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EFFECTS OF ALTERNATIVE SPRINKLER IRRIGATION PARAMETERS ON PESTICIDE MOVEMENT

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SUMMARY:

The relative reduction in potential ground-water contamination due to pesticides at several sites in Utah was determined by comparing alternative sprinkler system designs, water management practices and pesticides. Pesticide contamination of ground water can be reduced by careful selection of pesticides, properly designed irrigation systems and improved water management techniques. Procedures for selecting an appropriate sprinkler design and pesticides are presented.

KEYWORDS:

pesticides, sprinkler irrigation, partition coefficient, half-life, relative amount, irrigation schedule, uniformity coefficient, fraction of area adequately irrigated, distribution coefficient, soil texture, infiltrated water depths.

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ABSTRACT

The relative reduction in potential ground-water contamination due to pesticides at several sites in Utah was determined by comparing alternative irrigation system designs, water management practices and pesticides.

Alternative sprinkler irrigation distribution coefficients were used to estimate infiltration depths. The movement of pesticides through soils following sprinkler irrigations was simulated with one-dimensional model.

Pesticide contamination of ground water can be reduced by careful selection of pesticides, properly designed irrigation systems and improved water management techniques. Procedures for selecting an appropriate sprinkler design and pesticide are presented.

Key words: pesticides, sprinkler irrigation, partition coefficient, half life, relative amount, irrigation schedule, uniformity coefficient, fraction of area adequately irrigated, distribution coefficient, soil textures, infiltrated water depths.

INTRODUCTION

Pesticides minimize crop losses by insects, pathogens, weeds and other pests but can contaminate ground water. Potential contamination is of particular concern in areas where ground water is the source of culinary water.

There are more than 45,000 registered pesticides (USEPA, 1987). Almost 500 million kg of pesticides are used in the U.S. each year (Pimentel and Levitan, 1986). Of these, approximately 60 percent are herbicides, 24 percent are insecticides and 16 percent are fungicides. About 68 percent of these are used on agricultural land, where every dollar spent on pesticides returns about 4 dollars in agricultural production.

Zaki et al. (1982) found aldicarb, a carbamate pesticide, in ground water in Suffolk County, New York. More than 8,000 wells were tested. Aldicarb levels exceeded the state-recommended safety limits in 13 percent of these wells. According to Sum (1986) the USEPA reported that 17 pesticides were detected in the ground water of 23 states. Pesticide concentrations ranged from a trace to several hundred parts per million. In Oahu, Hawaii, pumping was discontinued at several essential wells due to ground-water contamination by pesticides used in pineapple production (Lau and Mink, 1987), probably nematicides (Oki and Giambelluca, 1987). In the Mahantango Creek watershed, Pennsylvania, atrazine was detected in 14 of 20 wells that were tested (Pionke et al., 1988).

About 50 percent of the U.S. population obtains drinking water from ground water (Leonard et al., 1988). This percentage is 63 percent in Utah (Waddell, 1987). Most rural residents totally rely on ground water for domestic needs.

Ground-water contamination by pesticides depends on such factors as agricultural practices, soils, plant uptake, geology, hydrology, climate, topography and pesticide properties.

This study examined how appropriate management (sprinkler irrigation system design and pesticide selection) affected potential pesticide contamination of ground water.

METHODOLOGY

Identification of Study Sites

Sites in the 29 counties of Utah that may be subject to ground-water contamination were identified and ranked (Eisele et al., 1989). First, a rapid screening procedure, DRASTIC (Aller et al., 1985), was used to identify sites with a high risk. Subsequently, a one-dimensional simulation model, CMLS (Nofziger and Hornsby, 1986, 1988), was used to simulate the movement of pesticides in unsaturated soils at locations where the risk of contamination was higher (Eisele et al. (1989). DRASTIC and CMLS rankings were compared by Ehteshami et al. (1991).

We identified six agricultural areas with a relatively high potential for ground-water contamination, based on the findings of Eisele et al. (1989) and Ehteshami et al. (1991). These study sites were located in Cache, Davis, Sevier, Utah, Washington and Weber counties of Utah.

Ground-water contamination potential from pesticides depends on agricultural practices, pesticide characteristics, time of pesticide application, and soil profile characteristics. For each of the selected sites, the data concerning these factors were obtained and the effects of alternative water management practices, pesticides and crops were simulated. Steps involed in the simulation procedure are illustrated in Figure 1. The infiltration was estimated using a distribution coefficient (Ha) approach (Hart and Reynolds, 1965). Estimated infiltration values were then used in CMLS to predict pesticide movement. The relative potential for ground-water contamination was determined and relative importance of each factor on ground-water contamination was assessed.

Simulation of Pesticide Movement Using CMLS

Based on the study of Eisele et al. (1989), we selected CMLS as the most appropriate pesticide transport model for this study. The following assumptions are used in CMLS (Nofziger and Hornsby, 1986, 1988):

1. All soil water residing in pore spaces participates in the transportation process. If this assumption is not valid and a preferential flow is present, a portion of the soil water will be bypassed during flow, and the model will underestimate the depth of the chemical front.

2. Water entering the soil redistributes instantaneously to field capacity. This assumption is more accurate for coarse-textured soils.

3. Water is removed by evapotranspiration from each layer in the root zone in proportion to the relative amount of water available in that layer. A uniform root distribution is assumed. This assumption is not strictly valid for many situations. More precise schemes for dealing with evapotranspiration would require information about the root distribution and the soil hydraulic properties.

4. Upward movement of soil water does not occur anywhere in the soil profile. Water is lost from the root zone by evapotranspiration and is not replenished from below.

5. The adsorption process can be described by a linear, reversible equilibrium model. If the sorption coefficient is described by non-linear isotherm, the partition coefficient decreases with increasing concentration of the chemical. Thus the depth to which the chemical will be leached will depend upon the concentration. This aspect is probably not significant for the concentration range of interest in most

agricultural applications. When adsorption equilibrium is not instantaneous, the chemical will be leached to a greater depth than predicted here. Irreversible sorption would result in less leaching.

6. The half-life time for biological degradation of the chemical is constant with time and soil depth. Degradation rate coefficients are dependent upon a variety of environmental factors, primarily temperature and soil-water content. Hence, seasonal changes in rate coefficients can be expected. Also, with decreasing microbial activity at greater soil depths, the degradation rate coefficient may decrease with depth. Sufficient data are not available to formulate mathematical relationships to describe these effects.

CMLS simulates: (a) the movement of the chemical and (b) the degradation of the chemical. Chemicals move only with soil-water movement. A volume balance approach is used to calculate water movement. At the beginning of the simulation, each layer in the soil profile is assumed to be at field capacity. Water is available for plants if the water content of any layer of the root zone exceeds the permanent wilting point, as expressed by the following relationship:

$$W_j^a = t_j \; (\theta_j \; - \theta_j^{pwp}) \qquad [1]$$

where W_j^a is the available water in the layer j (mm), t_j is the thickness of the layer j (mm), θ_j is the volumetric water content of layer j and θ_j^{pwp} is the volumetric water content at permanent wilting point of layer j. Total available water, W_{tot}^a in the root zone is the sum of available water of all the root zone layers. If W_{tot}^a exceeds the evapotranspiration (ET_{crop}) for a day, the depletion in each root zone layer is in proportion to available water amount in that layer as shown in the following equation:

$$\theta_j = \theta'_j - [ET_{crop} \ \ W_j^a] / [W_{tot}^a \ \ t_j] \qquad [2]$$

where θ'_j is the volumetric water content of layer j prior to adjustment. If the total available water is less than the evapotranspiration demand, water content in all layers of the root zone is assumed equal to:

$$\theta_j = \theta_j^{pwp} \qquad [3]$$

In equation 3, no effect of soil water content on ET when the volumetric water content of the soil is nearing wilting point is assumed. In the field, ET may actually decrease due to stress long before θ^{pwp} level is reached.

After an irrigation and/or rain occurs, the water content of each layer is adjusted, starting with the upper soil layer (j=1). Using the following equation, the soil-water deficit for that layer is determined:

$$swd_j = t_j \ (\theta_j^{fc} - \theta_j) \qquad [4]$$

where swd_j is the soil-water deficit of layer j (mm) and θ_j^{fc} is the volumetric water content of the layer at field capacity. If the infiltrating amount (irrigation and/or rain), I_j, is greater than swd_j, then:

$$\theta_j = \theta_j^{fc} \qquad [5]$$

$$I_{j+1} = I_j - swd_j \qquad [6]$$

If I_j is less than swd_j, then

$$\theta_j = \theta'_j + I_j / t_j \qquad [7]$$

$$I_{j+1} = 0 \qquad [8]$$

Due to adsorption processes, chemicals advance less far in depth than water. A reversible equilibrium and linear adsorption model simulates the retardation of the chemical movement. The following equations predict chemical movement:

if $W_p > 0$, $$d^s - d'^s = W_p / (RF \ \ \theta^{fc}) \qquad [9]$$

if $W_p \leq 0$, $$d^s - d'^s = 0 \qquad [10]$$

$$RF = 1 + (BD \ \ K_d / \theta^{fc}) \qquad [11]$$

$$K_d = K_{oc} \ \ OC \qquad [12]$$

where

W_p is the amount of water passing the depth d^s (mm), d^s is solute front depth (mm), d'^s is the solute front depth prior to the adjustment (mm), RF is the retardation factor, θ^{fc} is the soil-water content on a volume basis at field capacity, BD is soil bulk density (g/cm^3), K_d is the partition coefficient of the chemical in soil (ml/g soil), K_{oc} is the organic carbon partition coefficient (ml/g OC) and OC is the organic carbon content of the soil (OC fraction).

In the soil, chemicals are continuously exposed to degradation processes. Relative amount (RA), the fraction of the applied chemical remaining in a soil profile, is predicted by CMLS:

$$RA = e^{-tr \ln(2) / t_{1/2}} \qquad [13]$$

where tr is the travel time since the chemical was applied (days) and $t_{1/2}$ is the biological degradation half-life of the chemical (days).

In CMLS, the following parameters are used as input:

- soil properties (bulk density, water content at field capacity and permanent wilting point and soil organic carbon content)
- chemical properties of the pesticide (partition coefficient and degradation half-life)
- climatic and cultural factors (plant root depth, daily rainfall + irrigation and daily evapotranspiration amounts)

The outputs given by CMLS, among others, include, travel time (tr) for chemicals to move to selected depths and relative amount (RA) of pesticides remaining at those times in the soil profile.

The average sprinkler irrigation depth infiltrated over a field was estimated using the distribution coefficient (Ha) approach (Hart and Reynolds, 1965). Their approach recognizes that the average infiltration depth is a function of both uniformity coefficient (UC) and the percent of area that is at least adequately irrigated (F).

They assumed that the distribution of infiltrated water depths in an overlapped sprinkler pattern approximates the normal distribution. Then the average infiltrated or applied depth of water, Vi (mm) can be determined by:

$$Vi = Zreq/Ha \tag{14}$$

where Zreq is the required irrigation depth (mm) at a given date and Ha is the distribution coefficient (a fraction of the mean applied or infiltrated depth).

They reported Ha values for a range of UC values (60 to 99.9 %) for an assumed range of fractions of the field area ,F (50 to 100 %) that can be adequately irrigated. The UC is given by the following empirical relationship:

$$UC = 100\ (1.0 - \sum |z-m|\ /\ \sum z) \tag{15}$$

where UC is uniformity coefficient (%), z is the individual depth (mm) of catch observations from uniformity test, and m is the mean depth (mm) of observations.

In this study, Ha values reported by Hart and Reynolds (1965) were used to estimate average depth of water infiltrated in the soil profile. We assumed table combination of F (60, 70, 80, 90, and 100 percent) and UC (60, 80, and 96 percent). The UC values over 96 percent were omitted because irrigation uniformity higher than this is economically nonfeasible. It requires excessively close spacing of sprinklers.

For all combinations, the irrigation amount required in the soil profile was assumed 45 mm. The average infiltrated depth of irrigation (Vi) for each combination was computed by dividing Zreq (45 mm) by the appropriate Ha. For example, for UC value of 96 % and F of 100 %, the Ha value reported by Hart and Reynolds (1965) was 0.85. Therefore, the average infiltrated depth of irrigation (Vi) for a Zreq of 45 mm is 53 mm (45 mm / 0.85 = 53 mm). These infiltrated irrigation depths were used in CMLS to predict the relative amounts (RA) of pesticides for a known site, crop, irrigation schedule and system.

Out of many analyses performed for each of the 6 selected areas, only representative results are presented here. These illustrate the two methodologies presented in this paper. The first methodology illustrates the selection of a sprinkler irrigation system design for a range of pesticide RA values for a given site, crop and irrigation schedule. The second methodology illustrates the selection of a pesticide for a given irrigation system, schedule, site, crop, and desired RA. These methodologies can be used with other simulation models if the models more accurately represent preferential flow.

RESULTS AND DISCUSSION

Irrigation System Design and Pesticide Alternatives

Irrigation system design is a very important factor in estimating pesticide leaching in irrigated areas. Pesticide leaching can be significantly reduced by an efficient irrigation system design. Figure 2 illustrates how Ha and RA are affected by selecting the fraction (F) of area to be adequately irrigated for several UCs. For example, note that each of three combinations of UC and F (60%, 60%), (80%, 70%), and (96%, 100%), can yield the same RA (0.15) of hexazinone remaining at 2 m soil depth. Pesticide travel time to that depth was the same for all these combinations.

One can choose any one of these UC and F combinations to achieve the target RA of hexazinone. If one combination of UC and F does not yield an acceptable RA of hexazinone, a different might. The third combination of UC (96%) and F (100%) is environmentally relatively inoffensive although it might be expensive requiring much sprinkler hardware. This sprinkler irrigation system design was used for all subsequently discussed CMLS simulations.

A second situation exists when the irrigation system and schedule are in place and farmers must select an appropriate pesticide. Farmers usually have several pesticides to choose from. Each has different values of K_{oc} and $t_{1/2}$. To develop decision support nomograms, many simulations were performed in which K_{oc} varied from 1 to 100 ml/g OC and $t_{1/2}$ ranged from 10 to

100,000 days. We assumed alfalfa was irrigated using a 1986 irrigation schedule in Cache County, Utah. RA remaining when the pesticide reaches to soil depths of 0.5 m and 2.0 m were predicted (Figures 3 and 4 respectively).

Figure 3 illustrates that for low K_{oc} values, as the $t_{1/2}$ decreases, the RA remaining at 0.5 m soil depth also decreases. For a given $t_{1/2}$, as the K_{oc} increases, the RA remaining at 0.5 m soil depth decreases. For higher K_{oc} values (greater than 75 ml/g OC), the predicted RA is 0.00 regardless of $t_{1/2}$. This shows that all alfalfa pesticides having K_{oc} > 75 ml/g OC are safe to use in this situation even if $t_{1/2}$ is 100,000 days.

Figure 4 shows similar results for RA values for pesticides reaching a 2 m soil depth (below the alfalfa root zone). No pesticide with $t_{1/2}$ of 10 days or less percolates to 2 m. Short half life pesticides biodegrade long before they can percolate deeply at that site. The 2 m soil depth adsorbs more pesticides than the 0.5 m depth. In summary, only the pesticides with lower K_{oc} values (15 ml/g OC or less) and longer $t_{1/2}$ (>10 days) will leach below the alfalfa root zone. Ground-water contamination is more likely to occur from such pesticides.

Sensitivity Analyses

Soil physical properties were varied in a sensitivity analysis presented below. Also evaluated is the effect of assuming an average deep percolation value despite the fact that even the best sprinkler system applies significantly different amounts of water to different parts of a field.

Soil

The greater the clay and organic carbon content, the greater a soil's tendency to adsorb pesticides and the smaller the risk of ground-water contamination. Pesticides require more travel time when moving through heavy soils e.g. clay soils, than through lighter soils e.g. sand. The travel time, in turn, determines the time available for pesticide degradation via chemical and biological processes. Figure 5 illustrates how soil texture affects aldicarb RA values. Much more aldicarb reaches 2 m depth in sand than in the heavier soils.

Sprinkler Irrigation

In the preceding section, we have used the (Zreq / Ha) approach to determine a field average infiltrated depth of water. Here we examine how that value compares with a more detailed approach. We compare this field average infiltrated depth with the average of 10 normally distributed infiltrated depths. To do this, the field is divided into 10 incremental subareas of equal size. Using a normal distribution approach for the entire field, the appropriate infiltrated depth was determined for each subarea under a normal curve. Then these 10 infiltrated depths were averaged. Assumed were a uniformity coefficient (UC) of 60 percent, and 80 percent of the field area (F) adequately irrigated. A poor uniformity coefficient of 60 percent was selected because it demonstrates the greatest variations among infiltrated water depths. This combination of UC and F gives a distribution coefficient (Ha) of 0.578. The average infiltrated depth (Vi) for the entire field is computed by dividing Zreq (45 mm) by Ha (0.578). Thus Vi is estimated as 78 mm (Vi = 45 mm / 0.578 = 78 mm). This average infiltrated depth is used to develop a normal curve and then to compute each individual infiltrated depth cumulatively for each of the 10 subareas of the field.

These depths were then input into 10 different CMLS simulations. Aldicarb, one of the most mobile and commonly found pesticides in ground water was used (Table 1). The results of these simulations are shown in the first 10 rows of Table 1. The next row shows the average of the 10 detailed simulations. The final row shows the values computed by a single simulation using a 78 mm average infiltrated depth (the approach used in previous discussion). By comparing the last two rows, we are comparing the average of 10 detailed simulations with the value computed by a single simulation.

The results are very similar down to a depth of 1.5 m but are obviously different below that depth. This occurs because applied depth of water in each subarea is not uniform (16 to 140 mm). This nonuniformity produces some subareas with practically no deep percolation and pesticide movement and others with deep percolation and pesticide movement. Clearly, using the single average approach can give misleading results with increasing depth, if the uniformity coefficient is low. Underestimation can also become more important if preferential flow, not accounted for in the model, is present, and a portion of the soil water is bypassed during flow. The single depth approach is more accurate with higher irrigation uniformity (higher uniformity coefficients).

Figure 6 shows the influence of the different subarea percolation depths (Table 1) upon aldicarb movement. Clearly, pesticide will be much more prone to reach a water table at 2 m depth in some parts of a homogeneous field than in others.

SUMMARY

Procedures were developed for aiding environmentally safe pesticide/irrigation management. These required simulation of effects of sprinkler irrigation design, pesticide characteristics (partition coefficient and half life), and soil type on pesticide leaching. First is design of a sprinkler irrigation system for a particular site and pesticide. This enables discrimination of the uniformity coefficient - percent area adequately watered combos that avoid excessive pesticide movement. Second is selection of appropriate pesticides for a particular site, crop and sprinkler design. This permits determining the threshold partition coefficients or half lives for environmental safety in a particular site.

Analysis also revealed that using field average infiltration predicts inaccurate pesticide RA values at higher soil depths. However, for shallow soil depths or irrigation system of good uniformity, the field average approach is acceptable.

A combination of BMPs (best management practices) such as efficient sprinkler system design and management, and selection of less leachable pesticides will yield results with minimum potential for ground-water contamination and environmental hazards.

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TABLE 1: Pesticide Movement Comparison Under Sprinkler Irrigation.

No	PESTICIDE NAME	AREA %	d (mm)	RELATIVE AMOUNT REMAINING AT 1.0 m	1.5 m	2.0 m	3.0 m
1	ALDICARB	10	16	0.0000	0.0000	0.0000	0.0000
2	ALDICARB	10	37	0.0000	0.0000	0.0000	0.0000
3	ALDICARB	10	51	0.0001	0.0000	0.0000	0.0000
4	ALDICARB	10	63	0.0905	0.0001	0.0000	0.0000
5	ALDICARB	10	73	0.1649	0.0001	0.0001	0.0000
6	ALDICARB	10	83	0.1984	0.0905	0.0001	0.0000
7	ALDICARB	10	93	0.1984	0.1371	0.0686	0.0001
8	ALDICARB	10	104	0.2679	0.1649	0.1114	0.0001
9	ALDICARB	10	118	0.2679	0.1984	0.1371	0.0686
10	ALDICARB	10	140	0.3455	0.1984	0.1649	0.1114
	SUBAREAS AVERAGE		78	0.1534	0.0790	0.0482	0.0180
	ALDICARB	100	78	0.1649	0.0686	0.0001	0.0000

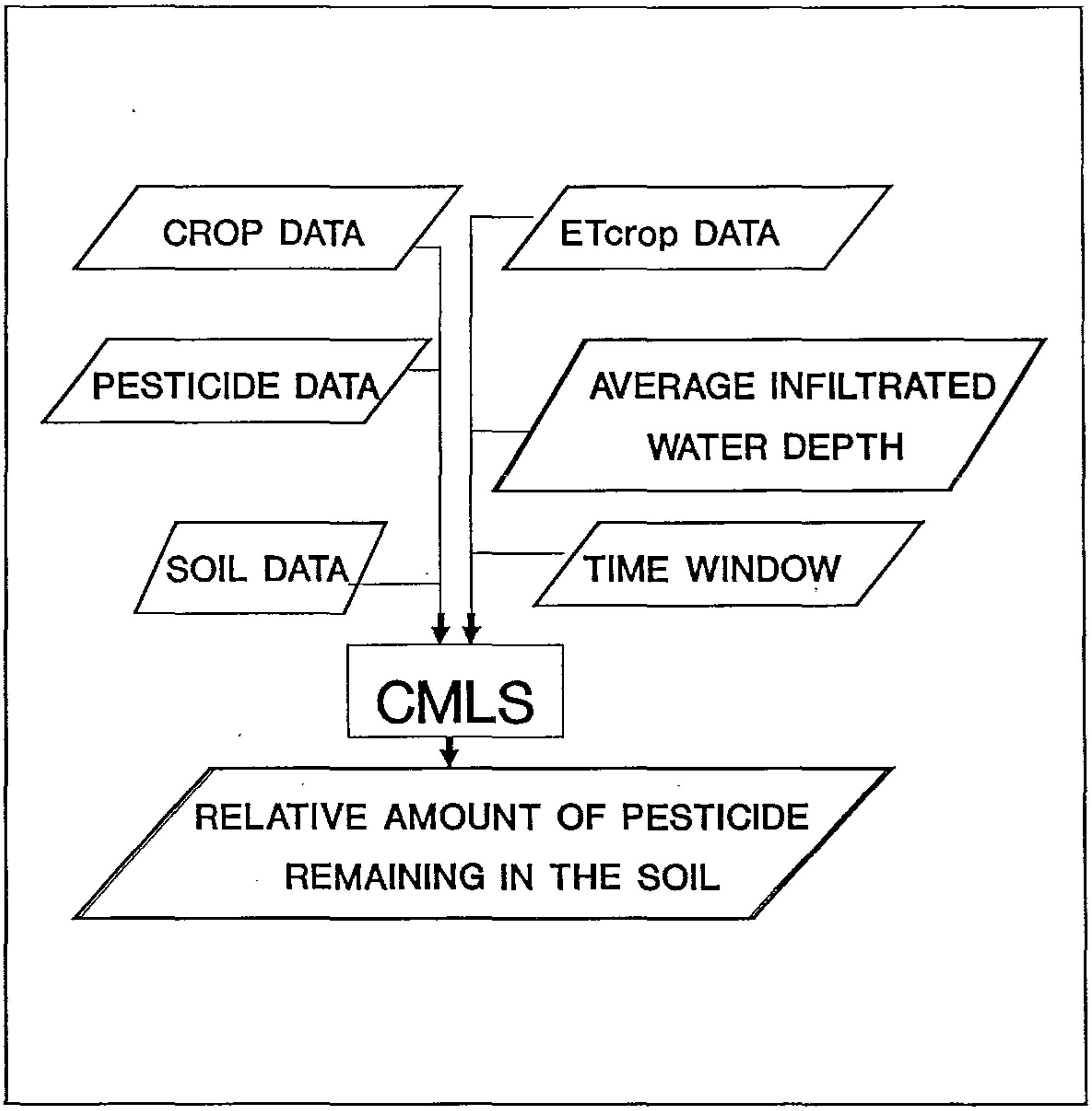

Figure 1: Schematic Representation of the Procedure to Estimate Relative Amount (RA) of Pesticides Remaining in the Soil.

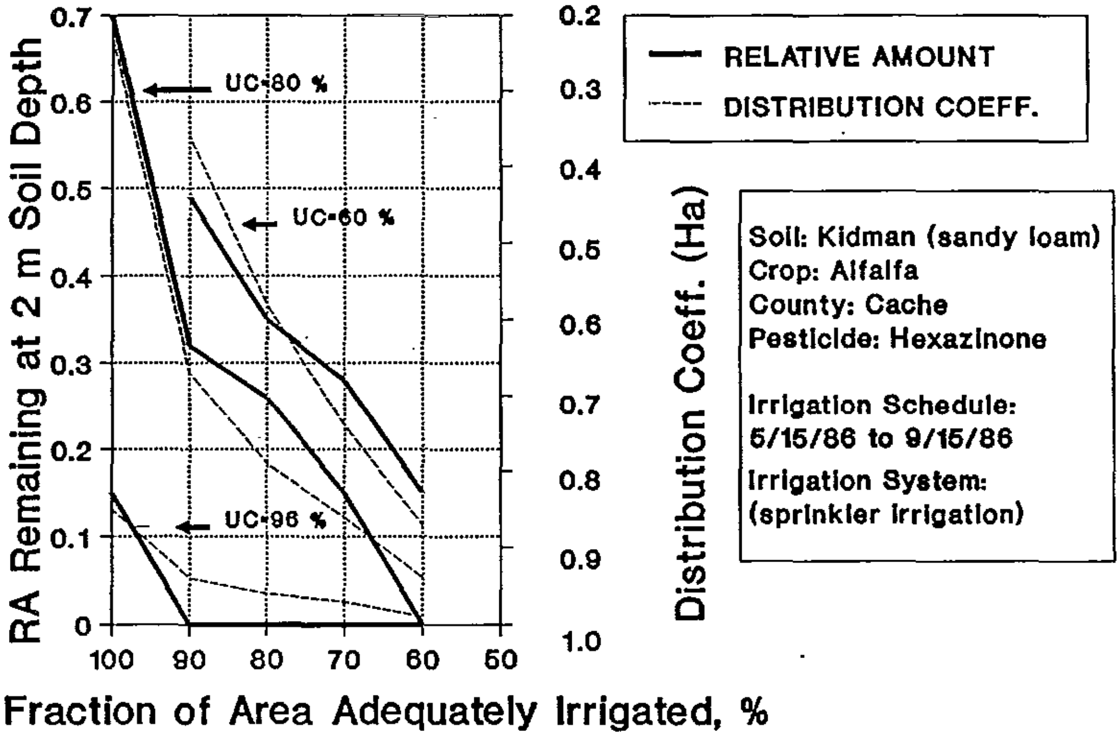

Figure 2: Effects of Uniformity Coefficient (UC) and Percent of Area Adequately Irrigated on Distribution Coefficient (Ha) and on the Relative Amount (RA) of Hexazinone Remaining at 2 m Soil Depth Under Alfalfa Irrigation Schedule.

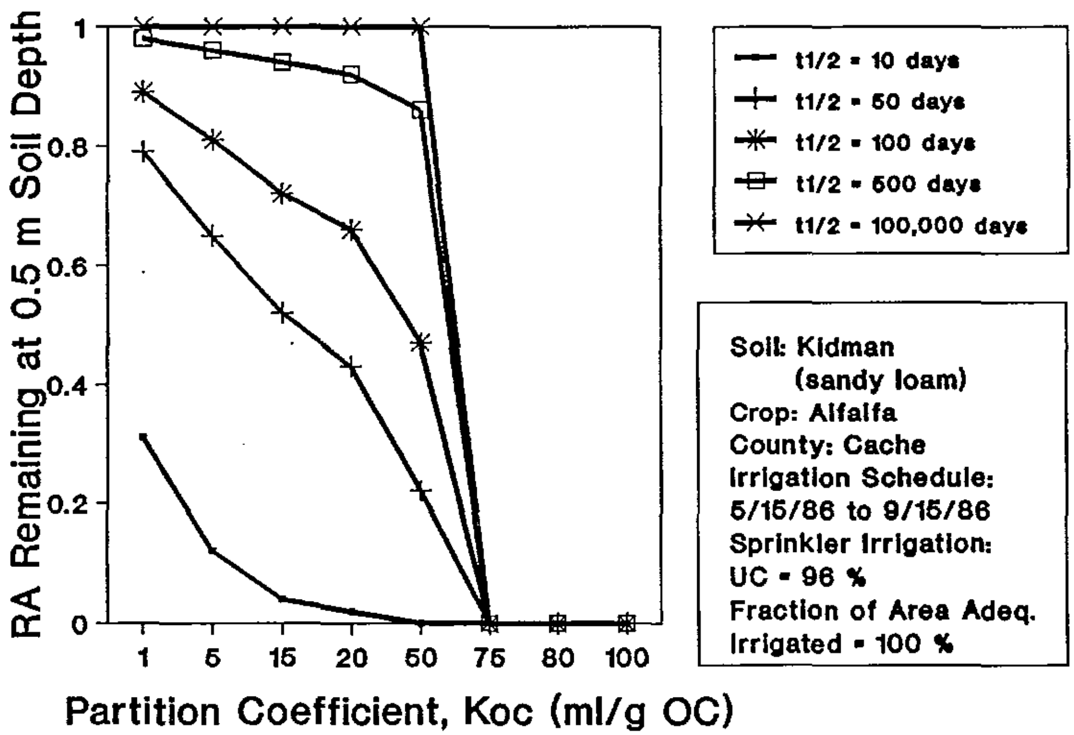

Figure 3: Effects of Pesticide Parameters on the Relative Amount (RA) Remaining when a Pesticide Reaches 0.5 m Soil Depth for Known Site, System and Irrigation Schedule.

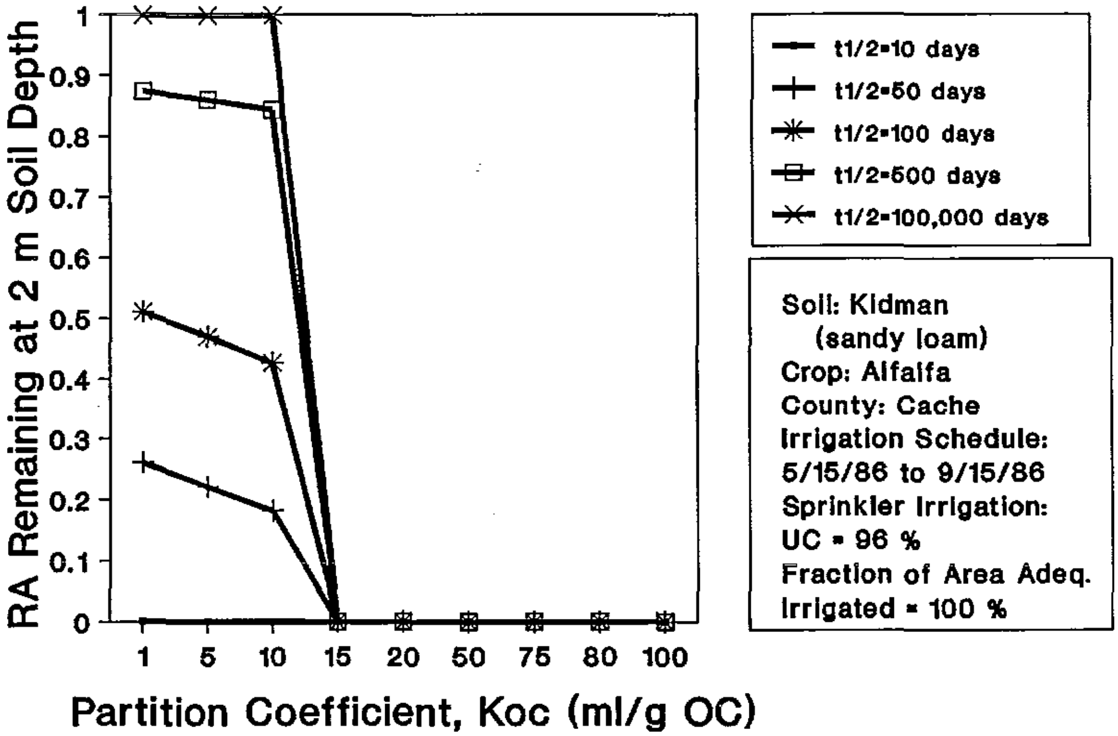

Figure 4: Effects of Pesticide Parameters on the Relative Amount (RA) Remaining when a Pesticide Reaches 2 m Soil Depth for Known Site, System and Irrigation Schedule.

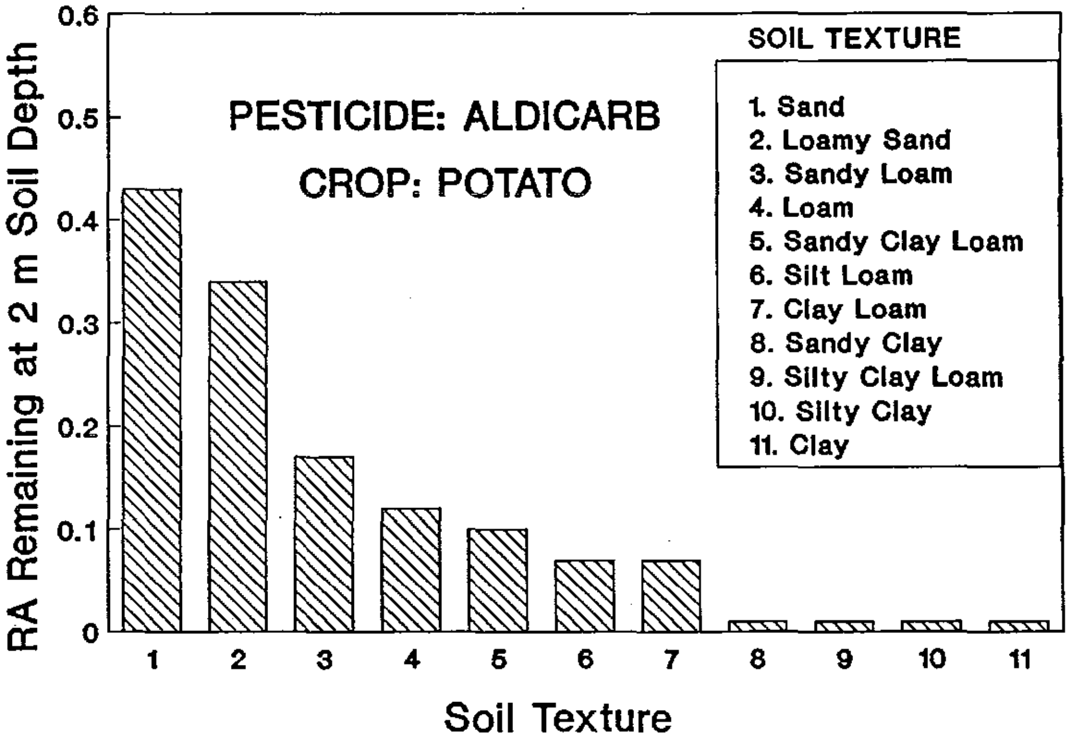

Figure 5: Effects of Various Soil Textures on Relative Amount (RA) of Aldicarb Remaining in the Soil when It Reaches to a Depth of 2 m.

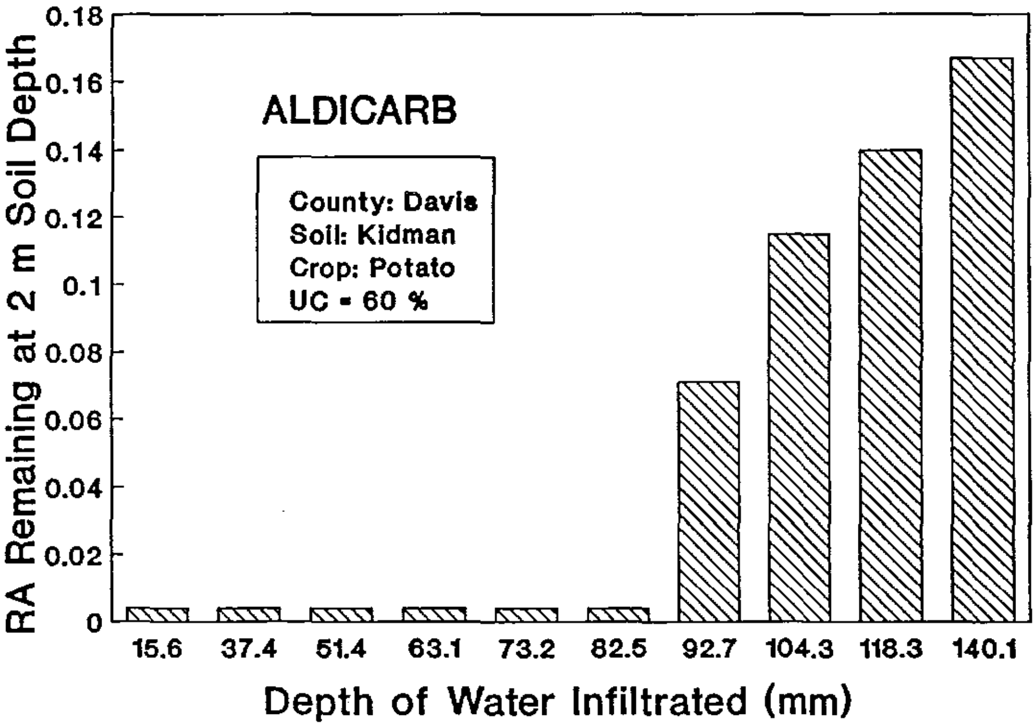

Figure 6. Effect of Various Infiltrated Water Depths on Relative Amount (RA) of Aldicarb Remaining in the Soil when It Reaches to a Depth of 2 m.