

GLEAMS: Groundwater Loading Effects of Agricultural Management Systems

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ABSTRACT

GLEAMS (Groundwater Loading Effects of Agricultural Management Systems) is a mathematical model developed for field-size areas to evaluate the effects of agricultural management systems on the movement of agricultural chemicals within and through the plant root zone. This paper describes the concepts of the hydrology, erosion, and pesticide components of GLEAMS. Results of sensitivity analysis and validation with observed bromide and pesticide data are given. The validation includes comparisons of model simulations and observed soil concentrations in the root zone, and simulated and observed leaching losses.

INTRODUCTION

Scientists of the U. S. Department of Agriculture, Agricultural Research Service, developed the CREAMS model (Knisel, 1980) to evaluate nonpoint source pollution from field-size areas. The CREAMS model was state-of-the-art and represented physical processes. The model had the capability to reflect differences in water, sediment, and chemical responses from different management practices. Data available from diverse physiographic and climatic regions were used to validate CREAMS (Foster and Ferriera, 1981; Knisel, 1980; Knisel et al., 1983; Lorber and Mulkey, 1982).

Following the original publication and distribution of CREAMS, a number of modifications have been made in the model to increase its applicability and to improve its useability. Some of the improvements were completely within the computer program and did not require changes in model input (i.e. representation of seasonally frozen soil, Knisel et al., 1985). Some modifications required changes in input, either as new variables or in streamlining (shortening) the input as well as increased comprehension of output (U. S. Department of Agriculture, Soil Conservation Service, 1984). Model improvements that required changes in input were kept minimal, but application of irrigation (Del Vecchio et al., 1983) did require additional input to consider water deficit management (U. S. Department of Agriculture, Soil Conservation Service, 1984).

Applications of CREAMS have been highly varied from simple to complex systems including extremes in climate and diverse management (Crowder and Young, 1985; Del Vecchio and Knisel, 1982; Dickey, 1985; Dumper, 1985; Nicks et al., 1984; Svetlosanov and Knisel, 1982). The applications were generally within the realm of expectations per model development, that is, evaluating effects of management practices on nonpoint pollutant loads as opposed to attempts to use the model as an absolute predictor. Although CREAMS was a 1980 state-of-the-art model, it has withstood the tests and continues to be a useful tool for evaluation of management practices. The many applications have showed model strengths and weaknesses, and areas of applicability and inapplicability from the different climatic, soil, cover, topographic, and management interactions.

Although CREAMS was developed to evaluate agricultural management systems, it is useful in evaluating forest management practices (Nutter et al., 1984) and associated risk analysis in pesticide usage (Nutter et al., 1986). Other forest applications have been made as well (Dumper et al., 1986; Svetlosanov and Knisel, 1982). Also, CREAMS has been used in design of landfill cover for low-level radioactive waste disposal sites (Hakonson et al., 1982; Lane, 1984).

Problems of pesticides in groundwater (Marti et al., 1984; Cohen et al., 1986) and increased emphasis on groundwater quality led to modifications of CREAMS to consider impact of agricultural management systems on groundwater loadings (Leonard et al., 1985). Since the CREAMS model had been validated and was accepted by many users, it was only logical to extend the model capability to estimate groundwater loadings rather than to redevelop an entirely new computer model. Thus, the approach followed in the development of GLEAMS was primarily to modify the hydrology, plant nutrient, and pesticide components of CREAMS to consider movement of water and chemicals within and through the root zone, and to improve the model representation of management practices. In view of the increased needs for risk analysis in groundwater quality, long-term simulation was considered desirable (Knisel and Leonard, 1986).

The GLEAMS model was developed to utilize the management-oriented physically-based CREAMS model (Knisel, 1980) and incorporate a component for vertical flux of pesticides. The purpose of this paper is to present the concepts of GLEAMS and the results of testing and sensitivity analyses. In-depth treatment of all processes is not given, but references are made to the CREAMS documentation (Knisel, 1980), where appropriate.

BACKGROUND

Groundwater quality concerns of the 1980's have resulted in the need for tools, such as computer models,

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to evaluate effects of interacting processes on chemical movement within and through soils. Models have been shown to be effective in evaluating nonpoint source pollution from agricultural areas (Crowder and Young, 1985; Dumper et al., 1986; Nicks et al., 1984). If these tools were beneficial for surface considerations, it is logical to extend the capabilities for the more complex rootzone/groundwater systems.

Complexities and limitations of some models do not lend to effective extensions or modifications. For example, the Agricultural Runoff Model, ARM, (Donigian and Crawford, 1976) required so much computer time for a one-year simulation, it was not feasible to consider routing pesticides through the rootzone. Simplifications of the surface processes would have been necessary before adding subsurface components.

Nofziger and Hornsby (1984) developed the CMIS model for use as a demonstration tool in extension and teaching in Florida. CMIS assumes surface runoff does not occur, and thus an erosion component is not needed. This is a valid assumption for the sandy soils of peninsular Florida in most situations where the water table is not at the soil surface. The model was developed for use during a growing season, but was not intended to evaluate effects of agricultural management practices on groundwater loadings.

The LEACHMP model developed by Wagenet and Hutson (1986) is a highly theoretical finite difference model that considers water balance and pesticide transport, but does not contain an erosion component. LEACHMP was not intended to consider the continued effects of management practices, but was generally intended for use during a growing season. It includes the capability to consider pesticide transformation products as well as the parent pesticide compound. This approach does not lump the metabolites together with the parent compound which is logical since the transformation products generally have considerably different characteristics than those of the parent compound. However, the model is not effective in evaluating impacts of management practices on groundwater loadings.

Carsel et al. (1985) developed the PRZM (Pesticide Root Zone Model) model for use in pesticide registration. PRZM uses an adaptation of the curve number method (U. S. Department of Agriculture, Soil Conservation Service, 1972) for estimating surface runoff from daily rainfall similar to that in CREAMS (Williams and Nicks, 1982). PRZM uses a modification of the Universal Soil Loss Equation developed by Williams and Berndt (1977) for the erosion/sediment yield component. The surface layer of soil in PRZM is relatively thick compared with other models and thus PRZM is less responsive to rainfall for surface runoff and erosion. Although this is sufficient for considering "worst case" potential leaching of pesticides for registration purposes, it may not consider the entire soil surface/root zone system for assessment of practices.

CREAMS (Knisel, 1980) is a computer-efficient model to simulate edge-of-field loadings of sediment and chemicals. It consists of three separate programs with intermediate output files from hydrology and erosion for use in the erosion and chemistry components, respectively. The intermediate files were generated for days with rainfall or irrigation with averages and accumulated values for the intervening time period. For

example, percolation below the rootzone was summed for all days between rainfall or irrigation. The cumulative values along with the number of days with percolation were included in the hydrology file to be read into the erosion component and rewritten by the erosion component for use in the chemistry component. Soil-water content for the total rootzone was averaged between dates of rainfall and included in the intermediate files, also. Actual and potential evaporation and plant transpiration were accumulated daily for the time interval between rainfall events for use in the chemistry component. Averages and cumulative data were not sufficient to estimate vertical flux and plant uptake of pesticides. Daily values by computational soil layer are needed for chemistry processes. The additional information on a daily basis cannot be manipulated efficiently with intermediate files due to increased read/write (I/O) time. Some initial data such as soil and water-retention characteristics are needed in each of the components of CREAMS, but since they are a one-time input, they are not included in the intermediate files which require duplication of input.

Soil evaporation and plant transpiration are calculated for each computational layer in CREAMS hydrology option I, but are summed for the rootzone between rainfall or irrigation events. Although water balance computations are made daily, information is disregarded in the intermediate files that are needed for solute transport computations in a rootzone model.

A surface layer of soil is set in the CREAMS model hydrology component to partition rainfall runoff and infiltration. The thickness of surface layer varies with rooting depth, crop, or other factors, but chemical extraction is not considered. A surface, or surface-active layer is then set at 1 cm thickness in the chemistry component. The difference between the two components may be a factor of 2 to 4 depending upon rooting depth. Also, the hydrology component considers porosity and water retention characteristics uniform throughout the rootzone. This is generally adequate for direct runoff, but may result in significant discrepancy for water movement through the rootzone. The pesticide component of CREAMS considers up to 10 pesticides simultaneously in a single computer run, whereas most models are limited to a single compound.

There are many features that make CREAMS a good management tool. Model validation and successful application in several land resource areas make it desirable to modify CREAMS to consider vertical flux of pesticides.

GLEAMS MODEL DESCRIPTION

The basic CREAMS model (Knisel, 1980) was modified to develop the present version of GLEAMS. Details of CREAMS are not given here, but only the changes are described in this report.

GLEAMS consists of a single computer program of interactive processes. Separate hydrology, erosion, and pesticide parameter files and output files are maintained. Separate parameter files enable the user to concentrate on one component at a time, but duplication of some parameters has been eliminated between the files. Separate output files provide considerable options on degree of output. Description of the model will be given by major component.

Hydrology Component

Since a layered soil system is needed for routing pesticides through the rootzone, hydrology option 1 of CREAMS with seven computational layers (Williams and Nicks, 1982) is used in GLEAMS. Soil profile description and crop data are used to estimate effective rooting depth. Soils data are input by soil horizon, and the model distributes values for porosity, water retention characteristics, and organic matter into the appropriate computational layers. The surface layer is set at 1 cm thick and the thickness of layer 2 is adjusted such that the total thickness of layers 1 and 2 is 1/6 of the effective rooting depth. This provides for consistent thicknesses between the hydrology and pesticide components. Although a surface layer of fixed thickness is assumed, it is recognized that infiltration control and interaction of runoff and chemical extraction occurs over some depth that varies with tillage, time since last tillage, rainfall, soil texture, soil water content, and soil cover, among other things. Relationships have not been established so therefore the surface layer is considered constant at 1 cm.

Water balance computations in GLEAMS are the same as those in CREAMS. The storage-routing technique used to simulate percolation from layer-to-layer in CREAMS is used here, but the volume of percolation by layer is retained for routing pesticides which will be described in that component.

Soil evaporation and plant transpiration are calculated layer-by-layer and their respective identities retained also for pesticide movement. The identification provides the concentration from the corresponding layer for upward movement and for pesticide uptake.

Erosion Component

The erosion component of GLEAMS is essentially the same as that in the CREAMS model (Foster et al., 1980). The only significant change is the calculation of sediment particle characteristics. Since development of the CREAMS erosion component, Foster et al. (1985) used additional data to better define aggregate sizes and their respective fractions in the detached soil. Specific gravity of the small and large aggregates, 1.8 and 1.6 g/cm³, respectively, were found to be satisfactory in the testing of CREAMS and were not changed in the revision (Foster et al., 1985).

The equivalent sizes of small and large aggregates were related to the fraction of clay in the matrix soil, MCL, (Foster et al., 1985) and the relations are repeated here since they are a part of GLEAMS. The equivalent diameter of small aggregates, DSG, and large aggregates, DLG, are:

$$\left. \begin{aligned} DSG &= 0.030 && \text{for } MCL < 0.25 \\ DSG &= 0.2 (MCL - 0.25) + 0.030 && \text{for } 0.25 \leq MCL \leq 0.60 \\ DSG &= 0.100 && \text{for } MCL > 0.60 \end{aligned} \right\} \dots\dots\dots [1]$$

$$\left. \begin{aligned} DLG &= 0.300 && \text{for } MCL \leq 0.15 \\ DLG &= 2 MCL && \text{for } MCL > 0.15 \end{aligned} \right\} \dots\dots\dots [2]$$

The fraction of primary clay, silt, and sand particles,

FCL, FSI, and FSA, respectively, and the fraction of small aggregates, FSG, and fraction of large aggregates, FLG, were related to the primary particles in the matrix soil (MCL, MSI, and MSA). The fraction of clay is given as

$$FCL = 0.26 MCL \dots\dots\dots [3]$$

$$\left. \begin{aligned} FSG &= 1.8 MCL && \text{for } MCL < 0.25 \\ FSG &= 0.45 - 0.6 (MCL - 0.25) && \text{for } 0.25 \leq MCL \leq 0.50 \\ FSG &= 0.6 MCL && \text{for } MCL > 0.50 \end{aligned} \right\} \dots\dots\dots [4]$$

$$FSI = MCI - FSG \dots\dots\dots [5]$$

$$FSA = MSA (1.0 - MCL)^5 \dots\dots\dots [6]$$

$$FLG = 1.0 - FCL - FSI - FSG - FSA \dots\dots\dots [7]$$

The primary particle composition of the five sediment classes were given by Foster et al. (1985) as:

Primary clay—
 $FCLCL = 1.0 \dots\dots\dots [8]$

Primary silt—
 $FSISI = 1.0 \dots\dots\dots [9]$

Small aggregate—
 $FCLSG = MCL / (MCL + MSI) \dots\dots\dots [10]$
 $FSISG = MSI / (MCL + MSI) \dots\dots\dots [11]$

Large aggregate—
 $FCLLG = (MCL - FCL - (FSG \times FCLSG)) / FLG \dots\dots\dots [12]$
 $FSILG = (MSI - FSI - (FSG \times FSISG)) / FLG \dots\dots\dots [13]$
 $FSALG = (MSA - FSA) / FLG \dots\dots\dots [14]$

Primary sand—
 $FSASA = 1.0 \dots\dots\dots [15]$

Foster et al. (1985) stated that under some conditions the fraction of clay in the large aggregates (FCLLG) in equation [12] is too small, and it was assumed that

$$FCLLG \geq 0.5 MCL \dots\dots\dots [16]$$

If the condition of equation [16] did not occur with equation [12], then

$$FCLLG = 0.5 MCL \dots\dots\dots [17]$$

and FSG is recomputed as

$$FSG = (0.3 + 0.5 S)(MCL + MSI) / (1.0 - 0.5 (MCL + MSI)) \dots\dots\dots [18]$$

S in equation [18] is defined as

$$S = FCL + FSI + FSA \dots\dots\dots [19]$$

Just as in CREAMS (Foster et al., 1980), Foster et al. (1985) assumed organic matter to be distributed among all sediment particle classes as clay is distributed. The same specific surface areas for sand, silt, clay, and organic matter are used as those in CREAMS, and the storm enrichment ratio is calculated accordingly.

Pesticide Component

The basic concepts of the pesticide component of GLEAMS are the same as those in CREAMS (Leonard and Wauchope, 1980) with some additions that enable the estimation of movement within and through the root zone. Since the principal objective of the GLEAMS model is to simulate the effect of management practices on pesticide movement within and through the root zone, it is essential to maintain sensitivity in the surface soil layer. Therefore, the pesticide concepts from CREAMS are retained in the GLEAMS surface component, and the user can examine both surface and subsurface response for all pesticides without requiring a model for surface and another model for subsurface responses.

A simple representation of the complete system is illustrated in Fig. 1. Pesticides may be aerially applied with some fraction intercepted by foliage and some fraction reaching the soil. Aerially-applied pesticides include herbicides, insecticides, acaricides, fungicides, and defoliant which have diverse chemical characteristics affecting adsorption/desorption onto soil and organic carbon, adsorption and washoff from living and dead plant tissue, and degradative characteristics. Degradation differs from foliage, surface soil, and rootzone. It is a function of soil water, temperature, and pH, among other factors. Volatilization of pesticides occurs from plant and soil surfaces, and the flux from the soil surface is a function of temperature, soil-water gradients, and chemical characteristics. Methods of application may vary from ground equipment to aircraft, and the material may be applied as solids, dispersions, emulsions or solutions.

Some pesticides, such as pre- and post-plant herbicides may be applied directly to the soil surface as a

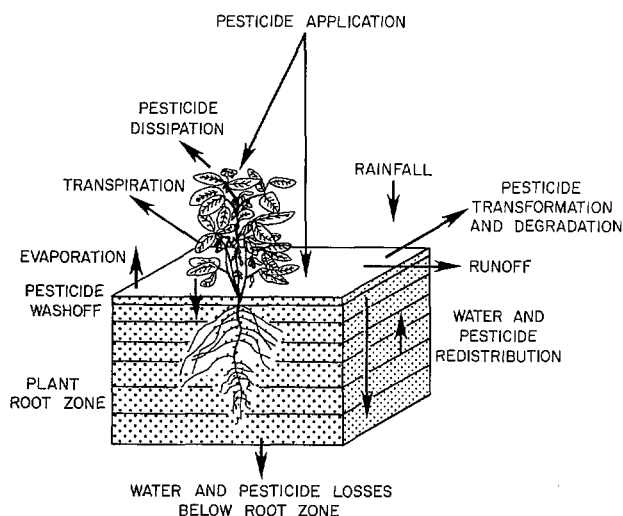


Fig. 1—The physical system and processes represented in GLEAMS.

liquid, whereas other herbicides, soil insecticides, and nematocides may be surface applied and incorporated into the soil by tillage. Mixing efficiency of tillage equipment is a function of type and depth of operation, and this affects the amount of pesticide in the surface soil available to all of the interacting processes.

Injection or banding of some pesticides is essential to reach such target pests as nematodes or other soil organisms that attack plant roots. Some may be systemic insecticides that must be taken up and translocated by the plants. Injected and banded pesticides may be applied as gases, liquids, granules, or powders. Applications made by banding are generally expressed as “broadcast equivalent” which is sufficient to denote the mass dispensed per unit area of field. However, representation of application in a computer model is practically impossible if a predictive model (absolute quantities) is expected. Macropore and crack flow are extremely significant for absolute predictions with banded applications, and for such management practices as in-row chisel planters in conservation tillage. Accelerated flow may be highly significant in this system, whereas retardance of flow and solute movement may be equally significant where a spatially continuous plow pan would be restrictive for a uniform aerial application as well as banding.

Exact representation of pesticide application and agricultural management practices is nearly impossible. However, depth of pesticide placement is necessary to properly account for location and magnitude of the processes. The CREAMS model only considered surface processes, and any injection below the surface 1 cm of soil was not considered. It is necessary in the GLEAMS model to specify the depth of incorporation or injection to obtain the correct initial distribution among the soil layers.

Pesticides washed from plants results in changed concentrations in the surface soil. Concentration in the surface determines the amount that is available for extraction into surface runoff and/or movement into the soil profile. The thickness of the surface control layer varies with soil density, surface roughness, soil water content, and pesticide characteristics. For example, a crusted surface layer in a no-till system, or a compacted surface due to continuous animal grazing of a pasture or rangeland, may have an effective surface layer only a few millimeters thick. On the other hand, a freshly tilled, cloddy surface of a heavy-textured soil, or a well aggregated soil surface, may have a relatively thick effective surface layer, possibly 2 to 3 cm. The effective layer for a cloddy surface may be 2 to 3 cm at the beginning of a rainfall event, but with “melting of the clods” and “running together” of the single grain particles may result in a much reduced surface active layer during the rainfall and subsequent runoff event. Soluble pesticides have a deeper zone of interaction than readily adsorbed ones. Also, when the soil surface is relatively dry, for example, a montmorillonitic clay that has considerable shrinkage cracks, the surface active layer is thicker than for the same soil in a wet, swelled condition.

In view of these complexities, good relationships have not been developed to calculate surface layer thickness. Therefore, the surface layer in GLEAMS is a constant 1 cm thickness.

Pesticide Degradation—The pesticide degradation component of GLEAMS is essentially the same as that in CREAMS (Leonard and Wauchope, 1980) with only a slight change. The relationships are repeated here in order to indicate the differences.

The same relationship is used for pesticide degradation on plant foliage, (Leonard and Wauchope, 1980)

$$PMF_t = PMF_o \text{ EXP } (-0.693t/F_{1/2}) \dots\dots\dots [20]$$

where PMF_o is the pesticide mass on the foliage on the day of application, PMF_t is the pesticide mass remaining on the foliage at time t , in days, and $F_{1/2}$ is the foliar half-life in days. Similarly, degradation in the soil is

$$C_{s(t)} = C_{s(o)} \text{ EXP } (-0.693t/S_{1/2}) \dots\dots\dots [21]$$

where $C_{s(o)}$ is the pesticide concentration in the soil on the day of application, $C_{s(t)}$ is the pesticide concentration in the soil at time t , and $S_{1/2}$ is the pesticide half-life in the soil. Equation [21] is used for each computational soil layer. $S_{1/2}$ is considered constant for the rootzone, but it is known to vary with soil water content, temperature, pH, and organic carbon, among other factors (Nash, 1980; Nash, 1984; and Rao et al., 1984), but not well enough for all pesticides to include general relationships in the model. $F_{1/2}$ and $S_{1/2}$ are for the "lumped" processes accounting for dissipation of pesticide.

GLEAMS operates with a daily time step since pesticide redistribution caused by water evaporation and transpiration are considered. Pesticide degradation must be calculated daily in order to adequately assess concentrations for evaporation and transpiration fluxes from the appropriate soil layers.

Pesticide Extraction Into Runoff—The pesticide runoff component of the CREAMS model (Leonard and Wauchope, 1980) was not changed in GLEAMS. The concepts and defining equations are repeated here for convenience and continuity for the complete pesticide component.

Leonard and Wauchope (1980) described pesticide distribution between the solution phase and the soil phase as a simple linear adsorption isotherm

$$K_d = \frac{C_s}{C_w} \dots\dots\dots [22]$$

where, at equilibrium, C_s is the concentration in the soil or solid phase, mg/kg, and C_w is the concentration in the solution phase in mg/L. The partitioning coefficient, K_d , is dependent upon pesticide characteristics, including water solubility, and soil organic carbon among other factors (Pionke and DeAngelis, 1980; Green and Karickhoff, 1986). Since organic carbon is soil or site specific, and it is desirable to use only pesticide-specific data as input, the K_{oc} is used as input for GLEAMS. The K_d is calculated in GLEAMS from the relation

$$K_d = K_{oc} \text{ OC}/100 \dots\dots\dots [23]$$

where K_{oc} is the linear adsorption coefficient for organic carbon, and OC is the organic carbon content of the soil expressed as a percent of the soil mass. Since some measured data are reported as organic matter rather

than organic carbon, organic matter is an input for GLEAMS and equation [23] is rewritten as

$$K_d = 0.0058 K_{oc} \text{ OM} \dots\dots\dots [24]$$

where OM is the organic content of the soil expressed as a percent of the total soil mass.

Data on K_{oc} are not available for all pesticides, and significant work has been done to provide methods of estimation. For example, Kenaga and Goring (1980) developed exponential relationship between K_{oc} and K_{ow} (octanol: water distribution coefficient) and between K_{oc} and water solubility. Rao and Davidson (1980) also used the concept of exponentiality between K_{oc} and K_{ow} for a number of pesticides not included by Kenaga and Goring (1980). The relation of Rao and Davidson (1980) is

$$\text{Log}_{10} K_{oc} = 1.02 \text{ Log}_{10} K_{ow} - 0.18 \dots\dots\dots [25]$$

Kenaga and Goring's (1980) relation with water solubility, WS, (mg/L) is

$$\text{Log}_{10} K_{oc} = 3.64 - 0.55 \text{ Log}_{10} \text{ WS} \dots\dots\dots [26]$$

Karickhoff (1984) suggested this method be used over a limited range of water solubility, e.g. 100 to 300 mg/L, due to the relative reliability of this method.

Pionke and DeAngelis (1980) related K_d to soil organic matter and also to soil or sediment specific surface area. This points out the difficulty in developing parameter sets for model testing or application. Green and Karickhoff (1986) stated: "A well-organized and well-documented data set which could be easily accessed by potential users would have tremendous value in the future." This is a clear indication of limited information available for use.

As defined in CREAMS (Leonard and Wauchope, 1980), the rate of change in pesticide mass, Z , in the soil surface is

$$-dZ = C_w f dt \dots\dots\dots [27]$$

where f is defined as the water flux and the other terms were previously defined. At saturation,

$$Z = (C_w \text{ POR}) + C_s (1 - \text{POR}) \text{ SSG} \dots\dots\dots [28]$$

where POR is porosity of the soil, g/cm³, and SSG is the soil specific gravity, g/cm³. Rearranging equation [22] and substituting into equation [28] gives

$$C_w = \frac{Z}{\text{POR} + K_d (1 - \text{POR}) \text{ SSG}} \dots\dots\dots [29]$$

The rate equation (equation [27]) can be rewritten as

$$-dZ = \frac{Z f dt}{\text{POR} + K_d (1 - \text{POR}) \text{ SSG}} \dots\dots\dots [30]$$

and integration from Z_o to Z , and from $t_o = 0$ to t give

$$Z = Z_o \text{ EXP } \left[\frac{-f t}{(\text{SSG}) K_d (1 - \text{POR}) + \text{POR}} \right] \dots [31]$$

where Z_0 is the mass of pesticide per unit volume of the surface soil layer at the beginning of the storm. The water flux, f , through the soil surface into the top layer during a storm is

$$f = \frac{P - Q - AWS}{t} \dots \dots \dots [32]$$

where P is rainfall depth, cm, Q is surface runoff depth, cm, AWS is soil water storage capacity to saturation, cm, and t is time (storm duration), h. AWS in equation [32] is the initial abstraction from rainfall necessary to fill the layer to saturation. Substituting equation [30] into equation [29] and eliminating t gives

$$Z = Z_0 \text{ EXP } \frac{-(P - Q - AWS)}{(SSG) K_d (1 - \text{POR}) + \text{POR}} \dots \dots [33]$$

In the CREAMS model, the limited data in the file from the erosion component required some assumption concerning the soil water content in the top layer of soil on the day of rainfall. The assumption was made that the surface 1 cm of soil was always dry (at 1500 kPa water content) on a day of rainfall. However, in GLEAMS, where the model operates as a single program, the actual soil water content estimated from the water accounting procedure is used on the day of rainfall.

The pesticide concentration in the soil at time t after application, PCS_t , from equation [21], is multiplied by the soil bulk density to give initial pesticide mass, Z_0 , on the day of rainfall

$$Z_0 = PCS_t (1 - \text{POR}) \text{ SSG} \dots \dots \dots [34]$$

and Z in equation [33] is

$$Z = C_{av}(1 - \text{POR}) \text{ SSG} \dots \dots \dots [35]$$

where C_{av} is the runoff-available pesticide concentration in the surface soil layer, mg/kg.

At the time of runoff, the surface layer of soil (layer 1), contains some pesticide residue determined after degradation, plus any addition due to foliar washoff, minus that removed by vertical translocation. The pesticide concentration units are expressed in mg/kg of dry soil. A pesticide is extracted by water flowing over the soil surface and by dispersion and mixing of the soil material by the flowing water and raindrop impact. At the interface between the soil matrix and the overland flow, some mass of soil is effective in supplying pesticide to the flow. The mass of pesticide, Y , in this mass of soil is the product of available concentration, C_{av} in equation [35], and the soil mass per unit volume of overland flow, B ,

$$Y = C_{av} B \dots \dots \dots [36]$$

As the pesticide equilibrates (instantly) between the soil mass and the overland flow

$$Y = C_w V + C_s B \dots \dots \dots [37]$$

where V is the volume of water per unit volume of runoff interface and the other terms were previously defined. Disregarding the volume occupied by the soil

mass compared to the larger volume of water, the total unit volume of runoff interface is 1, and

$$(1) C_w + C_s B = C_{av} B \dots \dots \dots [38]$$

Using the equilibrium distribution between the solution and the soil as expressed by equation [22], rearranging, and substituting into equation [38] for C_s , gives

$$C_w = \frac{C_{av} B}{1 + B K_d} \dots \dots \dots [39]$$

Substituting into equation [38] for C_w gives

$$C_s = \frac{C_{av} K_d B}{1 + K_d B} \dots \dots \dots [40]$$

It can be seen from equation [39] that when $K_d = 0$, then $C_w = C_{av} B$, while $K_d = 0$ in equation [40] gives $C_s = 0$. Also, if K_d is very large, C_s in equation [40] approaches C_{av} .

In the CREAMS model, the extraction coefficient, parameter B of equations [36] to [40], was entered into the parameter file by the user in the early versions of the model. However, most applications were made for somewhat adsorbed pesticides, and in the latest version of CREAMS, parameter B was deleted from the file and the model was modified to set the value at 0.1 for all pesticides. The extraction coefficient is known to vary with runoff conditions, pesticide solubility, and adsorptivity, among other factors. A functional relationship was developed for GLEAMS to relate the extraction coefficient to partitioning coefficient in the model as follows:

$$\left. \begin{array}{ll} B = 0.5 & \text{for } K_d \leq 1.0 \\ B = 0.7 - 0.2 K_d & \text{for } 1.0 < K_d \leq 3.0 \\ B = 0.1 & \text{for } K_d > 3.0 \end{array} \right\} \dots \dots [41]$$

Vertical Flux of Pesticide—The foregoing discussion of pesticide extraction into runoff provides the basis for pesticide movement through the soil surface layer and into and through the root zone.

Pesticide concentration in solution, C_w in equation [39], is the concentration in the water that is available for percolation into soil layer 2. The pesticide mass, PERCM, transported from layer 1 becomes

$$\text{PERCM}_1 = \text{PERC}_1 (C_w(1)) \dots \dots \dots [42]$$

where PERC_1 is the water mass percolated from soil layer 1. Using a simple pesticide accounting procedure, the pesticide mass percolated from layer 1 (PERCM_1) is subtracted from the pesticide mass in layer 1 and added to the pesticide mass in layer 2 after degradation,

$$\text{PMS}_1 = \text{PMS}_1 - \text{PERCM}_1 \dots \dots \dots [43]$$

and

$$\text{PMS}_2 = \text{PMS}_2 + \text{PERCM}_1 \dots \dots \dots [44]$$

where PMS is pesticide mass and the subscripts are

indices for the respective layers. The water mass percolated from layer 1, $PERC_1$, is

$$PERC_1 = f - FC_1 \dots\dots\dots [45]$$

where FC_1 is the field capacity of layer 1 and f is the infiltration into layer 1. For layers 2 through 7, the water mass percolated becomes

$$PERC_i = PERC_{(i-1)} + SW_i - FC_i \text{ for } i=2, 7 \dots\dots [46]$$

where SW_i is the initial soil water content of layer i before redistribution of percolate from the layer above.

For layers 2 through 7, there is not an extraction process analogous to that in surface runoff. Therefore, the procedure for those computational layers is different only in the calculation of the pesticide solution concentration, C_w .

The new pesticide mass in each layer, PMS_i , is divided by the soil mass in the layer, SM_i , to obtain the soil concentration, C_{si} ,

$$C_{si} = PMS_i / SM_i \text{ for } i=2, 7 \dots\dots\dots [47]$$

Using the partitioning concept in equation [22] and assuming instant pesticide equilibrium between the water and soil,

$$C_{wi} = PMS_i / (K_{di} SM_i + WM_i) \text{ for } i=2, 7 \dots\dots [48]$$

where WM_i is total water mass in layer i . This solution concentration of pesticide, C_{wi} , is used for calculation of pesticide percolation as well as plant uptake and that moved up in solution by evaporation which are given in the next section.

The pesticide concentration calculated by equation [48] for layer 7, C_{w7} , and the corresponding pesticide mass estimated as the product of concentration and percolated water mass, $PERC_7$,

$$PERC_{M7} = PERC_7 C_{w7} \dots\dots\dots [49]$$

represents the potential loadings to the vadose or groundwater zone.

Pesticide Transport with Sediment—Pesticide transport with sediment in the GLEAMS model is computed the same as in CREAMS (Leonard and Wauchope, 1980). An enrichment ratio is calculated in the erosion component based upon the specific surface area of the sediment leaving the field and the specific surface area of the matrix soil. This ratio is a measure of the enrichment of the clay and organic matter fractions in the sediment that enables the estimation of the transport capacity of the total sediment mass leaving the field. The enrichment ratio calculated in GLEAMS is a function of the sizes and composition of the large and small aggregates as given earlier in the erosion component of GLEAMS.

The pesticide mass transported with sediment is the product of sediment mass, enrichment ratio, and concentration of pesticide in the sediment phase as given by C_s in equation [40]. The pesticide concentration in the surface soil layer is used in the calculation of equation [40].

Pesticide Evaporation and Uptake—Pesticides that move readily with water (low K_d) may have significant quantities move up in the soil profile with evaporation or be taken up by plants. This is especially true for those with an intermediate or a long half-life. Inadequate consideration of these processes could lead to over-estimation of potential pesticide loadings to the vadose or groundwater zones.

In the hydrologic component of the GLEAMS model, soil water depletion, by layer, is partitioned between evaporation and transpiration as a function of leaf area index. This approximate partitioning does not affect the total water depletion by layer which is sensitive to the total water accounting procedure and thus to the runoff and percolation processes.

Since evaporation and transpiration are calculated by computational layer, and pesticide concentrations are estimated by layer, equation [48], plant uptake and upward movement can be estimated by layer. Upward movement by soil evaporation is considered to occur for one layer only, i.e. pesticide moved because of water evaporation is moved into layer $i-1$ only, and not all the way to and out of the surface layer. The supporting assumption is that movement from one layer occurs as an aqueous solution, whereas further movement would have to be as diffusion in a gaseous form. Such consideration would require significantly more information than is readily available or justified, based upon present knowledge of separate pesticide processes and associated field data.

The solution concentration for the appropriate layer, C_{wi} , is multiplied by the evaporation mass, EV_i , to calculate the pesticide mass moved from layer i , $PMEV_i$,

$$PMEV_i = EV_i C_{wi} \dots\dots\dots [50]$$

The pesticide mass is subtracted from the mass in layer i in a simple accounting procedure as

$$PMS_i = PMS_i - PMEV_i \dots\dots\dots [51]$$

and added to the pesticide residue in the $(i-1)$ th layer

$$PMS_{i-1} = PMS_{i-1} + PMEV_i \dots\dots\dots [52]$$

If evaporation occurs from layer 1, the pesticide is not moved out of the soil in the model since volatilization is not explicitly modeled.

Plant uptake of pesticides, e.g. systemic insecticides, is estimated for the appropriate layer as the product of solution concentration, C_w , and transpiration, TR , to give

$$PMTR_i = C_{wi} TR_i \dots\dots\dots [53]$$

where $PMTR_i$ is pesticide mass uptake. The uptake mass is subtracted from the residue in layer i as

$$PMS_i = PMS_i - PMTR_i \dots\dots\dots [54]$$

The foregoing discussion represents the concepts from the CREAMS model that have been changed and the new concepts evolving into the GLEAMS model. The plant nutrient component has not been finalized and incorporated into GLEAMS, and therefore the concepts of that component are not included in this paper.

TABLE 1. LOCATIONS, WATERSHEDS, PERIODS OF RECORD, CHEMICALS, AND REFERENCES FOR DATA USED IN TESTING GLEAMS MODEL

Location	Watershed/ plot	Soil	Drainage area, ha	Period of record	Pesticide or chemical	Reference no.
Watkinsville, GA	P-2	Cecil sandy loam	1.3	1973-75	Atrazine	(44)
Tifton, GA	Gopher Ridge Block B	Bonifay sand	0.36	1984	Atrazine	(28)
Lincoln, IA	Four Mile Creek, ISU-1	Tama silt loam	5.6	1976-78	Cyanazine Alachlor	(17)
	Four Mile Creek, ISU-2	Tama silt loam	7.6	1976-78	Cyanazine Alachlor	(17)
Coshocton, OH	Lysimeters Y101	Berks shaly silt loam*	0.0008	1980-84	Bromide	(39)

*Kelley et al. (1975) describe the soil around the lysimeters as Dekalb channery sandy loam, but the surface 40 cm has less sand and more silt than is allowed in a sandy loam. Soils on the nearby representative watersheds 129 and 135 are silt loams, thus Berks shaly silt loam characteristics were used in this application.

MODEL TESTING

The hydrology and erosion components of GLEAMS are basically the same as those in CREAMS, and since CREAMS has been extensively validated, it was not considered necessary to test those components specifically. Therefore, discussion of further testing of hydrology and erosion is treated very briefly with major emphasis on testing the pesticide component of GLEAMS.

Published data were used in testing so that reference to original works could be made and chemistry analytical procedures would not require detailed description, especially since data are used from several different sites and laboratories.

Consideration was given to climate, soil, and pesticide characteristics in the selection of test locations. The locations selected for testing represent a relatively wide range of climate and soils (Table 1). For example, climatic conditions range from frozen-soil conditions in Ohio (-3°C mean monthly February temperature) to the mild winters at Tifton, GA (11°C mean monthly February temperature), and from the sub-humid area of Lincoln, IA (791 mm/yr rainfall) to the humid region (1,163 mm/yr) at Tifton, GA (U. S. Department of Commerce, 1974a; 1974b).

Watkinsville, GA—Watershed P-2, a study site in the cooperative ARS/EPA pesticide project (Smith et al., 1978), was used in the validation of the CREAMS model (Knisel, 1980). The data represent a relatively complete data base for hydrology, sediment yield, and pesticide chemistry.

Tifton, GA—The very permeable, well drained sandy soils under a sprinkler irrigation system provides an opportunity to study pesticide leaching. The combination of low water retention characteristics and low organic carbon result in potentially high loadings to groundwater. Runoff from the sandy soil is negligible because of the high infiltration rate.

Lincoln, IA—Two single-crop research fields in the

Four Mile Creek Watershed near Lincoln, IA (Johnson, 1978), were selected for partial validation of the GLEAMS model. The two drainages (ISU1 and ISU2) had 2-year rotations of corn-soybeans with different crops on each watershed each year. Soils on the two watersheds have silt loam texture.

Coshocton, OH—The lysimeters at Coshocton provide the only readily available measured leaching data. Although the lysimeters contain a soil profile generally deeper than most CREAMS and GLEAMS model applications for surface runoff, they provide valuable validation data.

Results of Model Testing

Previous authors of root zone pesticide models have used data on very mobile compounds such as aldicarb for model testing/calibration/validation (Carsel et al., 1985; Wagenet and Hutson, 1986). While aldicarb moves readily in soil and provides a range of concentrations with depth and time in the root zone, aldicarb degrades through sulfoxide and sulfone intermediates, and each intermediate has K_{oc} values different from the parent compound and half-lives that vary with soil properties (Ou et al., 1986). Often field pesticide residue data on aldicarb are expressed only as total toxic residue or as the total of aldicarb plus intermediates, or metabolites. When using these data to test models that deal with only one species, the model is applied to the composite, assigning average parameter values for K_{oc} and half-life, or values thought to best represent the dominant aldicarb species (Carsel et al., 1985). The current version of GLEAMS considers a pesticide as a single species, but a new version will soon be available to consider multiple species.

Since GLEAMS contains the same features as CREAMS in simulating surface runoff, it is important to use test data allowing comparisons of observed versus simulated pesticide concentrations in the surface zone as well as in the lower layers of the rootzone. Preference was

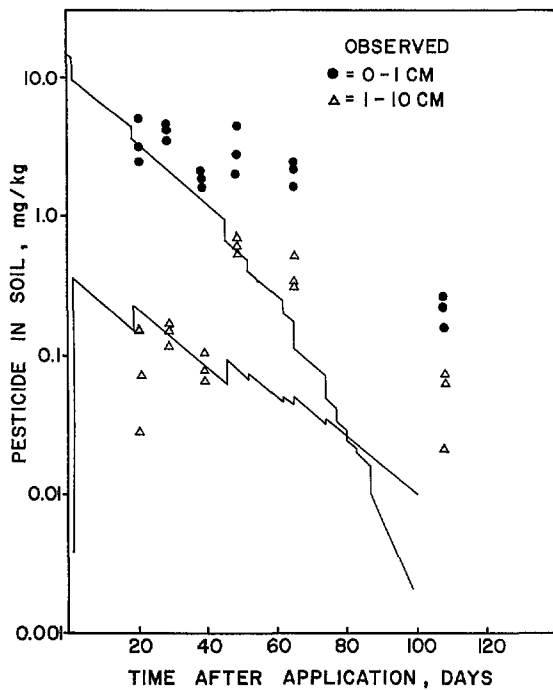


Fig. 2—Simulated and observed cyanazine concentrations in 0-1 cm and 1-10 cm soil layers, watershed ISU-1 near Lincoln, IA, 1977 (adapted from Johnson, 1978).

given to selecting data sets that had been used on several occasions in testing and calibrating other models.

Johnson (1978) conducted extensive research on pesticide runoff from silt loam soils of the Four Mile Creek Watershed in Iowa and provided data sets for calibration of the HSPF (Hydrologic Simulation Program Fortran) model (Donigian et al., 1983). Data from this study are shown in Fig. 2 and Fig. 3 for watersheds ISU-1 and ISU-2, respectively, along with

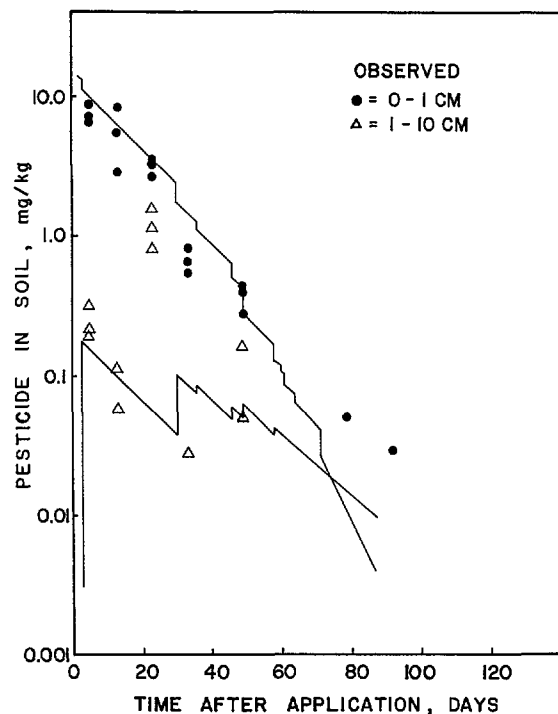


Fig. 3—Simulated and observed alachlor concentrations in 0-1 cm and 1-10 cm soil layers, watershed ISU-2 near Lincoln, IA, 1977 (adapted from Johnson, 1978).

GLEAMS model simulations. Watershed ISU-1 was planted in corn in 1977, and Fig. 2 shows cyanazine in the 0 to 1 cm and 1 to 10 cm layers as a function of time after planting and pesticide application. The watershed was sampled at three locations and all data points are plotted to show the spread in the data. From the model simulations, concentrations in the 0 to 1 cm and the 1 to 10 cm layers are plotted before and after rainfall events to show pesticide redistribution by percolation of water within and through the rootzone. In the model simulations, cyanazine dissipated from the 0 to 1 cm zone at a rate more rapid than observed. A soil half-life of 12 days was used in the model application based upon the work of Baker and Johnson (1979). A K_{oc} of 150 was selected from tabulated values (Green and Karickhoff, 1986). In this simulation, a half-life of 20 days would more nearly represent the data. However, the objective was to use available information and not to calibrate or fine tune to obtain a "best fit", but to show that the model represented the general behavior observed in the field.

After surface application, as shown by observations and simulation, cyanazine was translocated to the 1 to 10 cm zone. There it degraded, presumably with a half-life similar to that in the surface. Since the model assumes first-order pesticide decay, the plots from the model simulations are log-linear except for shifts due to translocation by percolation and some slight redistribution due to soil evaporation. The simulations represented the concentrations observed in the 1 to 10 cm zone well except for the period after day 49. Although information was not available (Johnson, 1978), other information indicates that about this time the corn was likely cultivated which redistributed the pesticide between layers.

Both model simulation and observations showed that no pesticide moved below the 10 cm depth during 1977. A small amount of runoff was observed during the growing season but none was simulated by the model. Rainfall events were numerous in 1977, but most were relatively small. The small events did move pesticide from the 0 to 1 cm zone during the season as reflected in the curvilinear nature of the plot for both simulated and observed.

Soybeans were planted on the adjacent watershed ISU-2 in 1977. Alachlor was applied on the soybeans on the date of planting. Fig. 3 shows results from GLEAMS simulation and observed concentrations in the 0 to 1 cm and 1 to 10 cm layers for watershed ISU-2. Alachlor, with a K_{oc} of 190 compared with 150 for cyanazine, did not leach quite as readily as cyanazine. The half-life of 12 days used in the simulation for alachlor more nearly matched the observations in the 0 to 1 cm surface layer. Simulated versus observed concentrations in the 1 to 10 cm layer were very close except for seemingly anomalous observations on the 23rd day after application. Neither the model nor observations indicated any movement below the 10 cm depth.

A summary of runoff and sediment yield for watersheds ISU-1 and ISU-2 are given in Table 2. Although chemical data were readily available only for 1977, rainfall and cropping data were available from January 1, 1976 through July 7, 1978. The full period of record was simulated for comparison with observed runoff and sediment yield. Although percolation below the root zone was not measured in the field, simulated

TABLE 2. GLEAMS MODEL SIMULATIONS AND OBSERVED RUNOFF AND SEDIMENT YIELD, AND SIMULATED PERCOLATION, WATERSHEDS ISU-1 AND ISU-2 NEAR LINCOLN, IA, 1976-1978

Year	Runoff		Percolation simulated, cm	Sediment yield	
	Observed, cm	Simulated, cm		Observed, t/ha	Simulated, t/ha
Watershed ISU-1					
1976	5.84	4.37	8.74	2.73	2.62
1977	1.17	4.92	0.00	4.67	3.25
1978	4.57	3.93	4.02	(NA)*	0.60
Watershed ISU-2					
1976	5.10	4.24	7.45	5.38	8.93
1977	0.09	4.97	0.00	0.17	8.25
1978	4.71	3.99	4.69	(NA)	1.30

*NA—Not available

values are given in Table 2 for water balance information. Precipitation was measured at a single location for the two watersheds, and the annual amounts were 53.9, 80.1, and 41.6 cm for 1976, 1977, and part-year 1978, respectively.

GLEAMS under-estimated runoff in 1976 for both watersheds although the values were within an acceptable range. In 1977, the model significantly over-estimated the runoff compared with the observed values. Simulations for the 1978 part-year more closely approximated the observed values. The sediment yield comparisons were quite good for both watersheds as shown in Table 2 even when considering the over- and under-estimates of runoff.

Annual summaries of rainfall, runoff, percolation, and sediment yield for watershed P-2 at Watkinsville, GA for the period May 19, 1973 to October 1, 1975 are shown in Table 3. GLEAMS model simulations are included in the table, also. The data show that the model slightly under-estimated runoff and sediment yield in 1973. Although this may be due to the initial estimate of beginning soil moisture, there were two large storms in one day only 9 days after simulation began, and these storms accounted for about half of the observed annual runoff for the year. Simulated runoff was well below the measured volume. Simulated sediment yield for 1973 was under-estimated, also, by about the same degree as that for runoff. Simulated runoff in 1974 was only slightly under-estimated, but sediment yield was drastically over predicted. The largest observed sediment-producing storms occurred in late June and again in late July, and the crop canopy may have been under-estimated and thus the soil loss ratio parameter in the erosion model. Both runoff and sediment yield were over-estimated for 1978. Again, as was pointed out for the Iowa location, hydrology and erosion parameters were not fine tuned to get a "best fit" for the pesticide component at Watkinsville.

Simulated and observed values for atrazine (Smith et al., 1978) on Watkinsville watershed P-2, 1973-75, are

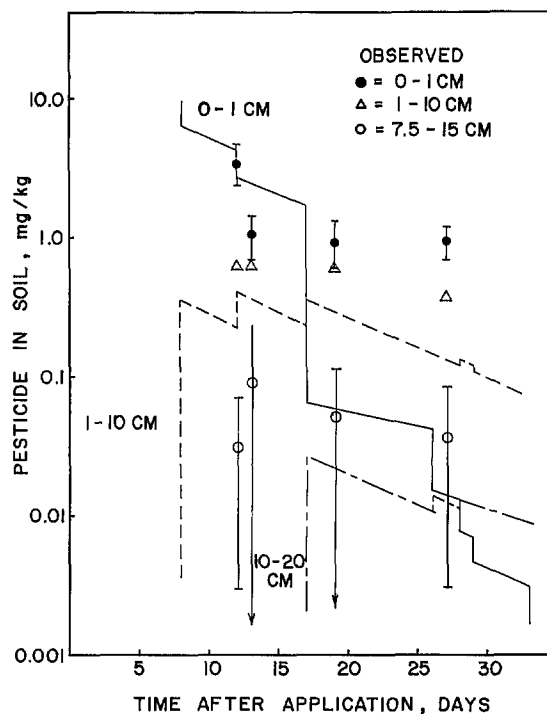


Fig. 4—Simulated and observed atrazine concentrations in the soil, watershed P-2, Watkinsville, GA, 1973 (adapted from Smith et al., 1978).

shown in Figs. 4, 5, and 6. In order to be consistent with earlier testing of the CREAMS pesticide component (Leonard and Wauchope, 1980), a value of 240 for K_{oc} was used along with a half-life of 7 days for atrazine. In Figs. 4-6, observations shown are means of 10 samples at different locations on the watershed. Representative standard deviations around the mean are indicated in the plots for 1973 (Fig. 4) and 1974 (Fig. 5). All standard deviations were not plotted since overlap would make the figures illegible.

In general, model simulations represented the field data relatively well considering variability in the field data. Both the observed and simulated data indicated movement of atrazine into the 3rd computational soil layer (10-20 cm), but only traces below this layer during the 30-day period shown in the figures. Two major differences were obvious between observed and simulated values. In 1973, concentrations simulated in the 0 to 1 cm soil layer after 17 days following atrazine application were well below those measured, and simulated values continued to decrease in response to subsequent rainfall. The rainfall on day 17 occurred in two storms totaling 10.85 cm. Runoff rates during these events were sufficient to cause considerable sheet and rill erosion, perhaps mixing the soil below the 1-cm depth and redistributing the pesticides.

TABLE 3. GLEAMS MODEL SIMULATIONS AND OBSERVED RUNOFF AND SEDIMENT YIELD, SIMULATED PERCOLATION, AND OBSERVED PRECIPITATION, WATERSHED P-2, WATKINSVILLE, GA, 1973-75

Year	Precipitation, cm	Runoff		Percolation cm	Sediment yield	
		Observed, cm	Simulated, cm		Observed, t/ha	Simulated, t/ha
1973	70.6	15.9	11.0	5.13	11.35	8.51
1974	102.3	12.1	10.2	13.54	0.94	6.50
1975	122.6	13.6	23.7	2.93	2.59	6.68

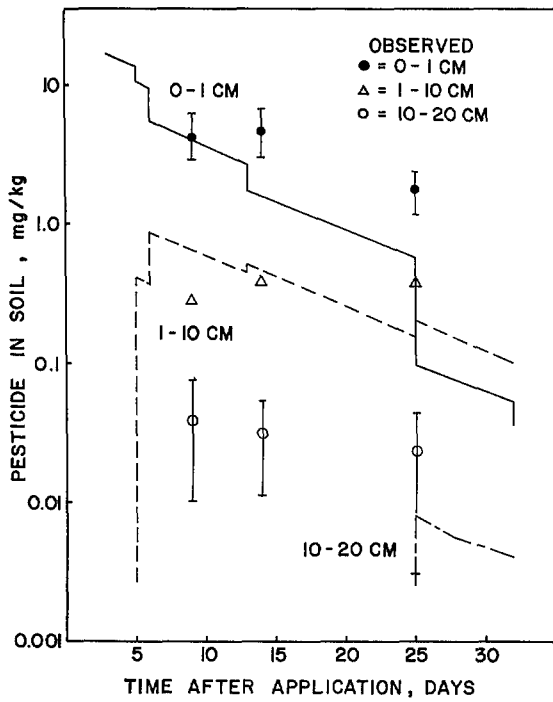


Fig. 5—Simulated and observed atrazine concentrations in the soil, watershed P-2, Watkinsville, GA, 1974 (adapted from Smith et al., 1978).

In 1974, atrazine observed in the 10 to 20 cm soil layer was much higher than simulated. However, residues were present at the first sampling date, presumably resulting from a carry-over from 1973.

Comparisons of atrazine data in a Bonifay sand profile from a 1985 study by Leonard et al. (1986) are presented in Fig. 7. The Bonifay is susceptible to leaching losses because of the sandy texture and low organic matter content (organic carbon 0.5%). A K_{oc} of 160 and a half-

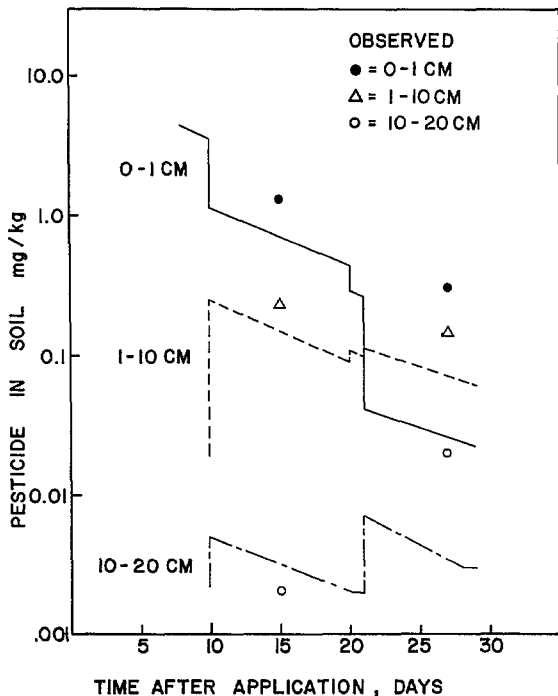


Fig. 6—Simulated and observed atrazine concentrations in the soil, watershed P-2, Watkinsville, GA, 1975 (adapted from Smith et al., 1978).

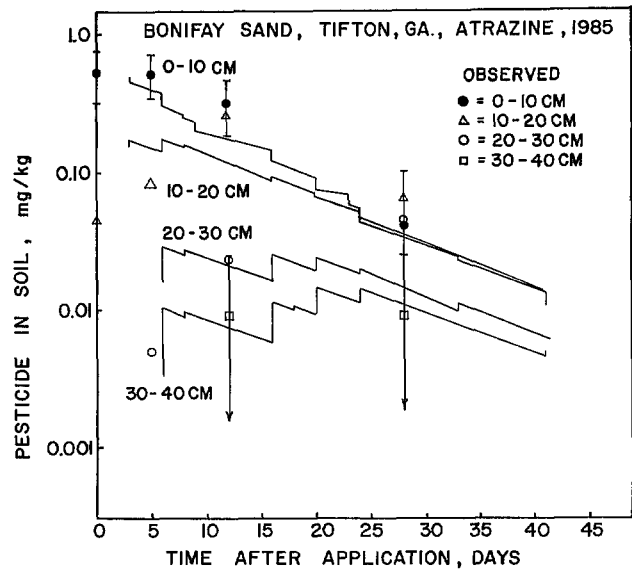


Fig. 7—Simulated and observed atrazine in a Bonifay sand, Tifton, GA, 1985 (adapted from Leonard et al., 1986).

life of 10 days were assumed for atrazine under these conditions. In this study, atrazine was incorporated to approximately 12 cm rather than surface applied as in the Watkinsville project. The 12-cm incorporation depth was determined by field sampling. A 15-cm incorporation depth was defined in the project design, but field and equipment conditions did not produce the desired depth. This known difference was taken into consideration when the parameter file was initially developed, and it was not a parameter adjustment for fine tuning. Soil sampling was by 10-cm increments, whereas the model simulations were for a 45-cm root zone which resulted in more depth resolution than that in the field data. Therefore, simulated values were composited by using approximate weighting factors to produce concentrations for depths corresponding to those sampled. The field plots received irrigation in addition to rainfall, and the irrigation volumes were added to the rainfall file for simulation rather than allow the model to "schedule" irrigation date and depth as would be done for simulation in a management mode.

The simulations and the observations indicated leaching of atrazine to at least the 30 to 40 cm depth (Fig. 7). Observations indicated traces of atrazine present below the 40-cm depth, also. Below the 40-cm depth, however, concentrations were not much greater than limits of detection and as a result, computed standard deviations were large. Standard deviations around means in Fig. 7 are from four samples only. The model simulation computed losses for the year of 3.75% of the atrazine application. Low levels of atrazine were known to reach shallow groundwater at 1.5 to 2.4 m below ground surface (Leonard et al., 1986).

The bromide ion (Br^-) may be assumed as a surrogate for a very mobile pesticide. Use of bromide allows an evaluation of water balance/solute transport components of GLEAMS without uncertainties of K_{oc} and half-lives, and complications due to metabolites. Simulations were performed to represent treatments and conditions during an experiment at Coshocton, OH, 1980-84 as reported by Owens et al. (1985). In their study, 168 kg Br^-/ha was broadcast on three lysimeters on June 19, 1980. The three monolith lysimeters (Y101B, Y101C, and Y101D)

TABLE 4. WATER BALANCE AND BROMIDE SIMULATIONS FROM THE GLEAMS MODEL, AND LYSIMETER DATA, COSHOCTON, OH, 1980-84

Component	Observation date			
	6/01/81	6/25/82	6/30/83	6/30/84
Observed (adapted from Owens et al., 1985)				
Precipitation, cm	106.56	216.68	320.60	414.61
Bromide leached, kg/ha*	42.66	73.73	89.97	95.40
Bromide uptake, kg/ha*	12.00	34.73	36.70	37.93
Simulated with 74-cm root zone				
Runoff, cm	8.58	20.41	29.53	37.91
Percolation, cm	33.49	61.20	88.78	114.49
Evapotranspiration, cm	60.41	135.11	199.63	263.17
Bromide leached, kg/ha	137.48	141.90	141.90	141.90
Bromide uptake, kg/ha	25.88	25.88	25.88	25.88
Simulated with 102-cm root zone				
Runoff, cm	9.22	21.92	31.76	40.57
Percolation, cm	32.55	59.54	86.00	111.20
Evapotranspiration, cm	60.46	135.19	199.79	263.34
Bromide leached, kg/ha	88.89	121.56	123.46	123.53
Bromide uptake, kg/ha	43.89	43.89	43.89	43.89
Simulated with 240-cm root zone				
Runoff, cm	9.60	23.09	33.40	42.34
Percolation, cm	30.56	57.89	80.39	107.39
Evapotranspiration, cm	60.47	135.31	199.88	263.51
Bromide leached, kg/ha	9.63	27.89	63.68	108.22
Bromide uptake, kg/ha	8.68	8.73	8.73	8.73

*Averages of values published for three lysimeters: Y101B, Y101C, and Y101D.

were 2.4 m deep and were planted in perennial pasture grasses. A more detailed description of the lysimeters may be found in Harold and Dreibelbis (1958). After bromide treatment, percolate volumes and bromide concentrations in percolate were determined for a 4-year period.

In the GLEAMS model simulations, bromide was assigned a K_{oc} of 0, and a half-life of 100,000 days (essentially infinite). The best information on the lysimeter soil characteristics (Kelly et al., 1975) indicate a 74-cm rooting depth for initial model simulation. However, the several profile descriptions given by Kelly et al. (1975) could easily be interpreted by a model user as having a rooting depth of 102 cm. Also, as an alternative for model validation using lysimeter percolate data, the full lysimeter depth might be used as the rooting depth, or 240 cm. It was considered desirable to examine model sensitivity to rooting depth and computational integrity for different thicknesses of soil layers, for both the hydrology water balance components and solute transport. Therefore, the three rooting depths (74 cm, 102 cm, and 240 cm) were simulated in separate runs of the GLEAMS model.

Bromide was applied on June 19, 1980. Cumulative data on bromide fate were reported by Owens et al. (1985) at approximately one year intervals to June 30, 1984. In the model simulation, nearly 140 kg Br⁻/ha had leached below the 74 cm depth by June 1, 1981 (Table 4). Adding plant uptake accounted for about 163 kg/ha during the same period. At the end of 2 years (June 25, 1982), the sum of bromide leached and plant uptake was 167.78 kg/ha, only 0.22 kg/ha less than that applied in 1980. Thus, an almost complete flushing of the 74-cm profile had occurred by the end of the second year. Comparing the bromide application with simulated uptake and leaching shows that the model is computationally stable and gives a chemical balance within round-off error. By the end of the second year, over 10,000 computational steps (layers by days) were performed in the model to provide the net results.

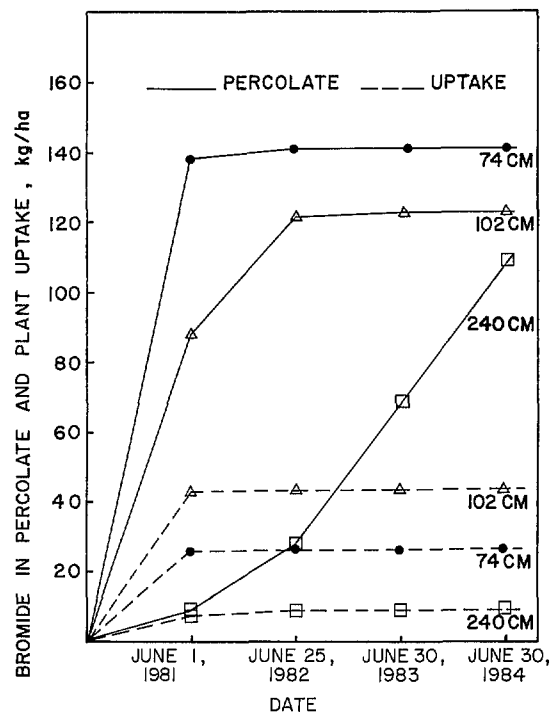


Fig. 8—Simulated bromide leaching losses and plant uptake for three rooting depths for lysimeters at Coshocton, OH, 1980-1984.

Model simulations with a 102-cm rooting depth resulted in complete flushing of the bromide at the end of the third year, whereas bromide leaching below the 240-cm root zone continued at the end of the 4-year period (Fig. 8). Regardless of rooting depth, most of the plant uptake of bromide occurred during the first year. This was the period when the bulk of the bromide was in the upper soil computational layers from which most of the water uptake occurred.

Simulation results compare favorably to observed results (Table 4) considering the different root zones. When the total lysimeter depth of 240 cm was used as the root zone, the amounts leached by the end of the fourth year was 18 kg Br⁻/ha compared with the average observed for the three lysimeters, 95 kg/ha. The three observations ranged from 61 to 138 kg/ha leached by the end of 4 years. Comparatively, 4-year simulated leaching ranged from about 108 to 142 kg/ha for the three rooting depths.

Owens et al. (1985) did not report lysimeter runoff volume, but runoff from two nearby watersheds varied considerably depending upon management averaging 1.6 and 12.4 cm/year for the study period. GLEAMS simulated runoff for the same period was 10.1 cm/yr. Average annual percolation observed from the three lysimeters ranged from 32.3 to 39.4 cm (Owens et al., 1985) compared with a simulated value of 27.8 cm averaged for the three rooting depths. The average annual observed watershed runoff plus lysimeter percolation was 34 cm compared with 38 cm simulated.

The present GLEAMS model simulates plant uptake as a final sink, that is, pesticide content in plants is not considered to be returned to the system. However, in nature, a significant amount of the bromide taken in through the transpiration stream would re-enter the soil through leaching of the crop canopy, particularly upon plant senescence, and diffusion from the plant roots,

resulting in a cycling process. This cycling process would return bromide to the soil profile over a much longer period than that presently simulated. The uptake/cycling processes will be represented as part of the plant nutrient submodel to be added in a later version.

The scatter and variability in the data from the test locations are similar to those observed by most investigators. Considerable variability in field pesticide data is inherent due to variability in pesticide application, spatial variability in soil properties, and sampling difficulties. The ability to adequately test models is affected by this variability and data uncertainty. Models provide much more resolution and fine structure in the simulated information than can ever be verified using field data. However, this fine structure is useful in addressing questions of logic. That is, are results consistent with our theory and/or concepts? Based upon the tests presented, GLEAMS appears to give logical outputs and represents field data within ranges of variability encountered.

Sensitivity Analyses

The reader is referred to the CREAMS documentation (Knisel, 1980) for fairly complete sensitivity analyses for surface response. However, the lysimeter data (Owens et al., 1985) provided an opportunity to examine sensitivity of the water balance components, bromide uptake, and bromide leaching to rooting depth. Although this was discussed briefly above (Table 4), further elaboration is desirable. Other sensitivity analyses were performed using base information presented in previous sections.

Using atrazine applied to Bonifay sand with the above test (Fig. 7) as baseline, values for K_{oc} , half-life, and pesticide incorporation depth were varied around those previously assigned in order to test sensitivity of model-simulated output to these parameters. Results expressed in terms of percent of the application leached below the 45-cm root zone are given in Fig. 9. Decreasing pesticide half-life results in decreased leaching losses, whereas decreasing K_{oc} results in increased losses, as would be

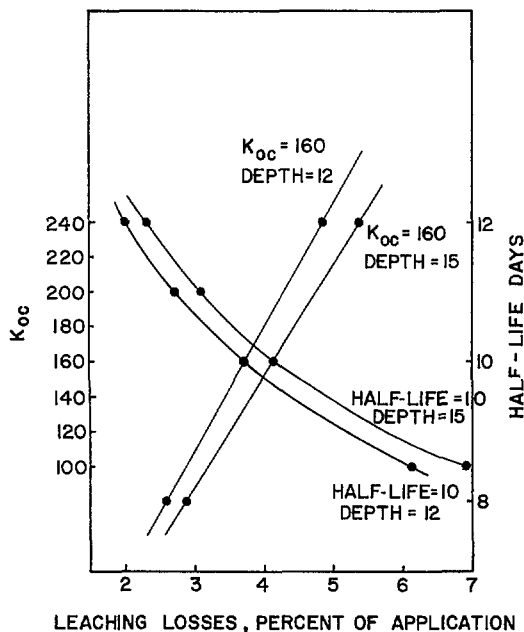


Fig. 9.—Potential pesticide leaching from Bonifay sand as affected by half-life, incorporation depth, and K_{oc} .

expected. Increasing incorporation depth results in increased losses because the pesticide is physically placed closer to the bottom of the root zone. In model applications, values selected for K_{oc} and half-life can have pronounced effects on simulation results. Unfortunately, values of these parameters are not known within a factor of 2 or 3 for specific sites without conducting extensive field studies (Rao and Davidson, 1982). Calibrating models with site-specific data does not overcome this inherent uncertainty if the model is to be applied at other sites.

Data in Table 4 indicate total simulated evapotranspiration is relatively insensitive to rooting depth. The main differences relative to rooting depth is the partitioning of water between surface runoff and percolation through the root zone. Accumulated runoff for each of the four years increases with increasing root depth. Although the surface soil layer is fixed in GLEAMS for all rooting depths, the remaining six computational layers are thicker for the deeper root-zones. The thicker layers are less sensitive to small changes in total water content, and runoff response increases as soil water increases. Since evapotranspiration was essentially the same for all rooting depths, and runoff increases with rooting depth, the only way a water balance can be achieved is by decreasing percolation and/or changing soil-water storage. This is shown in Table 4. If a model user wants to represent something like the lysimeter percolation and solute transport, then the deeper root zone should be used. If the user is more concerned with surface response, then the shallower root zone is recommended. In order to assess impacts of management practices on groundwater loadings at the bottom of the root zone, this sensitivity analysis indicates that rooting depth is not critical, but the user must be consistent in applications. GLEAMS is not a predictive model in the sense of absolute quantities, therefore simulated concentrations can not be interpreted as absolute values, and only differences between management practices should be assessed. Long-term observed data are not available to compare different practices on the same field.

Surface Response

In the development of the GLEAMS model, it was intended to retain as much of the surface sensitivity to management practices from CREAMS (Knisel, 1980) as feasible and be consistent with the additional concepts for root zone pesticide flux. The GLEAMS model description and testing presented in this paper has concentrated on chemical leaching and response at the bottom of the root zone. However, as a part of the model testing, surface response was simulated and a comparison of GLEAMS and CREAMS surface pesticide losses is given for Watkinsville, GA, since sample data, parameter, and output files are supplied with the CREAMS model. A comparison of observed and simulated surface runoff and sediment yield was given previously in Table 3. As shown in the summary comparison of Table 5, the combined runoff and sediment losses of atrazine simulated with GLEAMS is 0.75% of that applied for the simulation period, May 19, 1973 to October 1, 1975. This is lower than the 2.87% simulated with CREAMS, but actually closer to the total observed loss of 0.95% as given by Smith et al. (1978).

TABLE 5. COMPARISON OF SIMULATED AND OBSERVED SURFACE PESTICIDE LOSSES, WATKINSVILLE, GA, 1973-1975

Pesticide	Observed losses	GLEAMS simulation	CREAMS simulation
(----- Percent of application -----)			
Atrazine	0.95	0.75	2.87
Paraquat	7.23	5.46	4.98

Consistency of surface layer thickness between hydrology and pesticides, and the use of actual water content in the top layer for pesticide transport possibly account for most of the difference between the CREAMS and GLEAMS simulations. Although paraquat was not mentioned in the discussion on results of GLEAMS model testing above, it was included in the simulation for comparison with CREAMS. Neither the GLEAMS simulations nor the observed data indicated movement below the second computational soil layer. The total loss shown in Table 5 is almost entirely with sediment yield from the field. The GLEAMS simulation was 5.5% compared with 7.2% observed (Smith et al., 1978), and 5.0% loss simulated with the CREAMS model.

FUTURE MODEL MODIFICATIONS

The GLEAMS model has not been "completed", and some mention has been made in this paper thus far to indicate some known needed improvements or components. There are changes that will be made over some short time frame of about one year. Some of these have been formulated but have not been independently tested to the point of incorporating into the existing version of GLEAMS. The improvements include, but are not necessarily limited to, the following list.

- A plant nutrient component, completely different from that in CREAMS, has been formulated for incorporation. The component includes nitrogen fixation by legumes, land application of animal waste, distinction between ammonium and nitrate fertilizers and their uptake by crops, and improved nitrogen and phosphorus cycling algorithms.

- An optional climate generator will be included for daily rainfall, temperature, and radiation data. The climate generator used in the EPIC model (Williams and Renard, 1985) will be incorporated into GLEAMS.

- The pesticide component will be modified to consider degradation products (metabolites) to be used as an option for such compounds as aldicarb and fenamiphos.

- The erosion component will be revised to provide for internal model-driven parameter generation after incorporation of the plant nutrient component which generates growth/decay of crop residue. For example, the subfactor method of estimating soil loss ratio (Lafren et al., 1985) will be included as well as tillage/time functions for critical shear stress.

- Pest control/pesticide efficacy components will be incorporated into GLEAMS as they are developed to enhance the management capability of the model. For example, the present model version simulates pesticide concentrations by computational soil layer on a daily basis. This concentration/duration data can be used to estimate degree of pest control such as for various species of nematodes.

- The GLEAMS model will be interfaced with a groundwater model by the U. S. Geological Survey. The percolate output from GLEAMS will be used as input to the vadose or groundwater models as a part of the groundwater research program of the Southeast Watershed Research Lab, Tifton, GA.

SUMMARY

It is difficult to adequately summarize a diverse paper that presents model concepts, validation, and sensitivity. Therefore, a series of summary statements are enumerated to indicate the salient points.

- Concepts of the GLEAMS model were described to denote additions to the CREAMS model that enable the estimation of translocations and leaching of pesticides below the root zone as functions of pesticide characteristics and percolation.

- The GLEAMS model was tested, primarily for pesticide leaching at the bottom of the root zone, using readily-available data sets selected to provide a range of climate, soils, and pesticide characteristics.

- Results of model testing were presented to demonstrate that the model simulates pesticide and bromide movement and leaching generally within the range of variability of field data. Interpretations of model simulation results are given relative to the temporal and spatial variability of field observations and variability of pesticide characteristics.

- Sensitivity analyses were presented for some pesticide characteristics and for crop rooting depths. Pesticide half-life, adsorptivity, and depth of incorporation were shown to be sensitive in determining pesticide leaching.

LIST OF SYMBOLS

AWS	Depth of available water storage, cm
B	Soil mass per unit volume of water, g/cm ³
C _{av}	Runoff-available pesticide concentration, mg/L
C _s	Pesticide concentration in soil, mg/kg
C _w	Pesticide concentration in soil solution, mg/L
DLG	Diameter of large aggregates, mm
DSG	Diameter of small aggregates, mm
ET	Evapotranspiration, mm
EV	Depth of daily soil evaporation, mm
f	Water flux (depth) infiltrated into the soil surface, mm
F _{1/2}	Pesticide half-life on foliage, days
FC	Field capacity (depth), mm—depth of water soil can hold against the force of gravity after 24 h drainage (approximated by water content at 10 or 33 kPa)
FCL	Fraction of primary clay in the sediment
FCLCL	Fraction of clay in sediment as primary clay
FCLLG	Fraction of clay in large aggregate
FCLSG	Fraction of clay in small aggregate
FLG	Fraction of large aggregates in the sediment
FSA	Fraction of primary sand in the sediment
FSASA	Fraction of sand in sediment as primary sand
FSALG	Fraction of sand in large aggregate
FSI	Fraction of primary silt in the sediment
FSISI	Fraction of silt in sediment as primary silt
FSILG	Fraction of silt in large aggregate
FSISG	Fraction of silt in small aggregate
FSG	Fraction of small aggregates in the sediment
i	Subscript denoting soil layer, e.g. 1, 2, 3, etc.
K _d	Coefficient for partitioning pesticide between soil phase and solution (water) phase
K _{oc}	Coefficient for partitioning pesticide between organic carbon and water phase
K _{ow}	Coefficient for partitioning pesticide between octanol and water
MCL	Fraction of clay in the matrix clay
MSA	Fraction of sand in the matrix soil

MSI	Fraction of silt in the matrix soil
o	Subscript for initial time, t=0
OC	Organic carbon content of soil, %
OM	Organic matter content of soil, %
P	Precipitation depth, cm
PCS	Pesticide concentration in soil, mg/kg
PERC	Percolation depth, cm
PERCM	Pesticide mass in percolate water, kg/ha
PMEV	Pesticide mass evaporated, kg/ha
PMF	Pesticide mass on foliage, kg/ha
PMS	Pesticide mass in the soil, kg/ha
PMTR	Pesticide mass transpired (plant uptake), kg/ha
POR	Soil porosity, cm ³ /cm ³
Q	Runoff depth, cm
S	Sum of fractions of primary clay, silt, and sand in sediment
S _{1/2}	Pesticide half-life in soil, days
SM	Soil mass, kg/ha
SSG	Soil specific gravity, g/cm ³
SW	Initial soil water content in a soil layer before redistribution by rainfall or irrigation
t	Time, days
t	Subscript for time = t days after t = 0
TR	Depth of transpiration, cm
V	Volume of water per unit volume of runoff interface
WM	Water mass in a soil layer, kg/ha
WS	Pesticide water solubility, mg/L
Y	Pesticide mass in the soil, kg/ha
Z	Pesticide mass in the top soil layer, kg/ha

References

- Baker, J. L., and H. P. Johnson. 1979. The effect of tillage systems on pesticides in runoff from small watersheds. *TRANSACTIONS of the ASAE* 22(3):554-559.
- Carsel, R. F., L. A. Mulkey, M. N. Lorber, and L. B. Baskin. 1985. The pesticide root zone model (PRZM): A procedure for evaluating pesticide leaching threats to groundwater. *Ecological Modeling* 30(1985): 49-69.
- Crowder, B. M., and C. E. Young. 1985. Modeling agricultural non-point source pollution for economic evaluation of the Conestoga headwaters RCWP project. U. S. Department of Agriculture, Economic Research Service, Natural Resources Economics Division, Washington, D. C. ERS Staff Report No. AGES850614. 70 pp.
- Del Vecchio, J. R., and W. G. Knisel. 1982. Application of a field-scale nonpoint pollution model. Proceedings of the American Society of Civil Engineers Specialty Conference on Environmentally Sound Water and Soil Management, Orlando, Florida, July 20-23, 1982. pp. 227-236.
- Del Vecchio, J. R., W. G. Knisel, and V. A. Ferreira. 1983. The impact of irrigation on pollutant loads. Proceedings of the American Society of Civil Engineers Specialty Conference on Advances in Irrigation and Drainage: Surviving External Pressures. Jackson, WY, July 20-22, 1983. pp. 113-123.
- Dickey, G. L. 1985. Analysis of land treatment practices for water conservation. Proceedings of the Workshop, Planning and Management of Water Conservation Systems in the Great Plains States. U. S. Department of Agriculture, Soil Conservation Service, Lincoln, NB, October 21-25, 1985. pp. 345-375.
- Donigian, A. S., and N. H. Crawford. 1976. Modeling pesticides and nutrients on agricultural lands. U. S. Environmental Protection Technology Series, EPA-600/2-76-043, 317 pp.
- Donigian, A. S., J. C. Imhoff, and B. R. Bicknell. 1983. Predicting water quality resulting from agricultural nonpoint source pollution via simulation—HSPF. In: Schaller, F. W., and G. W. Bailey (Eds.) *Agricultural Management and Water Quality*. Iowa State University Press, Ames, pp. 200-249.
- Dumper, T. A. 1985. Simulating partial irrigation with the CREAMS model. Proceedings of the Workshop, Planning and Management of Water Conservation Systems in the Great Plains States, U. S. Department of Agriculture, Soil Conservation Service, Lincoln, NB, October 21-25, 1985, pp. 336-344.
- Dumper, T. A., W. G. Knisel, D. L. Hintz, and D. H. Plachy. 1986. Simulating the evaporation of windbreak species with the CREAMS model (Abstract). Presented at the International Symposium on Windbreak Technology, Lincoln, NB, June 23-27, 1986.
- Foster, G. R., and V. A. Ferreira. 1981. Deposition in uniform grade terraces. Proceedings of the American Society of Agricultural Engineers Conference on Crop Production with Conservation in the 80's, ASAE, St. Joseph, MI. pp. 185-197.
- Foster, G. R., R. A. Young, and W. H. Neibling. 1985. Sediment composition for nonpoint source pollution analyses. *TRANSACTIONS of the ASAE* 28(1):133-139, 146.
- Foster, G. R., L. J. Lane, J. D. Nowlin, J. M. Laflen, and R. A. Young. 1980. A model to estimate sediment yield from field-size areas: Development of model. In: Knisel, W. G. (Ed.), *CREAMS: A field-scale model for Chemicals, Runoff, and Erosion from Agricultural Management Systems*. U. S. Department of Agriculture, Science and Education Administration. Chapter 3, pp. 36-64.
- Green, R. E., and S. W. Karickhoff. 1986. Estimating pesticide sorption coefficients for soils and sediments. Unpublished Technical Report, prepared for publication by the U. S. Department of Agriculture, Agricultural Research Service, in supporting documentation for the Small Watershed Model (SWAM), 41 m/s pp.
- Hakanson, T. E., L. J. Lane, J. G. Steger, and G. L. De Poorter. 1982. Some interactive factors affecting trench cover integrity on low-level waste sites. Symposium on Site Characterization and Monitoring, Arlington, Virginia, June 16-17, 1982. Nuclear Regulatory Commission Report NUREG/CP-0028 CONF-820674, Volume 2.
- Harold, L. L., and F. R. Dreibeilbis. 1958. Evaluation of agricultural hydrology by monolith lysimeters. U. S. Department of Agriculture, Technical Bulletin No. 1179.
- Johnson, H. P. 1978. Development and testing of mathematical models as management tools for agricultural non-point pollution control. Annual Report 1977-1978. Department of Agricultural Engineering, Iowa State University, Ames, Unpublished, 645 pp.
- Karickhoff, S. W. 1984. Organic pollutant sorption in aquatic systems. *Journal of Hydraulic Engineering* 110: 707-735.
- Kelly, G. E., W. M. Edwards, L. L. Harold, and J. L. McGuinness. 1975. Soils of the North Appalachian Experimental Watershed. U. S. Department of Agriculture, Miscellaneous Publication No. 1296, 145 pp.
- Kenaga, E. E., and Goring, C. A. I. 1980. Relationship between water solubility, soil sorption, octanol-water partitioning and concentration of chemicals in biota. In: Eaton, J. G., P. R. Parrish, and A. C. Hendriks (Eds.) *Aquatic Toxicology*. American Society for Testing Materials, Philadelphia, PA. ASTM STP 707, pp. 78.
- Knisel, W. G. (Ed.). 1980. *CREAMS: A field-scale model for Chemicals, Runoff, and Erosion from Agricultural Management Systems*. U. S. Department of Agriculture, Science and Education Administration, Conservation Research Report No. 26, 643 pp.
- Knisel, W. G., G. R. Foster, and R. A. Leonard. 1983. *CREAMS: A system for evaluating management practices*. In: Schaller, F. W., and G. W. Bailey (Eds.), *Agricultural Management and Water Quality*, Iowa State University Press, Ames, pp. 178-199.
- Knisel, W. G., and R. A. Leonard. 1986. Impact of irrigation on groundwater quality in humid areas. Proceedings of the American Society of Civil Engineers, Water Forum '86: World Water Issues in Evolution, Long Beach, California, August 4-6, 1986, Volume 2, pp. 1508-1515.
- Knisel, W. G., D. C. Moffitt, and T. A. Dumper. 1985. Representing seasonally frozen soil with the CREAMS model. *TRANSACTIONS of the ASAE* 28(5):1487-1493.
- Laflen, J. M., G. R. Foster, and C. A. Onstad. 1985. Simulation of individual-storm soil loss for modeling impact of soil erosion on crop productivity. In: Proceedings of Preserve the Land, 1983 International Conference on Soil Conservation, Honolulu, Hawaii. January 1983. Soil Conservation Society of America, Ankeny, IA.
- Lane, L. J. 1984. Surface water management: A user's guide to calculate a water balance using the CREAMS model. Los Alamos National Laboratory, Los Alamos, NM. LA-10177-M Manual, 49 pp.
- Leonard, R. A., W. G. Knisel, D. A. Still, and A. W. Johnson. 1985. Modeling vertical flux of pesticides with CREAMS. Proceedings of the Non-point Pollution Abatement Symposium, Milwaukee, WI, April 23-25, 1985, pp. T-I-D-1 T-I-D-12.
- Leonard, R. A., A. Shirmohammadi, A. W. Johnson, and L. R. Marti. 1986. Pesticide transport in shallow ground water. ASAE Paper No. 86-2508, ASAE, St. Joseph, MI 49085, 43 pp.
- Leonard, R. A., and R. D. Wauchope. 1980. The pesticide submodel. In: Knisel, W. G. (Ed.). 1980. *CREAMS: A field-scale model for Chemicals, Runoff, and Erosion from Agricultural Management Systems*. U. S. Department of Agriculture, Science and Education Administration, Conservation Research Report No. 26, pp. 88-112.
- Lorber, M. N., and L. A. Mulkey. 1982. An evaluation of three runoff loading models. *Journal of Environmental Quality* 11:519-529.
- Marti, L. R., and J. DeKanel, and R. C. Dougherty. 1984. Screening for organic contamination of groundwater: Ethylene dibromide in Georgia irrigation wells. *Journal of Environmental Science* 18(12):973-974.

32. Nash, R. G. 1980. Dissipation rates of pesticides from soils. In: Knisel, W. G. (Ed.) CREAMS: A field-scale model for Chemicals, Runoff, and Erosion from Agricultural Management Systems. U. S. Department of Agriculture, Science and Education Administration, Conservation Research Report No. 26, Chapter 17, pp. 560-594.
33. Nash, R. G. 1984. Methods for estimating pesticide dissipation from soils. Unpublished Technical Report, Prepared for publication by the U. S. Department of Agriculture, Agricultural Research Service, in supporting documentation for the Small Watershed Model (SWAM), 44 m/s pp.
34. Nicks, A. D., J. B. Burt, W. G. Knisel, and G. A. Gander. 1984. Application of a field-scale water quality model for evaluation of a watershed land treatment project. ASAE Paper No. 84-2633, ASAE, St. Joseph, MI 49085, 33 pp.
35. Nofziger, D. L., and A. G. Hornsby. 1984. Chemical movement in soil: User's guide. University of Florida, Gainesville.
36. Nutter, W. L., P. B. Bush, D. G. Neary, R. McKenna, and J. Taylor. 1986. Pesticide runoff risk assessment for a Georgia Piedmont seed orchard. Forest Science (In press)
37. Nutter, W. L., T. Tkacs, P. B. Bush, and D. G. Neary. 1984. Simulation of herbicide concentrations in stormflow from forested watersheds. Water Resources Bulletin 20(6):851-857.
38. Ou, L. T., J. E. Thomas, K. S. V. Edvardsson, P. S. C. Rao, and W. B. Wheeler. 1986. Aerobic and anerobic degradation of aldicarb in aseptically collected soils. Journal of Environmental Quality 15:356-363.
39. Owens, L. B., R. W. Wan Keuren, and W. M. Edwards. 1985. Groundwater quality changes resulting from a surface bromide application to a pasture. Journal of Environmental Quality 14(4):543-548.
40. Pionke, H. B., and R. J. DeAngelis. 1980. Method for distributing pesticide loss in field runoff between the solution and adsorbed phase. In: Knisel, W. G. (Ed.) CREAMS: A field-scale for Chemicals, Runoff, and Erosion from Agricultural Management Systems. U. S. Department of Agriculture, Science and Education Administration, Conservation Research Report No. 26, Chapter 19, pp. 607-643.
41. Rao, P. S. C., and J. M. Davidson. 1980. Estimation of pesticide retention and transformation parameters required in non-point source pollution models. In: Overcash, M. R., and J. M. Davidson (Eds.), Environmental Impact of Nonpoint Source Pollution. Ann Arbor Science Publishers, Inc., Ann Arbor, MI, 67 pp.
42. Rao, P. S. C., and J. M. Davidson (Eds.). 1982. Retention and transformations of selected pesticides and phosphorus in soil-water systems: A critical view. U. S. Environmental Protection Agency, Washington, D. C., EPA-600/3-82-060.
43. Rao, P. S. C., V. E. Beriheiser, and L. T. Ou. 1984. Estimation of parameters for modeling the behavior of selected pesticides and orthophosphate, EPA-600/S3-84-019.
44. Smith, C. N., R. A. Leonard, G. W. Langdale, and G. W. Bailey. 1978. Transport of agricultural chemicals from small upland Piedmont watersheds. U. S. Environmental Protection Agency, Athens, GA, and U. S. Department of Agriculture, Agricultural Research Service, Watkinsville, GA. Final Report on Interagency Agreement No. D6-0381. Publication No. EPA 600/3-78-056, 363 pp.
45. Svetlosanov, V., and W. G. Knisel (Eds.). 1982. European and United States case studies in application of the CREAMS model. International Institute for Applied Systems Analysis, Laxenburg, Austria. CP-82-S11, 148 pp.
46. U. S. Department of Agriculture, Soil Conservation Service. 1972. National Engineering Handbook: Section 4, Hydrology. Washington, D. C., 548 pp.
47. U. S. Department of Agriculture, Soil Conservation Service. 1984. User's guide for the CREAMS computer model, Washington Computer Center Version. U. S. Department of Agriculture, Soil Conservation, Engineering Division, Washington, D. C. Technical Release 72, 160 pp.
48. U. S. Department of Commerce, National Oceanic and Atmospheric Administration. 1974. Climates of the states: Volume I—Eastern States plus Puerto Rico and the U. S. Virgin Islands. Water Information Center, Inc., Port Washington, NY, 486 pp.
49. U. S. Department of Commerce, National Oceanic and Atmospheric Administration. 1974. Climates of the states: Volume II—Western States including Alaska and Hawaii. Water Information Center, Inc., Port Washington, NY, 499 pp.
50. Wagenet, R. J., and J. L. Hutson. 1986. Predicting the fate of nonvolatile pesticides in the unsaturated zone. Journal of Environmental Quality, 15(4):315-322.
51. Williams, J. R., and H. D. Berndt. 1977. Sediment yield prediction based on watershed hydrology. TRANSACTIONS of the ASAE 20(6):1100-1104.
52. Williams, J. R., and A. D. Nicks. 1982. CREAMS hydrology model—Option 1. In: Singh, V. P. (Ed.) Applied Modeling in Catchment Hydrology, Proceedings of the International Symposium on Rainfall-Runoff Modeling. Water Resources Publications, Littleton, CO, pp. 69-86.
53. Williams, J. R., and K. G. Renard. 1985. Assessments of soil erosion and crop productivity with process models (EPIC). In: Follett, R. F., and B. A. Stewart. Soil Erosion and Crop Productivity, American Society of Agronomy, Madison, WI, pp. 68-103.