under laboratory conditions. Untreated soil from the 0- to 8-, 30- to 45- and 60- to 90-cm zones was collected in 1993 from plots utilized in the field study. Adsorption of fluometuron and norflurazon on soil was determined using a slurry

technique, and a distribution coefficient (K_d) was calculated for each location and depth combination. Dissipation of fluometuron and norflurazon under controlled conditions was determined for each location by depth combination by fortifying untreated soil with either fluometuron or norflurazon and incubating at 30 C. Fluometuron and norflurazon were quantified with High Pressure Liquid Chromatography (HPLC).

The ability of the Pesticide Root Zone Model (PRZM) and Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) model to simulate fluometuron and norflurazon movement was evaluated with site-specific environmental data. Tests of model performance were demonstrated using both site-specific and database sorption and dissipation parameters. Fluometuron or norflurazon concentration in total mass within each profile was chosen as a measure of model accuracy.

Fluometuron and norflurazon adsorption was correlated with organic matter but not with other soil characteristics. First-order kinetics described fluometuron dissipation under field and laboratory conditions ($r^2 \geq 0.82$). Fluometuron dissipation was more rapid under field conditions than under laboratory conditions in soil from two of three locations, and field dissipation was slower under dry conditions. Fluometuron half-life in soils from the 0 to 8 cm depth ranged from 9 to 28 days under field conditions and from 11 to 43 days in the laboratory. Fluometuron half-life was positively correlated with soil depth and inversely correlated with soil organic matter.

Norflurazon dissipation under controlled conditions in soil from the 0 to 8 cm zone was not different among locations, with half-life ranging from 63 to 167 days.

Norflurazon dissipation was more rapid under field conditions than under laboratory

conditions with half-life ranging from 7 to 79 days in the 0 to 8 cm soil horizon.

Neither norflurazon nor fluometuron was detected below 15 cm in the soil profile in any soil, based on data obtained from soil cores. Concentrations in the 8 to 15 cm soil zone were < 36 ppbw 112 days after treatment. Site-specific parameters increased PRZM accuracy in predicting total norflurazon concentration in the soil profile, and PRZM more accurately predicted norflurazon than fluometuron concentration when site-specific parameters were utilized. The use of database variables increased the accuracy of PRZM to predict fluometuron but decreased the accuracy of norflurazon predictions. PRZM provided more correct predictions of norflurazon concentration in the soil profile than did GLEAMS; however, both models performed similarly with regards to fluometuron. The use of site-specific data increased the accuracy of GLEAMS to predict fluometuron but not norflurazon. Simulations better predicted norflurazon concentration than fluometuron concentration when averaged over all model by parameter input combinations.

Soil organic matter and environmental conditions affected fluometuron and norflurazon dissipation. Migration of fluometuron or norflurazon into subsurface soil zones was minimal, and PRZM and GLEAMS demonstrated their utility to accurately predict herbicide behavior in these soils when correct input parameters were employed.

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PART I.
INTRODUCTION

Herbicides are used extensively to control unwanted vegetation. The advantages of using herbicides to control weeds are numerous and include reductions in the number of tillage operations needed to reduce weed interference, reduced human effort expended on hand weeding, increased flexibility in agricultural and horticultural management systems, and increased yields.

Preemergence herbicides are an important component in cotton (*Gossypium hirsutum* L.) management systems. Effective early season weed control is essential to cotton production since there are currently few selective postemergence broadleaf herbicides registered for use in this crop. Fluometuron is used throughout the cotton growing regions of the southeastern United States to control common cocklebur (*Xanthium strumarium* L.), pigweeds (*Amaranthus* spp.), and morningglories (*Ipomoea* spp.). Norflurazon is used alone or in combination with other preemergence herbicides to control spurred anoda [*Anoda cristata* (L.) Schlecht], spurge (*Euphorbia* spp.), velvetleaf (*Abutilon theophrasti* Medik.), prickly sida (*Sida spinosa* L.), nutsedges (*Cyperus* spp.), and annual grasses.

For a preemergence herbicide to be effective, it must provide weed control for a sufficient time to allow the crop to become established without competition from weeds; otherwise, crop yield may be reduced. Problems may be encountered when herbicides remain longer than required for weed control. Rotational crops may be injured, or herbicide residues may be moved from the site of application. A herbicide should remain in sufficient concentrations to provide weed control during crop production without

leaving potentially harmful residues.

Herbicide behavior in the environment is a function of adsorption, chemical degradation, microbial degradation, leaching, photodecomposition, and volatilization. Studies of herbicide fate in the soils may involve quantitation of resident concentrations, or simulations of profile concentrations.

Advances in programming and computer technology has led to the development of several sophisticated models which can be used to simulate pesticide movement into soil. The use of these models could provide a cost-effective environmental impact assessment, and serve to generate data by which pesticide inputs and management practices could be better employed.

The following sections of this dissertation will pertain to research conducted to determine the adsorption, dissipation under field and laboratory conditions, and simulated movement through the soil profile of norflurazon and fluometuron.

PART II

**LIQUID CHROMATOGRAPHIC DETERMINATION OF NORFLURAZON AND
ITS INITIAL METABOLITE IN SOIL¹**

¹William T. Willian and Thomas C. Mueller. *Journal of Assoc. Off. Anal. Chem.* 77(3):752-755

ABSTRACT

A rapid, sensitive method for the determination of norflurazon in four soils is described. Data on the initial soil metabolite is also obtained in soils with low organic matter. The method consists of extraction of soil samples with methanol, filtration, liquid chromatographic separation of methanol-soluble components by using a C₁₈ column, and fluorescence detection with excitation at 294 nm and emission measured at 398 nm. Recoveries from fortified soils were > 90% for norflurazon and > 80% for desmethylnorflurazon from the Shipps, Lexington, and Harkey soils. Average percent relative standard deviations over the soils examined was 5.5% for norflurazon and 8.7% for desmethylnorflurazon. The limit of detection for norflurazon was 10 ppbw soil, whereas the limit of detection for desmethylnorflurazon was 100 ppbw soil because of its smaller relative detector response.

INTRODUCTION

Norflurazon, 4-chloro-5(methylamino)-2-(3-(trifluoromethyl)phenyl)-3(2*H*)-pyridazinone, is a soil-applied herbicide registered for use in cotton, soybeans, citrus, and other crops (Figure 1). The typical application rate is from 1 to 2 kg ai ha⁻¹. Norflurazon has a water solubility of 28 µg ml⁻¹ and a vapor pressure of 2.0 X 10⁻⁸ mm Hg at 20 C (2). Norflurazon's initial soil metabolite, desmethylnorflurazon (DMN), has been identified in field dissipation studies (6).

The literature contains several methods using gas chromatography (GC) with an

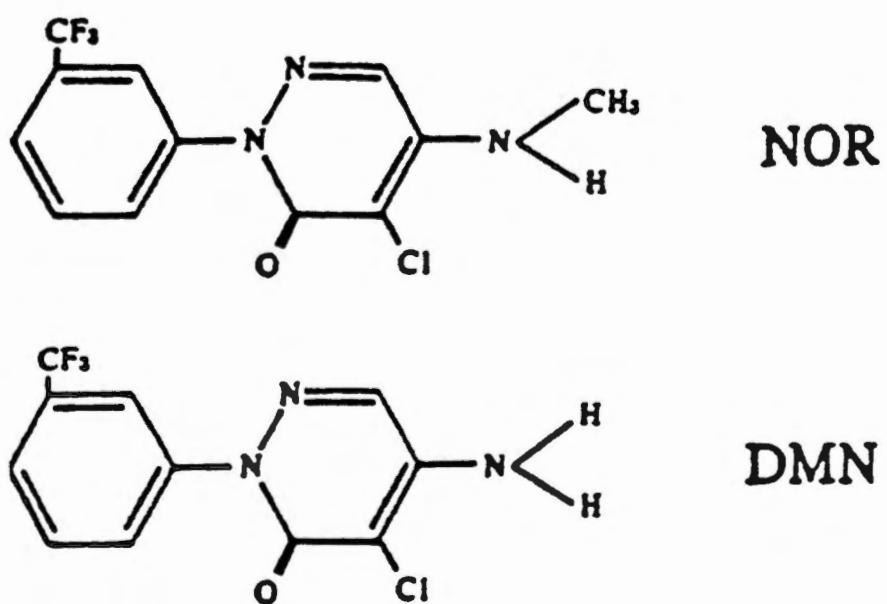


Figure 1. Norflurazon (NOR) and its initial soil metabolite desmethylnorflurazon (DMN).

electron capture detector for the determination of norflurazon in soil (1, 6, 8). Schroeder and Banks (6) extracted several Georgia soils (organic matter, 0.7-2.8%) with methanol or acetonitrile, with norflurazon recoveries ranging from 93 to 102%. Methods by Winkler et al. (8) and a method supplied by the manufacturer of this crop protection chemical (1) require extraction and cleanup techniques followed by GC analysis. Limits of detection of the GC methods were approximately 10 ppbw soil. Draper and Street (3) reported an isocratic liquid chromatographic method using UV detection of norflurazon and desmethylnorflurazon in fruit crops. The method involved liquid-liquid partitioning and other cleanup procedures. The minimum detectable quantity for each residue was $< 0.1 \mu\text{g g}^{-1}$ in the crops tested.

Analytical methodology was developed for the simultaneous determination of both norflurazon and its initial soil metabolite at ppm to ppb levels in soil, using an external standard liquid chromatographic (LC) method with fluorescence detection. The methodology could be used to support studies on the bioavailability and environmental fate of norflurazon.

MATERIALS AND METHODS

Apparatus and Reagents. (a) LC system. Waters liquid chromatograph, including Model 680 control unit, model 715 auto-injector, model 501 solvent delivery system, model 470 fluorescence detector set to excite at 294 nm, read emission at 398 nm, 4.0 sec filter, 1000X gain and ATTN = S (Milford, MA). Additionally, a Hewlett-Packard (San Fernando, CA) model 3396 integrator was used to interpret detector response. (b)

Analytical column. 25 cm X 4.6 mm id, 5 μ m, LC-C₁₈; in-line 1 cm X 1.5 mm pellicular C₁₈ guard column (Alltech, Chicago, IL). **Solvents.** LC grade (J.T. Baker Inc., Phillipsburg, NJ; and Burdick and Jackson, Muskegon, MI). (d) **Mobile phase.** Initially acetonitrile-water (50 + 50); 0 to 12 min linear gradient to 75 + 25 (v/v). (e) **Analytical standards.**-- Norflurazon and desmethylnorflurazon (Sandoz Crop Protection, Des Plaines, IL). Standards were > 99% pure and each was used without purification.

Soil Selection. Soil was selected to represent a range in texture, organic matter content, and pH (Table 1). Soil 1 is a Shipps clay (high clay content); soil 2 is a Lexington silt loam; soil 3 is a Harkey loam (high pH); and soil 4 is a Drummer silty clay loam (high organic matter content). All soils were passed through a 2 mm sieve prior to analysis. Soils 1, 2, and 3 represent soil upon which norflurazon is typically used. Soil 4 would not represent a soil upon which norflurazon is routinely applied but was included to determine the effect of relatively high organic matter content on norflurazon quantitation.

Fluorescence Detection. The fluorescence properties of norflurazon when dissolved in acetonitrile were examined previously (5). Fluorescence was maximum when excitation was at 294 and emission measured at 398 nm. Adding water or acidifying the solution had no effect on the observed fluorescence. The relative fluorescence was approximately 25% that of fluometuron (5), for which a similar method was reported (4).

Several acetonitrile-water combinations were evaluated for herbicide and metabolite retention time and capacity factor (k') (7). The accuracy of the analysis was examined by injection of a series of standards containing norflurazon and desmethylnorflurazon in methanol at 0.05 to 5 μ g ml⁻¹. Injection volume was 50 μ L.

Table 1. Soil characteristics.

Soil	Series	State	Sand, %	Silt, %	Clay, %	OM, % ^a	pH ^b
1	Shipps	Texas	1	32	67	1.4	7.3
2	Lexington	Tennessee	4	75	21	1.2	6.4
3	Harkey	New Mexico	36	39	25	1.1	7.2
4	Drummer	Illinois	14	53	33	4.8	6.3

^aDetermined by modified Debolt version of Walkley-Black method.

^bDetermined by 1:2 soil:water suspension technique.

This range corresponded to herbicide soil concentrations of 0.10 to 10 $\mu\text{g g}^{-1}$ by the described methodology.

Extraction. Analytical norflurazon and desmethylnorflurazon (in methanol) were added to duplicate 250 mL polyethylene bottles with screw-top caps (Nalgene, Rochester, NY) containing 40 g air-dried soil to achieve soil concentrations of 940 and 830 ppbw, respectively. The methanol was allowed to evaporate, bottles were capped, and the contents were mixed thoroughly by hand shaking. The bottles were then allowed to equilibrate statically for 24 h. Additional treatments included norflurazon and desmethylnorflurazon added to bottles containing no soil and each soil without norflurazon or desmethylnorflurazon. The four soils were examined for extraction efficiency in a single experiment. Methanol (80 ml) was added to each bottle before they were placed on a reciprocating shaker (Eberbach, Ann Arbor, MI) operated at 180 rpm for 16 h. A portion of the extract was filtered through two qualitative filter papers (Whatman, Clinton, NJ) directly into 4 ml autosampler vials.

RESULTS AND DISCUSSION

Injection of 50 μL of norflurazon and desmethylnorflurazon standards produced a peak with a retention time of 3.4 to 19.3 min, depending on mobile-phase composition (Table 2). At equal relative concentrations, methanol-containing mobile phases had greater capacity factors than those containing acetonitrile. Chromatographic concerns associated with the isocratic procedures for both methanol and acetonitrile mobile phases included poor resolution of norflurazon and desmethylnorflurazon (capacity factor

Table 2. Solvent systems and retention times for norflurazon (NOR) and desmethylnorflurazon (DMN).

Mobile phase	Solvent ratio (v/v)	Retention time, min		k'	
		NOR	DMN	NOR	DMN
Acetonitrile-water	80+20	3.45	3.79	0.35	0.49
Acetonitrile-water	70+30	4.00	4.58	0.57	0.80
Acetonitrile-water	60+40	4.92	6.11	0.93	1.40
Acetonitrile-water	50+50	7.03	9.55	1.76	2.75
Acetonitrile-water	40+60	12.83	19.29	4.03	6.56
MeOH-water	80+20	4.36	4.88	0.71	0.91
MeOH-water	70+30	7.12	8.31	1.79	2.26
Acetonitrile-water	Gradient 1 ^b	8.40	12.65	2.29	3.96
Acetonitrile-water	Gradient 2 ^c	7.43	10.43	1.91	3.09

^aCapacity factors for norflurazon and desmethylnorflurazon; t_0 for system = 2.55 min.

^bAcetonitrile-water (v/v), initial conditions of (45 + 55), 0 to 12 min linear gradient to (60 + 40).

^cAcetonitrile-water (v/v), initial conditions of (50 + 50), 0 to 12 min linear gradient to (75 + 25).

differences, < 0.5) and asymmetrical peak shape (peak tailing). The use of methanol also was coincident with higher system operation pressures, although the pressure could be reduced by the use of a column oven. To overcome these problems, gradient elution using acetonitrile-water (gradient 2) was used for subsequent quantitation (Figure 2). The norflurazon standard curve was linear in the 50- to 2000 ng ml⁻¹ range, with an r^2 of 0.99 ($n=7$, data not shown) but was nonlinear at higher concentrations (> 2000 ng ml⁻¹). With the proposed methodology, concentrations in soil extracts above this limit would not be expected from typical soil applications of norflurazon. The norflurazon analytical standards (in methanol) and methanol soil extracts were stable and showed no change in detector response when stored at room temperature for 8 wk. Soil extracts of desmethylnorflurazon were not stable, and detector response decreased by approximately 15% after 2 wk.

Norflurazon and desmethylnorflurazon recoveries from all soils ranged from 81 to 108% (Table 3). Desmethylnorflurazon was not determined in soil 4 because of background interference. The precision of the method was good, as indicated by an average relative standard deviation (RSD) of 6.8% for all recoveries (Table 3).

The standard curve for desmethylnorflurazon was linear from 0 to 5000 ng ml⁻¹, with $r^2 = 0.99$ ($n=8$, data not shown). The detector response from this initial metabolite was 10 times smaller than the parent compound. Fresh solutions were prepared three times to make sure the lower detector response was not a laboratory error. Each preparation of stock solutions and subsequent analysis produced the same result.

Desmethylnorflurazon possesses the same rings as the parent, but apparently, the loss of

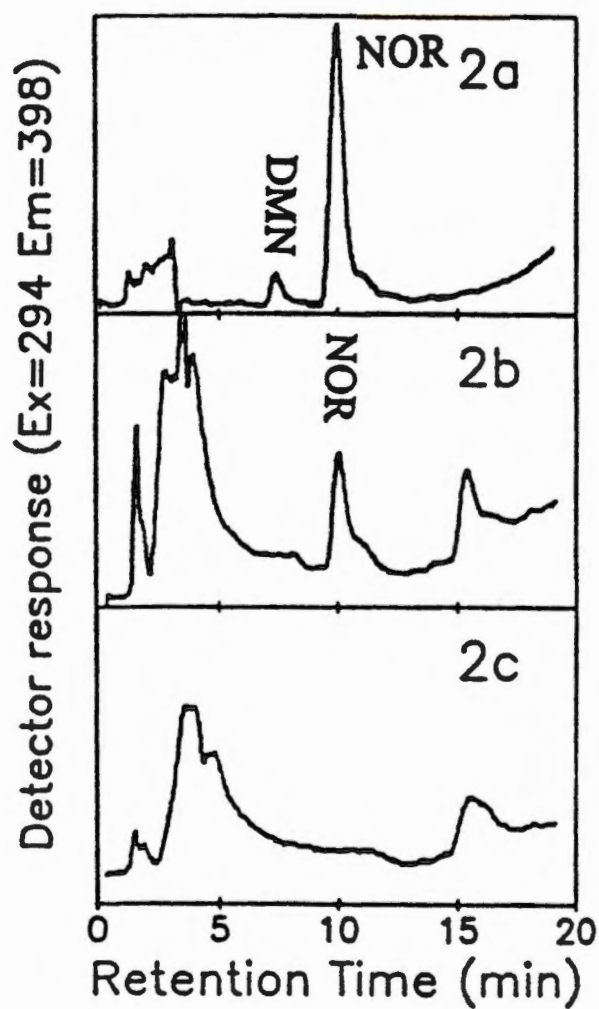


Figure 2. Liquid chromatograms of (a) analytical standard containing desmethylnorflurazon (DMN) at 1000 ng g^{-1} and norflurazon (NOR) at 1000 ng g^{-1} , (b) soil 1 extract containing norflurazon at 500 ng g^{-1} , and (c) soil 1 extract containing no norflurazon or desmethylnorflurazon.

Table 3. Recovery of norflurazon and desmethylnorflurazon from four soils with methanol extraction, gradient elution, and fluorescence detection^a.

Soil	Series	Parent or metabolite	Detected concn, ppbw	Recovery, %	RSD, %
1	Shipps	Norflurazon	860	92	7.0
		Desmethylnorflurazon	670	81	8.7
2	Lexington	Norflurazon	960	102	4.9
		Desmethylnorflurazon	900	108	17.2
3	Harkey	Norflurazon	930	99	1.2
		Desmethylnorflurazon	750	90	0.1
4	Drummer	Norflurazon	870	93	8.8
		Desmethylnorflurazon ^b	—	—	—

^aInitial concentrations: norflurazon, 940; desmethylnorflurazon, 830.

^bNot determined because of background interference.

the methyl group reduced fluorescence at the measured parameters because of changes in the electron distribution around the ring. This smaller detector response limits the utility of this method to quantitate trace amounts of desmethylnorflurazon.

Desmethylnorflurazon was not discernable in extracts from soil 4. This soil has a large organic matter content (for a mineral soil), which interfered with the small desmethylnorflurazon detector response. Norflurazon is not routinely used in soils with organic matter contents of > 4%, and we realize and acknowledge this limitation in the method to quantitate desmethylnorflurazon in this soil. This limitation could be addressed by cleanup procedures or other techniques.

Baseline interferences with the proposed methodology were minimal when compared with UV detection at 275 nm (data not shown). Other commonly used herbicides are not expected to interfere with norflurazon quantitation, because many do not fluoresce under these test conditions (5), with the exception of fluometuron. A previous paper has discussed fluometuron analysis (4). Chromatographic separation of fluometuron and norflurazon is possible, but the metabolites of each compound coelute with the other compounds of interest. This makes simultaneous determinations of the two herbicides difficult.

With a signal-to-noise ratio of two, a conservative limit of quantitation for this methodology is 10 ppbw of soil for norflurazon. Because of the weaker detector response, the limit of detection of desmethylnorflurazon is 100 ppbw of soil. The simple extraction method presented here combined with gradient elution and fluorescence detection would allow the rapid determination of these compounds in large numbers of

soil samples at levels suitable for many purposes. This method has been used for the analysis of > 400 soil samples from surface and subsurface soil samples, with the presented chromatograms being from actual field samples (Figure 2). For trace analysis of these compounds, or the analysis of these compounds in matrixes containing substantial amounts of organic material, modification of the extraction and chromatographic procedures may be necessary. The proposed methodology should serve as a good foundation for method development and refinement, and the sensitivity and selectivity provided by fluorescence detection should provide a sound basis for chemical analysis of these compounds.

ACKNOWLEDGMENTS

We thank Morris Merkle (Texas), Jill Schroeder (New Mexico), and Robert Hayes (Tennessee) for providing soil to perform the studies.

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PART III.

**ADSORPTION, DISSIPATION AND MOVEMENT OF FLUOMETURON IN
THREE SOUTHERN SOILS¹**

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ABSTRACT

Fluometuron adsorption, dissipation under field and laboratory conditions, and distribution within the soil profile was determined in three soils from Tennessee, Mississippi and Georgia representative of the cotton growing regions of the southeastern United States. Fluometuron adsorption was correlated with organic matter but not with other soil characteristics. First-order kinetics explained fluometuron dissipation under field and controlled conditions ($r^2 \geq 0.82$). Fluometuron dissipation was more rapid in soil from the 0- to 8-cm depth in the TN soil than in the MS soil under controlled conditions. Dissipation was more rapid under field conditions than under laboratory conditions at two of the three locations. Fluometuron half-lives in soils from the 0 to 8 cm depth ranged from 9 to 28 d under field conditions and from 11 to 43 d in the laboratory. Fluometuron dissipation in soils from 30- to 45- and 60- to 90-cm was not different among soils, with half-lives ranging from 58 to 99 d under laboratory conditions. Fluometuron half-life was positively correlated with soil depth and inversely correlated with organic matter. Field dissipation of fluometuron was slower under dry conditions. Fluometuron did not move below 15 cm in the soil profile in any soil and concentrations in the 8 to 15 cm soil zone were < 15 ppbw 112 days after treatment. These data indicate that organic matter, soil depth, and environmental conditions affect fluometuron dissipation, and fluometuron migration into subsurface zones is limited based on data obtained from soil core sampling. Nomenclature: Fluometuron, *N,N*-dimethyl-*N'*-(3-trifluoromethyl)phenyl]-urea; cotton, *Gossypium hirsutum* L.

Additional index words: Degradation, adsorption, herbicide, cotton, movement, depth, leaching.

INTRODUCTION

Fluometuron is a widely used substituted urea herbicide for control of weeds in cotton and sugarcane. Fluometuron may be applied PRE or POST at 1.12 to 2.24 kg ai ha⁻¹ and is more readily absorbed by plant roots than by foliage. Fluometuron inhibits photosynthesis in sensitive species by binding the D₁ protein in chloroplast thylakoid membranes, thus blocking electron transport (2). Fluometuron alone, or in combination with other herbicides, is an effective component in cotton management systems. Fluometuron alone reduced weed biomass and increased seed cotton yield (14). Banded applications of fluometuron reduced weed biomass 28 to 47% and broadcast applications further reduced weed biomass (15). Weed density reductions were dependent on fluometuron application, but not on cultivation. However, optimum seed cotton yield was obtained with a PRE application of fluometuron followed by one cultivation.

In recent years, awareness of a pesticide's potential for contamination of groundwater has increased. Herbicide degradation and movement in soil is a function of several biotic and abiotic factors including sorption, soil microorganisms, soil organic matter and clay content, and herbicide application rate. Fluometuron is considered moderately persistent and has been shown to be subject to leaching in low adsorptive soils (3). Fluometuron adsorption is influenced primarily by organic matter content, and to a

lesser degree by cation exchange capacity (CEC²) (5). Bouchard, et al. (3) found higher distribution coefficients (K_d ²) in the 10- to 20-cm soil zone than at 40- to 50-cm, due to lower organic matter contents in the lower soil zone. Darding and Freeman (7) used a bioassay to show that fluometuron phytotoxicity to oats was significantly correlated with organic matter content and CEC. Fluometuron is degraded by soil microorganisms, and cold, dry soil conditions may slow fluometuron dissipation. Rogers, et al. (12) found that increasing temperature favored more rapid fluometuron dissipation in three soils, with half-life ranging from 26 to 73 d. Fluometuron moved to a depth of 60-cm in all three soils; however, > 55% of applied remained in the upper 15-cm of the soil profile. Downward movement of fluometuron was greatest in a sandy soil with low organic matter. Mueller, et al. (11) observed decreasing degradation rates in subsurface soil. Decreased degradation was correlated with declines in microbial respiration and biomass, but not with adsorption.

The objectives of this research were to determine the adsorption, dissipation under field and controlled conditions, and herbicide distribution within the soil profile of fluometuron in three soils representative of the cotton growing region of the southeastern United States.

²Abbreviations: CEC, cation exchange capacity; K_d , adsorption distribution coefficient; DT₅₀, half-life or 50% disappearance in days; DAT, days after treatment; DMFM, desmethyl fluometuron; HPLC, high-pressure liquid chromatography.

MATERIALS AND METHODS

Field study. Field experiments were conducted in 1992 and 1993 at Jackson, TN on a Lexington silt loam (fine-silty, mixed thermic Typic Paleudalf), at Stoneville, MS on a Beulah silt loam (coarse-loamy, mixed thermic Typic Dystocrept), and at Griffin, GA on a Dothan loamy sand (Plinthic Paleudalf).

A randomized complete block design was utilized and treatments were replicated four times. Plots were eight rows wide, with rows spaced 102-cm wide, by 9 m long. Standard fertility and pest management practices were employed at each location, and all plots were tilled. Trifluralin was applied at 0.84 kg ai ha⁻¹ as a preplant incorporated treatment for grass control. Cotton was planted prior to fluometuron treatment. Fluometuron was applied at 1.7 kg ai ha⁻¹ in a water carrier with tractor mounted research sprayers in early to mid-May at each location. An intense rainfall event at GA in 1993 necessitated a separate, additional planting and herbicide application on July 15. An untreated check was included in the experimental design, and plots were maintained weedfree by hand-hoeing.

The eight row plots were divided into two four row sections and were sampled in one of two ways: surface (0 to 8 cm) soil samples were collected from each plot prior to and immediately following fluometuron application, and at 14, 28, 56, and 112 days after treatment (DAT²) at Griffin, Stoneville, and Jackson each year using hand-held soil probes. These samples were utilized to examine dissipation in the surface soil under field conditions.

Two full-length depth cores from the 0 to 120 cm zone were collected at 28, 56,

and 112 DAT at TN and MS with tractor-mounted hydraulic probes equipped with acetate tube liners in order to assess fluometuron distribution within the profile. A tightly compacted clay layer present at the GA location limited sampling depth to 60-cm. This sampling procedure was utilized to maintain the integrity of the subsurface zone data. Preliminary data indicated that unless the sample was collected on the day of application that subsurface soil zone samples were not contaminated using this sampling procedure (data not shown). All soil samples were stored at -10 C within 1 hr of a sampling event. Surface soil samples were passed through a 2-mm sieve prior to chemical analysis. Samples obtained from the 0 to 120 cm soil zone at TN and MS were divided into 0- to 8-, 8- to 15-, 15- to 30-, 30- to 45-, 45- to 60-, 60- to 90-, and 90- to 120-cm soil zones. Samples obtained from the 0-60 cm soil zone at GA were divided into 0- to 8-, 8- to 15-, 15- to 30-, 30- to 45-, and 45- to 60-cm soil zones. Each section was homogenized by being passed through a 2-mm sieve prior to analysis.

Soil (40 g oven dry weight basis) was placed into 250-ml polyethylene bottles equipped with screw-top caps and extracted with 80-ml methanol for 16 h using a reciprocating shaker. The soil extracts were allowed to settle for 1 hr and then filtered through two qualitative filter papers³ and an aliquot placed into 4-ml vials. Fluometuron was quantified with High Pressure Liquid Chromatography (HPLC²) utilizing fluorescence detection as described previously (10). Herbicide concentrations were based on detectable herbicide peak areas and soil moisture data. The analytical method utilized quantifies fluometuron and its initial metabolite, desmethyl fluometuron (DMFM²). Fluometuron

³Whatman, Clinton, NJ.

recoveries were > 90% in all soils and the limit of detection was 15 ppbw (data not shown).

Physical and chemical analysis of the soil was performed at Knoxville, TN (Table 1). Soil pH was determined using a 1:1 soil-water suspension (8). Particle size analysis was performed utilizing the hydrometer method (1). Total organic carbon (OC) was determined by total combustion and values were converted to percent organic matter by the equation % organic matter = $0.35 + (1.80 \times \% \text{OC})$. Cation exchange capacities were determined by direct displacement with distillation (6).

Adsorption study. Adsorption of fluometuron onto three soils was determined using a soil-slurry technique as described by Talbert and Fletchall (17). Untreated soil was collected in 1993 from plots at each location. Soil from the 0- to 8-, 30- to 45-, and 60- to 90-cm soil zones was air dried and screened through a 2-mm sieve, and 10 g of soil was added to 50-ml screw-top centrifuge tubes. Twenty ml of 0.01 M CaCl_2 solution containing either 0 or $1 \mu\text{g ml}^{-1}$ analytical fluometuron was added to each tube. Tubes were placed on a reciprocating shaker and equilibrated for 24 h at 22 C. Tubes were then centrifuged at 5000 g for 12 min and filtered through two pieces of qualitative filter paper and an aliquot placed into 4-ml screw-top vials. Fluometuron concentration was determined as previously described. Centrifuge tubes containing no soil were included, but no adsorption to the tubes was detected (data not shown). The differences in herbicide concentration from the initial was assumed to be adsorbed to the soil. A distribution coefficient (K_d) for fluometuron was determined for each state and depth combination. Fluometuron adsorption was analyzed utilizing a completely randomized

Table 1. Physical and chemical characteristics of three southern soils from Tennessee (Lexington silt loam), Mississippi (Beulah silt loam), and Georgia (Dothan loamy sand).

State	Depth	pH ^a	Organic matter ^b	CEC ^c	Sand ^d	Silt ^d	Clay ^d
	cm		%	cmol kg ⁻¹	%		
TN	0-8	6.0	2.3	9.4	10	73	17
TN	8-15	5.6	1.6	7.4	6	78	16
TN	15-30	6.4	1.0	12.9	11	63	26
TN	30-45	5.8	0.8	14.8	7	63	30
TN	45-60	5.1	0.7	12.7	9	63	28
TN	60-90	5.2	0.7	9.8	12	65	23
TN	90-120	4.7	0.5	10.5	32	47	21
MS	0-8	6.2	1.9	7.5	45	48	7
MS	8-15	5.7	1.0	8.6	43	49	8
MS	15-30	5.5	1.0	10.0	30	59	11
MS	30-45	7.0	1.1	12.4	15	75	10
MS	45-60	5.8	1.0	10.0	14	69	17
MS	60-90	6.8	1.0	12.8	26	62	12
MS	90-120	6.7	0.8	6.5	59	26	15
GA	0-8	5.2	1.3	2.6	73	18	9
GA	8-15	5.4	1.4	2.9	74	16	10
GA	15-30	5.0	1.2	3.0	67	17	16
GA	30-45	5.2	1.0	7.3	52	16	32
GA	45-60	4.8	0.6	6.7	42	16	42

^aSoil pH was determined using a 1:1 soil to water suspension.

^bSoil organic matter content was determined by total combustion and conversion of values to organic matter.

^cCation exchange capacity was determined by displacement and distillation.

^dParticle size analyses were performed using the hydrometer method.

design with three replications. Treatment means were separated by LSD at the 5% significance level.

Laboratory degradation study. Untreated soil with no prior fluometuron exposure was collected in 1993 from the same soils which were utilized for the field experiment. Soil from the 0- to 8-, 30- to 45-, and 60- to 90-cm soil zones was examined from the Tennessee and Mississippi locations, while the 0- to 8- and 30- to 45-cm zones were utilized from the Georgia location. All soils were passed through a 2-mm sieve. Moist soil (40 g oven dry weight equivalent) was added to 250-ml polyethylene bottles with screw-top caps. The soil water status was adjusted to -33 kPa, which was sufficient for microbial activity. Stock solutions ($40 \mu\text{g ml}^{-1}$) of analytical grade fluometuron in water were prepared. Bottles were fortified with 1 ml of fluometuron stock solution to achieve 1 ppmw soil. All bottles were then capped and thoroughly mixed to incorporate the herbicide into the soil. Soils were incubated in the dark at 30 C and transferred to a freezer (-10 C) after the appropriate interval. Soils were incubated for 0, 7, 14, 21, 28, 42, and 56 d after herbicide addition. At 28 d after herbicide addition, all bottles remaining in the incubator were vented for 30 min to prevent the development of anaerobic conditions inside the bottles. After the experimental time course was completed, all samples were simultaneously extracted and analyzed as previously described.

All field and laboratory degradation data were fit to first-order kinetics using the PCSAS NLIN procedure (4, 16). The natural logarithm of fluometuron concentration in ppbw soil was regressed against time in days, and the NLIN procedure calculated first-

order rate constant and 95% confidence intervals around the data. The corrected r^2 of each line was calculated by the equation $[1 - (\text{residual sums of squares}/\text{corrected total sums of squares})]$.

RESULTS AND DISCUSSION

Field study. Fluometuron was not detected in samples collected prior to herbicide application (data not shown). Differences in initial fluometuron concentration at locations was hypothesized to be a function of sampling depth, soil type, and individual sampling technique (Figures 1, 2). First-order kinetics empirically fit fluometuron dissipation in the 0- to 8-cm soil zone under field conditions, with $r^2 \geq 0.93$ (Table 2). In 1992, fluometuron dissipation was more rapid in GA than in MS (Figure 1). Fluometuron DT_{50} in TN, MS, and GA in 1992 was 12, 28, and 9 d, respectively. In 1993, fluometuron dissipation was not different among locations (Figure 2), with DT_{50} values ranging from 12 to 26 d (Table 2). Dissipation was more rapid at GA in 1992 than in 1993, probably due to greater rainfall at this location in 1992 (Table 3). The GA location was established in mid-July due to rainfall soon after the initial 1993 planting causing movement of fluometuron into adjacent plots. Fluometuron dissipation at MS and TN was not different among years (Table 2); however, fluometuron DT_{50} in the MS soil was 28 d in 1992, and 12 d in 1993. Precipitation at the TN location was similar both years, which may have contributed to similar DT_{50} values (Table 2).

In the sampling protocol which examined movement into the soil, fluometuron was not detected below 15 cm either year at any location (data not shown). Fluometuron

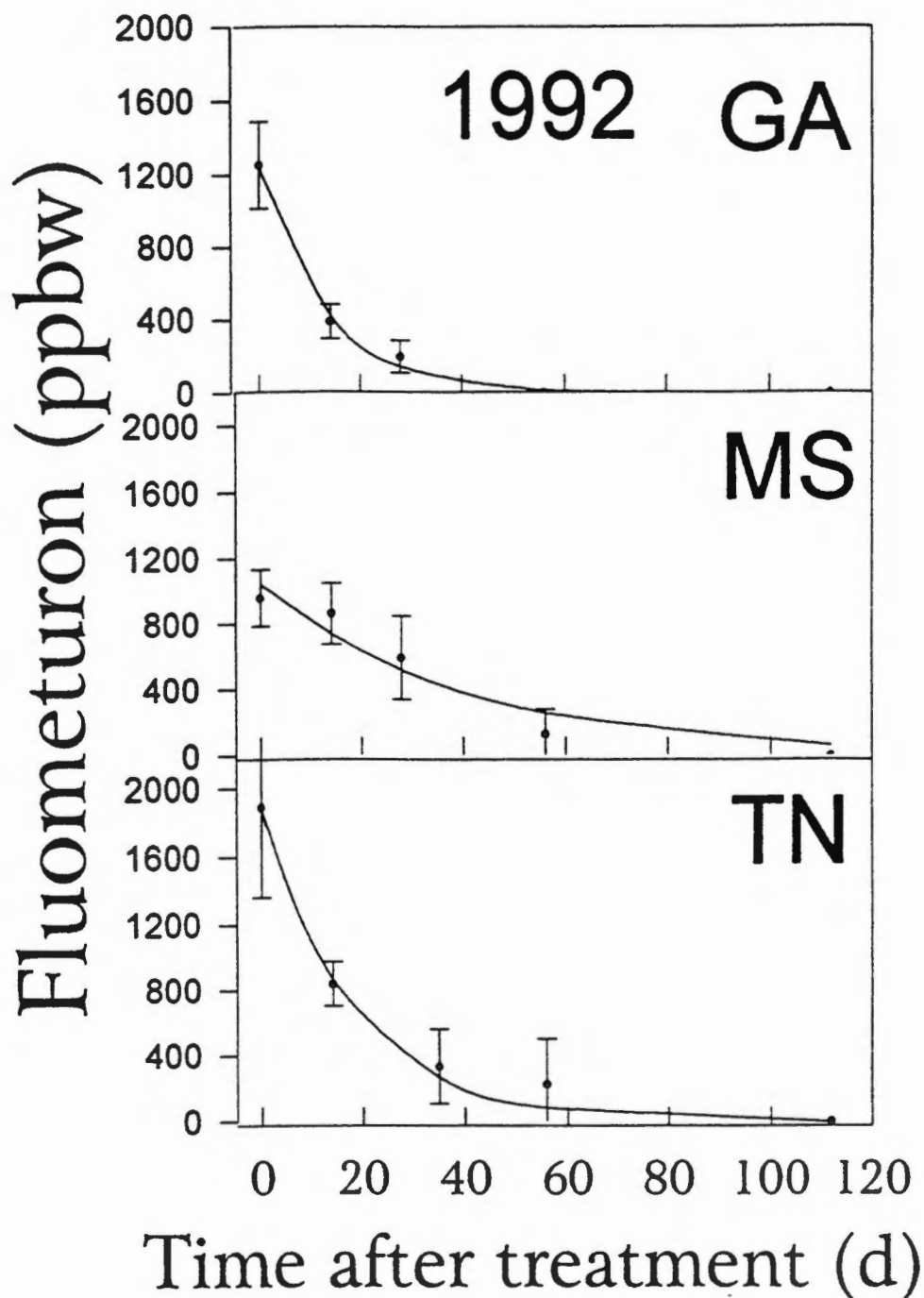


Figure 1. Change in surface fluometuron concentration over time in three southern soils in 1992. Data points are for the mean $\pm$ standard error of four replications. The predicted line for fluometuron is represented by the equation: $dC/dt = -kC$ (Table 2).

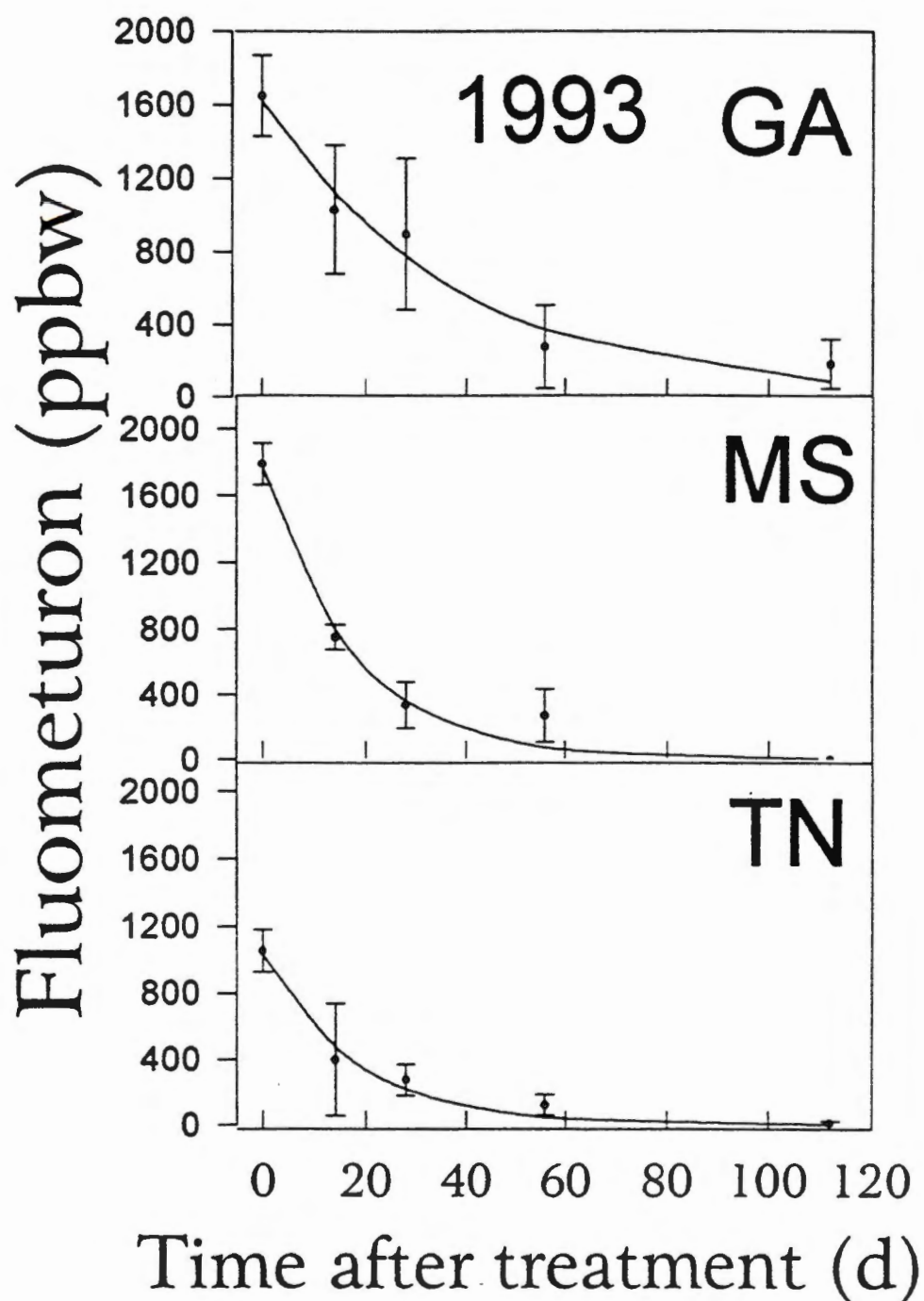


Figure 2. Change in surface fluometuron concentration over time in three southern soils in 1993. Data points are for the mean $\pm$ standard error of four replications. The predicted line for fluometuron is represented by the equation: $dC/dt = -kC$ (Table 2).

Table 2. Herbicide dissipation rate constants (K) and calculated half-lives (DT₅₀) for fluometuron in surface soils.

Soil	Year	K ^a	C.I. ^b	r ²	DT ₅₀
		day ⁻¹			d
TN	1992	0.056	(0.0386, 0.0726)	0.98	12
TN	1993	0.055	(0.0280, 0.0825)	0.98	13
MS	1992	0.025	(0.0042, 0.0456)	0.93	28
MS	1993	0.057	(0.0302, 0.0831)	0.98	12
GA	1992	0.077	(0.0598, 0.0950)	0.99	9
GA	1993	0.026	(0.0133, 0.0393)	0.97	26

^aFirst-order rate constant.

^bLower and upper limits of 95% confidence interval.

Table 3. Monthly precipitation during the 1992 and 1993 growing seasons at field locations in three southern states.

	Tennessee		Mississippi		Georgia	
	1992	1993	1992	1993	1992	1993
Month	cm					
April	6	14	5	17	5	7
May	6	18	6	17	4	7
June	28	11	14	10	10	4
July	13	3	10	7	10	5
August	8	6	11	8	22	0
September	11	11	8	11	9	8
October	3	4	2	7	7	2
November	8	17	12	7	31	9
Total	83	84	68	84	98	42

and DMFM concentrations at each depth were determined (Table 4). DMFM is the first metabolite of fluometuron, and it is liberated by microbial demethylation (10). In 1992, fluometuron concentration 28 DAT was greater at MS than at GA (Table 4). These data are in good agreement with 28 DAT fluometuron concentrations in the surface degradation study (Figure 1). DMFM concentration at GA in the 0 to 8 cm soil depth increased from 28 to 56 DAT, and decreased from 56 to 112 DAT (Table 4). At MS, DMFM disappearance was more rapid in 1993 than in 1992. This agrees well with fluometuron surface dissipation at this location (Table 2), since more rapid fluometuron dissipation in 1993 would promote faster DMFM formation and subsequent degradation. Fluometuron and DMFM concentrations in MS were < 55 ppbw 112 DAT (Table 4). Fluometuron concentrations at TN in 1992 were 303 ppbw 42 DAT, and had decreased to 24 ppbw by 112 DAT. At TN, the maximum DMFM concentration occurred earlier in 1993 than in 1992; however, faster DMFM formation was not an indication of more rapid fluometuron dissipation. Fluometuron concentrations in soil from the 8- to 15-cm depth were < 11 ppbw 56 DAT in all soils and was not detected 112 DAT. DMFM concentrations at the 8 to 15 cm depth in all soils were < 25 ppbw 112 DAT. In 1993, fluometuron concentrations 28 DAT at the 0 to 8 cm depth were 393, 231, and 600 ppbw at TN, MS, and GA, respectively. Fluometuron was not detected 112 DAT at TN and MS, but was 218 ppbw at GA, an indication that dry environmental conditions at GA in 1993 reduced fluometuron dissipation (Table 3). In 1993, fluometuron moved to the 8- to 15-cm depth by 28 DAT at TN, but was not present at 56 DAT (Table 4). Fluometuron was not detected at the 8- to 15-cm depth at GA until 56 DAT, and was

Table 4. Fluometuron (FLMT) and desmethyl fluometuron (DMFM) concentrations at various sampling intervals at two depths in three southern soils.

State	DAT ^a	Depth ^d	1992		1993	
			FLMT	DMFM	FLMT	DMFM
		cm	ng g ⁻¹			
TN	28	0-8			393 (198) ^b	204 (44)
TN	28	8-15			45 (90)	21 (42)
TN	42	0-8	303 (177)	438 (215)		
TN	42	8-15	34 (69)	108 (167)		
TN	56	0-8			208 (154)	142 (45)
TN	56	8-15			ND ^c	34 (48)
TN	63	0-8	172 (119)	498 (224)		
TN	63	8-15	ND	44 (34)		
TN	112	0-8	24 (29)	229 (164)	ND	26 (36)
TN	112	8-15	ND	20 (11)	ND	24 (35)
MS	28	0-8	375 (301)	151 (99)	231 (169)	75 (14)
MS	28	8-15	105 (150)	58 (98)	ND	6 (12)
MS	56	0-8	38 (75)	37 (74)	21 (40)	52 (15)
MS	56	8-15	ND	44 (88)	ND	ND
MS	112	0-8	53 (44)	144 (61)	ND	ND
MS	112	8-15	ND	16 (33)	ND	7 (9)
GA	28	0-8	201 (162)	ND	600 (249)	51 (62)
GA	28	8-15	ND	ND	ND	4 (8)
GA	56	0-8	50 (79)	99 (46)	275 (263)	95 (36)
GA	56	8-15	10 (18)	7 (11)	27 (53)	20 (29)
GA	112	0-8	15 (27)	55 (34)	218 (132)	63 (25)
GA	112	8-15	ND	23 (28)	13 (24)	ND

^aDAT, days after treatment.

^bValues are the mean of four replications followed by the standard error in parentheses.

^cND, fluometuron not detected.

^dFluometuron not detected at 15-30 cm or > depths.

present at 13 ppbw 112 DAT. Fluometuron was not detected in the 8- to 15-cm soil zone at MS in 1993.

Adsorption study. Fluometuron adsorption in the TN soil was greater at the 0 to 8 cm soil depth ($K_d = 1.69$) than at subsurface depths (Table 5). This decrease in adsorption at lower soil depths was probably due to decreasing organic matter in subsurface zones (Table 1), since adsorption in the 30- to 45-cm zone was not different from that at the 60- to 90-cm depth. Fluometuron adsorption at the 0- to 8-cm depth in the MS soil ($K_d = 1.73$) was greater than at 60- to 90-cm, but not different from 30- to 45-cm (Table 5). Fluometuron adsorption at GA did not decrease as depth increased, due to a more narrow range of organic matter between soil depths at this location (Table 1). Fluometuron adsorption at each soil depth was greater at the TN and MS sites compared to the GA site (Table 5).

Fluometuron adsorption was correlated with organic matter content but not with other soil characteristics (Table 6). These data agree with previous reports which indicate fluometuron adsorption decreases with decreasing organic matter (3). Fluometuron adsorption has previously been correlated with CEC (4, 9, 11, 13), but this research did not indicate a relationship.

Laboratory degradation study. First-order kinetics empirically fit the fluometuron dissipation data at each soil by depth regime under controlled conditions ($r^2 \geq 0.82$) (Table 7). Differences in dissipation rates were determined by comparing confidence intervals around first-order rate constants. The closed test system utilized decreased other dissipation mechanisms such as photodegradation, volatilization, and leaching; therefore,

Table 5. Distribution coefficients (K_d) of fluometuron adsorption onto three southern soils.

Depth	K_d		
	Tennessee	Mississippi	Georgia
cm	cm ³ per g soil		
0-8	1.69	1.73	0.56
30-45	0.64	1.11	0.31
60-90	0.35	1.00	0.22
LSD ^a = 0.71			

^aLeast significant difference at the 0.05 probability level.

Table 6. Correlation coefficients of various experimental parameters. Probabilities are denoted at the 5% (*) and 1% (**) significance level.

	pH	OM	CEC	Sand	Silt	Clay	K _d	DT ₅₀
Depth	0.13	-0.78*	0.56	-0.43	0.30	0.34	-0.52	0.94**
pH		0.17	0.58	-0.40	0.63	-0.53	0.61	0.04
OM			-0.33	0.09	0.09	-0.46	0.85**	-0.81*
CEC				-0.86**	0.76*	0.29	0.09	0.48
Sand					-0.92**	-0.27	-0.20	-0.31
Silt						-0.14	0.46	0.16
Clay							-0.61	0.38
K _d								-0.54

Table 7. Herbicide dissipation rate constants (K) and calculated half-lives (DT₅₀) for fluometuron under laboratory conditions.

Soil	Depth	K ^a	C.I. ^b	r ²	DT ₅₀
	cm	day ⁻¹			d
TN	0-8	0.064	(0.0393, 0.0886)	0.96	11
TN	30-45	0.011	(0.0078, 0.0152)	0.94	63 ^c
TN	60-90	0.007	(0.0030, 0.0111)	0.82	99 ^c
MS	0-8	0.016	(0.0116, 0.0207)	0.96	43
MS	30-45	0.012	(0.0079, 0.0163)	0.93	58 ^c
MS	60-90	0.008	(0.0062, 0.0104)	0.96	87 ^c
GA	0-8	0.031	(0.0191, 0.0420)	0.94	22
GA	30-45	0.010	(0.0050, 0.0159)	0.85	69 ^c

^aFirst-order rate constant.

^bLower and upper limits of 95% confidence interval.

^cDT₅₀ values > 56 d are estimated by extrapolation, since they are beyond the incubation timecourse.

degradation was assumed to be the primary loss mechanism.

Fluometuron dissipation in soil from TN and GA was greater at the 0- to 8-cm soil depth than at lower depths (Table 7). At MS, fluometuron dissipation was more rapid in the 0- to 8-cm soil zone than at a depth of 60 to 90 cm, but was not different from dissipation at the 30- to 45-cm horizon. Fluometuron concentrations at the 0- to 8-cm depth 56 DAT in TN, GA and MS soils were <15, 33, and 441 ng g⁻¹, respectively (Figures 3, 4, 5). These concentrations correspond to 0, 3, and 33% of initial concentration, respectively. Fluometuron DT₅₀ in the 0- to 8-cm soil zone was 11, 22, and 43 d for TN, GA and MS, respectively (Table 7). Fluometuron dissipation in soil from the 30- to 45-cm and 60- to 90-cm zone was not different among locations, with DT₅₀ values ranging from 58 to 99 d. Fluometuron dissipation rate in soil from 0 to 8 cm was more rapid in the TN soil compared to the MS soil, but was not different from the GA soil. Fluometuron DT₅₀ values > 56 d are estimated by interpolation, since they are beyond the incubation time course.

Fluometuron degradation under controlled conditions agreed well with DT₅₀ values in field degradation studies in the TN soil in 1992 and 1993, and the GA soil in 1993 (Table 2). At MS in 1992 and 1993, and at GA in 1992 fluometuron dissipated more rapidly in the field than under controlled conditions, an indication that environmental conditions or other loss mechanisms may have promoted more rapid dissipation at these locations.

Fluometuron DT₅₀ was positively correlated with soil depth and inversely related to organic matter (Table 6). Fluometuron half-life data were plotted versus soil organic

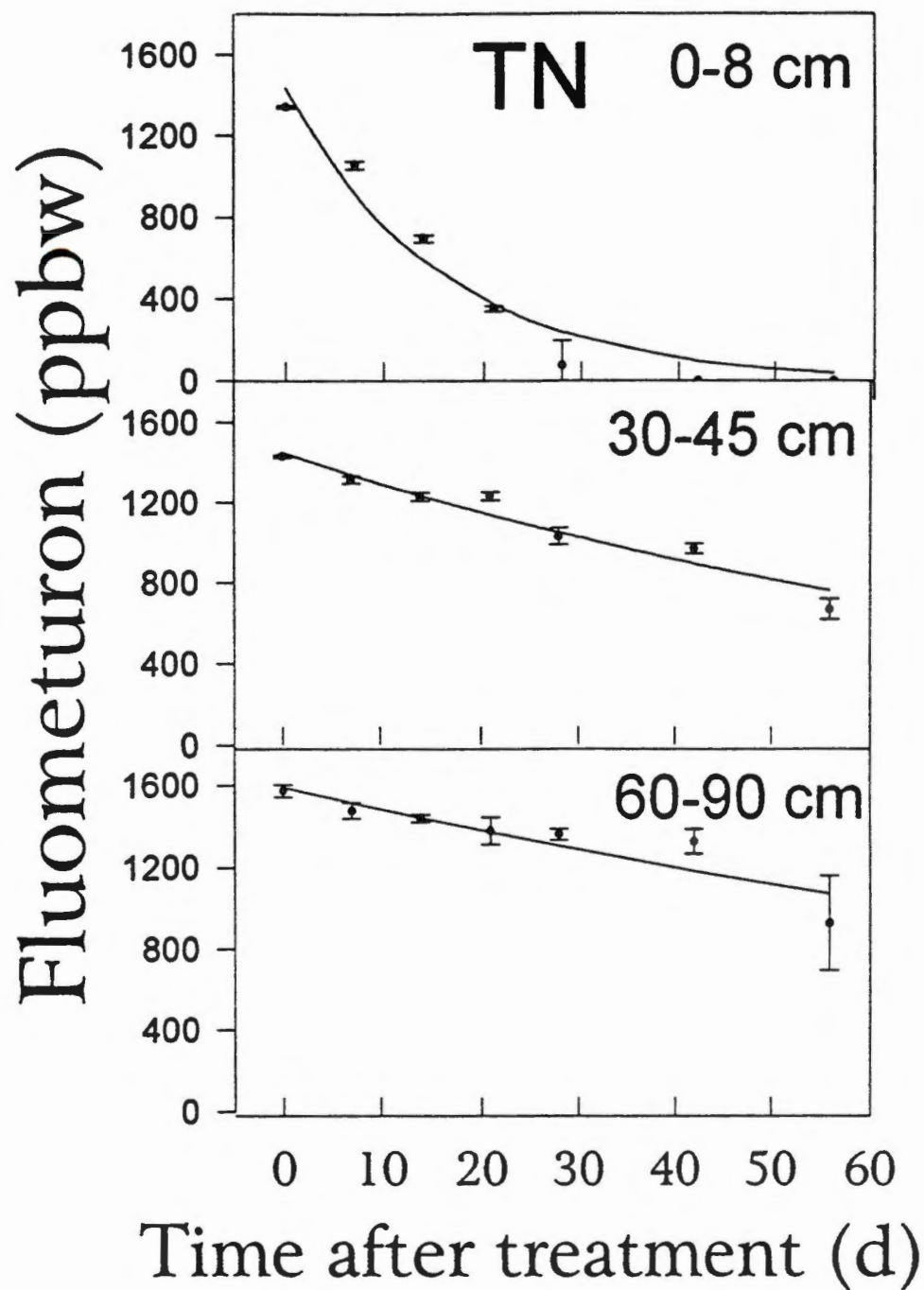


Figure 3. Change in fluometuron concentration over time in a Tennessee soil from 0 to 8, 30 to 45, and 60 to 90 cm at 30 C. Data points are for the mean $\pm$ standard error of three replications. The predicted line for fluometuron is represented by the equation: $dC/dt = -kC$ (Table 7).

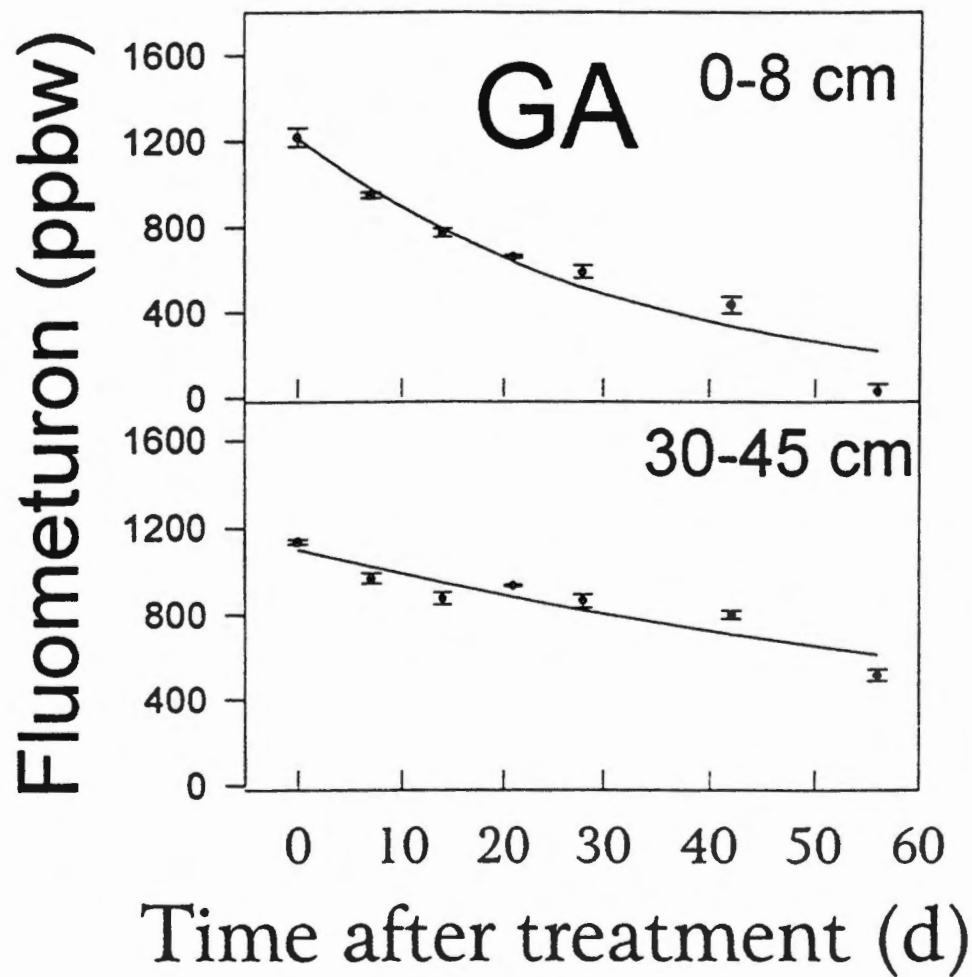


Figure 4. Change in fluometuron concentration over time in a Georgia soil from 0 to 8, and 30 to 45 cm at 30 C. Data points are for the mean $\pm$ standard error of three replications. The predicted line for fluometuron is represented by the equation: $dC/dt = -kC$ (Table 7).

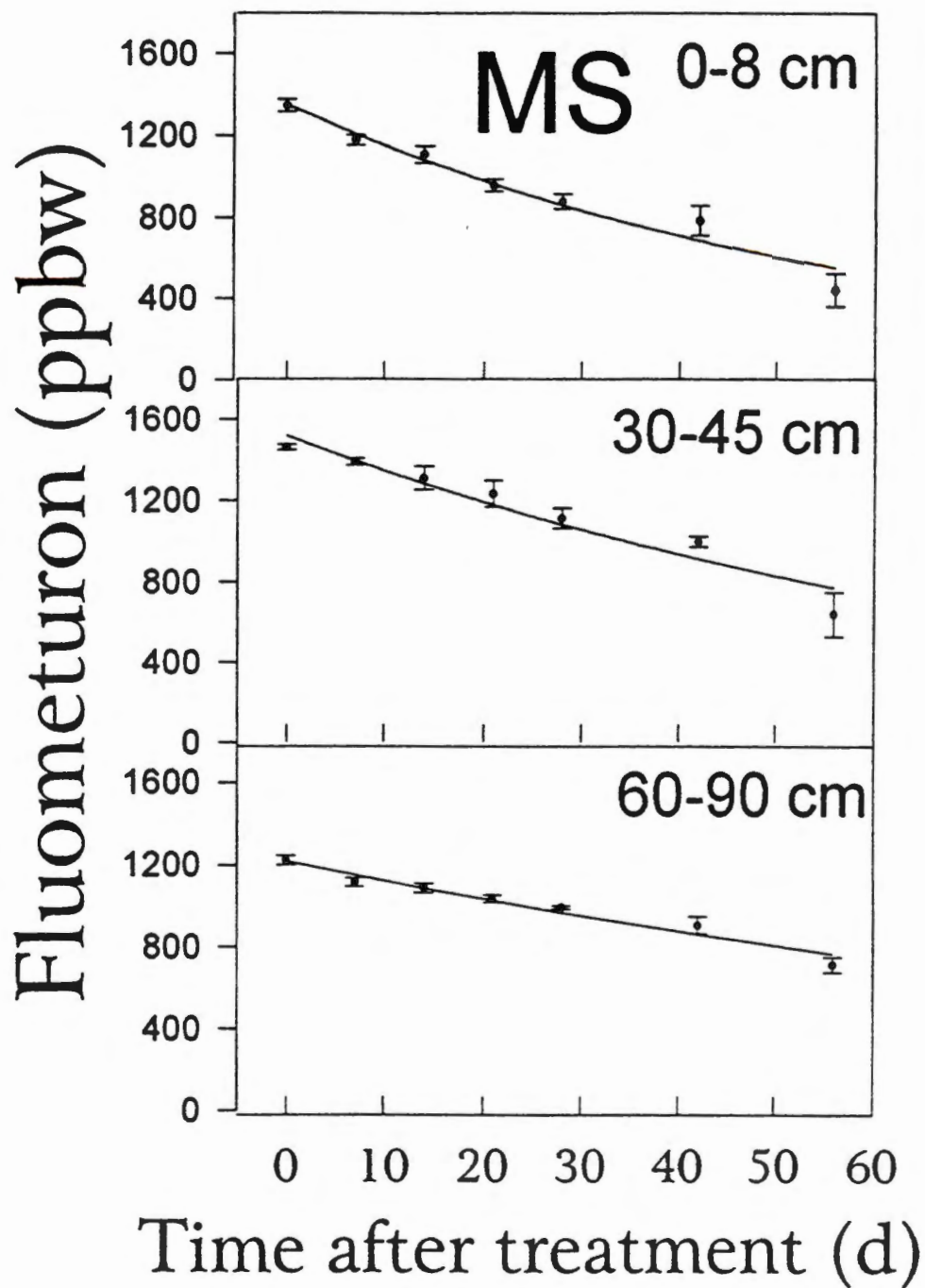


Figure 5. Change in fluometuron concentration over time in a Mississippi soil from 0 to 8, 30 to 45, and 60 to 90 cm at 30 C. Data points are for the mean $\pm$ standard error of three replications. The predicted line for fluometuron is represented by the equation: $dC/dt = -kC$ (Table 7).

matter, soil pH, and soil depth, to graphically illustrate these relationships (Figure 6). In this experiment, each organic matter unit (%) decrease caused fluometuron DT_{50} to increase by 44 d (Figure 6). Each cm increase in soil depth caused DT_{50} to increase by 0.96 d. A previous report indicated an inverse relationship between soil pH and fluometuron DT_{50} due to reduced microbial activity and/or lower microbial populations at lower pH (4). In this study, no relationship between DT_{50} and pH was evident, with a correlation coefficient of 0.02 (Figure 6). The reduced fluometuron dissipation as a function of organic matter may be due to lower microbial populations. Mueller, et al (11) reported decreased fluometuron degradation as soil depth increased, with decreased degradation being correlated with declines in microbial respiration and activity. At the 0- to 8-cm soil depth, organic matter at the TN location favored microbial activity, and thus fluometuron degradation, compared to the GA location (Table 1). The MS soil contained more organic matter in the 0- to 8-cm soil zone than did the GA soil; however, fluometuron adsorption was greater at MS, and this may have contributed to slower dissipation in this soil (Table 5).

In general, fluometuron concentration in soil was influenced by organic matter and soil depth. Decreased organic matter in subsurface soil zones contributed to decreased adsorption and slower dissipation. First-order kinetics explained fluometuron dissipation under both field and laboratory conditions; however, field dissipation was slower in most cases due to dry environmental conditions. Decreased adsorption in subsurface zones could potentially promote leaching of fluometuron; however, fluometuron movement in the soil profile did not significantly contribute to dissipation.

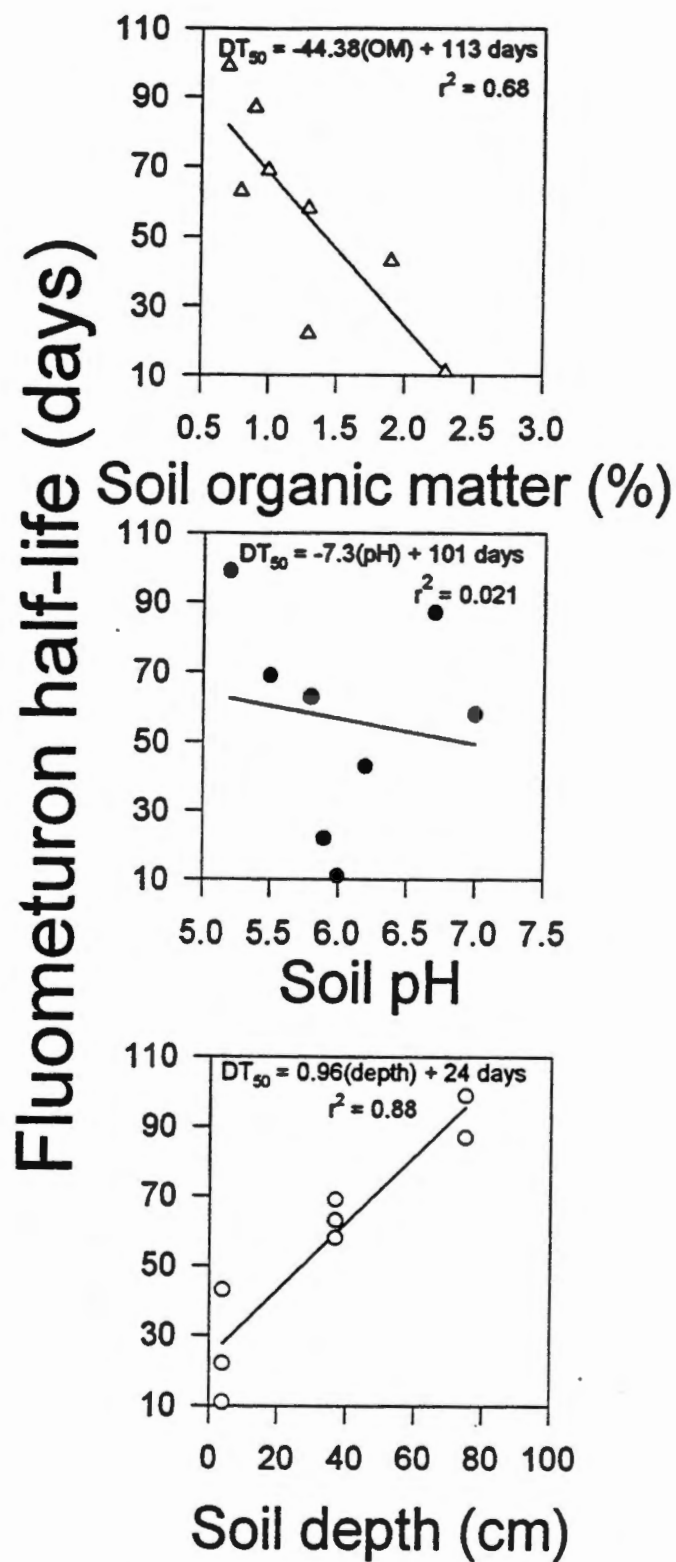


Figure 6. Fluometuron half-life (DT_{50}) in laboratory studies as influenced by soil organic matter, soil pH, and soil depth.

ACKNOWLEDGMENTS

The authors thank Ciba Plant Protection for supplying analytical fluometuron. This research was partially funded by the Southern Regional Pesticide Impact Assessment Project SR93-07-0-TN. Technical assistance by personnel at the West Tennessee Experiment Station, Jackson, TN; Delta Branch Experiment Station, Stoneville, MS; and Georgia Experiment Station at Griffin is gratefully acknowledged.

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PART IV.

NORFLURAZON ADSORPTION, DISSIPATION AND

MOVEMENT IN THREE SOUTHERN SOILS¹

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ABSTRACT

Norflurazon adsorption, dissipation under field and laboratory conditions, and distribution within the soil profile were determined in three soils representative of the cotton growing regions of the southeastern United States. Norflurazon adsorption was greater in soil from 0 to 8 cm in a Lexington silt loam (TN) and a Beulah silt loam (MS) than in a Dothan loamy sand (GA). Adsorption was directly related to organic matter. Norflurazon degradation under controlled conditions in soil from 0 to 8 cm from each state was not different among locations, with half-lives ranging from 63 to 167 d. Degradation at 30 C in soil from the 30- to 45-cm and 60- to 90-cm depths was not different among locations; and was slower at the 60- to 90-cm depth. Norflurazon dissipation was more rapid under field conditions than under laboratory conditions with half-lives ranging from 7 to 79 d in the 0 to 8 cm soil horizon. Dry field conditions slowed norflurazon dissipation. Norflurazon was not detected below 15 cm in the soil profile in any soil based on soil core data, and concentrations in the 8 to 15 cm soil zone were < 36 ppbw 112 days after treatment. These data indicate a significant effect of environment on norflurazon dissipation, and that norflurazon leaching was minimal in these soils under these field conditions.

Nomenclature: Norflurazon, [4-chloro-5-(methylamino)-2-(3-(trifluoromethyl)phenyl)-3(2*H*)-pyridazinone]; cotton, *Gossypium hirsutum* L.

Additional index words: Degradation, adsorption, herbicide, cotton, movement, depth.

INTRODUCTION

Norflurazon is a pyridazinone herbicide used for control of annual grasses, sedges, and broadleaf weeds in cotton, soybeans, cranberries, and tree and nut crops (2).

Norflurazon is usually applied PRE at 0.56 to 2.24 kg ai ha⁻¹ in cotton, with application rate dependent upon soil type. Norflurazon effectively controls susceptible species by inhibiting carotenoid biosynthesis (4). Inhibition of carotenoid formation results in chlorophyll degradation and subsequent plant chlorosis and necrosis (2). Norflurazon is an important component in cotton management systems in the southeastern cotton belt, especially in areas where spurred anoda [*Anoda cristata* (L.) Schlecht.], velvetleaf (*Abutilon theophrasti* Medik.), and nutsedges (*Cyperus* spp.) are problem weeds.

In recent years, awareness of the potential for contamination of groundwater by pesticides has increased. Herbicide degradation and movement in soil is a function of several biotic and abiotic factors including sorption, soil organic matter and clay content, and herbicide application rate. Norflurazon is considered slightly mobile in the soil system. Schroeder and Banks (16) detected > 60% of applied norflurazon in the top 8 cm of the soil profile 110 DAT²; and < 12% below 15 cm. Hubbs and Lavy (8) also observed limited mobility on soil thin layer plates. Norflurazon dissipation is biphasic in nature, with a rapid initial loss preceding a slow decline of the remaining herbicide (16). Norflurazon half-life in soil has ranged from 45 to 270 d, depending on soil and

²Abbreviations: DAT, days after treatment; HPLC, high-pressure liquid chromatography; K_d, adsorption distribution coefficient; DT₅₀, half-life or 50% disappearance time in days; CEC, cation exchange capacity.

environmental factors (2, 16). Norflurazon is considered moderately persistent, and may injure succeeding crops of wheat, corn or sorghum (9, 16). Repeated applications of 1.1 kg ai ha⁻¹ increased norflurazon residues and rotational crop injury in a loamy sand with < 1.0% organic matter (8). Residual carryover of norflurazon is related to organic matter content (15, 16), with reduced dissipation reported at higher organic matter contents. This is due, at least in part, to the adsorptive characteristics of organic matter which may reduce the amount of potentially degradable norflurazon. Cool, dry environmental conditions can slow norflurazon dissipation. Rahn and Zimdahl (14) extended norflurazon half-life in a sandy loam soil by lowering soil temperature.

The objectives of this research were to determine the adsorption, dissipation under field and controlled conditions, and herbicide distribution within the soil profile of norflurazon in three soils representative of the cotton growing region of the southeastern United States.

MATERIALS AND METHODS

Field study. Field experiments were conducted in 1992 and 1993 at Jackson, TN on a Lexington silt loam (Fine-silty, mixed thermic Typic Paleudult); Stoneville, MS on a Beulah silt loam (Coarse-loamy, mixed thermic Typic Dystocrept); and Griffin, GA on a Dothan loamy sand (Plinthic Paleudult) utilizing a randomized complete block design with treatments replicated four times. Plots were eight rows wide, with rows spaced 102-cm wide, by 9 m long. Standard fertility and pest management practices were employed at each location, and all plots were conventionally tilled. Trifluralin was applied at 0.84 kg

ha⁻¹ as a preplant incorporated treatment for grass control. Cotton was planted prior to norflurazon treatment. Norflurazon was applied at 1.7 kg ai ha⁻¹ in a water carrier with tractor mounted research sprayers. An untreated check was included in the experimental design, and plots were maintained weedfree by hand-hoeing.

The eight row plots were divided into two four row sections and were sampled in one of two ways: surface (0 to 8 cm) soil samples were collected from each plot prior to and immediately following norflurazon application, and at approximately 14, 28, 56, and 112 (DAT²) using hand-held soil probes. These samples were utilized to examine surface norflurazon dissipation.

Two full-length depth cores from the 0 to 120 cm zone were collected at TN and MS at approximately 28, 56, and 112 DAT with tractor-mounted hydraulic probes equipped with acetate tube liners in order to assess norflurazon distribution within the profile. A tightly compacted clay layer at the GA location prohibited sampling beyond 60 cm. This sampling procedure was utilized to maintain the integrity of the subsurface zone data. Preliminary data indicated that unless the sample was collected on the day of application that subsurface soil zone samples were not contaminated using this sampling procedure (data not shown). All soil samples were stored at -10 C within 2 hr of a sampling event. Surface soil samples were passed through a 2-mm sieve prior to chemical analysis. Samples obtained from the 0 to 120 cm soil zone at TN and MS were divided into 0- to 8-, 8- to 15-, 15- to 30-, 30- to 45-, 45- to 60-, 60- to 90-, and 90- to 120-cm soil zones. Samples obtained from the 0- to 60-cm soil zone at GA were divided into 0- to 8-, 8- to 15-, 15- to 30-, 30- to 45-, and 45- to 60-cm soil zones. Each section was

passed through a 2-mm sieve prior to herbicide analysis.

Moist soil (40 g oven dry weight basis) was placed into 250-ml polyethylene bottles equipped with screw-top caps. Norflurazon was extracted with 80-ml methanol for 16 h using a reciprocating shaker. The soil extracts were allowed to settle for 1 hr and then filtered through two qualitative filter papers³ into 4-ml vials. Norflurazon was quantified with High Pressure Liquid Chromatography (HPLC)² and fluorescence detection as previously described (19). Herbicide concentrations were based on detectable herbicide peak areas and were corrected for soil moisture. Norflurazon recovery was > 90% in all soils and the limit of detection was 10 ppbw (data not shown). Physical and chemical analysis of the soil was performed at Knoxville (Table 1). Soil pH was determined using a 1:1 soil-water suspension. Particle size analysis was performed utilizing the hydrometer method (10). Total organic carbon (OC) was determined using a carbon analyzer and values were converted to percent organic matter by the equation % organic matter = $0.35 + (1.80 * \% \text{ OC})$ (1). Cation exchange capacities (CEC²) were determined by direct displacement and distillation (6).

Adsorption study. Adsorption of norflurazon onto three soils was determined using a soil-slurry technique as described by Talbert and Fletchall (18). Untreated soil was collected in 1993 from plots utilized in the field study. Soil from the 0- to 8-, 30- to 45-, and 60- to 90-cm soil zones was air-dried and screened through a 2-mm sieve, and 10 g of soil was added to 50-ml screw-top centrifuge tubes. Twenty ml of 0.01 M CaCl₂ solution containing either 0 or 1 µg ml⁻¹ analytical norflurazon was added to each tube.

³Whatman, Clinton, NJ.

Table 1. Physical and chemical characteristics of the soils from Tennessee, Mississippi and Georgia used in the field and laboratory studies.

State	Depth	pH ^a	Organic matter ^b	CEC ^c	Sand ^d	Silt ^d	Clay ^d
	cm		%	cmol kg ⁻¹	%		
TN	0-8	6.0	2.3	9.4	10	73	17
TN	8-15	5.6	1.6	7.4	6	78	16
TN	15-30	6.4	1.0	12.9	11	63	26
TN	30-45	5.8	0.8	14.8	7	63	30
TN	45-60	5.1	0.7	12.7	9	63	28
TN	60-90	5.2	0.7	9.8	12	65	23
TN	90-120	4.7	0.5	10.5	32	47	21
MS	0-8	6.2	1.9	7.5	45	48	7
MS	8-15	5.7	1.0	8.6	43	49	8
MS	15-30	5.5	1.0	10.0	30	59	11
MS	30-45	7.0	1.1	12.4	15	75	10
MS	45-60	5.8	1.0	10.0	14	69	17
MS	60-90	6.8	1.0	12.8	26	62	12
MS	90-120	6.7	0.8	6.5	59	26	15
GA	0-8	5.2	1.3	2.6	73	18	9
GA	8-15	5.4	1.4	2.9	74	16	10
GA	15-30	5.0	1.2	3.0	67	17	16
GA	30-45	5.2	1.0	7.3	52	16	32
GA	45-60	4.8	0.6	6.7	42	16	42

^aSoil pH was determined using a 1:1 soil to water suspension.

^bSoil organic matter content was determined by total combustion and conversion of values to organic matter.

^cCation exchange capacity was determined by displacement and distillation.

^dParticle size analyses were performed using the hydrometer method.

Tubes were placed on a reciprocating shaker and equilibrated for 24 h at 22 C. Tubes were then centrifuged at 5000 g for 12 min and filtered through two pieces of qualitative filter paper and an aliquot placed into 4-ml screw-top vials. Norflurazon concentration was determined as previously described. Centrifuge tubes containing no soil were included but no adsorption to the tubes was detected (data not shown). The differences in herbicide concentration from the initial was assumed to be adsorbed to the soil. A distribution coefficient (K_d) for norflurazon was determined for each state by depth combination.

Laboratory degradation study. Untreated soil with no norflurazon exposure for 3 yr prior to establishment of field studies was collected in 1993 from the same soils which were utilized for the field experiment. Soil from the 0- to 8-, 30- to 45-, and 60- to 90-cm soil zones was examined from the Tennessee and Mississippi locations, while the 0- to 8- and 30- to 45-cm zones were utilized from the Georgia location. All soils were passed through a 2-mm sieve. Moist soil (40 g oven dry weight equivalent) was added to 250-ml polyethylene bottles with screw-top caps. The soil moisture status was adjusted to -33 kPa, which is sufficient for microbial activity. Stock solutions ($40 \mu\text{g ml}^{-1}$) of analytical grade norflurazon in methanol:water (20:80 v/v) were prepared. Bottles were fortified with 1 ml of norflurazon stock solution to achieve 1 ppmw soil. All bottles were then capped and thoroughly mixed to incorporate the herbicides into the soil. Soils were incubated in the dark at 30 C and transferred to a freezer (-10 C) after the appropriate interval. Soils were transferred at 0, 7, 14, 21, 28, 42, and 56 d after herbicide amendment. At 28 d after herbicide amendment, all bottles remaining in the incubator

were vented for 30 min to prevent the formation of anaerobic conditions inside the bottles. After the experimental time course was completed, all samples were simultaneously extracted and analyzed as previously described.

Statistical Analysis. Norflurazon adsorption was analyzed utilizing a completely randomized design with three replications. Treatment means were separated by LSD at the 5% significance level. All field and laboratory dissipation data were fit to first-order kinetics using the PCSAS NLIN procedure (17). The natural logarithm of norflurazon concentration in ppbw of soil was regressed against time in days, and the NLIN procedure calculated first-order rate constant and 95% confidence intervals around the data. The corrected r^2 of each line was calculated by the equation $[1 - (\text{residual sums of squares} / \text{corrected total sums of squares})]$. Norflurazon dissipation has been reported to be biphasic in nature, with a rapid initial loss preceding slower dissipation of the remaining herbicide (15), thus a two-compartment model was also utilized to describe norflurazon dissipation (7).

RESULTS AND DISCUSSION

Field degradation study. Norflurazon was not detected in samples collected prior to herbicide application (data not shown). Differences in initial norflurazon concentration were hypothesized to be a factor of soil type, sampling depth, and individual sampling technique (Figures 1, 2). First-order kinetics empirically fit norflurazon in the 0- to 8-cm soil zone at the TN and GA locations, ($r^2 \geq 0.83$), and at MS in 1993 ($r^2 = 0.94$) but not at MS in 1992 ($r^2 = 0.43$) (Table 2). The two-compartment model more accurately

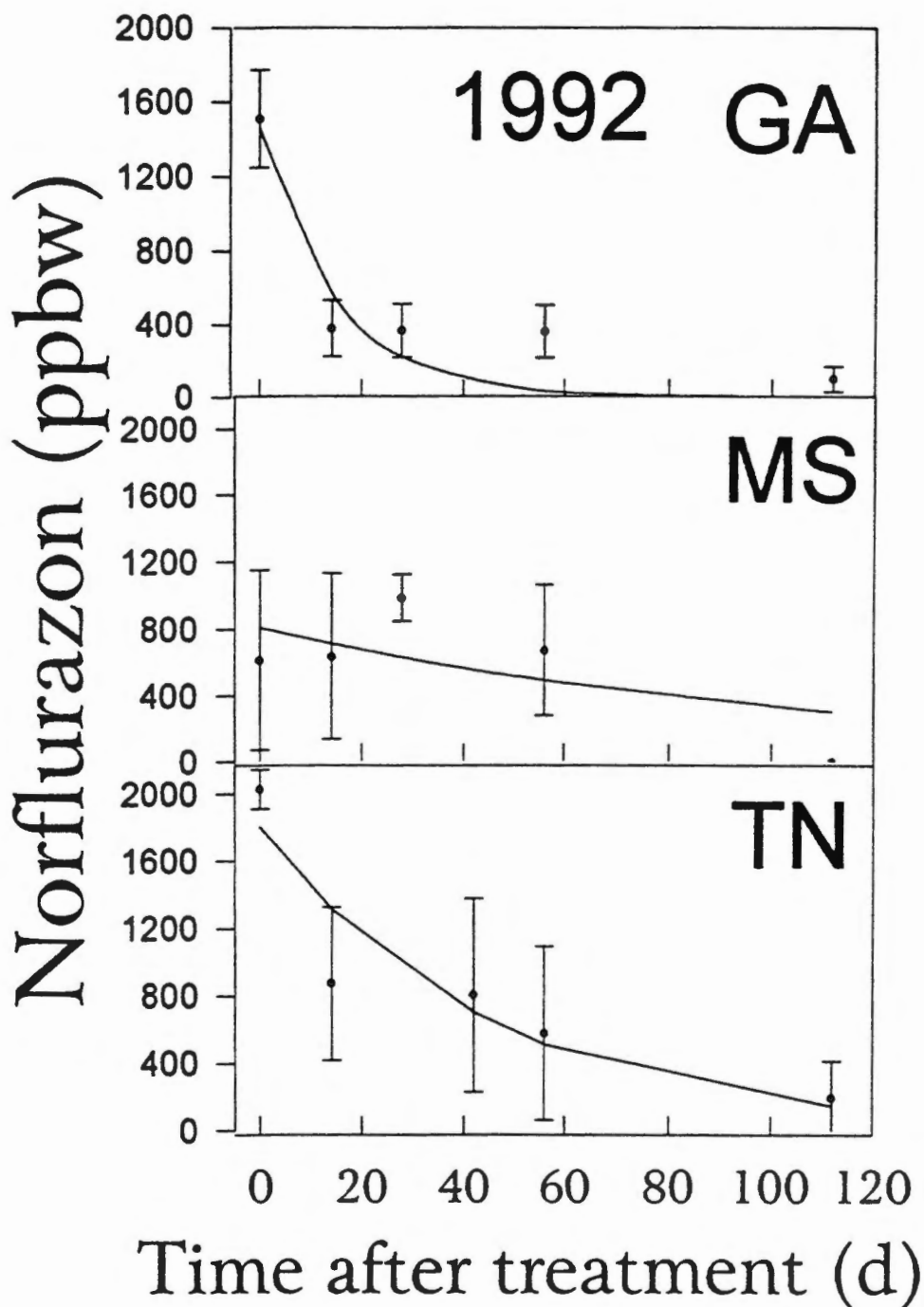


Figure 1. Change in surface norflurazon concentration over time in three southern soils in 1992. Data points are for the mean $\pm$ standard error of four replications. The predicted line for norflurazon is represented by the equation: $dC/dt = -kC$ (Table 2).

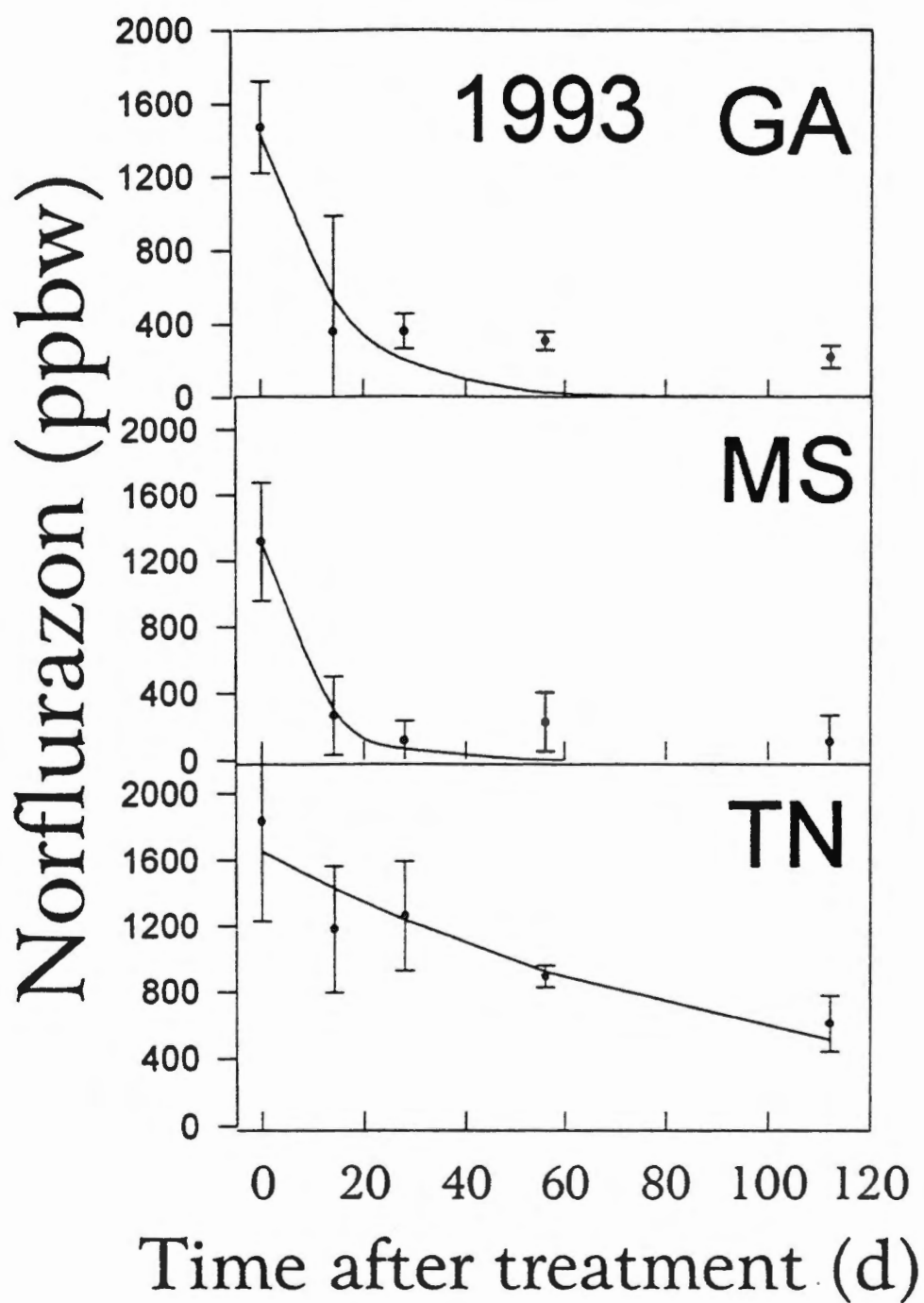


Figure 2. Change in surface norflurazon concentration over time in three southern soils in 1993. Data points are for the mean $\pm$ standard error of four replications. The predicted line for norflurazon is represented by the equation: $dC/dt = -kC$ (Table 2).

Table 2. Herbicide dissipation rate constants (K) and calculated half-lives (DT₅₀) in surface soils for norflurazon from field sites.

Soil	Year	K ^a day ⁻¹	C.I. ^b	r ²	DT ₅₀ d
TN	1992	0.023	(0.0062, 0.0388)	0.86	31
TN	1993	0.010	(0.0016, 0.0192)	0.87	66
MS	1992	0.009	(0.0178, 0.0353)	0.43	79
MS	1993	0.105	(0.0002, 0.2090)	0.94	7
GA	1992	0.067	(0.0135, 0.1470)	0.85	10
GA	1993	0.069	(0.0187, 0.1565)	0.83	10

^aFirst-order rate constant.

^bLower and upper limits of 95% confidence interval.

explained norflurazon dissipation under field conditions in 50% of the location by year regimes (data not shown). First-order kinetics more accurately explained norflurazon dissipation under field conditions in the remaining location by year regimes, and thus was considered a suitable technique for describing field dissipation. Norflurazon had little or no change in concentration from 20 to 60 DAT (Figures 1, 2). Dissipation was similar at GA both years, with < 25% of applied norflurazon remaining 14 DAT (Figures 1, 2). Norflurazon DT_{50}^2 values at GA were 10 d both years (Table 2). Norflurazon dissipation slowed from 14 to 112 DAT with < 225 ppbw remaining 112 DAT. Norflurazon dissipation at MS was more rapid in 1993 than in 1992, possibly due to greater precipitation at this location in 1993 (Table 3). Norflurazon dissipation at TN was similar both years, and norflurazon concentrations 112 DAT were 194 ppbw in 1992 and 611 ppbw in 1993 and DT_{50} values were 31 d in 1992 and 66 d in 1993 (Table 2).

In the sampling protocol which examined movement into the soil, norflurazon was not detected below 15 cm either year at any location (data not shown). Norflurazon concentrations at GA at 0- to 8-cm were 389 and 360 ppbw 28 DAT in 1992 and 1993, respectively (Table 4). Dissipation from 28 to 112 d was more rapid in 1992 than in 1993 which coincided with less rainfall in 1993 than 1992 at this location (Table 3). Concentrations 112 DAT were 120 and 251 ppbw in 1992 and 1993, respectively (Table 4). These data are in good agreement with 112 DAT concentrations from the surface dissipation study (Figures 1, 2). Norflurazon concentration in the 8- to 15-cm soil depth at GA was 15 ppbw 112 DAT in 1992 and was not detected in 1993 (Table 4). At the 0- to 8-cm depth, norflurazon dissipation at MS in 1993 was similar to concentrations

Table 3. Monthly precipitation during the 1992 and 1993 growing seasons at field locations in three southern states.

	Tennessee		Mississippi		Georgia	
	1992	1993	1992	1993	1992	1993
Month	cm					
April	6	14	5	17	5	7
May	6	18	6	17	4	7
June	28	11	14	10	10	4
July	13	3	10	7	10	5
August	8	6	11	8	22	0
September	11	11	8	11	9	8
October	3	4	2	7	7	2
November	8	17	12	7	31	9
Total	83	84	68	84	98	42

Table 4. Norflurazon concentration at various sampling intervals at two depths in three southern soils.

State	DAT ^a	Depth ^d cm	1992	1993
			norflurazon	
			ng g ⁻¹	
TN	28	0-8		1470 (370) ^b
TN	28	8-15		45 (60)
TN	42	0-8	329 (230)	
TN	42	8-15	ND ^c	
TN	56	0-8		941 (384)
TN	56	8-15		87 (101)
TN	63	0-8	710 (139)	
TN	63	8-15	ND	
TN	112	0-8	469 (591)	368 (74)
TN	112	8-15	ND	ND
MS	28	0-8	718 (342)	287 (102)
MS	28	8-15	81 (159)	36 (70)
MS	56	0-8	232 (48)	249 (136)
MS	56	8-15	37 (72)	29 (37)
MS	112	0-8	254 (192)	167 (125)
MS	112	8-15	36 (40)	ND
GA	28	0-8	389 (158)	360 (86)
GA	28	8-15	ND	ND
GA	56	0-8	118 (84)	295 (44)
GA	56	8-15	15 (18)	ND
GA	112	0-8	120 (210)	251 (108)
GA	112	8-15	15 (24)	ND

^aDAT, days after treatment.

^bvalues are means of four replications followed by standard error of the mean.

^cND, Norflurazon not detected.

^dnorflurazon not detected at 15-30 cm or > depths.

obtained from the surface dissipation study, with 167 ppbw remaining 112 DAT (Figure 2). Norflurazon was not detected at the 8- to 15-cm depth 112 DAT at either MS or TN. Norflurazon dissipation was slower at MS in 1992 with 254 ppbw remaining 112 DAT (Table 4). Dissipation was more rapid at TN in 1992 than in 1993. Cool and/or dry conditions can reduce norflurazon dissipation (13). Drier conditions at MS in 1992 may have slowed norflurazon dissipation (Table 3). Precipitation at TN was similar both years; thus, factors other than rainfall contributed to more rapid norflurazon dissipation in 1992.

Adsorption study. Norflurazon adsorption in a TN soil was greater at the 0- to 8-cm soil depth ($K_d = 9.50$) than at subsurface depths (Table 5). This decrease in adsorption at subsurface soil zones was probably due to decreased organic matter (Table 1), since adsorption in the 30- to 45-cm zone was not different from that at the 60- to 90-cm depth (Table 5). Norflurazon adsorption at the 0- to 8-cm depth in the MS soil was greater than at 60- to 90-cm, but not different from adsorption in the 30- to 45-cm zone. Norflurazon adsorption to the soil at GA was not different among soil depths, due to a more narrow range of organic matter between soil zones at this location (Table 1). At a soil depth of 0 to 8 cm, norflurazon adsorption was greater with the TN and MS soils as compared to the GA soil (Table 5). At the 30- to 45-cm depth, norflurazon adsorption was greater with the MS soil than either TN or GA soils. Norflurazon adsorption in 60- to 90-cm soil depth was greatest at MS, but no significant differences among location were detected (Table 5).

Norflurazon adsorption to soil was correlated with organic matter (Table 6).

Table 5. Distribution coefficients (K_d) for norflurazon adsorption onto three southern soils.

Depth	K_d		
	Tennessee	Mississippi	Georgia
cm	cm ³ per g soil		
0-8	9.50	9.29	2.46
30-45	2.67	5.89	1.29
60-90	1.72	4.37	0.87
LSD ^a = 3.88			

^aLeast significant difference at the 0.05 probability level.

Table 6. Correlation coefficients of various experimental parameters. Probabilities are denoted at the 5% (*) and 1% (**) significance level.

	pH	OM	CEC	Sand	Silt	Clay	K _d	DT ₅₀
Depth	0.13	-0.78*	0.56	-0.43	0.30	0.34	-0.55	0.80*
pH		0.17	0.58	-0.40	0.63	-0.53	0.55	0.58
OM			-0.33	0.09	0.09	-0.46	0.88**	-0.49
CEC				-0.86**	0.76*	0.29	0.05	0.57
Sand					-0.92**	-0.27	-0.21	-0.35
Silt						-0.14	0.46	0.43
Clay							-0.58	-0.19
K _d								-0.10

Norflurazon adsorption was not related with other soil parameters at any location. These data agree with previous reports that norflurazon adsorption increases as a function of increasing organic matter (3, 8, 13). Previous reports indicate a correlation between norflurazon adsorption and clay content (8); however, clay content was not correlated with norflurazon adsorption in these studies.

Laboratory degradation study. The closed test system utilized decreases important dissipation processes such as photodegradation, volatilization, and leaching; therefore, degradation was assumed to be the primary loss mechanism. First-order kinetics empirically fit the norflurazon degradation data in five of eight soil by depth regimes under controlled conditions ($r^2 > 0.70$) (Table 7). Confidence intervals were calculated around the first-order rate constant, and were used for comparison of differences in dissipation rate. Overlapping confidence intervals indicated no dissipation differences at the 5% significance level. Norflurazon dissipation has previously been reported to be biphasic, with rapid initial loss preceding slow loss of remaining norflurazon. Schroeder and Banks (16) reported no effect of application rate on norflurazon loss, which is one tenet of first-order kinetics. Concurrent research objectives include utilizing computer models to simulate norflurazon behavior. First-order dissipation rate constants are necessary inputs to the models (5, 12), and were calculated for this reason.

Norflurazon degradation rate in soil from 0 to 8 cm was not different among locations (Figures 3, 4, 5); however, degradation in the MS soil did not fit first-order kinetics, with $r^2 < 0.61$ (Table 7). The two-compartment model did not improve the overall fit of the dissipation data (data not shown), thus first-order kinetics were used to

Table 7. Herbicide dissipation rate constants (K) and calculated half-lives (DT₅₀) for norflurazon under laboratory conditions.

Soil	Depth	K ^a	C.I. ^b	r ²	DT ₅₀
	cm	day ⁻¹			d
TN	0-8	0.011	(0.0030, 0.0197)	0.75	63 ^c
TN	30-45	0.005	(0.0027, 0.0066)	0.89	139
TN	60-90	0.003	(0.0018, 0.0041)	0.89	222
MS	0-8	0.004	(0.0015, 0.0091)	0.42	167
MS	30-45	0.003	(0.0002, 0.0063)	0.60	220
MS	60-90	0.002	(0.0001, 0.0036)	0.59	315
GA	0-8	0.010	(0.0082, 0.0125)	0.97	69
GA	30-45	0.006	(0.0034, 0.0079)	0.89	116

^aFirst-order rate constant.

^bLower and upper limits of 95% confidence interval.

^cDT₅₀ values are calculated since they are beyond the incubation timecourse.

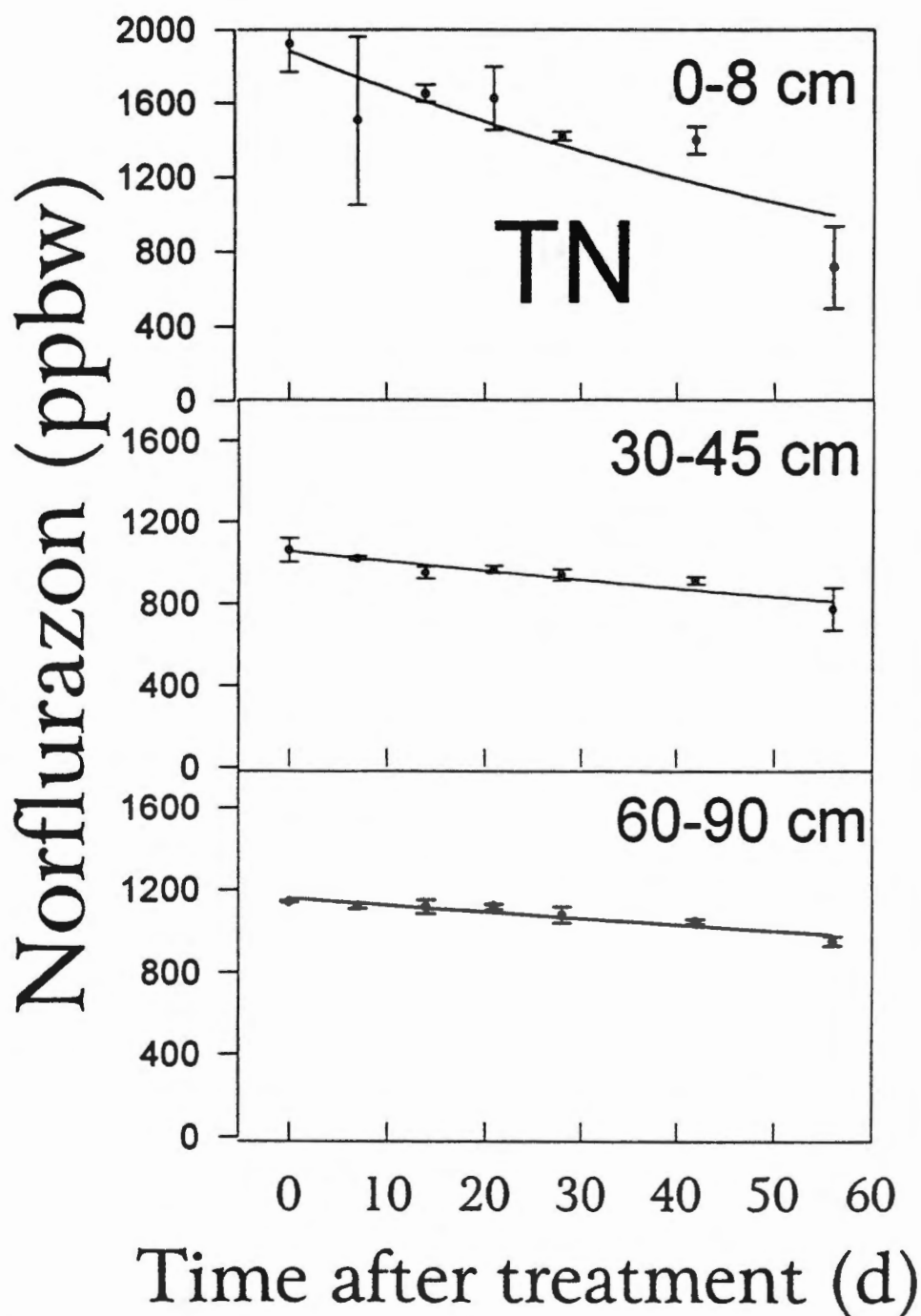


Figure 3. Change in norflurazon concentration over time in a Tennessee soil from 0 to 8, 30 to 45, and 60 to 90 cm at 30 C. Data points are for the mean $\pm$ standard error of three replications. The predicted line for norflurazon is represented by the equation: $dC/dt = -kC$ (Table 7).

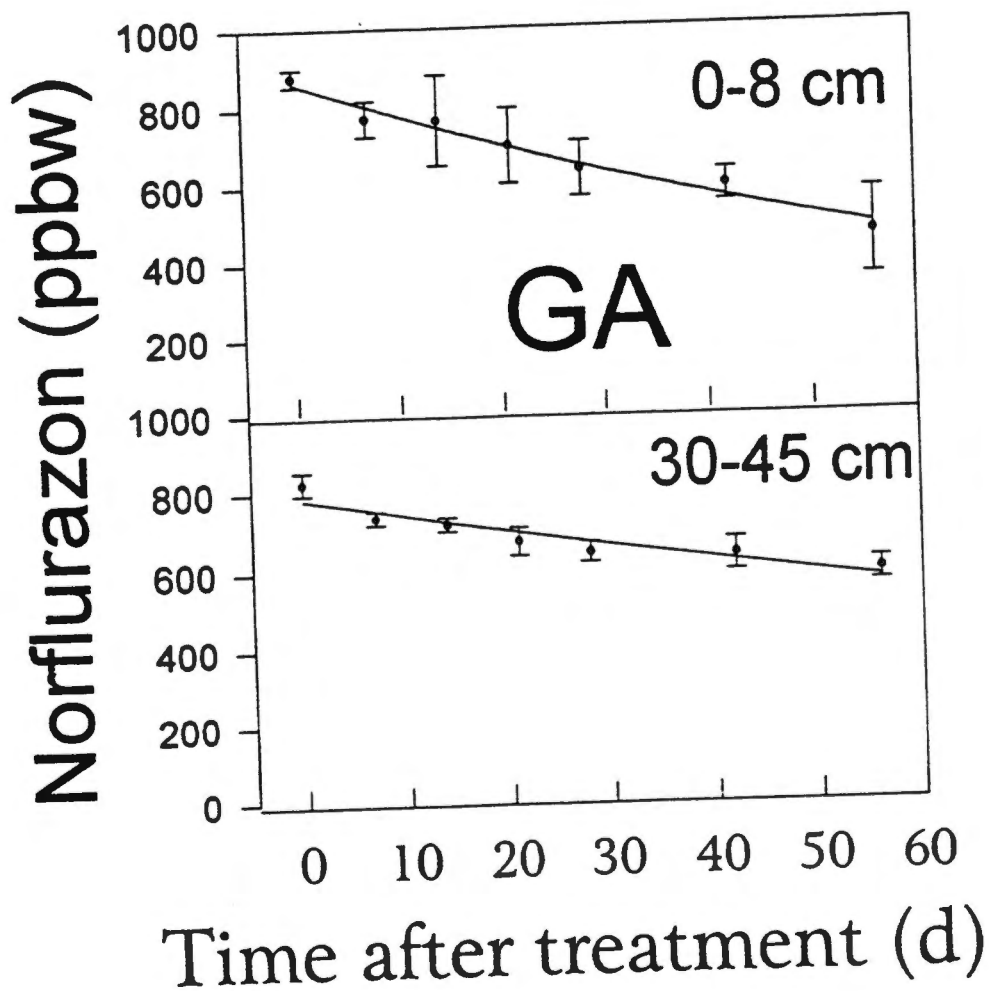


Figure 4. Change in norflurazon concentration over time in a Georgia soil from 0 to 8, and 30 to 45 cm at 30 C. Data points are for the mean $\pm$ standard error of three replications. The predicted line for norflurazon is represented by the equation: $dC/dt = -kC$ (Table 7).

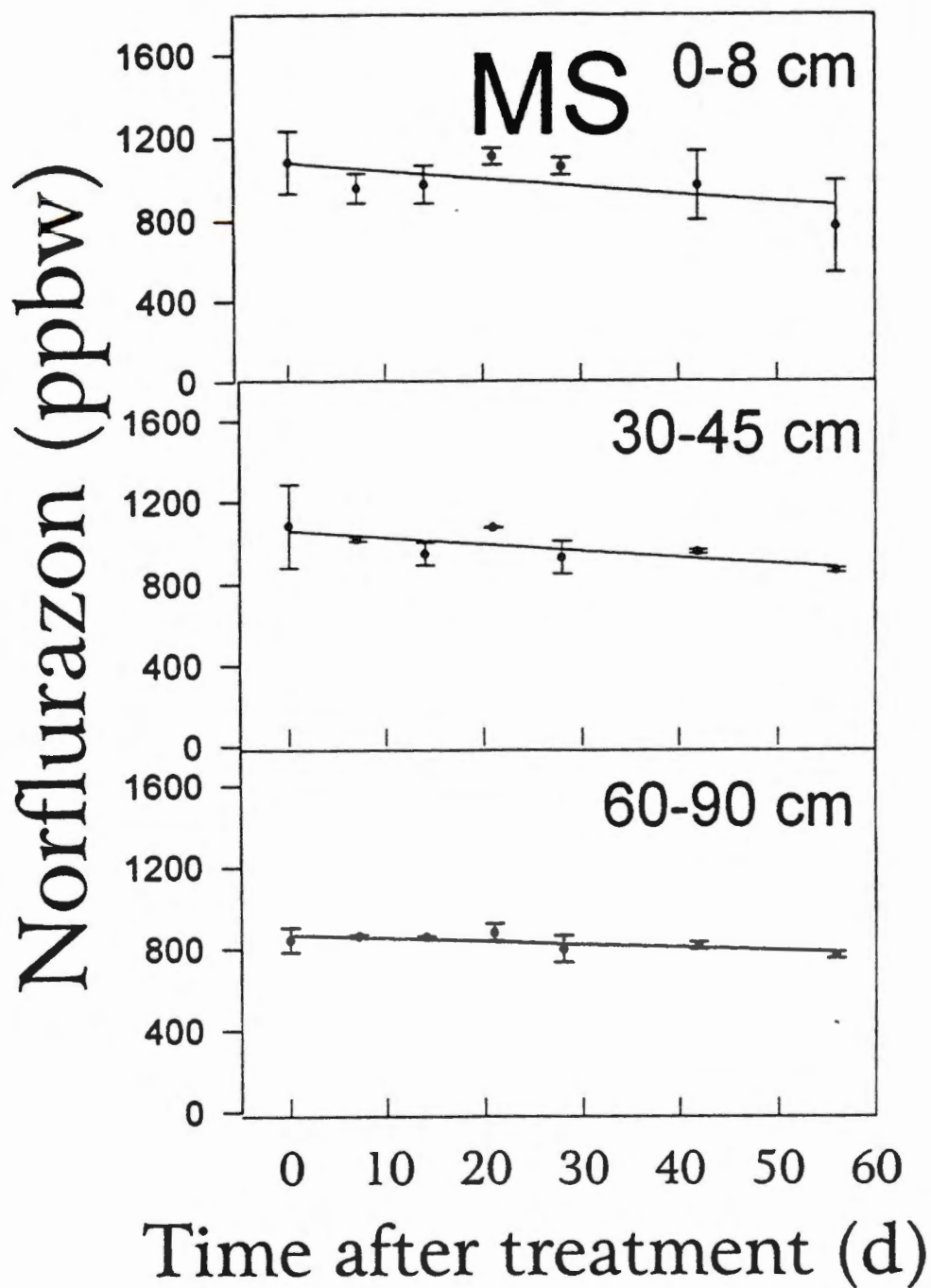


Figure 5. Change in norflurazon concentration over time in a Mississippi soil from 0 to 8, 30 to 45, and 60 to 90 cm at 30 C. Data points are for the mean $\pm$ standard error of three replications. The predicted line for norflurazon is represented by the equation: $dC/dt = -kC$ (Table 7)

.describe norflurazon degradation under laboratory conditions. The two-compartment model is applicable to situations where dissipation depends on the relative contribution of a rapid initial surface loss of herbicide and a slower loss of remaining herbicide due to adsorptive binding of herbicide residues to the soil (7). Norflurazon adsorption was not correlated with DT_{50} (Table 6) which may explain why this model did not improve the overall fit of the dissipation data. Norflurazon concentrations in soil from 0 to 8 cm 56 DAT were 719, 463, and 768 ppbw at TN, GA, and MS, respectively (Figures 3, 4, 5). Norflurazon DT_{50} in soil from the 0- to 8-cm depth was 63, 167, and 69 d for TN, MS, and GA, respectively (Table 7). Norflurazon DT_{50} under laboratory conditions was similar to DT_{50} values under field conditions in the TN soil; however, field dissipation was much more rapid than laboratory dissipation in the MS and GA soils (Table 2). Differences in field and laboratory DT_{50} values indicates that a process other than chemical or microbial degradation may contribute to field dissipation. Photodegradation soon after norflurazon application may have contributed to more rapid dissipation under field conditions. Norflurazon dissipation in the 30- to 45-cm and 60- to 90-cm soil zones were not different among location. No relationship between norflurazon DT_{50} and soil organic matter or norflurazon adsorption was evident, with a correlation coefficient < 0.01 (Table 6). As depth increased, norflurazon DT_{50} increased (correlation coefficient = 0.80).

Norflurazon half-life data were plotted versus soil organic matter, soil pH, and soil depth to graphically illustrate the relationships (Figure 6). In this experiment, each cm increase in soil depth caused DT_{50} to increase by 2.35 d. Slower norflurazon

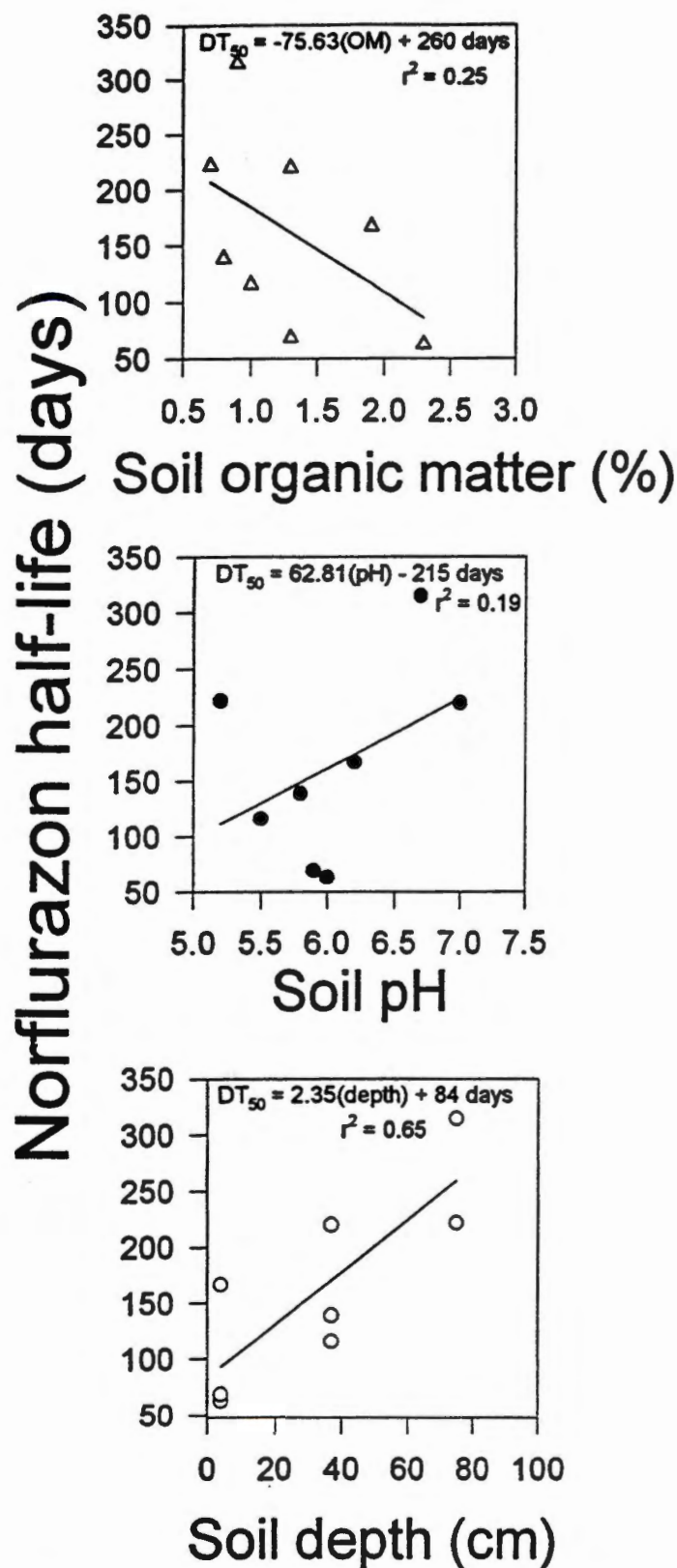


Figure 6. Norflurazon half-life (DT_{50}) in laboratory studies as influenced by soil organic matter, soil pH, and soil depth.

dissipation at lower depths would be anticipated under field conditions, due to reduced microbial activity in subsurface horizons. In laboratory experiments, norflurazon dissipated to 50% of initial concentration within 8 d of application to a dry soil surface (11), which indicates volatility or photodegradation may be important to norflurazon dissipation. Decreased organic matter in subsurface soil zones should promote less adsorption and thus, more rapid norflurazon dissipation. Slower dissipation in these soil horizons, compared to the surface horizon, may be attributable to reduced microbial activity.

In general, norflurazon concentration was influenced by organic matter and environmental conditions. Norflurazon half-life increased with depth, possibly due to decreased microbial activity. Leaching was not a significant contribution to norflurazon dissipation in this study, although our test system did not account for potential herbicide flux through the root zone in the saturated phase.

ACKNOWLEDGMENTS

The authors thank Sandoz Agro, Inc. for supplying analytical norflurazon. This research was partially funded by the Southern Regional Pesticide Impact Assessment Project SR93-07-0-TN. Technical assistance by personnel at the West Tennessee Experiment Station, Jackson, Tennessee; Delta Branch Experiment Station, Stoneville, Mississippi; and Georgia Experiment Station at Griffin is gratefully acknowledged.

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PART V.

**COMPARISON OF PRZM AND GLEAMS COMPUTER MODEL
PREDICTIONS WITH FIELD DATA FOR FLUOMETURON AND
NORFLURAZON LEACHING¹**

¹To be submitted for publication in Weed Science. Authors: William T. Willian, Thomas C. Mueller, Robert M. Hayes, Charles E. Snipes, and David C. Bridges. Res. Assoc., Asst. Prof., Prof., respectively, Dept. of Plant and Soil Sci., Univ. of Tennessee, Knoxville; Prof., Miss. State Univ., Stoneville; Assoc. Prof., Univ. of Georgia, Griffin.

ABSTRACT

The ability of the pesticide root zone model (PRZM) and groundwater loading effects of agricultural management systems (GLEAMS) model to simulate movement of two herbicides in soil was evaluated using site-specific environmental data. Site-specific and database soil and pesticide data were compared within each model. Field experiments examined fluometuron and norflurazon movement in three soils representative of the cotton growing regions of the southeastern United States. Measured herbicide concentrations were always greatest near the soil surface, and neither herbicide was detected below 15 cm, based on data obtained from soil cores. Site-specific parameters increased PRZM accuracy in predicting total norflurazon concentration in the soil profile. PRZM more accurately predicted norflurazon than fluometuron when site-specific sorption and degradation parameters were used. The use of database variables increased the accuracy of PRZM to predict fluometuron but decreased accuracy of norflurazon predictions. PRZM provided more correct predictions of norflurazon concentration in the soil profile than did GLEAMS; however, both models performed similarly with regards to fluometuron. The use of site-specific data increased accuracy of GLEAMS to predict fluometuron but not norflurazon. The use of database inputs decreased GLEAMS accuracy when averaged over all simulations. Simulations better predicted norflurazon concentration than fluometuron when averaged over all model by parameter input combinations. These data indicate that PRZM and GLEAMS have utility to predict herbicide movement into soil if correct input parameters are used in the simulations.

Nomenclature: Fluometuron, *N,N*-dimethyl-*N'*-(3-trifluoromethyl)phenyl}-urea;

Norflurazon, {4-chloro-5-(methylamino-2-(3-(trifluoromethyl)phenyl)-3(2*H*)-pyridazinone)}; cotton, *Gossypium hirsutum* L.

Additional index words: Degradation, dissipation, adsorption, herbicide, cotton, movement, depth, model validation.

INTRODUCTION

Pesticides have been used extensively in the United States to control a wide variety of pests, but in the last decade there has been an increased awareness and concern over potential contamination of the groundwater supply by pesticides. Currently, expensive field and laboratory experiments are needed to assess pesticide leaching potential. Advances in programming and computer technology have led to the development of several sophisticated models designed to simulate pesticide movement into the soil.

These models could provide a cost-effective environmental impact assessment, and generate data whereby pesticide inputs and management practices could be better employed. The inherent heterogeneity of soil and environmental parameters make model validation impossible under all conceivable scenarios; however, comparisons of simulated versus measured pesticide leaching data could prove a useful means of assessing model accuracy. The Pesticide Root Zone Model (PRZM)² and Groundwater Loading Effects

²Abbreviations: PRZM, Pesticide Root Zone Model; GLEAMS, Groundwater Loading Effects of Agricultural Management Systems; PRZM-SS, PRZM utilizing site-specific sorption and dissipation parameters; PRZM-DB, PRZM utilizing database sorption and dissipation parameters; GLEAMS-SS, GLEAMS utilizing site-specific sorption and dissipation parameters; GLEAMS-DB, GLEAMS utilizing database sorption and dissipation parameters; DT₅₀, half-life or 50% disappearance in days; DAT, days after treatment; HPLC, high-pressure liquid chromatography; K_a, adsorption coefficient; k, first-order rate constant; K_{oc}, sorption coefficient.

on Agricultural Management Systems (GLEAMS)² models predict a pesticide concentration in a soil profile with no error term associated with the prediction, thus, this is not a probability function.

This research focused on two commonly used cotton herbicides, comparing their simulated leaching potential to measured results from field and laboratory analyses. Simulation model predictions of pesticide movement necessitate a subjective decision as to the determination of a “correct” predicted value. In these experiments, any prediction within a factor of two of the measured concentration was considered a correct value. The PRZM and GLEAMS models were utilized to predict fluometuron and norflurazon movement and dissipation in three soils representative of the cotton growing regions of the southeastern United States.

Fluometuron is a widely used substituted urea herbicide for control of annual grasses and broadleaf weeds in cotton (2). Fluometuron is considered moderately persistent and has been shown to leach appreciably in low adsorptive soils (3). Fluometuron adsorption is influenced primarily by soil organic matter content, and to a lesser degree by cation exchange capacity (5). Fluometuron degradation is accomplished primarily by soil microorganisms (3), and cold, dry soil conditions may slow dissipation. Rogers, et al. (16) observed more rapid fluometuron dissipation at higher temperatures in three soils, with half-life (DT_{50})² ranging from 26 to 73 d. Fluometuron moved to a depth of 60-cm in all three soils, and downward movement was greatest in a sandy soil with low organic matter. Less than 50% of applied fluometuron moved to depths below 15 cm. Mueller et al. (14) reported slower fluometuron degradation rates in subsurface soil

compared to surface soils. Decreased degradation was correlated with declines in microbial biomass and respiration, but not with fluometuron adsorption.

Norflurazon is a pyridazinone herbicide for annual grass, sedge, and broadleaf weed control in cotton and soybeans (2). Norflurazon is considered slightly mobile in most soils but may persist to injure succeeding crops such as wheat, corn, or sorghum (10, 17). Residual carryover is directly related to soil organic matter, with slower dissipation at higher organic matter content (17). Schroeder and Banks (17) detected > 60% of applied norflurazon in the 0 to 8 cm soil zone at 110 days after treatment (DAT)²; and < 12% below 15-cm. Limited mobility has also been observed based on thin layer chromatography plate experiments (8). Soil and environmental factors influence norflurazon dissipation, and DT₅₀ ranges from 45 to 270 d (2, 17). Previous literature has often reported a range of norflurazon DT₅₀, which leads to a subjective decision as to which value to use. Concentrations of 0.05 to 0.34 μg g⁻¹ were found 365 DAT in five soils receiving 1.7 kg ai ha⁻¹.

PRZM and GLEAMS are mathematical models developed to evaluate the movement of agricultural chemicals into and through the soil profile. Detailed descriptions of these models have been previously published (6, 12). These models utilize pesticide, erosion, and hydrology parameters to simulate pesticide movement. Several researchers have used these and other models to predict pesticide fate in the environment. Mueller et al. (15) reported that both PRZM and GLEAMS accurately predicted resident concentrations of alachlor [2-chloro-*N*-(2,6-diethylphenyl)-*N*-(methoxymethyl)acetamide], metribuzin [4-amino-6-(1,1-dimethylethyl)-3-(methylthio)-1,2,4-triazin-5(4*H*)-one] and

norflurazon at 20 DAT. Both models overestimated movement in later simulations; however, both models accurately predicted depth to the limit of detection. PRZM more accurately predicted the location of norflurazon in the soil profile 84 DAT. Jones et al. (9) reported that PRZM simulations of aldicarb [2-methyl-2(methylthio)propionaldehyde-O(methyl-carbamoyl)oxime] and aldoxycarb [2-methyl-2(methylsulfonyl)propanal O-{methylamino}carbonyl}oxime] movement corresponded relatively well with measured field data; however, normal use rates of the pesticides were inadequate to analytically assess movement. Carsel et al. (7) found PRZM to effectively simulate metalaxyl [*N*-(2,6-Dimethylphenyl)-*N*-(methoxyacetyl)-alanine methyl ester] movement in two Florida soils. Bush et al. (4) used PRZM to accurately predict picloram [4-amino-3,5,6-trichloro-2-pyridinecarboxylic acid] movement in loamy soils of the Appalachian region. They observed that pesticide half-life, partition coefficient, and water infiltration parameters had the greatest impact on leachability of five pesticides evaluated. Zacharias and Heatwole (22) reported that PRZM and GLEAMS performed well in predicting total concentration within the soil profile for metolachlor [2-chloro-*N*-(2-ethyl-6-methylphenyl)-*N*-(2-methoxy-1-methylethyl)acetamide] and atrazine [6-chloro-*N*-ethyl-*N'*-(1-methylethyl)-1,3,5-triazine-2,4-diamine], but did not reliably predict concentration distributions.

MATERIALS AND METHODS

Field data. Field experiments were conducted in 1992 and 1993 at Jackson, TN on a Lexington silt loam (fine-silty, mixed thermic Typic Paleudalf), Stoneville, MS on a Beulah silt loam (coarse-loamy, mixed thermic Typic Dystocrept), and Griffin, GA on a

Dothan loamy sand (Plinthic Paleudalf) utilizing a randomized complete block design with treatments replicated four times. Plots were four rows wide, with rows spaced 102-cm wide, by 9 m long. Standard fertility and pest management practices were employed at each location, and all plots were tilled. Cotton was planted prior to herbicide treatment. Fluometuron and norflurazon were applied at 1.7 kg ai ha⁻¹ as a preemergence application. A hydraulic probe was utilized to obtain soil samples from a depth of 0 to 60 cm at approximately 28, 56, and 112 DAT to assess fluometuron and norflurazon distribution within the profile. Two full-length cores were collected from each plot and divided into 0- to 8-, 8- to 15-, 15- to 30-, 30- to 45-, and 45- to 60-cm soil zones and the two samples from each depth were composited. Soil samples were frozen at -10 C within 2 hr of a sampling event and later thawed, passed through a 2-mm sieve, and stored frozen until chemical analysis.

Soil (40 g oven dry weight basis) was placed into 250-ml polyethylene bottles equipped with screw-top caps. Fluometuron and norflurazon were extracted with 80-ml methanol for 16 h using a reciprocating shaker. The soil extracts were allowed to settle for 1 hr and then filtered through two qualitative filter papers and an aliquot placed into 4-ml vials. Fluometuron and norflurazon were quantified with HPLC² and fluorescence detection as described previously (13, 21). Herbicide concentrations were based on detectable peak areas and soil moisture data. Fluometuron and norflurazon recoveries were > 90% in all soils (data not shown). Limits of detection for fluometuron and norflurazon were 0.015 µg g⁻¹ and 0.01 µg g⁻¹, respectively. A complete description of the methods and results have been previously submitted.

Model simulations. Simulations were performed by using the PRZM (release 1) and GLEAMS (version 1.8.55) models. PRZM is a continuous compartmental model which is comprised of hydrology and chemical transport components. The hydrology component calculates erosion by the universal soil loss equation, and surface runoff by the SCS curve number technique. Generalized soil terms such as field capacity, wilting point, and saturation are used to predict percolation through the soil profile. Chemical transport is calculated by inclusion of several parameters including plant uptake, leaching, surface runoff, sorption, and soil degradation rate. PRZM is incapable of simulating volatilization processes and metabolite formation, but can be used to simulate multiple applications of a pesticide each year (6).

GLEAMS is also a continuous, field-scale hydrology and chemical transport model and thus shares many similarities with PRZM; however, erosion and evapotranspiration predictions are calculated differently. Additionally, the soil profile may be described by a maximum of five soil horizons (12). The GLEAMS output for fluometuron and norflurazon consisted of six computational layers: 0 to 1, 1 to 7.5, 7.5 to 15, 15 to 30, 30 to 45 and 45 to 60 cm. The data from the top two layers were combined to determine the herbicide concentration in the 0- to 7.5 cm zone. The number and thickness of each computational layer is derived by the GLEAMS model; therefore, direct comparison to measured data is sometimes not possible. To achieve a direct comparison to our data, soil depth was set at 60 cm for the GLEAMS simulations.

Site-specific data were obtained from each location to increase prediction accuracy. These data were generated independent of the field study. Measured daily

rainfall from each appropriate location was used in each simulation. Soil with no previous fluometuron or norflurazon exposure was collected prior to herbicide application to determine soil physical and chemical characteristics. Physical and chemical analysis of all soils was performed at Knoxville, TN (Table 1). Particle size and total organic carbon (OC) analyses were performed utilizing the hydrometer method (11) and dry combustion, respectively (1). Adsorption of fluometuron and norflurazon onto the three soils was determined using a soil-slurry technique as described by Talbert and Fletchall (19). Soil from the 0- to 8-, 30- to 45-, and 60- to 90-cm soil zones was air dried and screened through a 2-mm sieve, and 10 g of soil was added to 50-ml screw top centrifuge tubes. Twenty ml of 0.01 M CaCl_2 solution containing either 0 or 1 $\mu\text{g ml}^{-1}$ analytical fluometuron or norflurazon was added to each tube. Tubes were equilibrated for 24 h at 22 C by a reciprocating shaker and then filtered through two pieces of qualitative filter paper, and an aliquot removed for analysis. A distribution coefficient (K_d)² was determined for each state by depth combination (Table 1). Site-specific distribution coefficients for the remaining four soil horizons were interpolated from calculated data. Distribution coefficients for database input into PRZM were determined by converting published sorption coefficient (K_{oc})² (Table 2) values to K_d .

The degradation rates for fluometuron and norflurazon in the various soil zones were determined under laboratory conditions from dissipation data (Table 3). Both models utilize first-order kinetics to explain herbicide dissipation, with the PRZM model input as a first-order rate constant and GLEAMS as the soil half-life of the herbicide in days. Fluometuron and norflurazon degradation parameters were determined from the

Table 1. Soil profile parameters used in model simulations^a.

State	Depth	Bulk density	Organic carbon ^b				Fluometuron		Norflurazon	
				Sand ^c	Silt ^c	Clay ^c	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm	g cm ³		%			K _d			
TN	0-8	1.45	1.10	10	73	17	1.69	1.10	9.51	7.70
TN	8-15	1.40	0.68	6	78	16	1.34	0.68	7.30	4.73
TN	15-30	1.40	0.35	11	63	26	1.00	0.35	5.00	2.43
TN	30-45	1.45	0.25	7	63	30	0.64	0.25	2.67	1.75
TN	45-60	1.40	0.18	9	63	28	0.50	0.18	2.20	1.26
MS	0-8	1.25	0.87	45	48	7	1.73	0.87	9.29	6.09
MS	8-15	1.25	0.56	43	49	8	1.50	0.56	8.20	3.92
MS	15-30	1.30	0.64	30	59	11	1.30	0.64	7.00	4.48
MS	30-45	1.35	0.53	15	75	10	1.11	0.53	5.89	3.71
MS	45-60	1.35	0.22	14	69	17	1.00	0.22	5.10	1.54
GA	0-8	1.50	0.41	73	18	9	0.56	0.41	2.46	2.87
GA	8-15	1.50	0.37	74	16	10	0.45	0.37	2.21	2.56
GA	15-30	1.50	0.18	67	17	16	0.40	0.18	1.70	1.26
GA	30-45	1.55	0.17	52	16	32	0.31	0.17	1.29	1.19
GA	45-60	1.60	0.13	42	16	42	0.24	0.13	0.90	0.93

^aAbbreviations: PRZM-SS, PRZM using site-specific parameters; PRZM-DB, PRZM using database parameters; GLEAMS-SS, GLEAMS using site-specific parameters; GLEAMS-DB, GLEAMS using database parameters; K_d, adsorption coefficient.

^bSoil organic carbon content was determined by total combustion.

^cParticle size analyses were performed using the hydrometer method.

Table 2. Pesticide degradation parameters in k (first-order rate constant) and DT₅₀ (half-life in d).

State	Depth cm	Fluometuron				Norflurazon			
		—k—		—DT ₅₀ —		—k—		—DT ₅₀ —	
		SS ^a	DB ^b	SS	DB	SS	DB	SS	DB
TN	0-8	0.064	0.008	11	85	0.011	0.023	63	30
TN	8-15	0.049	0.008	25	85	0.009	0.023	77	30
TN	15-30	0.030	0.008	45	85	0.006	0.023	116	30
TN	30-45	0.011	0.008	63	85	0.005	0.023	139	30
TN	45-60	0.008	0.008	99	85	0.004	0.023	173	30
MS	0-8	0.016	0.008	43	85	0.004	0.023	167	30
MS	8-15	0.015	0.008	48	85	0.004	0.023	184	30
MS	15-30	0.014	0.008	53	85	0.003	0.023	202	30
MS	30-45	0.012	0.008	58	85	0.003	0.023	220	30
MS	45-60	0.010	0.008	73	85	0.003	0.023	266	30
GA	0-8	0.031	0.008	22	85	0.010	0.023	69	30
GA	8-15	0.025	0.008	37	85	0.009	0.023	87	30
GA	15-30	0.020	0.008	52	85	0.007	0.023	101	30
GA	30-45	0.010	0.008	69	85	0.006	0.023	116	30
GA	45-60	0.009	0.008	90	85	0.003	0.023	132	30

^aSS, site-specific parameters.

^bDB, database-generated parameters.

Table 3. Site-specific and database sorption parameters utilized in the GLEAMS model for fluometuron and norflurazon.

State	Fluometuron		Norflurazon	
	K_{oc}			
	GLEAMS-SS ^a	GLEAMS-DB ^b	GLEAMS-SS	GLEAMS-DB
TN	154	100	864	700
MS	199	100	1068	700
GA	137	100	600	700

^aGLEAMS model utilizing site-specific sorption parameters.

^bGLEAMS model utilizing database-generated sorption parameters.

same soil horizons utilized in the adsorption analysis, as previously described. All soils were passed through a 2-mm sieve and 40 g of moist soil was placed into 250-ml polyethylene bottles. The soil water status was adjusted to -33 kPa, which was sufficient for microbial activity. Bottles were fortified with $1 \mu\text{g g}^{-1}$ analytical fluometuron or norflurazon in water and methanol:water (20:80 v/v), respectively, and incubated in the dark at 30 C. Bottles were transferred to a freezer at 0, 7, 14, 21, 28, 42, and 56 d after herbicide amendment. All samples were simultaneously extracted and analyzed as previously described. The data were fit to first-order kinetics using the PCSAS NLIN procedure (18). The natural logarithm of herbicide concentration in ppbw soil was regressed against time in days, and the NLIN procedure calculated first-order rate constants (k^2). Fluometuron and norflurazon DT_{50} was calculated by the equation $(0.693 / k) = \text{DT}_{50}$. First-order rate constants and DT_{50} values for the remaining four soil horizons were interpolated from calculated data (Table 3). Other pesticide parameters were obtained from data bases (2), or calculated by using the PRZM model (6).

Tests of model performance were demonstrated using both site-specific and database adsorption and dissipation parameters. The SCS/CES pesticide properties for environmental decision making was utilized to obtain database values for fluometuron and norflurazon sorption and dissipation (20). Simulations performed with database values (PRZM-DB and GLEAMS-DB) were compared to data generated using site-specific soils data (PRZM-SS and GLEAMS-SS). The site-specific parameters allowed for differences among sorption and dissipation inputs throughout the soil profile examined. Database

parameters were consistent throughout the soil profile and thus did not account for differences in sorption or dissipation. Herbicide concentration in total mass within each soil profile was chosen as a measure of prediction accuracy. Predicted values are divided by measured values to obtain a predictive index value. Predicted index values within a factor of two from the measured concentration (index value >0.5 or <2.0) were considered accurate with regard to herbicide concentration.

RESULTS AND DISCUSSION

Neither fluometuron nor norflurazon was detected (limits of detection 15 and 10 ppbw, respectively) below 15 cm either year at any location and concentrations were always greatest in the 0 to 8 cm soil zone (Figures 1-12). PRZM-SS accurately predicted total fluometuron concentration within the soil profile at TN 42 DAT in 1992 and 28 DAT in 1993 but over predicted fluometuron concentration in later simulations (Table 4). PRZM overestimated fluometuron concentrations in subsurface soil zones at all simulation dates in 1992 and 1993 (Figures 1, 2). PRZM accurately predicted that the highest fluometuron concentration would be near the soil surface 28 DAT in 1993, but later simulations underestimated fluometuron concentration in the surface horizon. The PRZM model utilizing database values accurately estimated total fluometuron concentration within the profile up to 42 DAT but overestimated concentration at later simulations. Predicted index values within a PRZM simulation date were always lower when database values were utilized compared to site-specific parameters (Table 4). Database sorption values were lower than site-specific ones and thus, fluometuron adsorption may have been

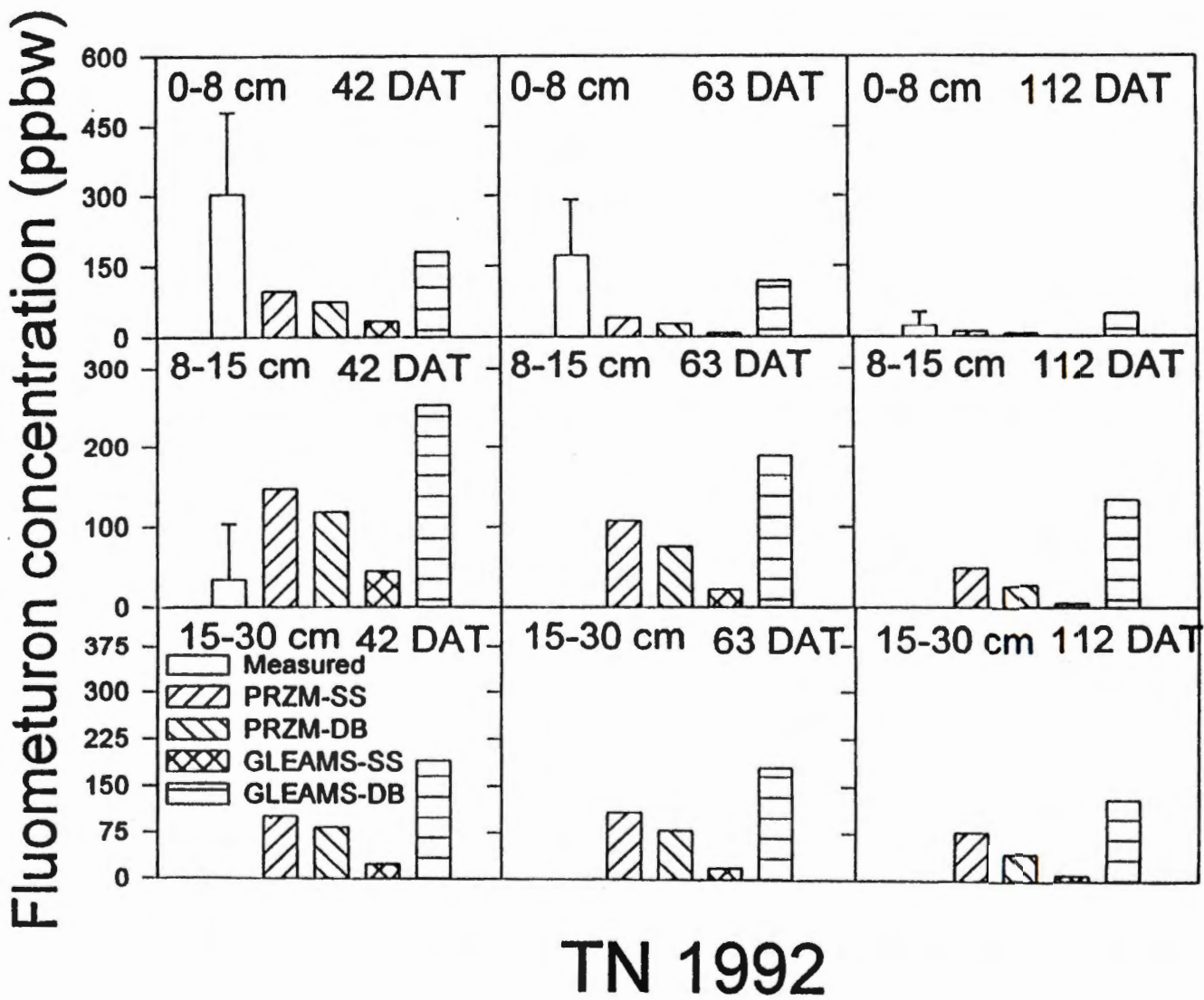


Figure 1. Measured and simulated fluometuron soil concentrations in a Tennessee soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1992. Measured concentrations are mean values of four replications $\pm$ standard error.

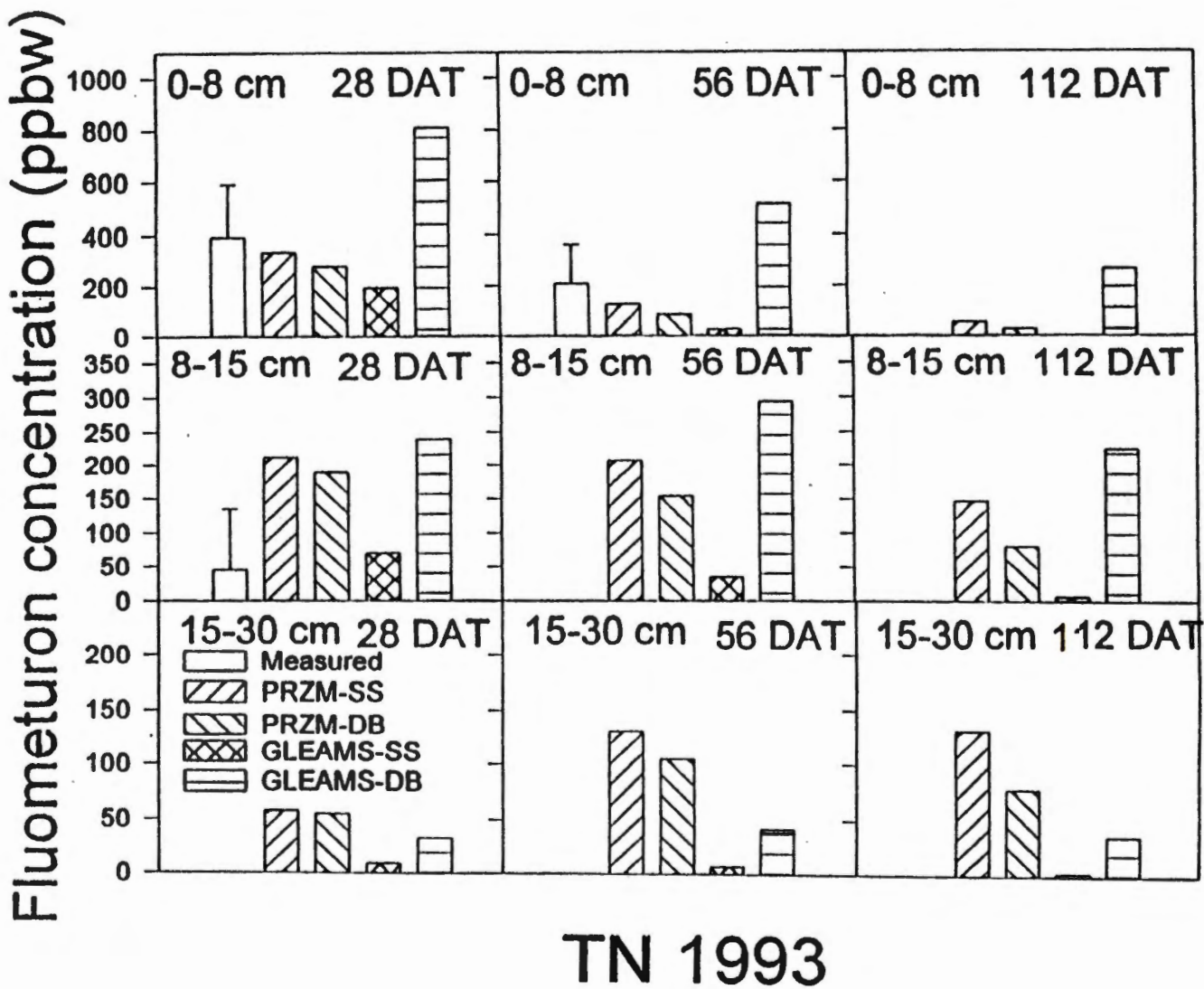


Figure 2. Measured and simulated fluometuron soil concentrations in a Tennessee soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1993. Measured concentrations are mean values of four replications $\pm$ standard error.

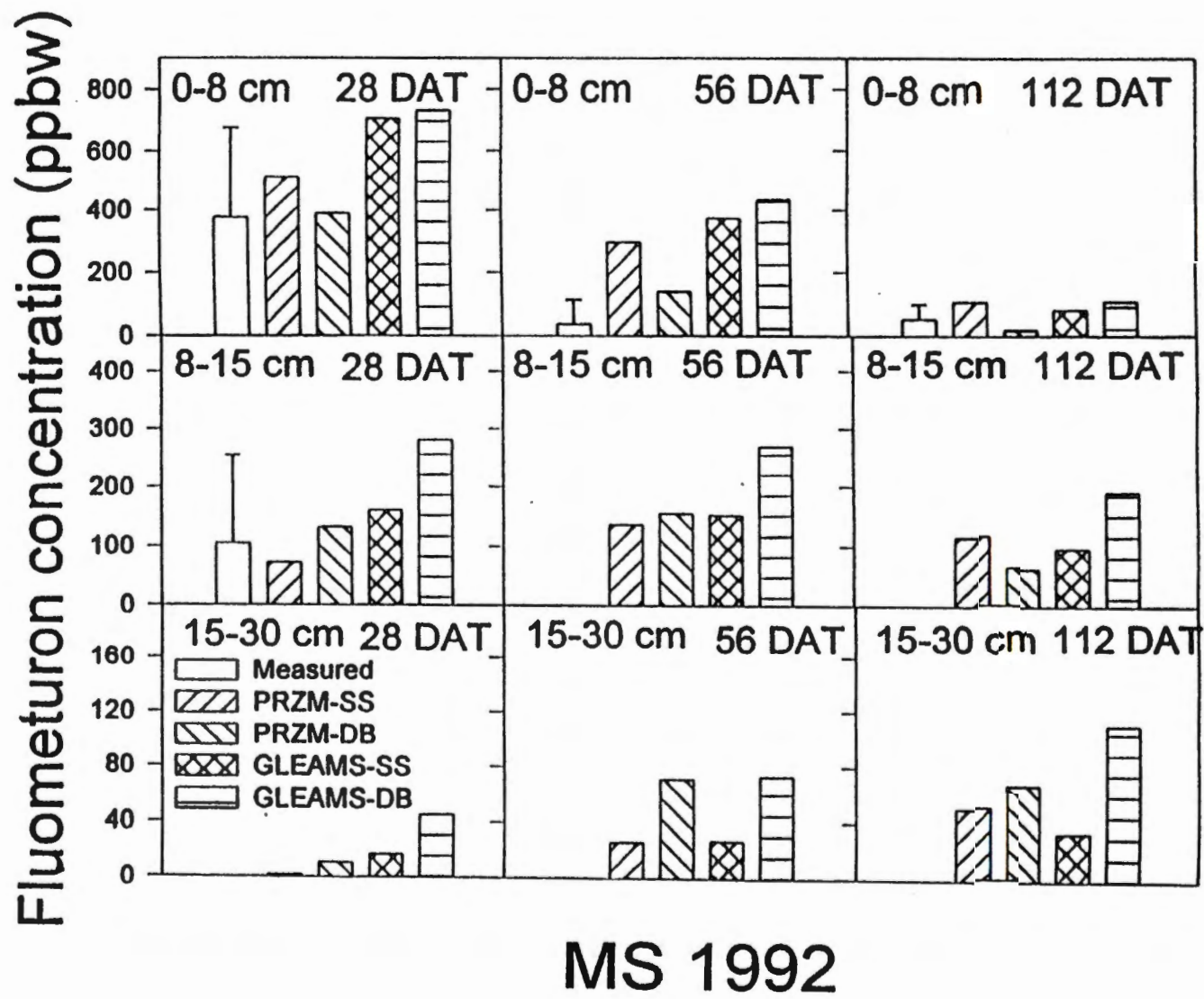


Figure 3. Measured and simulated fluometuron soil concentrations in a Mississippi soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1992. Measured concentrations are mean values of four replications $\pm$ standard error.

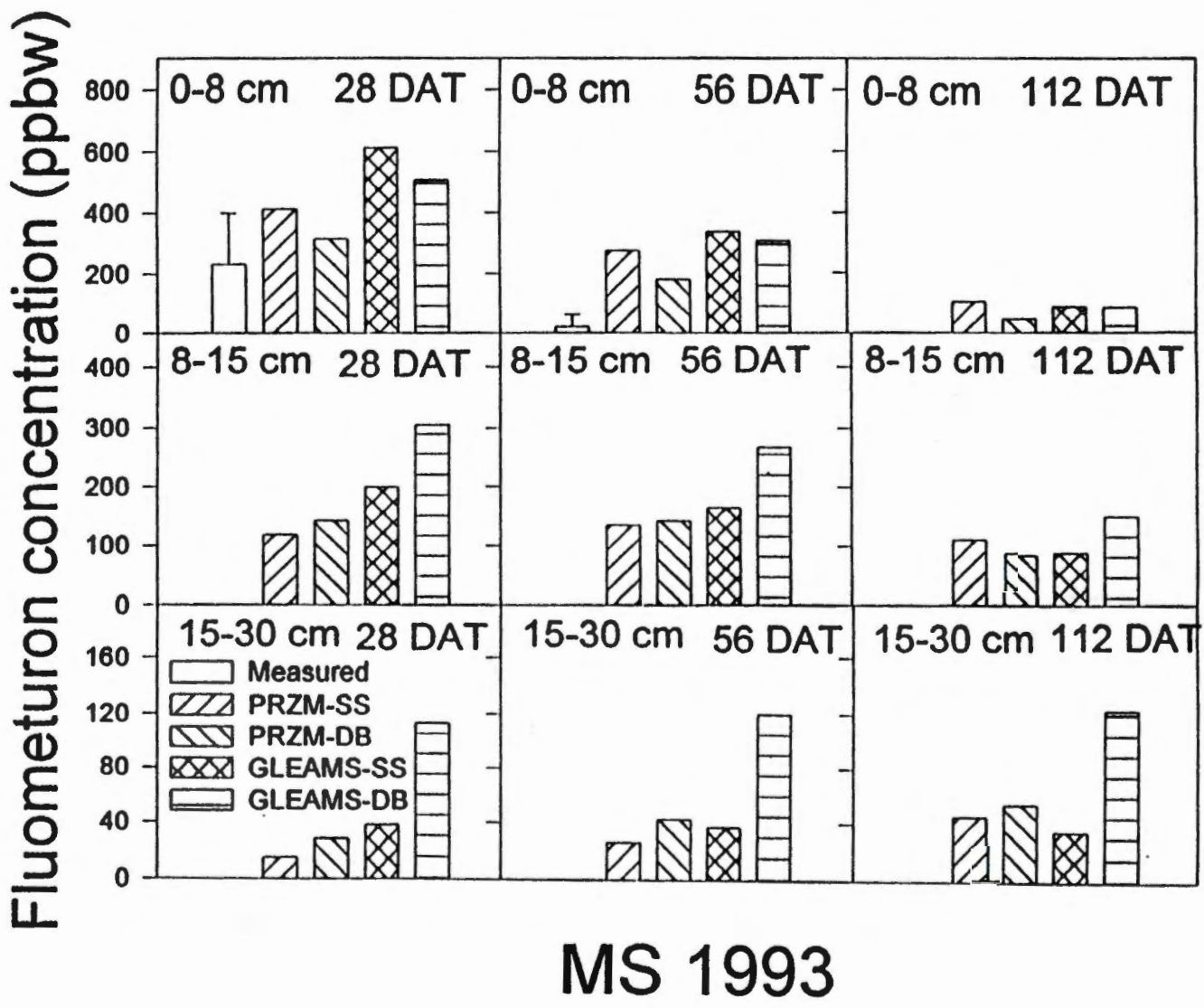


Figure 4. Measured and simulated fluometuron soil concentrations in a Mississippi soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1993. Measured concentrations are mean values of four replications $\pm$ standard error.

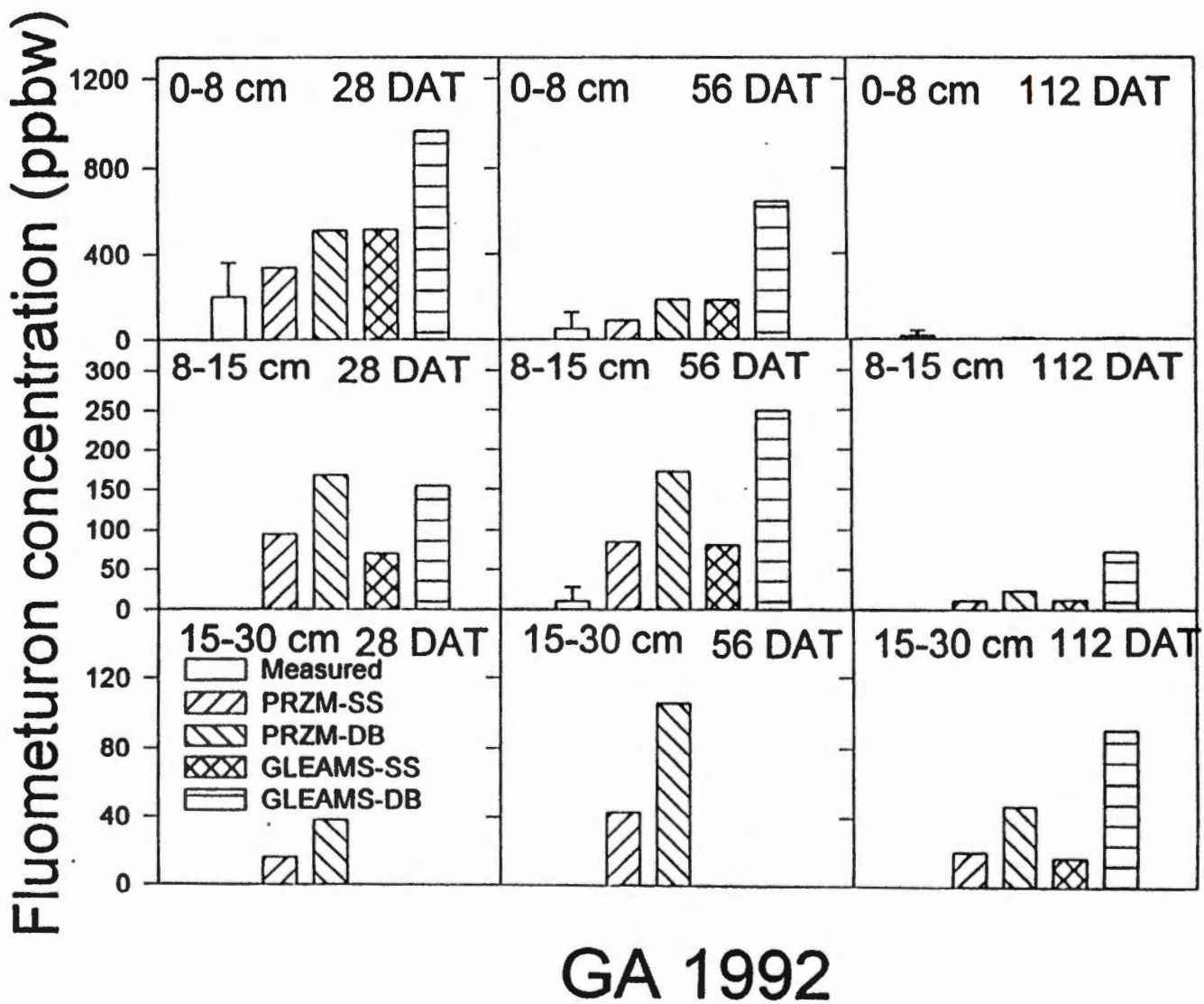
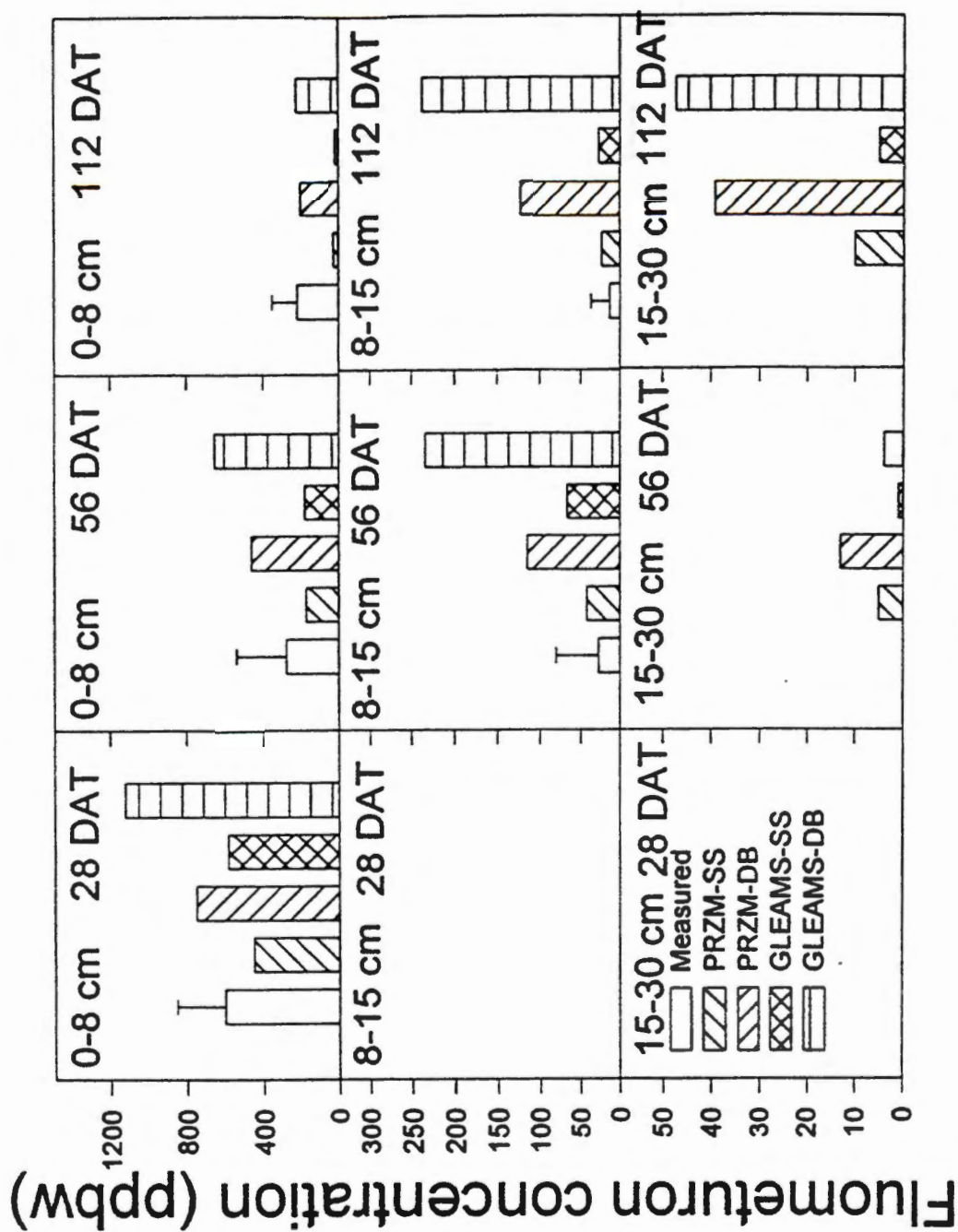
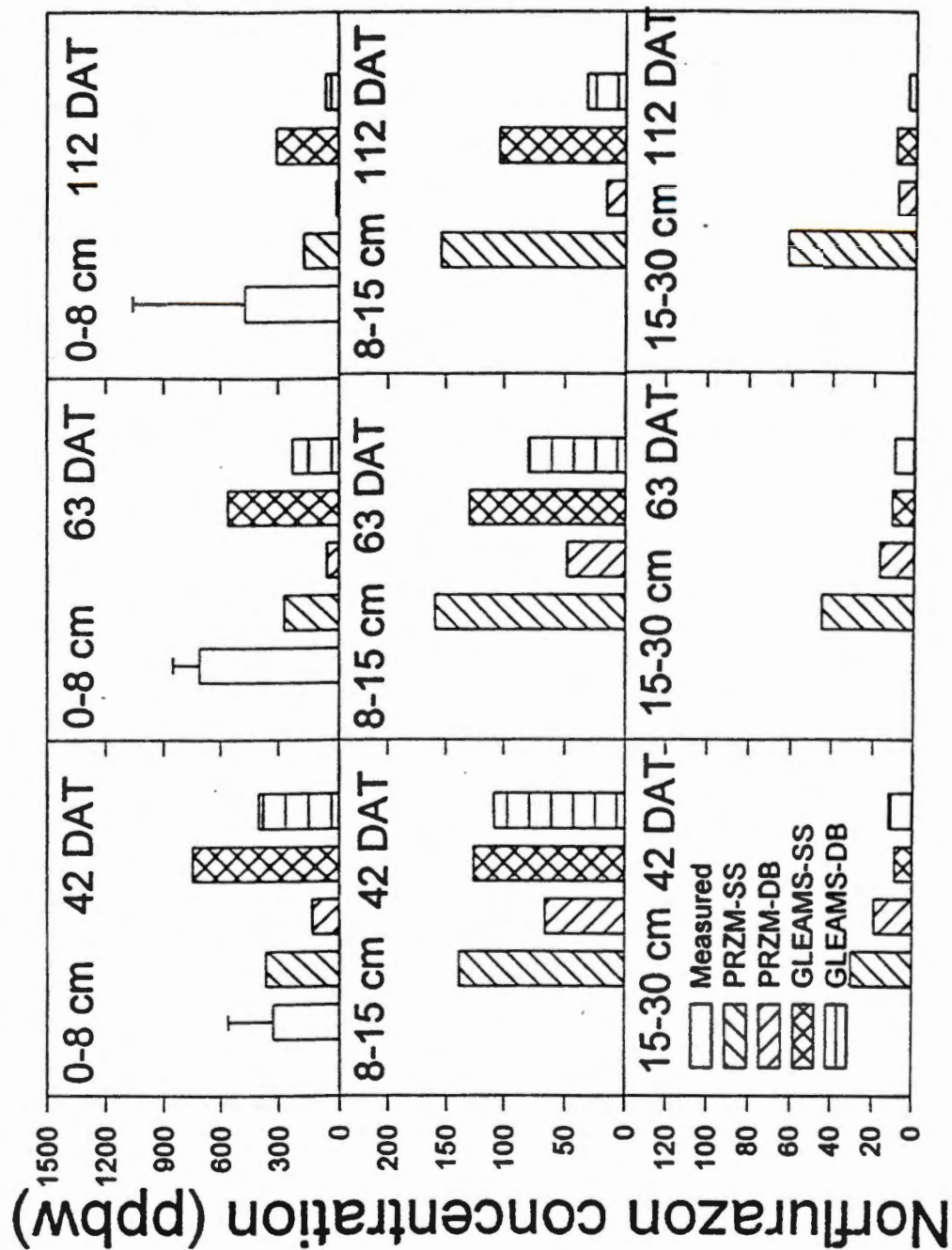


Figure 5. Measured and simulated flumeturon soil concentrations in a Georgia soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1992. Measured concentrations are mean values of four replications $\pm$ standard error.



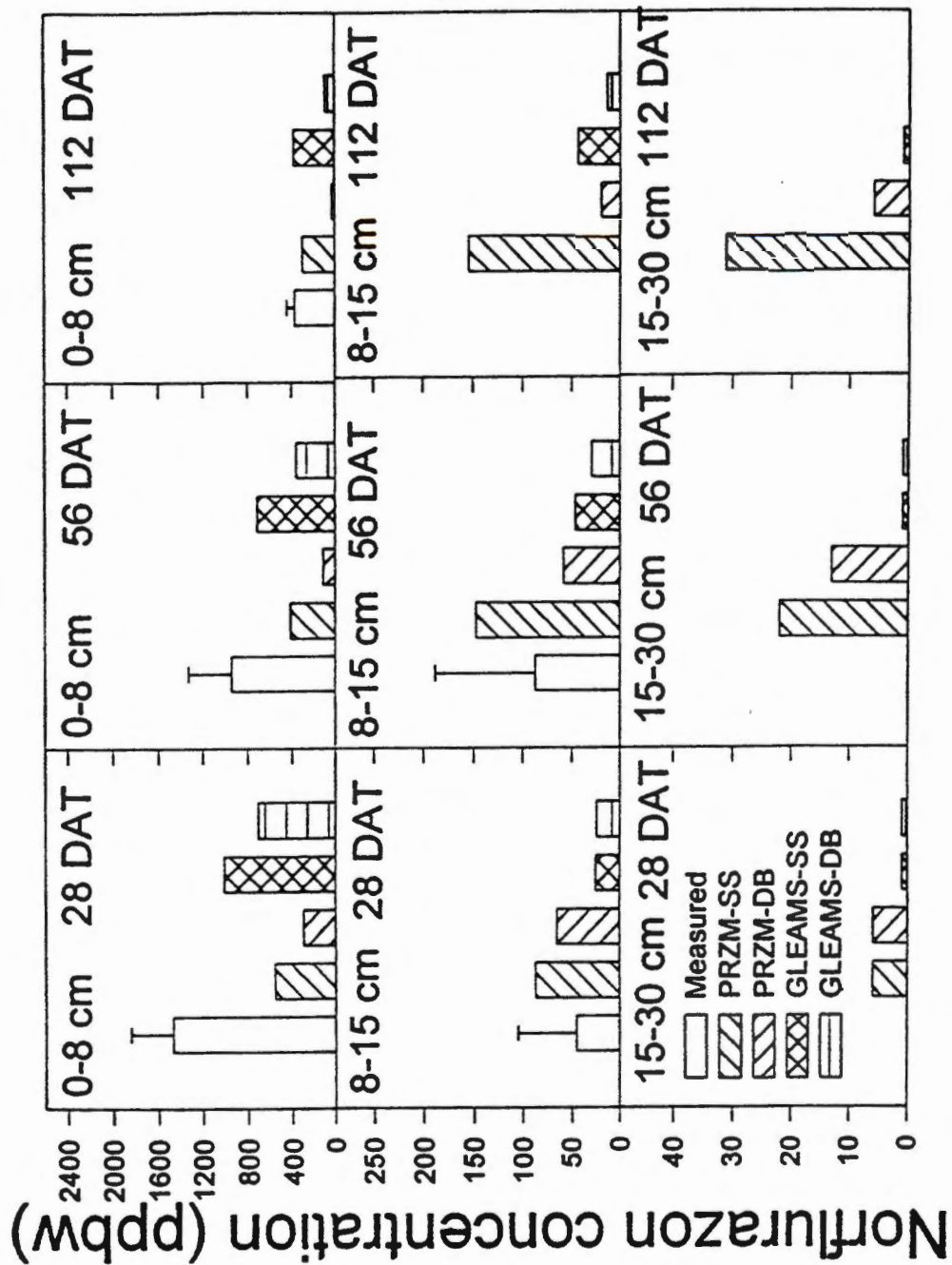
GA 1993

Figure 6. Measured and simulated fluometuron soil concentrations in a Georgia soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1993. Measured concentrations are mean values of four replications $\pm$ standard error.



TN 1992

Figure 7. Measured and simulated norflurazon soil concentrations in a Tennessee soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1992. Measured concentrations are mean values of four replications $\pm$ standard error.



TN 1993

Figure 8. Measured and simulated norflurazon soil concentrations in a Tennessee soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1993. Measured concentrations are mean values of four replications $\pm$ standard error.

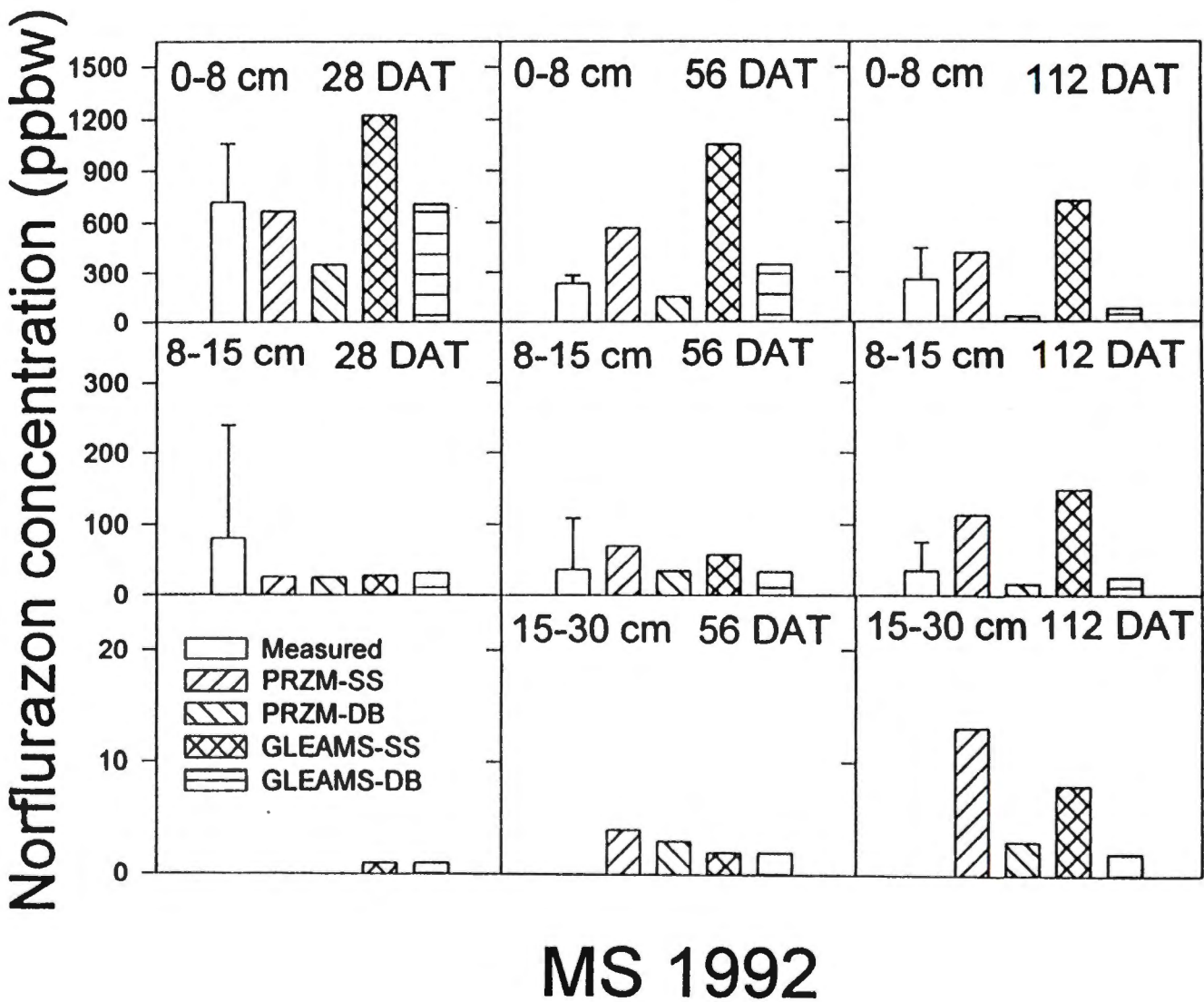


Figure 9. Measured and simulated norflurazon soil concentrations in a Mississippi soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1992. Measured concentrations are mean values of four replications $\pm$ standard error.

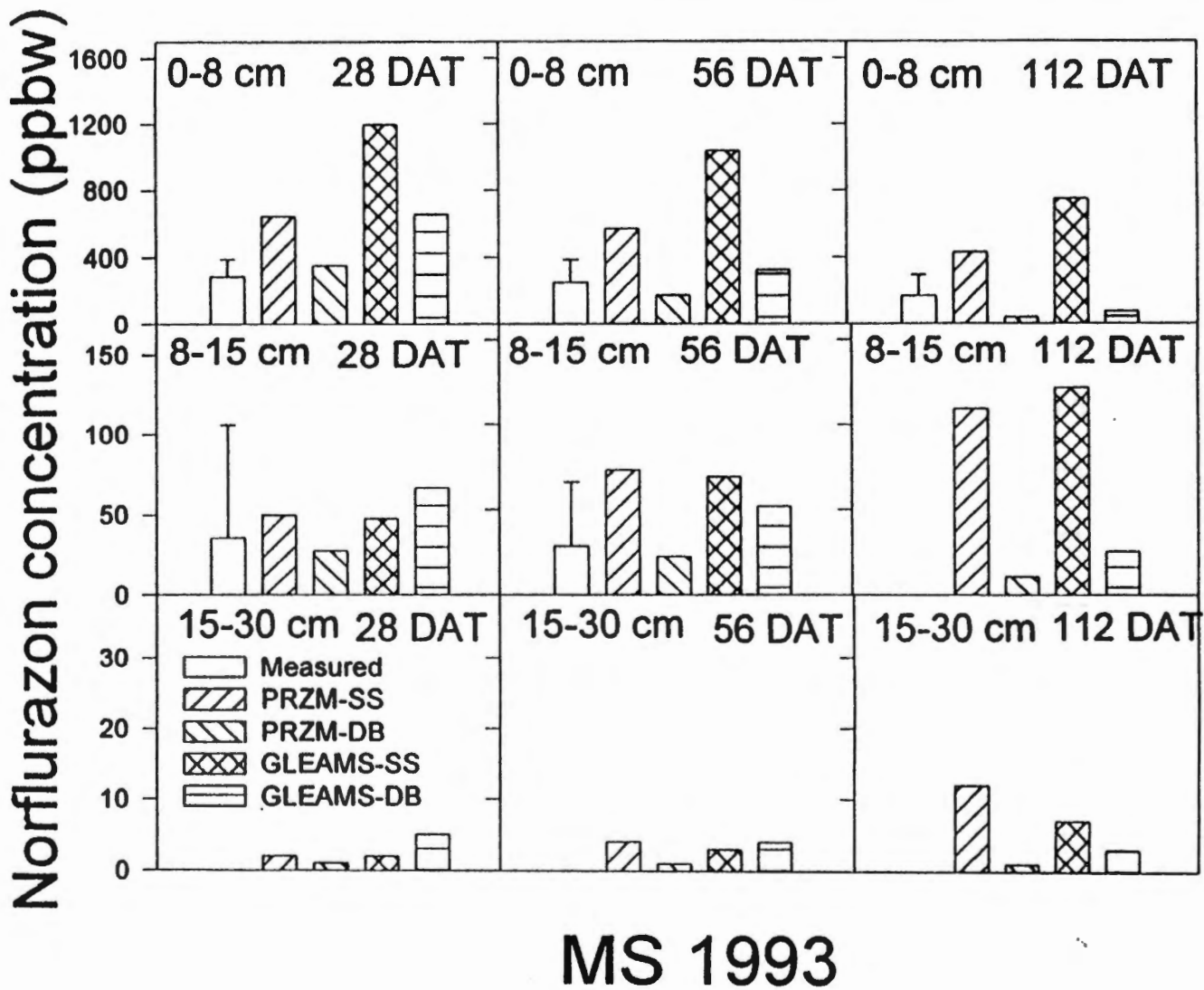


Figure 10. Measured and simulated norflurazon soil concentrations in a Mississippi soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1993. Measured concentrations are mean values of four replications $\pm$ standard error.

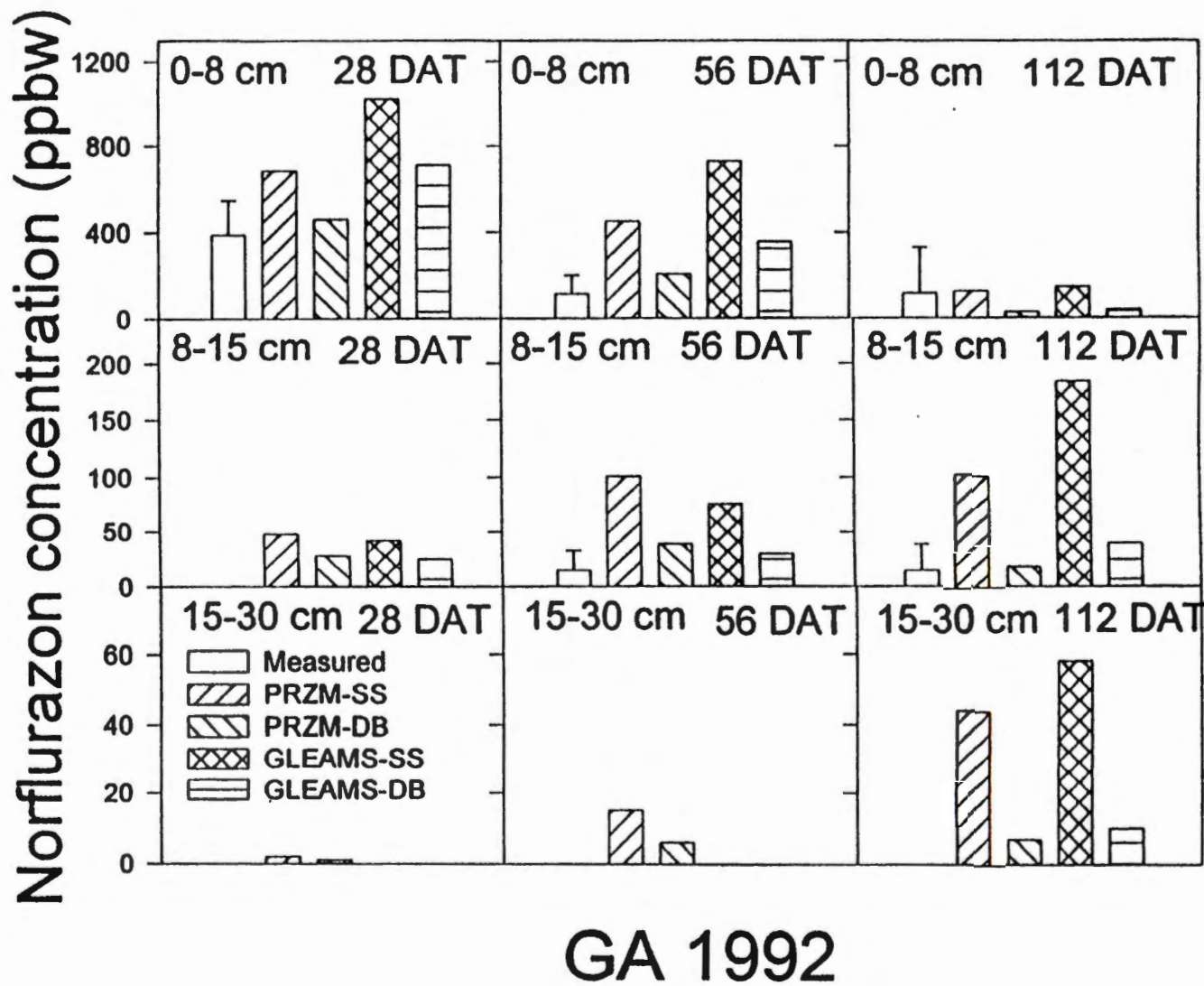


Figure 11. Measured and simulated norflurazon soil concentrations in a Georgia soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1992. Measured concentrations are mean values of four replications $\pm$ standard error.

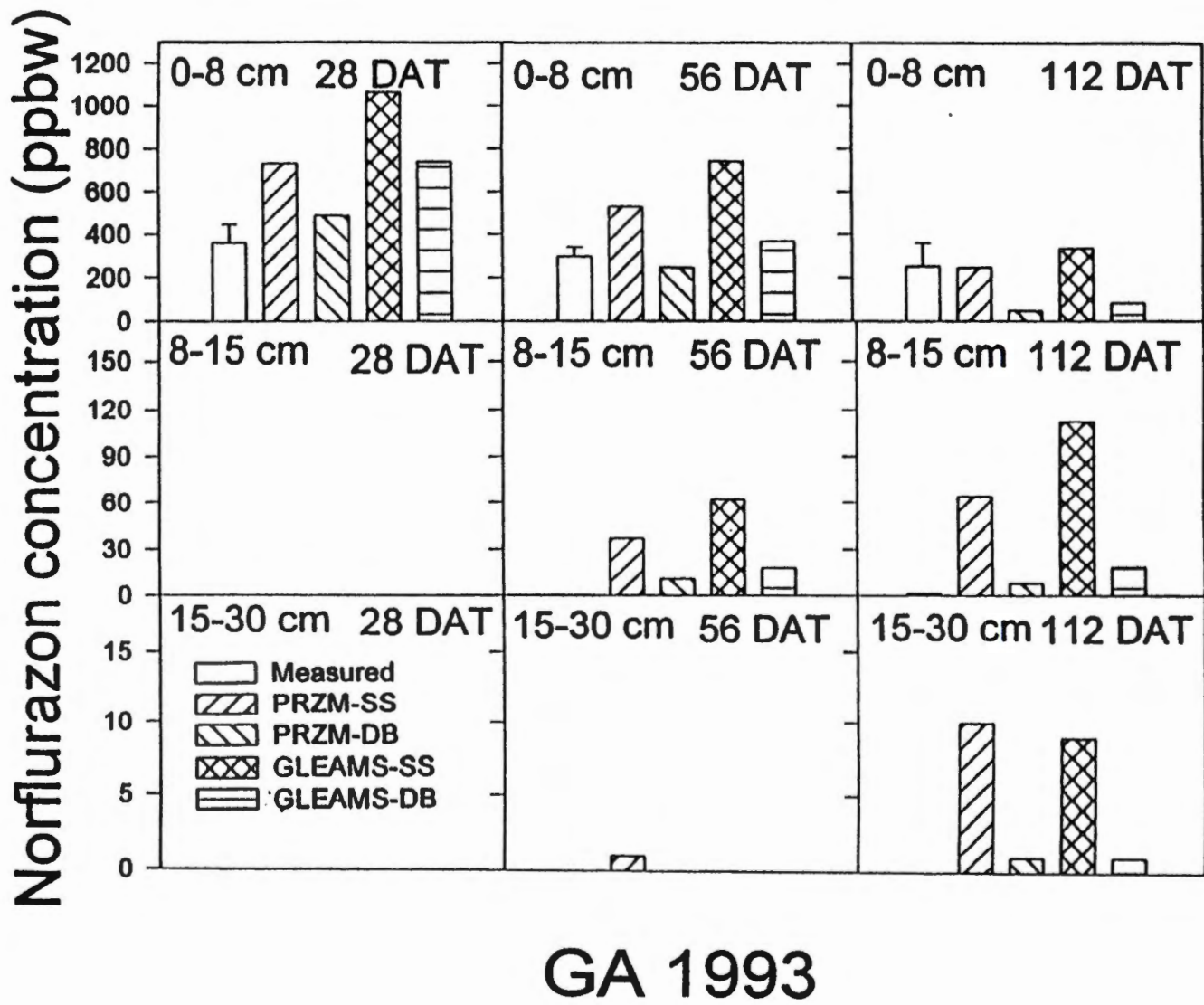


Figure 12. Measured and simulated norflurazon soil concentrations in a Georgia soil from 0 to 8, 8 to 15, and 15 to 30 cm in 1993. Measured concentrations are mean values of four replications $\pm$ standard error.

Table 4. Predicted index values (ratio of predicted to measured) for herbicide concentration (in total mass) in the 0 to 60 cm soil zone in Tennessee for fluometuron and norflurazon.

Year	Herbicide	DAT	Predicted index value ^a			
			PRZM-SS ^b	PRZM-DB ^b	GLEAMS-SS ^b	GLEAMS-DB ^b
1992	Fluometuron	42	1.70*	1.39*	0.38	2.66
1992	Fluometuron	63	3.17	2.34	0.42	4.34
1992	Fluometuron	112	18.3	10.6	1.08*	20.6
1992	Norflurazon	42	1.68*	0.71*	2.65	1.58*
1992	Norflurazon	63	0.73*	0.21	0.98*	0.45
1992	Norflurazon	112	0.97*	0.11	0.88*	0.22
1993	Fluometuron	28	1.51*	1.32*	0.63*	2.51
1993	Fluometuron	56	3.23	2.51	0.34	4.09
1993	Fluometuron	112	320.5	189.7	6.73	282.8
1993	Norflurazon	28	0.43	0.24	0.68*	0.49
1993	Norflurazon	56	0.57*	0.19	0.75*	0.38
1993	Norflurazon	112	1.34*	0.15	1.12*	0.28

^aValues marked with (*) meet criterion $0.5 \leq \text{index} \leq 2.0$.

^bAbbreviations: PRZM-SS, PRZM with site-specific adsorption and degradation parameters; PRZM-DB, PRZM with database adsorption and degradation parameters; GLEAMS-SS, GLEAMS with site-specific adsorption and degradation parameters; GLEAMS-DB, GLEAMS with database adsorption and degradation parameters.

underestimated (Table 2). GLEAMS underestimated fluometuron concentration in the profile 42 and 63 DAT in 1992 but accurately determined concentration 112 DAT in 1992 (Table 4). GLEAMS-DB overestimated fluometuron concentration at each simulation date in both years at TN. Fluometuron DT_{50} was 85 d in this model, which would simulate slower dissipation (Table 3). A limitation of the GLEAMS model is that it allows only one K_{oc} value for each simulation, thus if K_{oc} is constant, then sorption is modeled only as a function of organic carbon. GLEAMS-DB contained lower K_{oc} values (Table 2) and thus predicted greater fluometuron concentrations in subsurface soil horizons compared to GLEAMS-SS (Table 4).

In Mississippi, greater rainfall in April and May of 1993 promoted more rapid dissipation of fluometuron than in 1992 (data not shown). GLEAMS and PRZM accurately predicted fluometuron concentrations 28 DAT in 1992 but overestimated at 56 and 112 DAT (Table 5). Fluometuron concentration at the 0 to 8 cm depth was 375 ppbw 28 DAT. Each model correctly estimated that fluometuron concentration would be greatest in the surface horizon 28 DAT. Fluometuron was not detected below 8-cm at 112 DAT in 1992; however, simulations 112 DAT predicted fluometuron concentration would be greatest at either 8- to 15- or 15- to 30-cm. (Figure 3). All models overestimated fluometuron concentrations in Mississippi in 1993 (Table 5), although 28 and 56 DAT simulations correctly predicted fluometuron concentration would be greatest near the soil surface (Figure 4).

All simulations overestimated fluometuron concentration in the soil profile in Georgia in 1992 (Table 6). PRZM-SS and PRZM-DB predicted fluometuron to move to

Table 5. Predicted index values (ratio of predicted to measured) for herbicide concentration (in total mass) in the 0 to 60 cm soil zone in Mississippi for fluometuron and norflurazon.

Year	Herbicide	DAT	Predicted index value ^a			
			PRZM-SS ^b	PRZM-DB ^b	GLEAMS-SS ^b	GLEAMS-DB ^b
1992	Fluometuron	28	1.24*	1.12*	1.89*	2.31
1992	Fluometuron	56	12.3	11.7	14.8	22.2
1992	Fluometuron	112	6.30	6.05	4.47	9.88
1992	Norflurazon	28	0.87*	0.47	1.59*	0.93*
1992	Norflurazon	56	2.40	0.71*	4.20	1.44*
1992	Norflurazon	112	1.90*	0.17	3.03	0.36
1993	Fluometuron	28	2.36	2.15	3.77	4.57
1993	Fluometuron	56	21.2	19.0	26.7	39.5
1993	Fluometuron	112	167.8	147.7	127.9	277.3
1993	Norflurazon	28	2.16	1.18*	3.89	2.27
1993	Norflurazon	56	2.33	0.71*	4.01	1.37*
1993	Norflurazon	112	3.36	0.31	5.19	0.61*

^aValues marked with (*) meet criterion $0.5 \leq \text{index} \leq 2.0$.

^bAbbreviations: PRZM-SS, PRZM with site-specific adsorption and degradation parameters; PRZM-DB, PRZM with database adsorption and degradation parameters; GLEAMS-SS, GLEAMS with site-specific adsorption and degradation parameters; GLEAMS-DB, GLEAMS with database adsorption and degradation parameters.

Table 6. Predicted index values (ratio of predicted to measured) for herbicide concentration (in total mass) in the 0 to 60 cm soil zone in Georgia for fluometuron and norflurazon.

Year	Herbicide	DAT	Predicted index value ^a			
			PRZM-SS ^b	PRZM-DB ^b	GLEAMS-SS ^b	GLEAMS-DB ^b
1992	Fluometuron	28	2.27	3.69	2.86	5.48
1992	Fluometuron	56	4.86	11.5	4.30	14.7
1992	Fluometuron	112	7.78	20.7	5.52	29.2
1992	Norflurazon	28	1.88*	1.25*	2.72	1.89*
1992	Norflurazon	56	4.35	1.95*	6.08	2.92
1992	Norflurazon	112	2.46	0.46	3.45	0.75*
1993	Fluometuron	28	0.75*	1.25*	0.97*	1.87*
1993	Fluometuron	56	0.72*	1.96*	0.80*	2.89
1993	Fluometuron	112	0.32	1.75*	0.23	2.31
1993	Norflurazon	28	2.03	1.36*	2.95	2.05
1993	Norflurazon	56	1.91*	0.87*	2.70	1.30
1993	Norflurazon	112	1.28*	0.24	1.78*	0.41

^aValues marked with (*) meet criterion $0.5 \leq \text{index} \leq 2.0$.

^bAbbreviations: PRZM-SS, PRZM with site-specific adsorption and degradation parameters; PRZM-DB, PRZM with database adsorption and degradation parameters; GLEAMS-SS, GLEAMS with site-specific adsorption and degradation parameters; GLEAMS-DB, GLEAMS with database adsorption and degradation parameters.

the 30- to 45-cm depth by 28 DAT (data not shown). GLEAMS did not simulate fluometuron movement beyond 15-cm at 28 or 56 DAT, regardless of whether site-specific or database parameters were input (Figure 5). PRZM accurately predicted fluometuron concentration in the profile at GA in 1993 at each simulation date and utilization of database adsorption and degradation parameters did not decrease PRZM accuracy (Table 6). Use of either site-specific or database parameters in the GLEAMS model accurately predicted fluometuron concentration 28 DAT. Site-specific values increased prediction accuracy at 56 DAT; however, neither GLEAMS-SS or GLEAMS-DB accurately predicted fluometuron concentration 112 DAT. All models accurately predicted that fluometuron would not leach below 8-cm by 28 DAT, but over predicted movement in the soil profile at later simulations (Figure 6).

PRZM-SS accurately predicted norflurazon concentration 28 or 42 DAT in three of six simulations, overestimated twice and underestimated once. PRZM accurately simulated norflurazon concentration three of six times at 56 to 63 DAT, and overestimated norflurazon levels in the remaining three situations. Simulations performed 112 DAT were accurate in four of six situations and norflurazon concentration was overestimated twice. PRZM-SS correctly predicted total norflurazon concentration in the profile of TN soil in five of six simulations (Table 4). All models correctly predicted that the greatest concentration of norflurazon would be at the 0- to 8-cm depth (Figures 7, 8). Site-specific inputs increased PRZM accuracy in simulations with TN soil since PRZM-DB underestimated norflurazon concentration in five of six simulations. In the MS soil, PRZM-SS correctly predicted norflurazon concentration in two of three simulations in

1992 but overestimated in each simulation in 1993 (Table 5). GLEAMS-SS over predicted norflurazon concentration at the 0- to 8-cm depth at each simulation date in 1992 and 1993 (Figures 9-12). PRZM-DB performed similarly to PRZM-SS, correctly predicting in three of six simulations. In general, database parameters increased predictive capability at the GA site (Table 6). PRZM-SS gave a correct prediction 28 DAT in 1992 but over predicted norflurazon concentration at later simulations. PRZM has previously been reported to accurately predict norflurazon concentration in a Georgia soil in simulations < 20 DAT and over predict concentration in later simulations (15). PRZM-DB correctly predicted resident norflurazon in the profile 28 and 56 DAT both years but under predicted at the 112 DAT simulation.

GLEAMS-SS performed similarly to the corollary PRZM-SS model, accurately predicting norflurazon concentration within the profile in 7 of 18 simulations. At the TN location, site-specific parameters increased GLEAMS accuracy compared to GLEAMS-DB (Table 4). In contrast, database parameters more accurately predicted norflurazon behavior at MS and GA, achieving correct predictions in 7 of 12 simulations (Tables 5, 6). GLEAMS did not predict norflurazon behavior as accurately in the GA soil as did the PRZM model.

In summary, PRZM more accurately predicted norflurazon than fluometuron when site-specific sorption and degradation parameters were utilized. Site-specific data increased PRZM accuracy in predicting total norflurazon in the soil profile. PRZM simulated total norflurazon concentration in the profile more accurately than GLEAMS, but both models performed similarly with regards to fluometuron. Site-specific data

increased accuracy of GLEAMS to predict fluometuron but not norflurazon. These models better predicted norflurazon than fluometuron, when averaged over all simulations. These models have utility to predict herbicide movement into soil if correct input parameters are used in the simulations. Pesticide sorption and dissipation parameters are important inputs to these models. Model predictions utilizing database sorption and dissipation parameters should be evaluated with caution since differences in pesticide adsorption and dissipation at different soil depths is not accounted for. Pesticide adsorption and dissipation can be slower in subsurface than in surface soils (14) which may limit the utility of database parameters for predicting pesticide concentration in and movement through the soil profile.

ACKNOWLEDGMENTS

The authors thank Ciba Plant Protection and Sandoz Agro Inc. for supplying analytical fluometuron and norflurazon, respectively. This research was partially funded by the Southern Regional Pesticide Impact Assessment Project SR93-07-0-TN. Technical assistance by personnel at the West Tennessee Experiment Station, Jackson, TN; Delta Branch Experiment Station, Stoneville, MS; and Georgia Experiment Station, Griffin, GA is gratefully acknowledged.

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PART VI.
GENERAL SUMMARY

Fluometuron and norflurazon adsorption, dissipation under field and controlled conditions, and movement was examined in three soils representative of the cotton growing regions of the southeastern United States. Fluometuron and norflurazon adsorption was correlated with organic matter but not with other soil characteristics. First-order kinetics explained fluometuron dissipation under both field and laboratory conditions ($r^2 \geq 0.82$). Fluometuron dissipation was more rapid under field conditions than under controlled conditions in most situations. Dry conditions slowed fluometuron dissipation in the field. Fluometuron half-life in surface soils ranged from 9 to 28 days in the field and 11 to 43 days at 30 C. Fluometuron half-life was positively correlated with soil depth and inversely correlated with soil organic matter.

Norflurazon dissipation in surface soils under controlled conditions was not different among locations, with half-life ranging from 63 to 167 days. Norflurazon dissipation under field conditions was more rapid than under controlled conditions. The more rapid dissipation in the field was postulated to be a factor of environmental effects on norflurazon loss mechanisms.

Neither herbicide moved below 15 cm in the soil profile in any soil, thus leaching was not a significant contribution to fluometuron or norflurazon degradation. The PRZM and GLEAMS computer simulation models better predicted norflurazon concentration in the soil profile than fluometuron concentration when averaged over all model by parameter input combinations. Site-specific sorption and dissipation parameters increased PRZM accuracy in predicting total norflurazon concentration; however, database sorption

and dissipation inputs resulted in better predictions of fluometuron concentration. Site-specific data increased the accuracy of GLEAMS to predict fluometuron concentration but did not increase accuracy of norflurazon predictions. PRZM provided more correct predictions of norflurazon concentration than did GLEAMS; however, both models performed similarly with regards to fluometuron predictions.

These data indicate a significant relationship between soil organic matter content, environment, and fluometuron and norflurazon concentration. Fluometuron has the potential to leach in low sorptive soils; however, movement through the soil profile was minimal in all soils evaluated. Dry conditions slowed fluometuron and norflurazon dissipation in surface soils. PRZM and GLEAMS correctly predicted total herbicide concentration in the profile in < 60% of the simulations; however, site-specific inputs resulted in more accurate predictions than database-generated parameters. Further research is needed to evaluate the usefulness of these models for predictions of herbicide movement in these soils.

APPENDIX

Table A-1. Desmethyl fluometuron (DMFM) concentrations in soil from the 0- to 8-cm soil zone at Tennessee (TN), Mississippi (MS) and Georgia (GA). Values are the mean $\pm$ (standard error) of one treatment and four replications.

State	DAT	1992	1993
		DMFM	
		ng g ⁻¹	
TN	0	66 (18)	27 (44)
TN	14	137 (49)	259 (231)
TN	28		415 (189)
TN	42	265 (185)	
TN	56	292 (77)	268 (79)
TN	112	264 (139)	96 (56)
MS	0	0 (0)	12 (24)
MS	14	164 (79)	107 (76)
MS	28	199 (38)	65 (47)
MS	56	174 (27)	24 (30)
MS	112	0 (0)	10 (17)
GA	0	0 (0)	0 (0)
GA	14	147 (29)	43 (37)
GA	28	137 (27)	0 (0)
GA	56	72 (56)	52 (89)
GA	112	60 (13)	63 (19)

Table A-2. Measured and simulated concentrations of fluometuron at several soil depths from the 0- to 120-cm horizon at Tennessee in 1992.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm	ppbw				
42	0-8	303 (178)	97	75	34	180
42	8-15	34 (69)	146	118	45	253
42	15-30	ND ^a	102	84	24	191
42	30-45	ND	54	45	4	47
42	45-60	ND	25	22	1	20
42	60-90	ND	11	10	— ^b	—
42	90-120	ND	4	4	—	—
63	0-8	172 (119)	41	28	8	118
63	8-15	ND	107	76	22	188
63	15-30	ND	109	80	19	180
63	30-45	ND	71	54	4	50
63	45-60	ND	39	30	1	17
63	60-90	ND	18	15	—	—
63	90-120	ND	8	6	—	—
112	0-8	24 (29)	11	6	< 1	49
112	8-15	ND	49	27	5	133
112	15-30	ND	78	44	9	131
112	30-45	ND	74	43	2	34
112	45-60	ND	54	33	ND	11
112	60-90	ND	34	21	—	—
112	90-120	ND	17	11	—	—

^aND, not detected.

^bGLEAMS can simulate a maximum of five soil horizons.

Table A-3. Measured and simulated concentrations of norflurazon at several soil depths from the 0- to 120-cm horizon at Tennessee in 1992.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm		ppbw			
42	0-8	329 (230)	363	132	744	400
42	8-15	ND ^a	139	66	126	108
42	15-30	ND	30	19	9	12
42	30-45	ND	5	4	ND	1
42	45-60	ND	1	1	ND	ND
42	60-90	ND	ND	ND	— ^b	—
42	90-120	ND	ND	ND	—	—
63	0-8	710 (139)	271	61	559	227
63	8-15	ND	159	48	130	80
63	15-30	ND	45	17	11	10
63	30-45	ND	9	5	ND	ND
63	45-60	ND	2	1	ND	ND
63	60-90	ND	ND	ND	—	—
63	90-120	ND	ND	ND	—	—
112	0-8	368 (74)	170	13	304	66
112	8-15	ND	154	16	104	32
112	15-30	ND	62	9	10	4
112	30-45	ND	16	3	ND	ND
112	45-60	ND	3	1	ND	ND
112	60-90	ND	ND	ND	—	—
112	90-120	ND	ND	ND	—	—

^aND, not detected.

^bGLEAMS can simulate a maximum of five soil horizons.

Table A-4. Measured and simulated concentrations of fluometuron at several soil depths from the 0- to 120-cm horizon at Tennessee in 1993.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm		ppbw			
28	0-8	393 (198)	334	278	193	807
28	8-15	45 (90)	212	189	70	240
28	15-30	ND ^a	58	55	9	33
28	30-45	ND	10	10	1	3
28	45-60	ND	2	2	ND	ND
28	60-90	ND	ND	ND	— ^b	—
28	90-120	ND	ND	ND	—	—
56	0-8	208 (154)	126	87	28	511
56	8-15	ND	207	154	35	293
56	15-30	ND	131	105	7	42
56	30-45	ND	50	42	ND	2
56	45-60	ND	14	13	ND	ND
56	60-90	ND	3	3	—	—
56	90-120	ND	1	1	—	—
112	0-8	ND	56	28	< 1	263
112	8-15	ND	147	81	7	224
112	15-30	ND	132	79	3	37
112	30-45	ND	65	41	ND	1
112	45-60	ND	25	17	ND	ND
112	60-90	ND	8	6	—	—
112	90-120	ND	2	2	—	—

^aND, not detected.

^bGLEAMS can simulate a maximum of five soil horizons.

Table A-5. Measured and simulated concentrations of norflurazon at several soil depths from the 0- to 120-cm horizon at Tennessee in 1993.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm	ppbw				
28	0-8	1470 (370)	562	294	1010	713
28	8-15	45 (60)	87	65	25	24
28	15-30	ND ^a	6	6	1	1
28	30-45	ND	ND	ND	ND	ND
28	45-60	ND	ND	ND	ND	ND
28	60-90	ND	ND	ND	---	---
28	90-120	ND	ND	ND	---	---
56	0-8	941 (384)	410	112	716	356
56	8-15	87 (101)	147	58	46	29
56	15-30	ND	22	13	1	1
56	30-45	ND	2	2	ND	ND
56	45-60	ND	ND	ND	ND	ND
56	60-90	ND	ND	ND	---	---
56	90-120	ND	ND	ND	---	---
112	0-8	368 (74)	295	25	371	93
112	8-15	ND	153	19	43	13
112	15-30	ND	31	6	1	ND
112	30-45	ND	4	1	ND	ND
112	45-60	ND	ND	ND	ND	ND
112	60-90	ND	ND	ND	---	---
112	90-120	ND	2	2	---	---

^aND, not detected.

^bGLEAMS can simulate a maximum of five soil horizons.

Table A-6. Measured and simulated concentrations of fluometuron at several soil depths from the 0- to 120-cm horizon at Mississippi in 1992.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm		ppbw			
28	0-8	375 (301)	511	388	707	732
28	8-15	105 (150)	72	131	160	281
28	15-30	ND ^a	2	11	17	45
28	30-45	ND	ND	ND	2	7
28	45-60	ND	ND	ND	ND	1
28	60-90	ND	ND	ND	---	---
28	90-120	ND	ND	ND	---	---
56	0-8	38 (75)	295	136	373	436
56	8-15	ND	136	155	152	269
56	15-30	ND	26	71	27	73
56	30-45	ND	3	18	4	16
56	45-60	ND	ND	4	ND	3
56	60-90	ND	ND	1	---	---
56	90-120	ND	ND	ND	---	---
112	0-8	53 (44)	105	20	81	106
112	8-15	ND	116	63	97	192
112	15-30	ND	51	68	34	111
112	30-45	ND	14	43	4	20
112	45-60	ND	3	20	ND	2
112	60-90	ND	ND	8	---	---
112	90-120	ND	ND	2	---	---

^aND, not detected.

^bGLEAMS can simulate a maximum of five soil horizons.

Table A-7. Measured and simulated concentrations of norflurazon at several soil depths from the 0- to 120-cm horizon at Mississippi in 1992.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm	ppbw				
28	0-8	718 (342)	665	349	1225	704
28	8-15	81 (159)	26	25	28	32
28	15-30	ND ^a	ND	ND	1	1
28	30-45	ND	ND	ND	ND	ND
28	45-60	ND	ND	ND	ND	ND
28	60-90	ND	ND	ND	— ^b	—
28	90-120	ND	ND	ND	—	—
56	0-8	232 (48)	567	152	1055	348
56	8-15	37 (72)	70	35	58	34
56	15-30	ND	4	3	2	2
56	30-45	ND	ND	ND	ND	ND
56	45-60	ND	ND	ND	ND	ND
56	60-90	ND	ND	ND	—	—
56	90-120	ND	ND	ND	—	—
112	0-8	254 (192)	418	30	721	76
112	8-15	36 (40)	113	16	149	25
112	15-30	ND	13	3	8	2
112	30-45	ND	1	ND	ND	ND
112	45-60	ND	ND	ND	ND	ND
112	60-90	ND	ND	ND	—	—
112	90-120	ND	ND	ND	—	—

^aND, not detected.

^bGLEAMS can simulate a maximum of five soil horizons.

Table A-8. Measured and simulated concentrations of fluometuron at several soil depths from the 0- to 120-cm horizon at Mississippi in 1993.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm	ppbw				
28	0-8	231 (169)	412	314	612	506
28	8-15	ND ^a	118	142	199	305
28	15-30	ND	15	28	38	113
28	30-45	ND	1	3	6	31
28	45-60	ND	ND	ND	1	7
28	60-90	ND	ND	ND	— ^b	—
28	90-120	ND	ND	ND	—	—
56	0-8	21 (40)	273	178	335	304
56	8-15	ND	135	143	165	267
56	15-30	ND	26	43	37	120
56	30-45	ND	3	7	5	29
56	45-60	ND	ND	1	1	7
56	60-90	ND	ND	ND	—	—
56	90-120	ND	ND	ND	—	—
112	0-8	ND	105	46	87	86
112	8-15	ND	111	84	89	151
112	15-30	ND	46	55	35	123
112	30-45	ND	12	22	5	34
112	45-60	ND	2	7	ND	4
112	60-90	ND	ND	2	—	—
112	90-120	ND	ND	ND	—	—

^aND, not detected.

^bGLEAMS can simulate a maximum of five soil horizons.

Table A-9. Measured and simulated concentrations of norflurazon at several soil depths from the 0- to 120-cm horizon at Mississippi in 1993.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm	ppbw				
28	0-8	287 (102)	642	351	1194	656
28	8-15	36 (70)	50	28	48	67
28	15-30	ND ^a	2	1	2	5
28	30-45	ND	ND	ND	ND	ND
28	45-60	ND	ND	ND	ND	ND
28	60-90	ND	ND	ND	---	---
28	90-120	ND	ND	ND	---	---
56	0-8	249 (136)	569	174	1035	324
56	8-15	29 (37)	73	23	69	52
56	15-30	ND	4	1	3	4
56	30-45	ND	ND	ND	ND	ND
56	45-60	ND	ND	ND	ND	ND
56	60-90	ND	ND	ND	---	---
56	90-120	ND	ND	ND	---	---
112	0-8	167 (125)	428	41	747	74
112	8-15	ND	109	11	121	26
112	15-30	ND	12	1	7	3
112	30-45	ND	1	ND	ND	ND
112	45-60	ND	ND	ND	ND	ND
112	60-90	ND	ND	ND	---	---
112	90-120	ND	ND	ND	---	---

^aND, not detected.

^bGLEAMS can simulate a maximum of five soil horizons.

Table A-10. Measured and simulated concentrations of fluometuron at several soil depths from the 0- to 60-cm horizon at Georgia in 1992.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm	ppbw				
28	0-8	201 (162)	339	510	513	967
28	8-15	ND ^a	95	168	70	154
28	15-30	ND	16	38	ND	ND
28	30-45	ND	2	6	ND	ND
28	45-60	ND	ND	1	ND	ND
56	0-8	50 (79)	91	185	182	644
56	8-15	10 (18)	85	172	81	249
56	15-30	ND	43	106	ND	ND
56	30-45	ND	16	54	ND	ND
56	45-60	ND	5	21	ND	ND
112	0-8	15 (27)	2	5	1	5
112	8-15	ND	11	23	12	73
112	15-30	ND	20	47	17	91
112	30-45	ND	21	56	13	65
112	45-60	ND	15	49	8	41

^aND, not detected.

Table A-11. Measured and simulated concentrations of norflurazon at several soil depths from the 0- to 60-cm horizon at Georgia in 1992.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm	ppbw				
28	0-8	389 (158)	684	460	1021	713
28	8-15	ND ^a	48	28	42	25
28	15-30	ND	2	1	ND	ND
28	30-45	ND	ND	ND	ND	ND
28	45-60	ND	ND	ND	ND	ND
56	0-8	118 (84)	450	209	731	357
56	8-15	15 (18)	101	39	76	30
56	15-30	ND	15	6	ND	ND
56	30-45	ND	2	1	ND	ND
56	45-60	ND	5	21	ND	ND
112	0-8	120 (210)	127	29	148	39
112	8-15	15 (24)	102	18	184	40
112	15-30	ND	44	7	58	10
112	30-45	ND	12	2	18	3
112	45-60	ND	3	ND	4	1

^aND, not detected.

Table A-12. Measured and simulated concentrations of fluometuron at several soil depths from the 0- to 60-cm horizon at Georgia in 1993.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm	ppbw				
28	0-8	600 (249)	450	749	583	1121
28	8-15	ND ^a	ND	ND	ND	ND
28	15-30	ND	ND	ND	ND	ND
28	30-45	ND	ND	ND	ND	ND
28	45-60	ND	ND	ND	ND	ND
56	0-8	275 (263)	170	460	178	650
56	8-15	27 (53)	42	115	66	234
56	15-30	ND	5	13	1	4
56	30-45	ND	ND	ND	ND	ND
56	45-60	ND	ND	ND	ND	ND
112	0-8	218 (132)	28	201	17	224
112	8-15	13 (24)	23	123	27	237
112	15-30	ND	10	39	5	47
112	30-45	ND	3	9	1	5
112	45-60	ND	ND	2	ND	ND

^aND, not detected.

Table A-13. Measured and simulated concentrations of norflurazon at several soil depths from the 0- to 60-cm horizon at Georgia in 1993.

DAT	Depth	Measured	PRZM-SS	PRZM-DB	GLEAMS-SS	GLEAMS-DB
	cm	ppbw				
28	0-8	360 (86)	732	488	1063	738
28	8-15	ND ^a	ND	ND	ND	ND
28	15-30	ND	ND	ND	ND	ND
28	30-45	ND	ND	ND	ND	ND
28	45-60	ND	ND	ND	ND	ND
56	0-8	295 (44)	530	246	741	368
56	8-15	ND	37	11	62	18
56	15-30	ND	1	ND	ND	ND
56	30-45	ND	ND	ND	ND	ND
56	45-60	ND	ND	ND	ND	ND
112	0-8	251 (108)	245	51	333	85
112	8-15	ND	64	8	112	19
112	15-30	ND	10	1	9	1
112	30-45	ND	1	ND	ND	ND
112	45-60	ND	ND	ND	ND	ND

^aND, not detected.

Table A-14. Model parameters for PRZM simulations.

Parameter	Units	Value
Pan factor		0.75
Snow factor		0
Pan evaporation flag		1
Minimum depth evaporation extracted	cm	60
Initial crop number		1
Erosion flag		1
Number of crop simulations		1
Crop number		1
Maximum interception crop storage	cm	0.10
Maximum active crop root depth	cm	20
Maximum areal crop coverage	%	60
Runoff curve number		88
Number of cropping periods		1
Number of pesticide applications		1
Depth of pesticide incorporation	cm	1
Pesticide application model		1
Soil core depth (total)	cm	120
Plant uptake efficiency factor		0
Total number of simulation compartments		7
Bulk density flag		1
Soil moisture calculation flag		1
Pesticide sorption calculation flag		0
Switch for soil hydraulics		1
Soil partition calculation model		0

Table A-15. Model parameters for GLEAMS simulations.

Parameter	Units	Value
Effective saturated conductivity	cm h ⁻¹	0.20
Initial plant available water fraction		0.75
Soil evaporation parameter		3.3
SCS curve number		70
Hydraulic slope of field		0.0001
Effective rooting depth	cm	60
Specific surface area (clay)	m ² g ⁻¹	20
Specific surface area (organic matter)	m ² g ⁻¹	1000
Depth of incorporation	cm	1
Coefficient of plant uptake		0
Coefficient of pesticide transformation		1
Code of pesticide application		1

VITA

William Thomas (Todd) Willian was born in Munfordville, Kentucky on February 7, 1965. He is the son of Mr. and Mrs. Kermit Willian. He was raised on a family-owned burley tobacco and dairy farm. He attended Hart County High School and graduated in May 1983. He entered Berea College in September 1983 and received a Bachelor of Arts in Biology in May 1988. He enrolled in the graduate program at the University of Tennessee in August 1989 and earned a Masters of Science in Plant and Soil Science with emphasis in Weed Management under the direction of Dr. G. Neil Rhodes, Jr. and Dr. William A. Krueger. He began his Ph.D. research in May 1992 and later accepted the position of Research Associate in Plant and Soil Science under the guidance of Dr. Thomas C. Mueller. Todd is the author or co-author of 17 abstracts and 7 research reports and has twice made presentations at the Milan No-Till Field Day. Todd received the Tennessee Agricultural Chemicals Association (TACA) Outstanding Graduate Student Award in 1994, and has twice placed first in the TACA poster contest. He has served as Graduate Student Representative in the Department of Plant and Soil Science, and has been an active member of the University of Tennessee Weed Team. Upon graduation he plans to work in the agrichemical industry in the area of Research and Development.