

PESTICIDE BEHAVIOR IN SOILS AND GROUNDWATER PROTECTION IN PEANUT MANAGEMENT SYSTEMS

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INTRODUCTION

Most current production guides for any crop, like peanuts, now include sections dealing with the environmental ramifications of producing the commodity. These discussions primarily focus on pesticide use and groundwater quality. A number of pesticides, including some commonly used in peanut production, have been detected in groundwater in several states. Some detections can be attributed to point sources. Other contamination is believed to be the result of normal usage over broad areas and therefore considered nonpoint source pollution. Growing public concern, media coverage, and increased governmental attention has prompted much debate about agricultural pesticides as a nonpoint source threat to groundwater. Minute concentrations of pesticides in potable water sources may or may not pose real health hazards. However, the threat perceived by the public is real and must be addressed appropriately.

While the benefits to crop production from proper pesticide use have been well documented, pesticide behavior in the soil environment is not as clearly understood. Mechanisms of pesticide degradation and transport through soil have been the focus of many research papers in the past 10 years. Some relationships are well established but much work is still needed to predict with reasonable accuracy and consistency what compounds and conditions will lead to groundwater contamination.

Producers will increasingly have to be aware of, and responsible for, the potential fate of any pesticide used in their pest management strategies. Characterizing and predicting the behavior of organic chemicals in the soil environment is crucial for proper pesticide use. Avoiding the use of mobile compounds in unsuitable soil situations will help minimize possible off-site damage, public concern, and may prevent the loss of pesticides as a crop production tool.

PESTICIDES IN GROUNDWATER

The discovery of the insecticide aldicarb in shallow groundwater on Long Island, New York in the early 1980's drew increased attention to the potential for pesticides to leach to groundwater (Cohen *et al.*, 1984). Detection of the nematicide ethylene dibromide in domestic and municipal wells in several

states provided further evidence that some pesticides could leach to groundwater supplies (Marti *et al.*, 1984; Frink and Hankin, 1986; U.S. EPA, 1987). These reports and others lead to national concerns that pesticide use could result in significant contamination of ground and surface waters (Garner *et al.*, 1986). Since a major portion of the U.S. population depends on groundwater as their primary potable water source (97% of the rural population, 40% of those served by public water supplies) the impacts of contamination could be far reaching and costly (Nielsen and Lee, 1987).

Several national and regional studies have been conducted to investigate pesticide residue levels in groundwater. The United States Environmental Protection Agency (U.S. EPA) reported detection of 46 pesticides in groundwater from 26 states as the result of normal agricultural use (U.S. EPA, 1988). Misuse and point sources were attributed to the detection of 32 pesticides in 12 states. Several of the chemicals found in this and other studies are among the most widely used in crop production. The herbicide atrazine has been detected in groundwater from at least 13 states and alachlor and metolachlor, important herbicides in peanut production, have been detected in a minimum of 15 and five states, respectively (U.S. EPA, 1977, 1978, 1988; Cohen *et al.*, 1984, 1986; Ritter, 1986; Southwick *et al.*, 1988; Maas *et al.*, 1992; Koterba *et al.*, 1993). According to the latest available national pesticide use data bases, an estimated 760,000 kg ai alachlor and 460,000 kg ai metolachlor were applied to 43 and 41%, respectively, of the total 1987 U.S. peanut acreage (Gianessi and Puffer, 1990, 1992). The insecticide aldicarb has been reported in groundwater in seven states (U.S. EPA, 1988). Approximately 270,000 kg ai of aldicarb was applied to 41% of the 1987 U.S. peanut acreage (Gianessi and Puffer, 1992). While total use of these pesticides changes from year to year, the 1987 national data demonstrates the wide use in peanuts of some pesticides that have come under scrutiny. Results of the 1988 U.S. EPA study are similar to other national assessments and differ primarily in data acquisition (U.S. EPA, 1987; Hind and Evans, 1988).

Many states with reported nonpoint source detections are not in the peanut growing region and most detections are not above currently established maximum contaminant limits (MCL) or health advisory limits (HAL) (U.S. EPA, 1988). The distinction between point and nonpoint source pollution is sometimes questionable and further complicates the picture. However, enough data is available to confirm that nonpoint source pesticide contamination does occur and greater steps need to be taken to protect groundwater supplies.

GENERAL REGULATORY ENVIRONMENT

The two primary statutes regulating pesticides in the U.S. are the Federal Insecticide, Rodenticide, and Fungicide Act (FIFRA) and the Federal Food, Drug, and Cosmetic Act (FFDCA) (Johnson and Stangel, 1990; McClain, 1991). The U.S. EPA registers and licenses pesticide products sold and used

in the U.S. under FIFRA. Specific tolerances for pesticide residues in food or feed are established under FFDCA. These tolerances must be set before any pesticide is registered for use. The Food and Drug Administration and the United States Department of Agriculture monitor domestic and imported foods to ensure compliance with established tolerances.

While both FIFRA and FFDCA are focused on assuring a safe food supply, FIFRA also has elements related to groundwater protection. Scientific data supplied by registrants to U.S. EPA must demonstrate that each pesticide can be used without "unreasonable adverse effects" on people or the environment (Johnson and Stangel, 1990). These data may include results from small laboratory degradation experiments to multi-site field-scale groundwater monitoring studies. Excessively mobile and persistent compounds (those that easily move through soils to groundwater) are flagged and may not be given registration. The environmental fate data requirements help prevent the use of compounds that may present an unacceptable threat to groundwater. Unfortunately, these data can be lacking in the sense that not all pesticide/soil/climatic combinations can be evaluated. Further, controlled studies often do not reflect actual field conditions.

Other statutes more specifically address nonpoint source pollution and water quality such as the Clean Water Act, Safe Drinking Water Act, and the Coastal Zone Management Act (Conner *et al.*, 1987; McClain, 1991). Sections of the Clean Water Act call for each state to identify nonpoint source pollution problem areas and devise management programs to attain applicable water quality standards (McClain, 1991). The Coastal Zone Management Act requires states to develop and enforce nonpoint source control programs in coastal areas under a legislatively set timetable (Griffin, 1991). Soil runoff from agricultural land is a nonpoint source specifically identified for management. Maximum contaminant levels for chemicals in potable water are set under the Safe Drinking Water Act (McClain, 1991). Passage and reauthorization of these acts point to an increased trend toward regulation for groundwater protection and water quality (Danielson, 1992).

As indicated above, states are often given the task of groundwater protection. States are not allowed to enforce groundwater protection rules which are less stringent than those set by federal law. However, they can and do develop rules more strict than federal guidelines (McClain, 1991; Danielson, 1992). The greatest regulatory impacts likely to be felt by commodity producers is through state actions in development of nonpoint source pollution or groundwater protection control programs. Legislation passed in Kansas is an example. The Kansas Board of Agriculture has been authorized to identify management areas and develop control programs for specific chemicals deemed to threaten groundwater (Murphy and Fahnestock, 1993). A plan adopted in 1992 established an atrazine management area in northeast Kansas and limits atrazine use in the Delaware River Basin. Nebraska developed Natural Resource Districts (NRD) in 1986 to reduce groundwater contamination from nitrates (Murphy and Fahnestock, 1993). One of the 23 Nebraska NRDs has implemented a three-phase plan that controls how and when nitrogen applications are made. Phase I (groundwater

nitrate levels of 0 to 12.5 ppm) requires nitrogen fertilizer applicators to take NRD certification classes and bans fall and winter applications before March 1 on sandy soils. Phase II (groundwater nitrate levels of 12.5 to 20 ppm) includes sampling of irrigation water and 90-cm soil samples for nitrates and submission of annual reports. Phase III (groundwater nitrate levels >20 ppm) bans fertilizer application on all soils before March 1 and specifies that spring fertilizer applications must be split with less than 50% applied preplant and the rest sidedressed.

States are currently developing State Pesticide Management Plans to more strictly monitor and regulate use of specific pesticides. These pesticides will be identified by the U.S. EPA, but states will have to develop and enforce the plans. For example, in the peanut producing state of North Carolina, State Management Plans are currently planned for five pesticides. The list of five pesticides will be submitted to North Carolina from U.S. EPA in the first or second quarter of 1994 (H. Wade, pers. commun., 1994). A generic State Pesticide Management Plan has already been prepared and will be the basis of subsequent plans (NCDA, 1993; U.S. EPA 1993, 1994a,b). The generic State Pesticide Management Plan, December 1993 draft, outlines the monitoring and regulatory strategies the state may use. Activities ranging from the encouragement of best management practices (BMPs) to establishment of prohibition areas are outlined. Each state must prepare a management plan for those pesticides specified by the U.S. EPA. If a plan is not developed then the pesticide cannot be sold in that state.

Regional, state, and local controls will likely become more widespread. It is hoped, however, that continued research and education programs can lead to adequate groundwater protection without extensive regulatory restrictions on producers (Contant, 1990; Helfrich and Weigmann, 1990).

PESTICIDE BEHAVIOR AND FATE

Overview of Relevant Properties

The behavior and fate of pesticides in the environment is governed by specific properties of the pesticide and those of the surrounding media (Weber, 1972, 1992; Hance, 1980; Swann and Eschenroeder, 1983; Sawhney and Brown, 1989). These interrelated properties determine the major pathways in which organic chemicals (pesticides) are moved about and how long they may persist.

Pesticides. Important characteristics of a pesticide (and its metabolites) affecting environmental behavior and fate include ionization, water solubility, presence of reactive groups, volatility, longevity, and extractability (Weber, 1987).

Ionization, the acid-base characteristics of a compound, determines whether it has acidic, basic, or nonionic properties. Acidic chemicals may be predominately active in the soil environment as anions and basic compounds as cations depending on their pK_a (acid dissociation) values and soil pH (Weber, 1970, 1980, 1982b, 1987). Nonionic chemicals have no specific

charge properties. Cationic species may interact with negatively charged surfaces of soil constituents, expressed as the cation exchange capacity (CEC), becoming adsorbed and limiting movement (Weber, 1970, 1991). Anionic species may be excluded from CEC sites and some micropores due to charge repulsion, actually enhancing movement (Thomas and Swoboda, 1970). Soils with high levels of oxides and hydrous oxides of iron, aluminum, or manganese may exhibit positive charge at low pHs and can adsorb anionic species (Watson *et al.*, 1973).

Water solubility, typically measured at 25 C and atmospheric pressure, determines how readily a chemical enters the aqueous phase. Compounds with high water solubilities are typically more susceptible to leaching than those with low solubilities as they may readily move with soil water. Those with very low solubilities may precipitate out of solution as the soil water content changes. They may also be retained by hydrophobic portions of soil constituents, such as those associated with organic matter (Chiou, 1989).

Reactive groups on a pesticide, such as $-\text{PO}_3^-$ or $-\text{AsO}_3^-$, can strongly complex with clay minerals in soil rendering them nearly immobile (Woolson, 1975). In addition, $-\text{NO}_2$ groups can participate in hydrogen bonding to proteinaceous matter (Weber 1987).

A chemical's volatility, represented by vapor pressure, indicates propensity of the compound to enter the gaseous phase. Some amide, anilide, carbamate, and thiocarbamate pesticides have moderate to high volatility (vapor pressure $>100 \times 10^{-6}$ mm Hg at 25 C) and may be lost if not incorporated into the soil. Loss of fumigant pesticides through volatilization readily occurs because of their high vapor pressures (>1 mm Hg at 25 C). In addition to vapor pressure, Henry's Law constants, effective diffusion coefficients, and fugacity have been used to characterize volatilization of pesticides from soil (Mackay, 1979; McCall *et al.*, 1983; Jury, 1986).

Longevity of a pesticide refers to the length of time it remains in the parent molecular form after application. This is generally expressed as half-life ($T_{1/2}$) and, for most compounds, is in the range of a few weeks to a few months. Considering this property only, compounds with long $T_{1/2}$ present a greater threat to water quality than those with short $T_{1/2}$. The longer an undegraded pesticide remains in the soil environment the greater the likelihood of significant rainfall events moving it toward ground or surface waters. A host of biological and chemical processes affect a chemical's longevity and degradation is often strongly influenced by soil pH (Weber, 1994).

Extractability refers to the partitioning of a chemical into nonpolar organic solvents and is related to the lipophilic character of the chemical. This is most often expressed as the octanol-water partition coefficient or K_{ow} . Pesticides with high K_{ow} values are generally of low water solubility and are frequently associated with lipophilic matter in soils (Weber, 1988). This factor may also be used to evaluate the potential for a chemical to biomagnify or enter the food chain (Kenaga and Goring, 1980).

Soils. Many pesticides are applied directly to the soil. Portions of those applied to plant foliage inevitably reach the soil. Knowledge of a pesticide's

chemical properties is crucial, but knowledge of key soil properties is also important. It is not possible to accurately assess the threat a particular chemical may pose to groundwater at a given location by examining chemical properties alone. Key soil properties include (a) clay content and type, (b) organic matter content, (c) hydrous oxide content, (d) soil structure, and (e) soil reaction or pH (Weber, 1987).

Clay content and type is related to movement of pesticides through soil in several ways. From a physical standpoint, soils with low clay contents are often coarse-textured and porous. Coarse-textured soils (sands) generally have low water-holding capacities and high water transmission rates which permit rapid movement of water and, potentially, dissolved chemicals (Smith and Cassel, 1991). Conversely, soils with significant amounts of expanding lattice-type 2:1 clays, such as montmorillonite, may shrink considerably when dry and leave large channels for water flow (Buol *et al.*, 1989). These channels are often open from the surface to a depth of 1 m or more. Chemically, clays can affect the mobility of certain types of pesticides because of their large surface area and net negative charge or CEC. Kaolinitic clays, predominant in peanut soils of the Southeast, have a low charge density (*ca.* 10-100 mmol charge/kg clay) and are 1:1 silica tetrahedral:aluminum octahedral layered, nonexpanding type clays with appreciable pH dependent charge (Bohn *et al.*, 1985). Vermiculite and montmorillonite have higher charge densities and are both 2:1 silica tetrahedral:aluminum octahedral layered clays. Vermiculite and montmorillonite have charge densities of *ca.* 1200-1500 and 800-1200 mmol charge/kg clay, respectively. Montmorillonite has extensive expanding properties but vermiculite does not. Experiments relating pesticide adsorption to clay has frequently been done with montmorillonite and/or kaolinite (Weber, 1970, 1982a; Aharonson and Kafkafi, 1975; Terce and Calvet, 1978; Loux *et al.* 1989a,b; Moliner and Street, 1989; Che *et al.*, 1992; Laird *et al.*, 1992). Some studies have demonstrated increased adsorption of pesticides with increasing soil clay content (Strek and Weber, 1982; Shea and Weber, 1983; Peter and Weber, 1985). Solution pH is cited as having a major influence on pesticide adsorption with adsorption increasing as solution pH decreases (Weber, 1970; Aharonson and Kafkafi, 1975; Terce and Calvet, 1978; Loux *et al.* 1989a,b; Moliner and Street, 1989; Che *et al.*, 1992).

Soil organic matter content (OM) is often considered the most important edaphic variable affecting pesticide movement. This amorphous material is primarily comprised of carbon (43-62%), hydrogen (3-5%), oxygen (29-51%), and nitrogen (1-5%) (Gieseking, 1975). Some components of OM, about 15%, are readily identifiable as proteins, enzymes, carbohydrates, alcohols, esters, aldehydes, lignins, and waxes. However, the bulk of OM is not readily identifiable because it has undergone extensive biochemical alteration. These humified materials, called humus, are grouped into three broad categories (humic acid, fulvic acid, and humin) and comprise about 85% of the total OM (Kononova, 1966; Schnitzer, 1991). From 17 to 96% of the total organic carbon in the soil is found in the humic and fulvic acid fractions and can be removed from soil by extraction with dilute NaOH

(Kononova, 1966; Weber *et al.*, 1986a; Schnitzer, 1991). These organic acids contain high molecular weight compounds in the 5000 to 100,000 range that have aromatic rings, aliphatic chains, and many ionizable carboxyl and hydroxyl groups which give rise to pH-dependent charge (Kononova, 1966; Schnitzer, 1991). The combination of both lipophilic and hydrophilic groups in OM means that it may react with pesticides of diverse chemistries. Many studies have been conducted where increased adsorption of a pesticide has been highly correlated with OM content (Weber *et al.*, 1969; Streck and Weber, 1982; Kozak *et al.*, 1983; Shea and Weber, 1983; Peter and Weber, 1985; Blumhorst *et al.*, 1990; Guo *et al.*, 1991, 1993; Chen *et al.*, 1992; Mallawatantri and Mulla, 1992). Several of these studies demonstrated a decrease in pesticide mobility with increasing OM. It has also been postulated that pesticides could bind to dissolved organic matter, thus enhancing transport through the soil, but experimental results have been inconsistent (Kan and Tomson, 1990; Pennington *et al.*, 1991).

Oxides and hydrous oxides of aluminum and iron are commonly found in southeastern soils in mineralogical forms such as hematite, goethite, gibbsite, and boehmite (Brown *et al.*, 1978; Buol *et al.*, 1989; Sposito, 1989). Depending on their oxidation state, these minerals can give rise to the distinctive red, orange, yellow, brown, and gray colors in soils. Some soils may also have relatively high amounts of manganese oxides. The charge associated with hydrous oxides is pH dependent and could be positive, neutral, or negative. At low soil pH levels, oxides tend to have a net positive charge enabling them to adsorb anionic pesticide species (Watson *et al.*, 1973).

Soil structure, the arrangement of void and solid space in a soil, may determine the degree of interaction that occurs between a pesticide and the previously mentioned soil constituents. Large channels and continuous macropore networks may be responsible for chemical movement through soil to greater depths and at higher rates and concentrations than expected. The contribution of this phenomena, termed preferential flow, to movement of pesticides has received considerable attention in the past 5 years. Laboratory studies have shown that preferential flow can occur in soils due to established macropore networks or heterogeneous textural layers (Czapar *et al.*, 1992; Radulovich *et al.*, 1992; Kung, 1993). Field studies with pesticides and tracers have also reported preferential flow as a result of macropore networks (Everts and Kanwar, 1990; Ghodrati and Jury, 1990; Jabro *et al.*, 1991; Rice *et al.*, 1991). Some field studies comparing no-till versus conventional-till practices have shown greater preferential flow in no-tillage situations. Most authors have attributed this to development of undisturbed macropore networks (Shipitalo *et al.*, 1990; Gish *et al.*, 1991; Hall *et al.*, 1991). Other studies have shown greater preferential flow with conventional tillage (Levanon *et al.*, 1993). This topic needs further attention to evaluate differences in preferential flow of surface-applied versus incorporated chemicals in conventional tillage systems. While the total quantity of pesticide that moves by way of preferential flow is probably minor (compared to that moving by advection through the soil micropores), these small quantities can be critical. Many pesticides can be accurately quantified at

concentrations in the high part per trillion to low part per billion range and the current trend in U.S. EPA is to regulate pesticides in groundwater to these low levels. Therefore, preferential movement of minute amounts of a pesticide to groundwater could jeopardize its registration status.

Soil reaction or pH affects several of the soil and pesticide parameters as previously discussed. Pesticides with ionizing characteristics, net charge of clays, organic matter, and hydrous oxides, and sorption and longevity of a pesticide can all be influenced by soil pH.

Climate. Climatic factors such as precipitation, air and soil temperature, and solar radiation affect pesticide behavior and fate. The influence of these factors will be explained in the discussion of dissipation pathways.

Overview Of Dissipation Pathways

By establishing the relevant properties of pesticides and soils it is possible to better predict what dissipation processes will most affect organic chemicals once in the environment. These interrelated processes can be classified in two general groups, transfer and transformation processes (Weber, 1991). Transfer processes move a chemical within the system but do not change the chemical structure. Transformation processes change the structure of the chemical and result in reaction products. Some of these processes are represented in Fig. 1.

Transfer Processes. Transfer processes (CS intact, Fig. 1) include (a) absorption and exudation by plants and organisms; (b) adsorption by soil

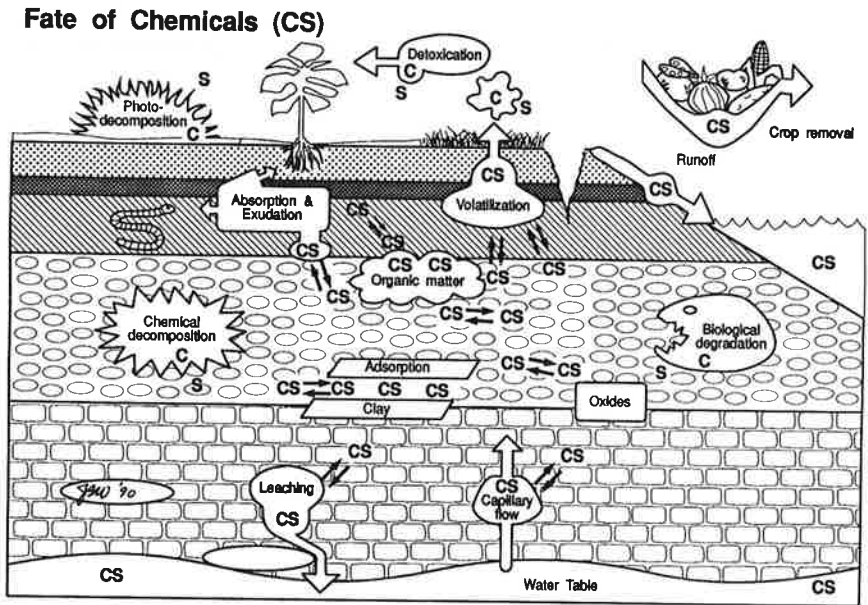


Fig. 1. Dissipation of pesticides (chemicals = CS) in the environment. Transfer processes are shown with the pesticide (CS) intact. Transformation processes are shown with C separated from S, indicating that the chemical has been degraded (from Weber, 1991).

constituents such as organic matter, clays, and hydrous oxides; (c) volatilization into gaseous forms that escape the soil; (d) movement downward with percolating water (leaching); (e) movement upward with evaporating water (capillary movement); (f) movement over the soil surface in runoff water; and (g) removal in harvested crops.

Absorption, exudation, and retention of pesticides by plants and other organisms are relatively minor processes. The amount of a mobile herbicide normally taken up and retained by plants rarely exceeds 5% of the total amount applied and even in fields with heavy weed pressure would probably not exceed 10% (Best and Weber, 1974; Weber, 1991). Best and Weber (1974) examined the dissipation of atrazine and prometryn in a 5-month study and found that from 0.6 to 4.3% of the herbicides were taken up and retained by plants. Higher herbicide concentrations occurred in plants grown in higher pH soils. The pH effect is characteristic of compounds with acidic or basic properties. Since only small quantities of pesticides are actually taken up by crop plants even smaller amounts are generally transferred into harvestable parts.

Adsorption by soil constituents such as clays, hydrous oxides, and organic matter can be a major process regulating pesticide movement (Weber, 1972, 1982a,b; Watson *et al.*, 1973; Guo *et al.*, 1991, 1993). However, the degree of interaction between pesticides and these soil constituents depends on the properties of both, as previously discussed. Adsorption is generally a more important determinant of movement for some families of chemicals; such as the quaternary nitrogen pesticides, organic arsenic and phosphoric acid pesticides, dinitroanilines and triazines, than it is for other families; such as carboxy and hydroxy acids, amides, and anilides.

Volatilization is a significant transfer pathway for many chemicals. Evidence is seen as crop injury in fields from pesticides which were applied to soils some distance away and the presence of pesticides in rainwater (Zabik and Seiber, 1993). Gaseous losses of applied pesticides is dependent upon soil moisture content and temperature (Glotfelty *et al.*, 1984, 1989; Jury, 1986; Glotfelty and Schomburg, 1989; Whang *et al.*, 1993). Up to 90% of some moderately volatile pesticides surface applied to moist soils have been lost in as little as 3 days (Glotfelty *et al.* 1984). Differences in volatilization due to tillage practices have also been reported. Whang *et al.* (1993) concluded that several pesticides had volatile losses up to four times greater in no-till versus conventional-till systems.

Leaching, along with capillary flow, is the most important transfer pathway when considering groundwater protection. Downward movement of dissolved pesticides by bulk flow with water is termed leaching. Water input is the obvious driving force, and the amount and timing of precipitation or irrigation events can influence the degree of leaching that occurs. The first precipitation event after application may be the most influential single event influencing depth of chemical movement, especially for surface applied chemicals. Light rain or irrigation after application may move surface-applied chemicals a short distance into the soil matrix and reduce the potential of rapid macropore transport from subsequent precipitation (Shipitalo *et al.*, 1990). Heavy rain

after application may limit pesticide contact with the soil matrix as the water, with dissolved pesticides, moves through large macropores (Gish *et al.*, 1991). This phenomenon is probably less important for incorporated compounds that are already mixed with the soil. The amount and rate at which a chemical leaches is dependent not only on water flow, but adsorption-desorption, volatilization, and other processes. Upward movement of pesticides with water may also occur due to the adhesive-cohesive capillary forces between water molecules and the soil matrix. In areas with high evapotranspiration rates, such as in a rapidly growing crop, net water movement may be upwards even with frequent rainfall events (Smith and Cassel, 1991). Diffusion or movement of the dissolved chemical within the solution phase is important in both cases. Nearly all of the pesticide and soil properties previously discussed can influence leaching of a pesticide. This topic has been thoroughly reviewed in texts by Goring and Hamaker (1972), Hartley and Graham-Bryce (1980), Hance (1980), Sawhney and Brown (1989), and Honeycutt and Schabaker (1994).

Because of the complexity of factors that affect pesticide movement through soil it is difficult to accurately predict the movement of a specific compound through a specific soil under any given set of conditions. The breakdown of the parent pesticide compound into metabolites further complicates matters. Metabolites have different chemical properties than the parent and may behave quite differently in the soil. Metabolites may or may not pose a greater health risk than the parent.

Numerous assumptions are made when attempting to predict pesticide movement. The flux of water through a soil is typically estimated with Darcy's law or Richards' equation, gaseous diffusion of the pesticide with Fick's law, partitioning of pesticide vapors between the soil solution and air with Henry's law, and sorption of the pesticide to soil constituents with the Freundlich equation [see Jury *et al.* (1991) for a discussion of these and other commonly used process models]. A number of other relationships or variations of established ones are also being used (Jury and Ghodrati, 1989). Each of these relationships are generalizations and may or may not be good descriptions of field situations.

Movement of an organic solute (dissolved pesticide) in an aqueous mobile phase (soil water) through a stationary solid matrix (soil) is analogous to analytical chromatography, except that the soil media is a heterogeneous mixture unlike that used in chromatographic studies (Weber, 1991). While it is not a perfect analogy because of the physical and chemical heterogeneity associated with the soil matrix, relative chromatographic principles have been applied to pesticide movement through soil. Retardation factors (R_f values), defined as the distance a solute moves through a solid medium divided by distance the solvent front moves through the solid medium, have been used to compare the mobility of different pesticides in soil. Gerber *et al.* (1970) used inclined thick-layer soil trays, Helling (1971a,b,c) used thin layer soil plates, and Weber *et al.* (1986b) used soil leaching columns to evaluate relative pesticide mobility in soils. Relative pesticide mobility data obtained with these methods has been consistent with that from other studies, with the more mobile pesticides tending to have larger R_f values and the less mobile pesticides lower R_f values (Helling, 1971a,b,c; Weber *et al.*, 1986b).

Movement of pesticides over the soil surface in runoff water as a dissipation pathway has also received considerable attention. Pesticides may move over the soil surface dissolved in the aqueous phase or sorbed to suspended sediments. This may be a major transfer pathway in situations where a heavy rainfall occurs shortly after pesticide application. A review of literature by Wauchope (1978) indicated that, for the majority of pesticides, less than 0.5% of those applied were lost in surface runoff. Losses greater than 2.0% were rare, but as much as 13.0% loss has been reported (Bailey *et al.*, 1974; Wauchope, 1978). Surface texture, slope, surface cover, and pesticide application method influence this phenomenon. Comparisons of no-till and conventional-till fields indicate that more pesticides are lost in surface runoff from bare soil, conventional-till fields (Kenimer *et al.*, 1987; Hall *et al.*, 1991). Soils with high clay contents and slow infiltrating surfaces tend to have greater pesticide runoff losses than coarse-textured soils (Hubbard *et al.*, 1989). Postemergence applications of atrazine have been shown to be lost at significantly slower rates than atrazine applied preemergence (Pantone *et al.*, 1992).

Transformation Processes. Transformation of a pesticide infers the breaking apart or disassembling of the parent molecules. These breakdown products are generally known as metabolites. A number of chemical and biochemical processes act to degrade pesticides in the soil environment. Transformation processes (C separated from S; Fig. 1) include (a) alteration by target organisms (plants, insects, etc.); (b) photodecomposition by sunlight; (c) chemical decomposition by hydrolysis, oxidation, or reduction reactions; and (d) biological degradation by microorganisms.

Pesticides that are taken up by plants and other target organisms may be biochemically altered by enzymes such as multifunctional oxidases or cytochrome P450s. These alterations may render the pesticide more toxic or less toxic to the target organism. Hydrolysis, conjugation, dehalogenation, and demethylation are some of the types of reactions reported to alter pesticides in plants (Gronwald, 1989; McFadden *et al.*, 1989; Owen, 1989; Dean *et al.*, 1990). As previously stated, most pesticides are taken into plants in very low quantities, so biochemical degradation of pesticides by plants is relatively minor except where plant populations and uptake are extremely high.

Photodecomposition is the nonbiological degradation of pesticides involving sunlight. This subject has been discussed by Crosby (1976) and Miller *et al.* (1989). Radiant energy from the sun can energize a variety of reactions such as oxidation, reduction, hydrolysis, and isomerization. Ultraviolet light, with wavelengths of 200 to 420 nm, is responsible for most photodegradation. This process may be of major or minor importance, depending on susceptibility of the chemical, method of application, and length of exposure. Degradation losses may be only a few percent for soil-incorporated chemicals to over 90% for surface-applied or aquatic compounds.

Nonbiological chemical decomposition may be a major degradation pathway for some chemicals through oxidation, reduction, and hydrolysis reactions. Unfortunately, many studies evaluating pesticide degradation make no attempt to separate degradation caused by biotic and abiotic processes. This makes it difficult to establish the importance of nonbiological

chemical alteration for many compounds (Wolfe *et al.*, 1989). Kearney and Kaufman (1975, 1976, 1988) and Sawhney and Brown (1989) have reviewed this topic and indicated that hydrolysis may be the most common abiotic degradation process. Studies with atrazine and other *s*-triazines have shown hydrolysis to be the major abiotic degradation process (Li and Felbeck, 1972; Plust *et al.*, 1981; Ghadiri *et al.*, 1984). Acid catalysis under low pH conditions increases the rate of the reaction. Carbamates like chlorpropham, esters of acid pesticides like 2,4-D, and sulfonylurea pesticides like chlorsulfuron are also hydrolyzed in soils. Dinitroaniline pesticides have been shown to undergo reduction in anaerobic conditions and some bipyridyl, diphenylether, and phenol compounds also undergo reduction (Probst *et al.*, 1975; Calderbank and Slade, 1976; Kaufman, 1976; Matsunaka, 1976).

Biological degradation by soil organisms, predominately microflora and microfauna, is an important degradation pathway for most pesticides. The subject has been reviewed by Kearney and Kaufman (1975, 1976, 1988), Hill and Wright (1978), Tortensson (1980), and Alexander and Scow (1989). A wide range of biologically mediated reactions have been reported for many different types of pesticides. These include oxidation of phenoxyalkanoic acids, triazines, substituted ureas, carbamates, benzoic acids, and phenols; reduction of dinitroanilines, diphenylethers, and phenols; dealkylation of phenylureas, carbamates, and dinitroanilines; decarboxylation of benzoic acids, picolinic acids, and phenoxyalkanoics; hydrolysis of phenylcarbamates, substituted ureas, and organoarsenicals; and dehalogenation of chlorinated aliphatics and diphenylethers. Reactions may be mediated by indigenous enzymes, exogenous enzymes, or both. In general, factors affecting the microbial biomass affect the extent and rate of biological degradation of pesticides. Moisture content, temperature, pH, and oxidizable organic carbon sources are all important parameters. Most conducive conditions for microbial growth and pesticide decay are found near the soil surface. Biotic degradation of pesticides typically decreases with soil depth (Walker and Welch, 1989; Walker *et al.*, 1989).

Most pesticide-environmental research concentrates on fate and behavior of the parent pesticide molecules. It is important to remember that metabolites of many pesticides can display markedly different mobility and toxicity characteristics from the parent compound. Evaluation of major metabolite behavior and fate is now routinely conducted for most compounds.

TYPES OF METHODS FOR EVALUATING THREAT TO GROUNDWATER FROM PESTICIDES

Many methods for evaluating the mobility of pesticides in soils and the subsequent threat to groundwater have been proposed. Approaches to the problem are diverse and range from simple to very complex. Most evaluation models or systems can be classified in two broad groups, process-based, and index-based. Process-based methods are used to estimate quantitative rate, depth, and concentration information about chemical movement in soil.

Index-based methods generally provide relative, qualitative estimations of chemical movement or groundwater susceptibility. Bonazountas (1983) and Jury and Ghodrati (1989) detail aspects of modeling pesticide movement in soils. The time and cost involved in generating model validation data have contributed to the scarcity of adequate data sets for rigorous validation of current systems. In some cases one model is "validated" with another model even when neither one has been proven to accurately predict pesticide transport in the field. This data limitation greatly hampers the development of reasonably accurate methods to assess the threat to groundwater from pesticides and their metabolites.

Process-Based Methods

Process-based models often utilize traditional differential equations, complex algorithms, and/or compartmental subroutines to predict the concentration and depth a chemical will move. These models are typically developed as simulation models. A simulation model attempts to represent pesticide transformation and transport according to specific quantitative process laws (Swann and Eschenroeder, 1983; Jury and Ghodrati, 1989). Some are compartmentalized, breaking the soil continuum into discrete zones where movement is modeled in each section. Rainfall infiltration, saturated and unsaturated water flow, gas phase transport, hydrodynamic dispersion, microbial population densities, and pesticide decay rates are some of the parameters typically modeled in sub-routines of a process-based simulation model. They are frequently used to evaluate possible movement of chemical spills or leaks or as research aids in contaminant transport theory. The GLEAMS (groundwater loading effects of agricultural systems) and PRZM (pesticide root zone model) are examples of process-based simulation models currently used to predict pesticide transport (Carsel *et al.*, 1984; Leonard *et al.*, 1987; Davis *et al.* 1990). Both GLEAMS and PRZM utilize a large number of input variables (60 or more) to determine chemical movement. Numerical solutions for specific chemical concentration and distribution with depth are calculated. Unfortunately, because they require a large number of parameters, they cannot be used without significant experimentation to calibrate unknown values. Many input parameters are routinely estimated with limited experimental data so validity of the component process models is never established (Jury and Ghodrati, 1989). Tests of these models against field data, and one another, show that they predict tracer and pesticide movement with varying degrees of accuracy (Nelson, 1991; Sichani *et al.*, 1991; Smith *et al.*, 1991; Mueller *et al.*, 1992; Parrish *et al.*, 1992). Predictions are often within two to three orders of magnitude of actual field data. GLEAMS and PRZM both tend to predict pesticide movement and concentration less accurately with increasing soil depth and increasing time after pesticide application.

Index-Based Methods

Index-based models utilize key soil and/or pesticide factors to qualitatively evaluate and rank a pesticide's potential to reach groundwater. Knowledge and experience of the researchers, evaluation of existing data sets, and

characterization or ranking of key parameters are typically the basis for an index-based system. They are often developed as screening tools to group large numbers of soils or pesticides into similar mobility categories. These systems yield relative differences instead of numerical solutions for chemical depth and concentration. Fewer input variables are required in index-based screening models but many of the same ones incorporated in process-based simulation models are used. Behavior assessment models (Jury *et al.*, 1983, 1984, 1987), physical property index systems (Gustafson, 1989), thin-layer chromatography tests (Helling, 1971a,b,c), and parameter ranking systems (Aller *et al.*, 1985; Rao *et al.*, 1985; Nielsen and Lee, 1987; Weber, 1990, 1991b; Kovach *et al.*, 1992; Warren and Weber, 1992, 1993, 1994; Weber and Warren, 1993) are examples.

The DRASTIC system typifies the index-based ranking system approach of determining groundwater pollution potential and has been widely used (Aller *et al.*, 1985). DRASTIC was developed by the National Ground Water Association for the U.S. EPA to give a relative measure of aquifer vulnerability to contamination. Seven factors affecting chemical movement to groundwater, along with a set of rating and weight values, have been established by expert opinion. The factors considered to be the best predictor of aquifer vulnerability to chemicals are D = depth to water table, R = net aquifer recharge, A = aquifer media, S = soil media, T = topography or slope, I = impact of vadose zone, and C = hydraulic conductivity of the aquifer. Relative weights developed for use with agricultural pesticides are used in conjunction with ratings for each of the seven factors to give a single index of contamination potential. The DRASTIC pollution potential index (Dppi) is calculated as

$$Dppi = D_R D_W + R_R R_W + A_R A_W + S_R S_W + T_R T_W + I_R I_W + C_R C_W \quad \text{Eq. (1)}$$

where R denotes rating, W the weight, and D, R, A, S, T, I, C correspond to the seven factors defined above. Calculated scores are only used as a relative indication of vulnerability and do not signify contaminant depths or concentrations. DRASTIC scores calculated for each county in the U.S. using the pesticide weights (typical range 69 to 245) identify the Southeastern Coastal Plain region of the U.S., including the peanut growing areas, as vulnerable to groundwater contamination (Nielsen and Lee, 1987).

DRASTIC, by itself, includes only soil and hydrogeologic factors and does not include pesticide information of any kind. Nielsen and Lee (1987) combined DRASTIC scores and pesticide use estimates in an attempt to more clearly delineate areas in the U.S. most vulnerable to groundwater contamination. The Southeastern Coastal Plain region was again identified as one of the most likely areas for contamination.

By not accounting for use rates and differences in pesticide chemistry, DRASTIC, used alone, assumes all chemicals behave the same in a given soil. The Nielsen and Lee method accounts only for average use patterns and assumes all pesticides at a given rate behave the same in a given soil. This could lead to over or under estimation of groundwater contamination potential for specific chemicals.

DRASTIC also excludes soil organic matter content as a factor affecting

pesticide movement. The importance of organic matter in controlling pesticide mobility has been well documented (Harris and Warren, 1964; Weber *et al.*, 1969; Streck and Weber, 1982; Kozak *et al.*, 1983; Shea and Weber, 1983; Peter and Weber, 1985; Blumhorst *et al.*, 1990; Guo *et al.*, 1991, 1993; Mallawatantri and Mulla, 1992). Studies comparing DRASTIC values to field data and predicted values from other systems have found little correlation (Banton and Villeneuve, 1989; Warren and Weber, 1992; Koterba *et al.*, 1993).

A PESTICIDE AND SOIL RANKING SYSTEM (PSRS) FOR EVALUATING THE RELATIVE POTENTIAL OF NONPOINT SOURCE GROUNDWATER CONTAMINATION

The most recent version of a ranking system based on work by Warren and Weber (Weber, 1990; Warren and Weber, 1992, 1993, 1994; Weber and Warren, 1993; McLaughlin *et al.* 1994) is explained and presented here. The system may be used to evaluate the relative nonpoint source groundwater contamination potential of pesticide and soil combinations. It is proposed for use as a simple screening tool that could aid crop producers, extension agents, farm chemical dealers, or consultants in selecting pesticides with the least probability of reaching groundwater for a particular soil. Seven pieces of information are utilized: (a) texture of the soil profile to 91 cm (3 ft), (b) acidity or pH of the soil profile to 91 cm, (c) percent organic matter or humic matter in the plow layer, (d) longevity or half-life ($T_{1/2}$) of the pesticide, (e) rate of pesticide application, (f) fraction of applied pesticide that reaches the soil, and (g) index of pesticide retention by soil or K_{oc} . The leaching potential index of any soil is computed using the first three pieces of information and the leaching potential index of any nonfumigant pesticide is computed using the last four pieces of information. Pesticide and soil leaching potential indices are calculated individually, then used together for systematic comparison of pesticide and soil combinations.

Soil Leaching Potential (SLP)

The leaching potential of a soil depends on many parameters but the ones that are of greatest importance to pesticide movement are organic matter (OM), texture, and acidity or pH (Weber, 1987, 1990, 1991; Blumhorst *et al.*, 1990). Soil leaching potential values, representing the relative ease with which a soil allows transport of pesticides, are calculated using data on the these three soil properties and a series of weight and rating tables (Tables 1-3). The relative importance of each differs with each pesticide but, for this system, weight factors of 10 for OM, 10 for texture, and 3 for acidity (on a scale of 10) have been assigned to these three key soil properties. Weight factors indicate the relative importance of each property in determining a soils ability to influence pesticide mobility. This system considers OM and texture to be highly influential with acidity less important. Rating factors

Table 1. Rating and weight coefficients for the soil texture value (Tv).

Soil texture	Rating	Weight factor	Tv
Coarse sand, sand, fine sand, very fine sand, loamy coarse sand, loamy sand, loamy fine sand	10	10	100
Loamy very fine sand, coarse sandy loam, sandy loam, fine sandy loam	8	10	80
Very fine sandy loam, loam, silt loam, silt	6	10	60
Sandy clay loam, clay loam, silty clay loam	4	10	40
Sandy clay, silty clay	2	10	20
Clay or muck	1	10	10

Table 2. Rating and weight coefficients for the soil acidity value (Av).

pH	Rating	Weight factor	Av
>7	10	3	30
6-7	6	3	18
5-6	3	3	9
<5	1	3	3

Table 3. Rating and weight coefficients for the soil organic matter (or humic matter) value (OMv).

OM	HM	Rating	Weight factor	OMv
----- % -----				
<2	<1	10	10	100
2-4	1-2	6	10	60
4-6	2-4	3	10	30
>6	>4	1	10	10

(also on a scale of 10) are used to relate the ranges within each of the three properties to pesticide leaching. For example, sandy soil textures, low OM levels, and high soil pH's would receive the highest ratings.

The relative amounts of sand, silt, and clay in a soil (soil texture) is largely responsible for governing how readily water can move through it. The movement of dissolved chemicals (those that travel with the aqueous phase) can be modeled by characterizing water movement in soil. Water can be expected to move most readily through sand, less readily through loams, and least in clays and muck. Sands typically have the lowest water-holding capacity with clays holding the greatest amounts. Water in excess of a soil's

water-holding capacity is available for leaching (Brady, 1984; Smith and Cassel, 1991). Sandy soils typically have a large macropore network allowing rapid infiltration and percolation, especially after significant precipitation events. This implies that leaching would begin sooner and proceed more rapidly in a sandy soil than in a clayey soil. Besides affecting water movement, soil texture is also directly related to clay content of a soil. As previously discussed, the clay fraction of soils is chemically active and can influence pesticide movement through adsorption reactions. Sandy soils have lower clay contents and are less likely to participate in adsorption reactions than loamy or clayey textured soils. A rating scheme for various soil textures, ranging from 1 to 10, is shown in Table 1. The textural component (T_v) of the SLP is obtained by computing a weighted average rating for texture through the soil profile to 91 cm depth. It is assumed that the properties of the upper 91 cm (3 ft) rooting zone will, in large part, determine a pesticide's threat to groundwater. Those compounds that readily move below this zone during a typical growing season would be expected to have a high probability of reaching groundwater. The weighted average rating is multiplied by the weight factor for texture, 10, and the product is used as the textural component of the SLP.

Soil acidity (pH) affects the mobility and/or the degradation of pesticides to varying degrees depending on the chemical and biological properties of the compound (Weber, 1987). The mobility of pesticides with acidic or basic properties often increases as soil pH increases (Weber, 1982b). Chemical degradation of many pesticides is generally greater under acidic conditions (Weber, 1990). It is therefore assumed that low soil pH conditions would lessen the possibility of pesticide movement to groundwater. A rating scheme for various soil pH levels is given in Table 2. The acidity component (A_v) of the SLP is obtained by computing a weighted average rating for pH through the soil profile to 91 cm depth. The weighted average rating is multiplied by the weight factor, 3, and the product is used as the acidity component of the SLP.

Organic matter (OM) adsorbs pesticides to varying degrees depending on the chemical properties of the compound and is considered the single most important determinant of organic chemical movement in soil. The relationship between pesticide mobility and soil OM is well documented (Harris and Warren, 1964; Talbert and Fletchall, 1965; Weber *et al.*, 1969; Weber 1972, 1990; Streck and Weber, 1982; Kozak *et al.*, 1983; Shea and Weber, 1983; Peter and Weber, 1985; Chiou, 1989; Blumhorst *et al.*, 1990; Guo *et al.*, 1991, 1993; Chen *et al.*, 1992; Mallawatantri and Mulla, 1992). Adsorption increases and mobility decreases as OM content of the soil increases. A rating scheme for various OM (or humic matter) levels of the surface horizon (plow layer) is given in Table 3. Only the surface zone is evaluated because data are readily available on soil OM levels in the plow layer. Further, soil testing laboratories typically report OM or humic matter as part of a routine analysis of surface soil samples submitted by growers. Organic matter levels are generally greatest in the surface zone of Southeastern Coastal Plain soils. The organic matter component (OM_v) of the SLP is the product of the rating

and the weight factor, 10.

Once values are determined for texture, acidity, and OM the soil leaching potential value (SLPv) for a given soil is calculated as

$$\text{SLPv} = \text{Tv} + \text{Av} + \text{OMv} \quad \text{Eq. (2)}$$

where texture value (Tv) is the product of the depth-weighted average texture rating to 91 cm depth and the texture weight factor, 10; acidity value (Av) is the product of the of the depth-weighted average acidity rating to 91 cm depth and the acidity weight factor, 3; and OM value (OMv) is the product of the OM rating for the surface layer and the OM weight factor, 10. The lowest possible SLPv is 23 and the highest 230. For ease of interpretation, all resulting SLPv's are converted to an SLP index (SLPi), on a scale of 0 to 100, by normalization:

$$\text{SLPi} = [(\text{SLPv} - 23)/(230 - 23)] \times 100 \quad \text{Eq. (3)}$$

The relationship between SLPv and SLPi may be expressed as a linear equation which allows rapid conversion between SLPv and SLPi:

$$\text{SLPi} = (0.4832) (\text{SLPv}) - 11.1069, R^2 = 1.000 \quad \text{Eq. (4)}$$

A soil receiving a higher SLPi is presumed more susceptible to leaching or less likely to retard pesticide movement than a soil with a lower SLPi. See Table 4 for a sample SLPi calculation for a Norfolk loamy fine sand (fine-loamy, siliceous, thermic Typic Kandiudult). Table 5 lists the Tv, Av, OMv, SLPv, and SLPi values, as well as two surface horizon properties, for 82 soils prevalent in the southeastern U.S. peanut-growing region. The SLPi may be readily calculated for any soil in the Southeastern Coastal Plain region using the method described. Information needed to calculate the SLPi for a given soil can be found in local county soil survey reports, which are published by the National Cooperative Soil Survey, or obtained from local Soil Conservation Service (now the Natural Resources Conservation Service) offices.

Pesticide Leaching Potential (PLP)

The leaching potential of a pesticide depends on its physicochemical properties including ionization, water solubility, volatility, reactive groups, and longevity in soil, water, and air as previously discussed. The chemical properties of a pesticide determine its reactivity with, and mobility through, soil constituents. This is estimated from sorption isotherm studies which provide K_f values (from Freundlich isotherms) or K_d values (from pesticide sorbed/pesticide in solution distribution studies). A K_f value is an estimate of the sorption capacity of a given pesticide for a given soil at an equilibrium concentration of 1 nmol/mL, as obtained using the Freundlich equation

$$x/m = KC^{1/n} \quad \text{Eq. (5)}$$

where x/m is the amount of pesticide sorbed (nmol/g), K is a constant, C is the equilibrium pesticide concentration in solution (nmol/mL), and $1/n$ is a constant. K_f values may be obtained from sorption isotherm studies by

Table 4. Sample calculation of the soil leaching potential index (SLPi) for a Norfolk loamy fine sand (fine-loamy, siliceous, thermic Typic Kandiudult).

Norfolk pedon description from soil survey of Bladen County, NC:

Ap 0 to 18 cm; grayish brown (10YR 5/2) loamy fine sand; weak fine granular structure; very friable; common fine roots; medium acid; clear smooth boundary.

E 18 to 38 cm; light yellowish brown (2.5 Y 6/4) loamy fine sand; weak fine granular structure; very friable; few fine roots; very strongly acid; clear smooth boundary.

BE 38 to 46 cm; yellowish brown (10YR 5/4) fine sandy loam; weak fine subangular blocky structure; friable; few fine roots; few faint clay bridges between sand grains; very strongly acid; clear smooth boundary.

Bt1 46 to 89 cm; yellowish brown (10YR 5/8) sandy clay loam; weak medium subangular blocky structure; friable; slightly sticky; few fine pores; few distinct clay films on faces of peds; very strongly acid; gradual wavy boundary.

Bt2 89 to 127 cm; yellowish brown (10YR 5/8) sandy clay loam; few medium distinct yellowish red (5YR 4/8) mottles; weak medium subangular blocky structure; friable; slightly sticky; few fine pores; few distinct clay films on faces of peds; very strongly acid; gradual wavy boundary.

Texture component value (Tv) calculation:

The Tv is calculated using the soil pedon description and Table 1. Each horizon description includes the horizon thickness and USDA texture. In this example, both the Ap and E horizons have a loamy fine sand (lfs) texture and combined thickness of 38 cm. The BE horizon has a fine sandy loam (fsl) texture and is 8 cm thick. The Bt1 and Bt2 horizons have a sandy clay loam (scl) texture and combined thickness to 91 cm (the depth limit for calculation) of 45 cm. A weighted average rating is calculated by multiplying the combined thickness of all layers with the same texture by the appropriate rating for that texture (Table 1), summing the values, then dividing by 91 cm. The weighted average rating is then multiplied by the weight factor for texture, 10, and the product is used as the texture component (Tv) of the SLPv. These steps are summarized below for the Norfolk loamy fine sand.

<u>Total layer thickness</u>	<u>Texture</u>	<u>Table 1 rating</u>	<u>Weighted average rating</u>	<u>Weight factor</u>	<u>Product</u>
38 cm	lfs	10	$\frac{[(38 \times 10) + (8 \times 8) + (45 \times 4)]}{91}$	10	= 68
8 cm	fsl	8			
45 cm	scl	4			
			= 6.8	x	= 68

Table 4 (Continued)

Acidity component value (Av) calculation:

The Av is calculated in the same manner as the texture component, except the acidity information from the pedon description and Table 2 are used. Each connotative acidity description refers to a specific pH range. For example, medium acid = 5.6-6.0 and very strongly acid = 4.5-5.0. Definitions for pH descriptive terms may be found in county soil surveys or obtained from the Soil Conservation Service. The mean pH of each range is used for calculation purposes. In the Norfolk pedon, the Ap horizon is medium acid (ca. pH = 5.8) and 18 cm thick. All other horizons, to 91 cm, are very strongly acid (ca. pH = 4.8) and have a combined thickness of 73 cm. This information and Table 2 is used to calculate a weighted average acidity rating in the same manner as for texture. This is multiplied by the weight factor for acidity, 3, and the product used as the acidity component (Av) of the SLP value. These steps are summarized below for the Norfolk loamy fine sand.

Total layer thickness	pH	Table 2 rating	Weighted average rating	Weight factor	Product
18 cm	5.8	3	$\frac{[(18 \times 3) + (73 \times 1)]}{91} = 1.4$	x 3	= 4
73 cm	4.8	1			

Organic matter or humic matter component value (OMv) calculation:

A weighted average rating to 91 cm depth is not required for the OM component. Only the percent OM in the surface horizon is used. OM percentages for the surface horizon of soils are found in the chemical and physical properties table of county soil surveys or can be obtained from Soil Conservation Service offices. Table 3 lists the ratings for soil OM levels. The rating is multiplied by the weight factor for OM, 10, and the product used as the OM component (OMv) of the SLP value. These steps are summarized below for the Norfolk loamy fine sand.

Total layer thickness	% OM	Mean % OM	Table 3 rating	Weight factor	Product
18 cm	0.5-2.0	1.25	10	x 10	= 100

SLP value (SLPv) and SLP index (SLPi) calculation:

The SLPv is the sum of the Tv, Av, and OMv terms. The SLPv is then normalized to a 0 to 100 scale. This normalized value is the SLPi and is the one number used to represent the leaching potential of a soil.

$$\text{SLPv} = \text{Tv} + \text{Av} + \text{OMv} = 68 + 4 + 100 = 172$$

$$\text{SLPi} = (0.4832) (\text{SLPv}) - 11.1069 = (0.4832) (172) - 11.1069 = 72$$

Table 5. Soil Leaching Potential (SLP) component values, indices, typical surface texture, and average surface OM (%) for 82 soils commonly used in peanut production in the Southeastern U.S.

Soil series	SLP component values			SLP		Surface texture	Surface mean OM %
	Tv (0-100)	Av (0-30)	Omv (0-100)	Value (23-230)	Index (0-100)		
Ailey	87	9	100	196	84	s	0.50
Alaga	100	9	100	209	90	fs	0.75
Albany	100	6	100	206	88	s	1.50
Alpin	100	4	100	204	87	fs	1.00
Altavista	52	5	100	157	65	fs	1.75
Augusta	45	9	100	154	63	l	1.25
Autryville	93	9	100	202	86	ls	0.75
Aycock	45	9	60	114	44	vfsl	2.50
Baymeade	93	4	100	197	84	fs	0.75
Bigbee	100	9	100	209	90	ls	1.25
Blanton	100	9	100	209	90	fs	1.25
Bojac	87	10	100	197	84	ls	0.75
Bonifay	100	6	100	206	88	fs	2.00
Bonneau	98	9	100	207	89	ls	1.25
Butters	93	6	100	199	85	ls	1.25
Cainhoy	100	4	100	204	87	s	0.50
Carnegie	32	9	100	141	57	sl	1.50
Caroline	48	7	100	155	64	fsl	1.25
Chapanoke	50	9	100	159	66	sil	1.25
Chipley	100	9	60	169	71	s	3.50
Conetoe	93	8	100	201	86	ls	1.25
Cowarts	75	9	100	184	78	ls	0.50
Craven	32	4	100	136	55	sil	1.25
Dogue	42	5	100	147	60	sl	0.75
Dothan	55	9	100	164	68	sl	0.50
Dragston	85	4	100	189	80	lfs	0.75
Dunbar	42	4	60	106	40	fsl	3.00
Duplin	33	6	100	139	56	sl	1.25
Echaw	100	5	100	205	88	ls	0.50
Emporia	60	4	100	164	68	fsl	1.75
Esto	27	4	100	131	52	ls	0.50
Exum	47	5	100	152	62	vfsl	1.25
Faceville	32	9	100	141	57	fsl	1.25
Foreston	85	5	100	190	81	ls	1.25
Freemanville	38	10	100	148	60	sl	0.50
Fuquay	97	8	100	205	88	lfs	1.25
Goldsboro	55	5	100	160	66	sl	1.25
Greenville	40	9	100	149	61	sl	0.75
Gritney	30	5	60	95	35	fsl	2.50
Johns	62	9	100	171	72	fsl	1.25
Kalmia	70	4	100	174	73	lfs	1.25
Kenansville	93	9	100	202	86	s	1.25
Lakeland	100	9	100	209	90	s	0.50
Leefield	98	11	100	209	90	fs	1.50
Lenoir	25	8	60	93	34	l	3.00
Lucy	95	8	100	203	87	ls	0.75
Lynchburg	48	5	60	113	43	fsl	2.75
Malbis	57	9	100	166	69	sl	0.75
Marlboro	40	9	100	149	61	ls	1.25
Miccosukee	50	8	30	88	31	fsl	4.00

Table 5 (Continued)

Soil series	SLP component values			SLP		Surface texture	Surface mean OM %
	Tv (0-100)	Av (0-30)	Omv (0-100)	Value (23-230)	Index (0-100)		
Munden	90	9	100	199	85	ls	0.75
Myatt	57	4	100	161	67	fsl	1.25
Nahunta	45	5	60	110	42	vfsi	3.00
Nankin	30	4	100	134	54	sl	0.75
Nansemond	83	4	100	187	79	fsl	1.50
Nimmo	88	4	100	192	82	ls	1.50
Norfolk	68	4	100	172	72	lfs	1.25
Ocilla	83	8	100	191	81	lfs	1.50
Onslow	72	11	100	183	77	lfs	1.25
Orangeburg	58	9	100	167	70	ls	0.75
Pactolus	100	9	100	209	90	fs	1.25
Peawick	33	4	100	137	55	sil	1.25
RedBay	63	3	100	166	69	ls	1.00
Rumford	92	9	100	201	86	lfs	0.75
Seabrook	100	9	100	209	90	s	1.25
Shellbluff	40	8	100	148	60	cl	1.75
Slagel	55	4	100	159	66	fsl	1.25
State	53	6	100	159	66	lfs	0.50
Suffolk	68	7	100	175	73	ls	0.75
Tarboro	100	9	100	209	90	s	0.75
Tetotum	62	8	100	170	71	fsl	1.25
Tifton	63	7	100	170	71	ls	0.50
Tomahawk	93	4	100	197	84	s	1.75
Troup	100	9	100	209	90	fs	0.50
Uchee	87	9	100	196	84	ls	0.75
Valhalla	95	9	100	204	87	fs	0.75
Varina	73	4	100	177	74	ls	1.25
Wagram	87	9	100	196	84	s	1.25
Wando	100	18	100	218	70	fs	0.50
Wickham	48	7	100	155	64	fsl	1.25
Yamassee	52	6	60	118	46	fsl	2.25
Yeopim	53	4	100	157	65	l	1.25

% of soils with SLP indices 75 - 100 = 45

% of soils with SLP indices 50 - 75 = 45

% of soils with SLP indices 25 - 50 = 10

% of soils with SLP indices 0 - 25 = 0

^aAll data used to calculate SLPi's and to determine which soils are used for peanut production was obtained from selected SCS/USDA/USGS county soil surveys from each of the following states: Virginia, North Carolina, South Carolina, Georgia, Florida, and Alabama. USDA textural abbreviations: v = very, f = fine, s = sand, c = clay, l = loam, si = silt.

equilibrating selected soils (1 to 10 g) with varying aqueous concentrations (0.1, 1, 5, 10, 50, 100 nmol/mL) of a given pesticide, plotting the logarithm of the amount sorbed ($\log x/m$) versus the logarithm of the equilibrium concentration ($\log C$) and reading the K_f value ($\log x/m$) off of the plot at an equilibrium concentration of 1 nmol/mL ($C = 1$ nmol/mL).

K_d values may be obtained by equilibrating the concentration of pesticide normally applied in the field to soil in a saturated condition, separating the water from the soil, and determining the concentration of pesticide in the solution. The distribution coefficient (K_d) is calculated by dividing the amount sorbed by the soil (nmol/g) by the amount of pesticide in solution (nmol/mL). K_f and K_d values are normally divided by the fraction of organic carbon in the soil to yield K_{oc} values which are reported in the literature (Kenaga, 1980; Kenaga and Goring, 1980; Wauchope, 1988; Hornsby and Augustijn-Beckers, 1991; Weber, 1991a, 1994; Weber and Warren, 1993). A K_{oc} value may be thought of as the ratio of sorbed pesticide/weight of soil organic carbon to the amount of pesticide in soil solution at equilibrium and reflects the tendency of a pesticide to sorb with the soil matrix through one or several sorption processes (Lyman, 1990). Units associated with these K values (K_f , K_d , K_{oc}), which are commonly not specified, are mL/g or L/kg. The relationship between pesticide movement through soil and K_{oc} is inverse in that chemicals with small K_{oc} values are more mobile and chemicals with large K_{oc} values are less mobile (Weber, 1991a).

The length of time that a parent pesticide compound is present to be leached is estimated by $T_{1/2}$ values (decay kinetics usually assumed to be first order), as determined from laboratory studies in air, soil, and water at various pH levels, and in the field at several locations. The $T_{1/2}$ value is a composite index of some of the major transformation processes for a given pesticide (Fig. 1).

The application factors, rate and fraction of applied pesticide reaching the soil are also important. Differing amounts of each pesticide active ingredient (ai) are required to control the target weeds, insects, or diseases. All else being equal, a higher application rate (R) increases the chances of leaching since more pesticide is available to be leached. Pesticides may be preplant incorporated into the soil, applied preemergence to the soil surface, or applied post-over-top or post-directed to growing plants. In order to leach through the soil, a pesticide first has to reach the soil. Those applied to plants may be absorbed by the plant or degraded by sunlight. Direct soil incorporation provides the greatest opportunity for leaching because all of the pesticide is placed directly in the soil. For foliar applications the fraction (F) of applied pesticide not intercepted by crop plants can be generally estimated as:

$$F = 1 - (W_c / W_r) \quad \text{Eq. (6)}$$

where canopy width (W_c) is the width of the crop foliage canopy at the time of application and row width (W_r) is the row spacing used at planting. Measurements for both W_c and W_r must be in the same units. Note that this is an idealized estimation. Assumptions are that any foliar applied pesticide

is uniformly distributed over the row and inter-row area, pesticide hitting the canopy remains there, and that the canopy completely covers the soil underneath. Weed canopy interception is not accounted for in the estimation.

The $T_{1/2}$, K_{oc} , and application parameters described above are used to calculate the pesticide leaching potential value (PLPv) of a pesticide by:

$$PLPv = T_{1/2} \times R \times F / K_{oc} \quad \text{Eq. (7)}$$

where $T_{1/2}$ is the pesticide half-life measured in days, R is the application rate measured in kg ai/ha, F is the fraction of pesticide reaching the soil during application which may be estimated using Eq. (6), and K_{oc} is the pesticide soil sorption coefficient. The equation is arranged to reflect the assumptions that the potential for a pesticide to leach through soil to groundwater is directly proportional to $T_{1/2}$, R, and F and inversely proportional to K_{oc} . PLPv's can range over several orders of magnitude depending on the pesticide. Nearly all lie within 0.0001 and 1000 which are defined as the lower and upper limits, respectively. For ease in interpretation, all PLPv's are converted to a PLP index (PLPi), scaled from 0 to 100, by normalization:

$$PLPi = \frac{(\log PLPv - \log 0.0001)}{(\log 1000 - \log 0.0001)} \times 100 \quad \text{Eq. (8)}$$

The relationship between PLPv and PLPi may be expressed as a linear equation which allows rapid conversion between PLPv and PLPi:

$$PLPi = (14.2857) (\log PLPv) + 57.1429, R^2 = 1.000 \quad \text{Eq. (9)}$$

A pesticide receiving a higher PLPi is presumed to be more likely to leach than one with a lower PLPi. While the K_{oc} and $T_{1/2}$ values for each pesticide can be provided in tables (Tables 6 and 7), the R and F values will be dependent upon specific application conditions. They may change from field to field or in the same field over time. The same pesticide may be applied at varying rates depending on the crop, pest pressure, soil type, and application method. Because crop canopies cover more of the soil as they grow the amount of a foliarly applied pesticide reaching the soil may also change. The R and F factors must be determined by the grower. PLP parameters and indices for pesticides commonly used in peanut production are presented in Tables 6 and 7. Indices for the 40 pesticides were calculated using average rates of application, commonly reported K_{oc} and $T_{1/2}$ values, and estimated fractions hitting the soil (1.00 for soil, 0.90 for granular over row, and 0.50 for foliar applications). They are presented only to demonstrate the system and show typical PLPi's for a variety of compounds. More than one PLPi may be given for an ionizing pesticide if there is adequate information about its behavior at high and low soil pH levels. K_{oc} and $T_{1/2}$ values may significantly change for ionizing compounds with changes in soil pH. To properly use this system the PLPi should be calculated for specific conditions of use.

A PLPi is not calculated for fumigant pesticides, but they are assumed to have a low probability of contaminating groundwater. Even though most fumigants are used at high application rates, their vapor pressures are so

Table 6. Pesticide leaching potential (PLP) parameters, values, and indices for herbicides commonly used in peanut production.*

Common name	Trade name [®]	Applic. method	K _{oc}	T _{1/2} days	Avg rate kg ai/ha	Soil frac. (F) (0-1)	PLP	
							Value	Index
Acifluorfen	Blazer	Post	100	20	0.49	0.5	0.049	38
Alachlor	Lasso	PPI	170	20	2.80	1.0	0.330	50
		Pre	170	20	2.80	1.0	0.330	50
		Crack	170	20	2.80	1.0	0.330	50
Benfen	Balan	PPI	10000	40	1.47	1.0	0.006	25
Bentazon	Basagran	Post	30	20	0.98	0.5	0.327	50
Chlorimuron	Classic	Post	110	30	0.009	0.5	0.001	14
2,4-DB	Butyrac 200	Post	100	14	0.25	0.5	0.018	32
Ethalfuralin	Sonalan	PPI	10000	60	0.73	1.0	0.004	23
Fenoxaprop	Bugle	Post	100	9	0.17	0.5	0.008	27
Imazethapyr	Pursuit	Post	20	90	0.071	0.5	0.159	46
		PPI	20	90	0.036	1.0	0.162	46
Metolachlor	Dual	PPI	150	40	2.52	1.0	0.672	55
		Pre	150	40	2.52	1.0	0.672	55
		Crack	150	40	2.52	1.0	0.672	55
		Layby	150	40	2.52	1.0	0.672	55
Paraquat	Starfire	Crack	1.0E+6	500	0.15	1.0	<0.0001	<1
		Post	1.0E+6	500	0.15	0.5	<0.0001	<1
Pendimethalin	Prowl	PPI	10000	90	0.98	1.0	0.009	28
Sethoxydim	Poast (+)	Post pH7	100	5	0.26	0.5	0.007	24
		Post pH4	600	5	0.26	0.5	0.001	14
Vernolate	Vernam	PPI	260	7	2.58	1.0	0.069	41

Table 6 (Continued)

Common name	Trade name [®]	Applic. method	K _{oc}	T _{1/2} days	Avg rate kg ai/ha	Soil frac. (F) (0-1)	PLP	
							Value	Index
Tank Mixes:								
(Betazon+acifluorfen)	Storm	Post	30	20	0.56	0.5	0.187	47
(Benefin+vernolate)	(Balan+Vernam)	PPI	10000	40	1.47	1.0	0.006	
(Ethalfluralin+vernolate)	(Sonalan+Vernam)	PPI	260	7	2.58	1.0	0.069	41
(Bentazon+2,4-DB)	(Basagran+Butyrac 200)	Post	30	20	0.98	1.0	0.069	41
(Bentazon+acifluorfen)	(Basagran+Blazer)	Post	100	14	0.13	0.5	0.009	50
[(Bentazon+acifluorfen)+2,4-DB]	(Storm+Butyrac 200)	Post	30	20	0.84	0.5	0.280	49
			100	20	0.28	0.5	0.028	
			100	20	0.56	0.5	0.187	47
			100	20	0.28	0.5	0.028	
(Paraquat+2,4-DB)	(Starfire+Butyrac 200)	Post	1.0E+6	500	0.15	0.5	<0.0001	31
			100	14	0.20	0.5	0.014	
(Paraquat+bentazon)	(Starfire+Basagran)	Post	1.0E+6	500	0.15	0.5	<0.0001	47
			30	20	0.56	0.5	0.187	
(Fenoxaprop+bentazon)	(Bugle+Basagran)	Post	100	9	0.20	0.5	0.009	
			30	20	0.98	0.5	0.327	50
(Fenoxaprop+acifluorfen)	(Bugle+Blazer)	Post	100	9	0.22	0.5	0.010	
			100	20	0.49	0.5	0.049	238

^a PLPI's calculated using typically reported K_{oc}, T_{1/2}, and rate values (R) for the southeastern U.S. region and estimated fractions reaching the soil (F) (ppi, pre, at crack = 1.0, post to foliage = 0.5). These tables are intended only to demonstrate the PLPI scheme and show PLPI's that might be expected with the most commonly used peanut pesticides. A PLPI should be calculated for specific application conditions where rate applied (R) and fraction reaching the soil (F) may be quite different from the data in this table.

Table 7. Pesticide leaching potential (PLP) parameters, values, and indices for insecticides, miticides, fungicides, and nematocides commonly used in peanut production.^a

Common name	Trade name [®]	Applic. method	K _{oc}	T _{1/2} days	Avg rate kg ai/ha	Soil frac. (F) (0-1)	PLP	
							Value	Index
Insecticides & miticides:								
Acephate	Orthene	Post	2	3	0.56	0.5	0.420	52
Aldicarb	Temik	At plant	30	30	1.18	1.0	1.180	58
Carbaryl	Sevin	Post	300	10	1.35	0.5	0.022	33
Carbofuran	Furadan	Over row	22	50	2.30	0.9	4.699	67
Chlorpyrifos	Lorsban	Dir. spray	6000	30	1.76	0.9	0.008	27
		Over row	6000	30	2.35	0.9	0.011	29
	Spectracide	Over row	1000	40	2.20	0.9	0.079	41
Diazanone	Di-Syston	At plant	600	30	1.18	1.0	0.059	40
Esfenvalerate	Assana	Post	5300	35	0.06	0.5	0.002	4
Ethoprop	Mocap	Over row	70	25	2.80	0.9	0.901	56
Fonofos	Dyfonate	Over row	870	40	2.30	0.9	0.095	42
Malathion	Numerous	Post	1800	2	12.78	0.5	0.007	26
Methomyl	Lannate	Post	72	30	0.76	0.5	0.158	46
Phorate	Thimet	At plant	1000	7	1.18	1.0	0.008	27
		Over row	1000	7	2.35	0.9	0.015	31
Propargite	Comite	Post	4000	56	0.20	0.9	0.003	21
Fungicides:								
Benomyl	Benlate	Post	1900	67	0.14	0.5	0.002	19
Carboxin	Vitavax	Post	260	3	1.26	0.5	0.007	26

Table 7 (Continued)

Common name	Trade name [®]	Applic. method	K _{oc}	T _{1/2} days	Avg rate kg ai/ha	Soil frac. (F) (0-1)	PLP	
							Value	Index
Chlorothalonil	Bravo	Post	1380	30	1.26	0.5	0.014	31
Iprodione	Rovral	Post	700	14	1.12	0.5	0.011	29
Mancozeb	Dithane-M45	Post	2000	70	1.47	0.5	0.026	34
Metham sodium	Vapam	Inject	10	7	35.64	1.0	Fumigant	—
PCNB	Terraclor	Over row	5000	21	8.41	0.9	0.032	36
Thiophanate-methyl	Topsin-M	Post	1830	10	0.30	0.5	0.001	14
(PCNB+metalaxy)	Ridomil PC	At plant	5000	21	2.10	1.0	0.009	—
			50	35	0.21	1.0	0.147	45
		Over row	5000	21	3.70	0.9	0.014	—
			50	35	0.37	0.9	0.233	48
(Methyl isothiocyanate + 1,3-dichloropropene)	Vorlex	Inject	6	7	8.67	1.0	Fumigant	—
			32	10	17.35	1.0	Fumigant	—
Fungicide/insecticide mixes:								
(PCNB+chlorpyrifos)	Terraclor	Over row	5000	21	11.21	0.9	0.042	37
	Lorsban		6000	30	2.29	0.9	0.010	—
(PCNB+fensulfothion)	Terraclor	Over row	5000	21	8.41	0.9	0.032	—
	Dasinit		400	24	3.36	0.9	0.182	47
(PCNB+ethoprop)	Terraclor	Over row	5000	21	11.21	0.9	0.042	—
	Mocap		70	25	3.36	0.9	1.081	58
(PCNB+fonofos)	Terraclor	Over row	5000	21	8.41	0.9	0.032	—
	Dyfonate		870	40	2.24	0.9	0.093	42

Table 7 (Continued)

Common name	Trade name [®]	Applic. method	K _{oc}	T _{1/2} days	Avg rate kg ai/ha	Soil frac. (F) (0-1)	PLP	
							Value	Index
Nematicides:								
Aldicarb	Temik	At plant	30	30	3.36	1.0	3.362	65
Fenamiphos	Nemacur	At plant	100	50	2.82	1.0	1.408	59
1,3-Dichloropropene	Telone II	Inject	32	10	53.82	1.0	Fumigant	—
(1,3-Dichloropropene+chloropicrin)	Telone C-17	Inject	32	10	57.64	1.0	Fumigant	—
			62	1	12.19	1.0	Fumigant	—
Metham sodium	Vapam	Inject	10	7	35.64	1.0	Fumigant	—
(Methyl isothiocyanate + 1,3-dichloropropene)	Vorlex	Inject	6	7	8.67	1.0	Fumigant	—
			32	10	17.35	1.0	Fumigant	—

^aPLP_i's calculated using typically reported K_{oc}, T_{1/2}, and rate values (R) for the southeastern U.S. region and estimated fractions reaching the soil (F) (inject, at plant = 1.0, over row = 0.9, post to foliage = 0.5). These tables are intended only to demonstrate the PLP_i scheme and show PLP_i's that might be expected with the most commonly used peanut pesticides. A PLP_i should be calculated for specific application conditions where rate applied (R) and fraction reaching the soil (F) may be quite different from the data in this table.

large (vapor pressure $>>1$ mm Hg) that they do not readily partition into soil solution and escape the soil. High rates of fumigants applied to coarse soils followed by large water inputs could result in groundwater contamination in some cases, however. Surface-applied water may seal pores preventing rapid diffusion of fumigant vapors out of the soil and may actually force the fumigant downward with entrapped air.

Tank mixes and pre mixes of pesticides are frequently used to broaden the spectrum of pest control from a single application. A PLPi for each component of a mixture should be calculated. The highest PLPi calculated for any one component should be used to represent the PLPi of the mix.

Groundwater Contamination Potential (GWCP)

The susceptibility of a soil or pesticide to leaching is useful information to determine their relative ranking. But both soil and pesticide properties are involved in determining the relative risk of groundwater contamination. For instance, a pesticide with a large PLPi will pose a greater risk to groundwater when used on a soil with a large SLPi, but is less of a threat if used on a soil with a small SLPi. Similarly, a pesticide with a small PLPi may pose a greater risk to groundwater when applied to a soil with a high SLPi than when applied to a soil with a lower SLPi. These relationships may be evaluated numerically by determining the groundwater contamination potential index (GWCPi) as

$$GWCPi = (SLPi + PLPi)/2 \quad \text{Eq. (10)}$$

As with the SLPi and PLPi, the GWCPi is scaled from 0 to 100. Pesticide and soil combinations receiving a larger GWCPi represent combinations where groundwater is more likely to be contaminated than combinations with lower GWCPi values. These values do not mean that there will or will not be contamination from pesticides but serve only as a relative screening tool with which to evaluate the use of different pesticides on different soils. By finding the SLPi and PLPi in Tables 5, 6, and 7 and using Equation (10), pest control strategies can be ranked according to their GWCPi. The strategy with the lowest GWCPi should be chosen. In this way the chances of contaminating groundwater may be reduced.

A spreadsheet program for use with popular spreadsheet software has been developed so that one only needs to enter the soil type, pesticide, application method (soil or foliar), and rate. The program calculates and displays the resulting SLPi, PLPi, and GWCPi. Many soil and pesticide scenarios may be quickly evaluated using the program.

Using PSRS: A Herbicide Selection Example

For example, if a peanut grower chooses to use a herbicide as part of an annual grass control program, several options are available. Crop production manuals, weed control guides, or extension information may suggest that the grower select from the following herbicide treatments: (a) preplant incorporated (ppi) in soil using alachlor, benefin, ethalfluralin, metolachlor, pendimethalin, or vernolate; (b) preemergence (pre) to soil using alachlor or metolachlor; or (c) postemergence (post) to the crop using fenoxaprop or

sethoxydim (York *et al.*, 1993).

From local soil survey reports or information from the local SCS office, it is determined that the predominate soil in the field of interest is classified as Lakeland sand (siliceous, coated, thermic Typic Quartzipsamment). The SLPi for the Lakeland, listed in Table 5, is 90.

Using the K_{oc} and $T_{1/2}$ values in Table 6, the specific rates and fractions reaching the soil determined by the grower, and the PLP equations, the grower (or extension agent, consultant, chemical dealer, etc.) determines the PLPi for each herbicide. The PLPi calculated for each herbicide in this example is as follows: alachlor = 50, benefin = 25, ethalfluralin = 23, fenoxaprop = 27, metolachlor = 55, pendimethalin = 28, sethoxydim = 24, vernolate = 41. Since the soil type, Lakeland sand, remains constant, selecting the herbicide with the lowest PLPi will result also in the lowest overall GWCPi (Eq. 10). The GWCPi for each herbicide and Lakeland sand combination is as follows: alachlor = 70, benefin = 57, ethalfluralin = 56, fenoxaprop = 58, metolachlor = 72, pendimethalin = 59, sethoxydim = 57, and vernolate = 66. A postemergence application of sethoxydim has the lowest GWCPi, 57. This relative ranking does not mean that there will not be measurable levels of sethoxydim reaching groundwater. It does mean the chances of significant concentrations of sethoxydim reaching groundwater should be lower than for any of the other herbicide selections for this soil. Herbicides with the least potential to contaminate groundwater should be selected for use. In this example, several options had a similar GWCPi to sethoxydim used postemergence. It then depends on grower preference, cost, and efficacy as to which lower risk option is most appropriate.

This system, or one generating similar information, could be used to identify those pesticides or application methods that minimize the potential for groundwater contamination. The majority of growers do not use groundwater protection information as a routine part of the pesticide selection process. However, most growers are aware of current issues surrounding pesticide use and the environment. Ready access to this type of information may help assure the effective use of pesticides in pest control programs while reducing off-site concerns.

EVALUATION OF SOILS COMMONLY USED IN PEANUT PRODUCTION

Soils used in peanut production are found primarily in the Coastal Plain region of the Southeastern U.S. These soils typically have thickened sandy surface horizons that facilitate rapid water infiltration and movement (Buol, 1973; Smith and Cassel, 1991). Sandy textured, well drained surface horizons are needed for proper peanut growth and efficient harvesting (Henning *et al.*, 1982). Unfortunately, some of the same properties which make these soils good for peanut production also make them conducive to pesticide transport. Sandy textured soils frequently have low water-holding capacities and high water transmission rates which are favorable for leaching. These

soils generally have low organic matter contents and low activity subsoil clays. This means the potential to attenuate or retard pesticide movement through adsorption reactions is limited. Several studies have identified the Coastal Plain region of southeastern states as one of the most susceptible areas in the country to groundwater contamination (Aller *et al.*, 1985; Nielsen and Lee, 1987; Moreau and Danielson, 1990; Danielson *et al.*, 1993). Other studies have documented the occurrence of rapid pesticide movement or groundwater contamination associated with these and other coarse-textured soils (Hubbard, 1989; Guo *et al.*, 1991, 1993; Maas *et al.*, 1992). Pesticide use is also typically high in the coastal regions of each of these states (Pait *et al.*, 1992). The combination of these factors should encourage peanut growers and those in the supporting agrochemical and agrosearch sectors to be sensitive to potential problems associated with pesticide use in peanut production.

Results from an analysis of soils commonly used in peanut production, using the pesticide and soil ranking system (PSRS), is consistent with the previous assessment of peanut soils. SLPi's were determined for 82 soils found throughout the peanut growing Southeast (Table 5). These soils are representative of those most commonly used in peanut production and were calculated from county soil survey data from selected counties in Virginia, North Carolina, South Carolina, Georgia, Florida, and Alabama. As shown in Table 5, 45% of these soils had a SLPi of >75 while only 10% had SLPi <50. The SLPi data reflect the sandy profile textures to 91 cm and low levels of organic matter generally associated with the peanut soils.

EVALUATION OF PESTICIDES COMMONLY USED IN PEANUT PRODUCTION

Peanuts are currently produced in a pesticide intensive manner. Herbicide treatments, as many as five separate applications, are commonly applied preplant incorporated, preemergence, at cracking, and again postemergence (Linker and Coble, 1990). Systemic insecticides are used on the majority of peanut acreage and subsequent foliar insecticide applications for corn earworm and spider mites are commonly used. Fungicides are also heavily used with growers generally applying prophylactic treatments on a regular schedule throughout the season (Bailey, 1993). Tables 6 and 7 list 40 herbicides, insecticides, fungicides, and nematicides commonly used in peanut production. Key properties of each pesticide, used for evaluation with the PSRS system, are given also.

Herbicides

Herbicides and herbicide tank mixes frequently used in peanuts are listed in Table 6. Reported longevity of these materials ranges from 5 days to >100 days with the average $T_{1/2}$ of most compounds being about 1 month. Soil sorption or K_{oc} values range from 20 to more than 10,000. The carboxy acid,

aminosulfonyl, and amide herbicides have lower soil sorption values than the dinitroaniline and quaternary nitrogen herbicides which readily sorb to soil. PLPi's, calculated with average rate and estimated fractions hitting the soil, range from <1 to 55. These indices will change as the actual rate selected or fraction reaching the soil changes and must be determined for specific use situations. However, it can be generally noted that herbicides used at lower rates, those applied postemergence, and those with high soil sorption values (dinitroanilines) have the lowest PLPi's. Metolachlor and alachlor have the highest PLPi which reflects their higher application rates and lower K_{oc} values.

The PLPi assigned to a tank mix or pre mix is governed by the component in the mixture with the highest PLPi. PLPi's are calculated separately for each component in a mixture, then compared. For example, a pre mix of the herbicides bentazon and acifluorfen (Storm) has two active ingredient components. The calculated PLPi for bentazon in this case is 47. The PLPi for acifluorfen is 35. The overall PLPi for the tank mix is, therefore, 47.

Insecticides and Miticides

Organophosphate and carbamate pesticides make up the majority of commonly used peanut insecticides (Table 7). The $T_{1/2}$ of these compounds ranges from 2 to 56 days, with the average $T_{1/2}$ being just under 4 weeks. Soil sorption values range from less than 10 to 6000. PLPi's range from 4 to 67 with aldicarb, carbofuran and ethoprop receiving the highest scores. These three insecticides have low K_{oc} values, $T_{1/2}$ of 1 month or longer, and are typically used at rates above 1 kg ai/ha. Aldicarb and carbofuran have both been reported in groundwater from normal agricultural use (U.S. EPA, 1988).

Fungicides

Organometallic compounds, chlorinated hydrocarbons, carbamates, basic compounds, and other miscellaneous water soluble compounds are used as fungicides in peanuts (Table 7). Most have relatively short half-lives and large or moderately large K_{oc} values. PLPi's calculated from average application data range from 14 to 48. As a group, the fungicides have lower PLPi's than the herbicide or insecticide and miticide groups.

Nematicides

Several of the peanut nematicides are fumigants. Because of their high vapor pressures (>1 mm Hg) they do not easily bind with soil materials or partition into soil solution. This enables them to readily volatilize and escape the soil without presenting a large threat to groundwater. All fumigants are assumed to present a low risk to groundwater in the PSRS system. Aldicarb and fenamiphos are used also as nematicides in peanuts. As previously noted, aldicarb has properties which result in a large PLPi. Fenamiphos has a reported K_{oc} value of 100, a moderately long $T_{1/2}$, and is used at rates near 3 kg ai/ha which also results in a relatively large PLPi.

MINIMIZING THREAT TO GROUNDWATER IN PEANUT PRODUCTION SYSTEMS

Peanuts are currently produced in a pesticide intensive manner on soils generally conducive to pesticide transport. The majority of treatments are preventative and may not always be needed. Any practice that reduces the number of applications or the quantity of active material per application will help reduce contamination risk. Fortunately, many newer pesticides currently on the market utilize lower rates of highly active materials. Extension programs, such as the peanut leaf spot advisory system in North Carolina and Virginia, may prevent unnecessary pesticide applications further reducing total use (Bailey, 1993). One important traditional cultural tool that is not being effectively utilized by many peanut growers is rotation. Inadequate rotations increase pest pressure which in turn increase the need for control measures such as pesticides.

Scouting and other sound integrated pest management (IPM) practices also may help reduce the pesticide load by correctly identifying pests and infestation levels (Linker and Coble, 1990). If pest problems are correctly diagnosed the most appropriate control strategy may then be used. As stated by Linker and Coble (1990), growers must be willing to substitute knowledge for pesticides. Growers should be able to effectively reduce pesticide applications, and environmental risk, without jeopardizing yield. Yet, little information on effective alternative methods exist. Pest management programs have concentrated on pesticide based strategies. So growers reluctance is, in part, due to inadequate information about risk and alternatives. Profitability of sound IPM programs must be stressed if grower adoption is to be expected. Napier and Forster (1982) and Napier and Napier (1991) concluded that farmers attitudes are dominated by short-run profit motives, even at the expense of productivity. Thus, profitability, the perception of profitability, or the availability of economic incentives may determine the rate of adoption of contamination control practices (Danielson, 1992).

Besides reducing the total pesticide load, information about environmental behavior of the products used must be readily available to growers. There is little chance that a producer, environmentally conscious or otherwise, could make a sound choice between pesticide options without some basis of selection. Current criteria for selection include price, efficacy, length of control, rotational restrictions, and application flexibility. Data on these and other aspects are provided from a number of sources enabling producers to make informed choices. Groundwater protection information from systems such as PSRS or other screening or simulation models should be provided also as another basis of selection. Unfortunately, current systems have not been rigorously validated and/or do not lend themselves to ready use and interpretation. Without proper field validation, evaluation of groundwater contamination potential with any model is a theoretical exercise. Research must continue to develop and validate reasonably accurate, easy to use systems that can generate appropriate recommendations useful to producers.

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