Assessing Hydrogeologic Risk Over Large Geographic Areas

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ASSESSING HYDROGEOLOGIC RISK OVER LARGE GEOGRAPHIC AREAS

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INTRODUCTION

A system of analysis designed to predict groundwater sensitivity over relatively large areas (e.g., county, state) would provide a useful tool for land-use decision making. Such a system would aid in predicting present and future impacts of land-use activities such as agricultural use of pesticides or commercial and residential development. A system for groundwater sensitivity analysis should be mappable, allow comparison of relative groundwater sensitivity between different locations, and use all available pertinent data in making the best possible evaluation.

Several standardized rating systems have been developed. Michigan's Site Assessment System, Florida's Sole-Source Aquifer Method, New Jersey's Priority Ranking System, the Environmental Protection Agency's (EPA) Hazard Ranking and Surface Impoundment systems, and LeGrand's methods are systems designed to rate sensitivity of existing contamination sites. Illinois and Wisconsin have developed statewide sensitivity maps based on geologic conditions in their states. Fuller (1986) developed a method to predict sensitivity based on specific hydrogeologic and soil conditions. EPA's DRASTIC method[1] predicts hydrogeologic sensitivity down to the county level and requires less data than Fuller's method (see Trojan, 1986, for review). Several states have considered adapting or modifying DRASTIC for use in their groundwater management strategies. In personal communication with state agencies several other methods were also identified. These included[2] LEACH, VOLAT, CMIS, PRZM (Hawaii Department of Health), HELP, and SAFE (Idaho Department of Health and Welfare), as well as several modifications of existing systems.

Most rating systems assess groundwater susceptibility by assigning relative scores to various hydrogeologic, climatic, contaminant, or water-use "factors." Examples of "factors" used in various systems include depth to water, recharge to aquifers, soil characteristics (texture, infiltration, etc.), topography, aquifer conductivity, aquifer composition, and contaminant toxicity and mobility. Scores for the various factors are added to give a site score or index. This index, when compared to indices from other sites, indicates relative groundwater susceptibility. Factors may be weighted if some are perceived to have a greater impact on contamination susceptibility than others.

Existing rating systems are generally easy to use, can be easily mapped, and consider most of the important factors in evaluating groundwater sensitivity. Their major limitations are failure to identify specific goals and uses of a rating system, inflexibility due to the standardized nature of most rating systems, and failure to provide the system user with the most

[1] Acronyms describing hazard rating systems were taken from original literature and where available, are defined in the glossary.

[2] See footnote 1 and glossary for acronym definitions.

accurate and informative assessment of groundwater sensitivity.

This paper presents a method for conducting a concise, accurate, and effective analysis of groundwater susceptibility to contamination over large geographic areas. As an example of its application, the methodology is used to develop a sensitivity analysis for Winona County, Minnesota.

CHAPTER I: SYSTEM DESIGN

NEED FOR A NEW APPROACH

Assessing hydrogeologic sensitivity to contamination over large geographic areas is an important part of groundwater management strategies being developed in several states. Because sensitivity over an area as large as a county or state is often extremely variable, groundwater hazard ratings which identify regions or sites sensitive or tolerant to various land management practices within this area are an important tool for preventing or controlling groundwater contamination.

Development of hazard ratings is a data intensive process. A questionnaire was distributed to each of the 50 states requesting information on the availability of data for use in a groundwater rating system. Results indicate a wide variation in data availability (figure 1).

Several western states are unable to use existing rating methods because of limited hydrogeologic data, while many eastern states can make more detailed evaluations than allowed by current methods. Comments in the responses suggest that many states are frustrated in their attempts to develop a meaningful approach to assessing groundwater susceptibility. Many comments centered around difficulties in assessing diverse hydrogeologic environments using a standardized approach.

Based on survey data, we concluded that no single system would be uniformly applicable to every state. It is apparently imperative that each state develop it's own rating system based on needs and capabilities for assessing groundwater sensitivity. However, it is equally important that limitations inherent in existing rating systems be adequately addressed.

Figure 1: ***Results of a questionnaire distributed to each state regarding data availability. Number of states responding = 33.***

1. Do you have an adequate means of assessing recharge to the major aquifers in your state?

Yes	No
10	23

2. If not, please describe the quality of the following:

	Good	Fair	Poor
- low-flow data	16	7	5
- precipitation data	25	2	1
- evaporation data	6	17	5
- water-level data from observation wells	9	14	5

3. What is your current method of determining recharge?

	Number of states
- modeling	7
- water budget	5
- water level data	5
- low-flow, base-flow separation	5
- percent of precipitation	3
- use of tracers	1

4. Do you have adequate data for depth to seasonal high water table or to the top of confined aquifers throughout your state?

Yes	No
14	19

5. Do you have soil association maps, soil surveys, or some other means of describing unsaturated zone profiles throughout your state?

Yes	No
27	6

6. To what degree do these soil descriptions allow the following analysis?

	Good	Fair	Poor
- Comparisons of infiltration characteristics among soils	15	16	2
- Identification of impeding beds or layering in the unsaturated profile	10	16	7
- Comparison of water-holding capacities, including thickness, among soils	13	13	4

7. To what degree do soil surveys or association maps describe the following characteristics of the soil or the unsaturated zone?

	Good	Fair	Poor
- organic matter content	11	19	3
- clay content and type	15	16	2
- cation exchange capacity	3	17	13
- textural analysis with depth	12	12	5
- hydraulic conductivity as a function of moisture content	4	9	20

8. Are there adequate maps of land-use throughout the state (e.g., forest, agriculture, etc.)?

Yes	No
17	16

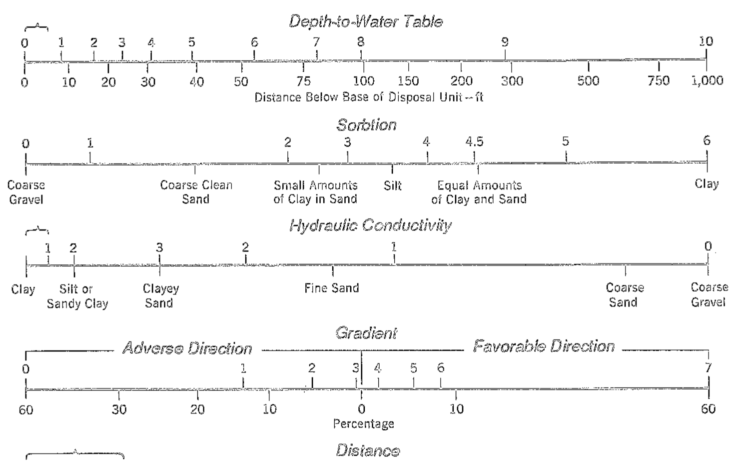
FORMAT

It is theoretically incorrect to predict groundwater sensitivity solely through evaluation of hydrogeologic factors. This will be shown in the ensuing discussion. Groundwater contamination is the result of diverse physical processes, best described by mathematical equations. Because this is impractical on a large scale the concept of a rating system has been developed. A rating system must satisfy three requisites to provide an accurate assessment of groundwater sensitivity.

First, the system must allow for each site in an evaluation area to be assessed to the fullest degree possible with available data. A site assessment is a point estimate of groundwater sensitivity. This estimate represents hydrogeologic sensitivity for the region adjacent to the sampling point. The size of the region is a function of sampling density. Sites are likely to differ in the amount of data available for use in their evaluation.

To allow a comparison of relative risk between sites, site scores can be adjusted for the number of factors entered into an evaluation. New Jersey uses this approach in it's Priority Ranking System, where a total of up to 19 factors may be evaluated at a site. Because sites vary in the number of factors entered into their analysis, the final site score equals the sum of factor scores divided by the maximum possible factor score at that location (New Jersey Geological Survey, 1983).

Figure 2: Rating chart for pollution susceptibility in recharge areas (LeGrand, 1964).



TOTAL POINTS	SUSCEPTIBILITY TO POLLUTION
0 – 4	Imminent
4 – 8	Probable or possible
8 – 12	Possible but not likely
12 – 23	Very improbable
23 – 35	Virtually impossible

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Second, a site index should utilize all relevant site information. LeGrand (1983) utilized this concept by incorporating identifiers into a site index. Using the LeGrand methodology, an example index may consist of the following:

20-4-5-6-5-K where

20 = site score (sum of scores for four factors)
4 = score for distance to water supply
5 = score for depth-to-water
6 = score for water-table gradient
5 = score for permeability/sorption
K = indicates the site is located in a karst region

Identifiers are variables which are used to indicate the presence of processes or conditions which may influence contaminant transport but not to change or adjust scores. Examples include karst aquifers or sites located on floodplains. These conditions are difficult to quantify because of their diversity over space and time. Including them as identifiers alerts a system user to the potential existence of sites with extreme hydrogeologic risk and gives the index user a high degree of site information. This concept can be greatly enhanced with computer programs. (See Chapter III)

Due to data restrictions and natural variability, indices can rarely be evaluated by numerical methods. They are consequently evaluated through the use of general characteristics (e.g., texture for infiltration.) However, these more general methods fail to account for spatial variability, thus restricting the accuracy of assessments. Therefore, the third requisite characteristic is an ability to reduce inaccuracies through the use of scales and correction terms in determining an index score.

LeGrand (1964) first developed the concept of scales for analyzing site index scores (figure 2). Scales suggested a mathematical relationship between an index value and hydrogeologic risk. The utilization of fractional scores lent increased accuracy to the analysis.

The concept of a "correction term" is also introduced to obtain a more accurate estimate for an index score. Because the determination of scores for factors is of a general nature, additional data improving the accuracy of an original estimate may be added. For example, infiltration is important because it affects the amount of water that moves below the root zone (i.e., it affects recharge). Texture is often used as an indication of average infiltration rates for geologic materials. Average rates are, in turn, applied to a rating scale to determine the score for a factor (figure 3).

In addition to texture, infiltration may be affected by pore-size distribution, moisture content, land-use, degree of macroporosity, climate, soil structure, layering, topography, and vegetation. If data exist on any of these variables, the original infiltration rates can be mathematically adjusted to include the effects of these "correction terms".

As an aid in clarifying terms, consider an example of a loam soil underlain by clay at a depth of 2.5 feet, located in an agricultural region where tillage is common. Recharge is assumed to occur in the spring following snowmelt. Intense summer storms may lead to localized recharge, but this effect will not be considered in the determination of factor score due to difficulties in quantifying the variable. If infiltration is a factor being evaluated, initial permeability is 1×10^{-6} meters per second (figure 3). In spring, freeze-thaw cycles reduce soil aggregate size and leave the soil surface crusted. Tillage reduces this effect and increases permeability. The clay layer reduces permeability. Assume these are quantifiable effects and the final permeability is 1.4×10^{-7} meters per second. Therefore the index score is 5.0 and using Figure 3, the corrected index score is 3.3. An identifier letter specifies the assumption of no summer recharge. The final factor index then becomes 3.3ab;c;d, where 3.3 is the index score, *a* and *b* the correction terms, and *c* the identifier. The letter *d* identifies the aquifer being evaluated. Because there may be more than one aquifer in an evaluation area or even at a specific site, it is necessary to identify the aquifer of concern. The correction terms and identifiers are arbitrary and would have assigned meanings in a computer program, where they could be rapidly retrieved to illustrate the role each factor plays in an analysis.

A site index represents hydrogeologic sensitivity for a region adjacent to the site. When assessing sensitivity over large geographic areas, several thousand site evaluations are required to accurately establish and delineate relative sensitivities across this area.

Considering the preceding requisites we have developed two forms of a fundamental equation used to compute a hazard index:

equation 1a

$$HI = R((WaAab...n + WbBab...n + ... WnNab...n)/(Wa + Wb + ... Wn))$$

equation 1b

$$HI = ((WrRab...n + WaAab...n + ... WnNab...n)/(Wr + Wa + ... Wn))$$

where

HI	=	*hazard index for a site*
R	=	*water available for recharge*
Wa,Wb,Wn	=	*weights for factors A,B, and N*
A,B,...N	=	*scores for factors A, B, and N*
a,b,...n	=	*identifiers and correctors on factors*

Equation *1b* differs from *1a* in use of the term *R* as a factor rather than a multiplier. Version *1b* may be selected as an alternative to *1a* or in applications where the term cannot be adequately evaluated throughout the analysis area. Factor scores are determined from rating

Figure 3: Rating scales for permeability and infiltration (Fetter, 1980; Harrold et al., 1976; Silka and Swearingen, 1978; United States EPA, 1985)

Rating Score	0	1	2	3	4	5	6	7	8	9	10
-log permeability (meters per second)	12	9	8	7	6.5	6	5.5	5	4.5	4	2

clays,peat *clay-loams* *silt-loams* *sandy-loams* *coarse sand*
silt *loams* *fine sand* *gravel*
unfractured shale *fractured shale* *fractured-limestone*
cemented-sandstone *poorly-cemented-sandstone* *karst*
igneous/metamorphic rock *fractured igneous and metamorphic*
siltstone *evaporites, basalt*

scales. The terms used in these equations will be developed in the following section.

Dividing by factor weights (maximum potential site score may be substituted) fulfills the requisite of obtaining an average score for factors. The use of "identifiers" and "correction terms" fulfills the requisite of providing information and accuracy for a site evaluation.

STEPS IN SYSTEM DESIGN

Figure 4 is an algorithm useful in developing a rating system. The algorithm presents a five-step approach:

1. Identifying specific objectives,
2. Determining factors,
3. Determining data availability,
4. Developing scales, weights, correction terms, and identifiers, and
5. Implementing, testing and refining the system.

1. Setting Specific Goals

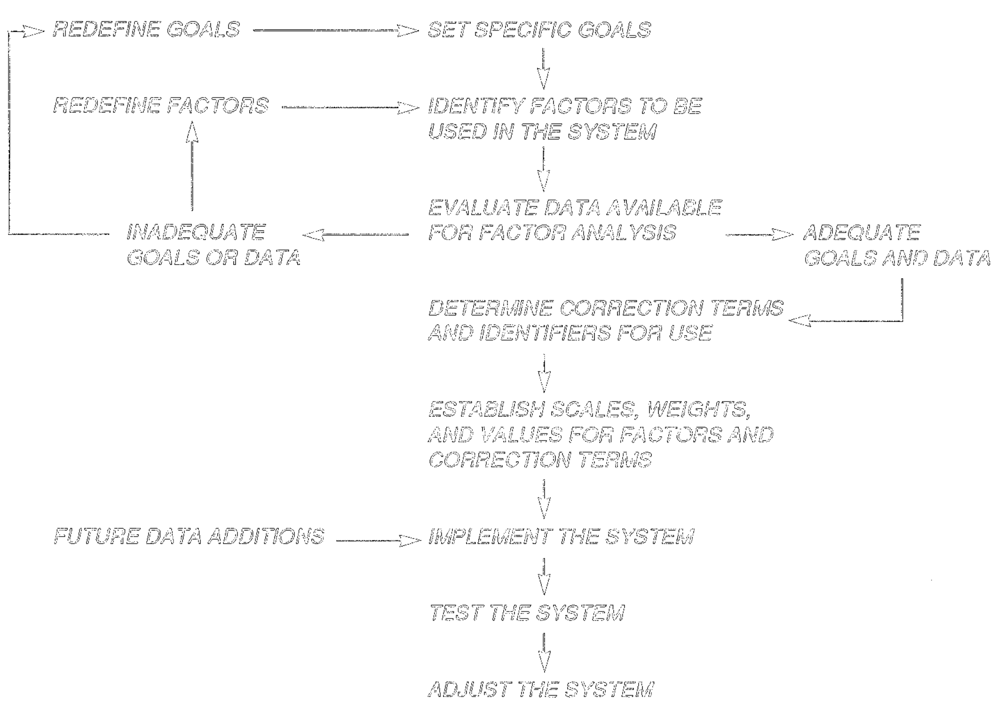
What information does a hazard index and accompanying map provide and what can these tools be used for? Without clear, concise answers to these questions it is impractical to begin examining the problem of groundwater sensitivity. Identification of goals and objectives is the critical step in developing an evaluation system (Perry *et al.*, 1987).

Specific application of a rating system must be clearly envisioned. Many states are attempting to build a generic method for predicting groundwater susceptibility to contamination, or utilize an existing method for this purpose. This definition of goals is inadequate. It is too general and does not define any specific application.

Other states are attempting to utilize existing methods to assess susceptibility to specific contaminants. However, these methods are generally inadequate to allow this type of analysis.

Development of a hazard rating is complex due to the tremendous range of potential contaminants. Their fate in the environment is highly varied and often strongly influenced by attenuation factors such as organic matter content, cation exchange capacity, clay content and type, and pH of soils. Existing systems generally do not evaluate these attenuation factors, although they must be considered in a susceptibility analysis. DRASTIC, for example, is widely used to predict the relative likelihood that recharge will occur over areas down to 40 acres in size. It cannot evaluate susceptibility of contamination for specific contaminants unless it is modified to consider the specific characteristics of those particular contaminants and their attenuation rates.

Figure 4: Algorithm for developing a rating system



The specific applications and/or goals of a rating system need to be specified before a system can be developed. Examples of specific goals include:

- *develop a system to assess potential recharge over a geographic area;*
- *develop a system to assess groundwater susceptibility to a class of pesticides; (See Chapter II)*
- *develop a system to identify areas which are likely to have sites favorable for development of a low-level radioactive waste site;*
- *identify aquifers with large water transport capacities.*

Each of these goals requires a minimum generic analysis as well as different detailed analysis to achieve the desired objective.

2. Determining Factors

All factors considered important to a sensitivity analysis should be included in the system design, even if they cannot initially be evaluated. Equations *1a* and *1b* have the flexibility to allow these factors to be incorporated once their evaluation becomes possible. For example, it may be desirable to design a system consisting of "base" factors (i.e., those general factors that influence recharge potential). These factors do not necessarily lead to a specific analysis of contamination potential. Examples of base factors include depth-to-water, recharge, topography, permeability, and infiltration.

During an analysis, specific factors can be added to a base system to achieve specific contaminant analysis and, after completing the desired objective, these specific factors can be deleted from the system. An example of a specific analysis would be the assessment of aquifer sensitivity to pesticide contamination. In addition to the base factors, a specific factor such as organic matter content of soils would be included to consider the attenuation of pesticide movement.

After choosing factors, it is necessary to determine how each factor will be evaluated. For example, infiltration may be assessed by textural classification related to average permeabilities. Identifiers and correction terms should be utilized if they can be quantified, affect a factor score, or are important in identifying a particular characteristic of an area.

3. Assessing Data Availability

The system designer must determine which of the chosen factors can be evaluated, how they will be evaluated, where they can be evaluated, and at what scale and level of confidence the analysis will occur.

Figure 5 lists exemplary data sources for factor evaluation. The most common sources are geologic maps, soil maps, and soil surveys. Well logs can be valuable sources of information. Climatological data often needs to be converted from traditional temperature and precipitation data into water balance parameters such as evapotranspiration. (See Chapter II) Reports from research studies can add a great deal of local information.

Data form and quality must be determined. Acceptable forms for data structure include length (m,ft.) for depth-to-water; length per unit time (m/y,m/h) for recharge, permeability, and infiltration; percent slope for topography; and organic matter content (percent of total sample weight), cation exchange capacity (meq/100g), pH, and/or clay content (type and percent) for attenuation. Available data must be converted to units acceptable for incorporation into computer programs. (See Chapters II and III)

For spatial scale decisions, the user may wish to section the evaluation area into subareas. Evaluating the same factors in each subarea provides increased spatial detail and provides a visual identification of the ensuing analysis. An alternative or supplement to factor division is to evaluate a specific aquifer in a subarea. Subarea division should include correction terms and identifiers.

To clarify terms, an *evaluation area* represents a large geographic area such as a state or county where hydrogeologic sensitivity is to be evaluated. Across this area, *sensitivity* is likely to be highly variable due to differences in geologic, soil, climatic, and land-use characteristics. Sensitivity across this area is determined from *site evaluations* which represent point estimates of hazard rating. The number of site evaluations is a function of data availability and variability in sensitivity.

When *site indices* are compiled, a final map can be prepared by developing *hazard-rating* contours across the entire evaluation area. From these contours *relative hazard risk* can be established by creating ranges of scores indicative of groundwater sensitivity (e.g., scores of 0-10 represent low risk, 11-20 moderate, etc.). Thus, *regions* of similar risk will emerge and can be identified on the final map. Chapter IV illustrates a practical application of this procedure.

Subareas represent subdivisions of the evaluation area where similar analysis can be made or a specific aquifer is being evaluated. They are designed to simplify the evaluation process.

Once goals and data availability have been determined, initial goals should be re-evaluated.

Figure 5: Sources of Information for Rating Systems (Trojan, 1986)

Type of Data	*Typical Sources*	*Factors Evaluated*
Property survey	*County records, property owner*	*Waste characteristics*
Well drillers' logs	*Well driller, property owner, state records*	*Geologic, water use*
Water level measurement	*Drillers' logs, topographic and ground water maps*	*Geologic, water use*
Topographic maps	*United States Geological Survey, state sales offices*	*Geologic, water use*
Air photos	*United States Department of Agriculture, United States Forest Service*	*Water use*
County road maps	*State agencies*	*Water use*
Ground water report	*United States Geological Survey, state agencies*	*Geologic, water use*
Soil surveys	*United States Department of Agriculture*	*Geologic, water use*
Geologic maps	*United States Geological Survey, state surveys*	*Geologic, water use*
Waste character	*Owner/operator, state or federal permits SIC code*	*Waste*
	SAX Toxicity	*Waste*
	Health departments	*Water quality, water use*
	Water companies	*Water quality, water use*
Monitor wells	*United States Geological Surveys, water management agencies*	*Water quality, water use*
Climatological	*National Oceanic and Atmospheric Administration, National Weather Service, local weather data, state climatologist*	*Climate, recharge*
Demographic	*State planning agencies, census bureau*	*Land-use, population*
	Local colleges or universities	*Geologic, waste*
	Professional organizations	*Geologic*

Geologic factors: depths, permeability, sorption, topography, aquifer data, soil data, bedrock data

Water use factors: well number and location, quantity used, distances to waste source

Waste factors: type and quantity, toxicity, mobility, persistence, location

Climatic factors: precipitation, evapotranspiration, runoff

If they cannot be met, goals must be readjusted, factors altered, or study area redefined. This recycling step provides an iterative re-evaluation procedure critical to project success (Perry *et al.*, 1985, 1987). To satisfy equations *1a* and *1b*, at least one factor must be evaluated throughout the evaluation area. In addition, specific factors may be required to meet a specific objective (e.g., organic matter content when considering attenuation of pesticides.)

Successful completion of stated objectives often requires redefining the evaluation area. Alaska, for example, has virtually no reliable data for a significant portion of the state (Alaska Department of Environmental Conservation, personal communication). In such a case it is best to delete subareas where no evaluation can be made and assume that they are not likely to be significantly impacted by humans.

In the event that deleted subareas are impacted, incidents are generally localized and best addressed through site-specific techniques. As human impacts increase, data generally becomes available and evaluations for those subareas can be made. Unfortunately, such evaluations often come after groundwater problems become evident. Close attention to growth and development in un-analyzed subareas, and to data availability, will help alleviate such problems.

Five procedures would assist states in improving existing data and increasing it's accessibility.

1. Integrate all data into a central, computerized data base. The data base should include existing data and files from all state, county, and municipal agencies and should be supplemented with detailed maps. From a geologic and soil perspective, this data would include well logs, soil tests, research results, data from contamination sites, and existing maps and surveys. Illinois and Minnesota are two examples of states which have recognized this need and are developing centralized data-information centers (Illinois Environmental Protection Agency, 1986; Minnesota Pollution Control Agency, personal communication).

2. Digitize existing maps. Maps are useful in displaying visual data but are not useful in computer programs to provide quick and accurate results. Digitization remedies this inability.

3. Complete unfinished maps and surveys. With a centralized data base, unfinished projects can be identified and prioritized. Once completed, maps should be digitized.

4. Update existing data. With a centralized data base, new information can be identified and incorporated into existing files. An example of an update is the inclusion of new well logs for aquifer delineation.

5. Identify data needs and develop the necessary means of acquiring needed data. Once steps 1-4 are completed and the specific goals of a rating system have been established and refined, it will be possible to identify, prioritize, and determine how to satisfy data needs.

Funding and time constraints are severe problems for state agencies. The steps for improving data accessibility require expenditures, but future benefits will far outweigh the costs. Completion of these steps, particularly 1–3, should precede design of a rating system.

4. Developing Scales, Weights, Correction Terms, and Identifiers

A factor score is determined by applying field data to a rating scale (See figures 2,3,7,8,9). This score is multiplied by a "weight" to adjust for the perceived importance of a factor. Because scales and weights are used to determine a

Figure 6: Depth-to-water scores for six rating systems (Trojan, 1986)

DRASTIC

range(ft)	rating
0 - 5	10
5 - 10	9
10 - 30	7
30 - 50	5
50 - 75	3
75 - 100	2
100+	1

LeGrand (1983)

range(ft)	rating
0	9
0 - 2	8
3 - 8	7
9 - 15	6
16 - 25	5
26 - 35	4
36 - 60	3
61 - 90	2
91 - 200	1
200+	0

EPA Hazard Ranking System

range(ft)	rating
0 - 20	3
21 - 75	2
76 - 150	1
150+	0

New Jersey

range(ft)	rating
0 - 20	3
20 - 50	2
50 - 100	1
100+	0

Fuller*

range(m)	rating
0 - 1	0
2 - 3	1
3 - 4	2
4 - 6	3

Wisconsin*

range(ft)	rating
50+	10
20 - 50	5
0 - 20	1
6+	4

* lower rating indicates increased risk

Figure 7: Transformation of seven rating systems to scaler form for depth-to-water

	Rating Score	0	1	2	3	4	5	6	7	8	9	10
1.	feet		100	75	50		30		15	10	5	0
2.	feet	1000	290	100	80	55	40	30	24	16	9	0
3.	feet		200		70		30		13		1.8	0
4.	feet	19.7					9.8					0
5.	feet	150			80				18			0
6.	feet	100			55				18			0
7.	feet	50					20					0

1=DRASTIC
2=LeGrand (1964)
3=LeGrand(1983)
4=Fuller
5=EPA Hazard Ranking System
6=New Jersey
7=Wisconsin

Figure 8: Composite depth-to-water scale for seven rating systems

Rating Score	0	1	2	3	4	5	6	7	8	9	10
depth-to-water (feet)	263.9*	120.6	78.9	53.9	40.5	30.3	22.5	15.3	9.1	3.8	0

*composit of four systems

Figure 9: Rating scales for recharge and topography

Rating Score	0	1	2	3	4	5	6	7	8	9	10
Recharge (inches per year)	0	1	2	3	4	5	6	7	8	9	10

Rating Score	0	1	2	3	4	5	6	7	8	9	10
slope (%)	100	50	40	30	20	15	10	7	4	2	0

factor score, it is critical that they are understood and accurately represent hydrogeologic risk.

Scales range from low to high hazard or risk, with 0-10 being a logical choice. A scale is continuous, including fractional scores, and avoids the rigidity of classes. For example, scores used for depth-to-water in six rating systems are shown in figure 6. In figure 7 each of these has been transformed into a scale, providing a more precise assessment of score.

Scales can be developed using data from existing contamination sites, research work, literature, experienced specialists, and existing rating methods. Figure 8 combines the scales from figure 7 to derive a single rating scale. In figure 9, we present rating scales for recharge and topography; a permeability scale was presented in figure 3.

These scales are derived from other rating systems and literature discussed in Chapter II. They are correct in design, but data presented in this paper are inadequate for development of scales applicable to the wide variety of conditions that might occur in evaluation of localized areas. Each rating scale must be locally derived and supported with sufficient literature and data before its use can be justified.

Equations *1a* and *1b* indicate that weights may be used to adjust a factor score based on the score's perceived relative importance in a given analysis. DRASTIC for example, assigns weights of 1-5 for each factor (U.S. EPA, 1985). We do not recommend the use of weights other than 1; we suggest that adjustments to factor scales and correction terms are more appropriate. Weights adjust a score to stress the importance of a particular factor. For example, consider two sites, *A* and *B*, evaluated with three factors having weights of 1. If a fourth factor with a weight of 3 can be evaluated at site *A* but not at site *B*, the relative scores between the two sites will be biased because of the higher input for factor 4 into the denominator of equation *1a* or *1b* for site *A*. If weights must be used, they should be as accurate as possible, including the use of fractional values.

Multiplying a factor value by a correction term improves it's accuracy. Application of this improved factor value to a rating scale results in a more accurate factor score. Correction terms are determined from mathematical relationships or scales, which in turn are derived from literature and research work.

Care must be taken to use only correctors which have a unique effect on a specific factor. For example, correcting for both tillage and shrink-swell capacity of soils is incorrect because tillage reduces the effect of shrink-swell. Examples of correction terms are presented in the discussion in Chapter II, and their application is demonstrated in the sensitivity analysis of Winona County, Minnesota which is presented in Chapter IV.

Identifiers are subjective variables which influence a factor but cannot be adequately quantified. Applying an identifier to a site index requires that the identified characteristic be prevalent at or near a site, that it significantly affect the factor, and that it's effect on the factor be understood. Examples are shown in Chapter II.

An Illustrative Exercise

To illustrate the application of the first four steps in system design, assume a rating system is being developed to predict the sensitivity of groundwater to pesticide contamination. Factors include depth-to-water, recharge, attenuation-in-the-soil-zone, topography, infiltration, and permeability-of-the-vadose-zone.

Depth-to-water (in meters) is calculated by subtracting water table elevations from surface elevations using geologic maps, well logs, and topographic maps. Recharge, (in centimeters per year), is determined from climatic records as precipitation minus evapotranspiration. Infiltration and permeability (as meters per hour) are determined from average permeabilities of identified geologic materials. A soil atlas and geologic maps provide data for these factors.

In localized areas, soil surveys provide information on shrink-swell capacity of soils. This is used as a correction term for infiltration. Assume land-use maps exist for portions of the evaluation area and that they indicate the presence of forested or agricultural land. Also assume an extensive floodplain occurs in the evaluation area. Attenuation (ug organic matter/g soil) is derived from scales which consider the relationship between average organic matter content of soils and pesticide adsorption.

Figure 10 summarizes, and figure 11 presents a visual model of the proposed analysis. Scales from figures 3, 8, and 9 may be used in the exercise. An attenuation scale may be developed from data presented in Chapter II. Factor values are applied to scales to determine score. These are then applied along with predetermined factor weights to equation *1a* or *1b* to arrive at a site index.

Figure 10: Example of potential factor analysis to meet a specific analysis objective

Objective: Develop a rating system designed to assess groundwater susceptibility to contamination by pesticides

Factor	*Method of Analysis*
1. Depth-to-water	*Distance, in units of length, from a designated point near the land surface or contamination source to the top of confined aquifers or the seasonally high water table.*
2. Permeability-of-the-vadose-zone	*Permeability, in units of length per unit time, of the layer with the lowest conductivity in the vadose zone.*
3. Water available for recharge	*(Precipitation + Irrigation - Evapotranspiration) in units of length.*
4. Topography	*Percent slope and slope position.*
5. Infiltration	*Permeability or infiltration rate of the root zone, with units of length per unit time.*
6. Attenuation	*Organic matter content of the root zone, as a percent of soil weight.*

Factor	*Possible Correction Terms and Identifiers*
1. Depth-to-water	*Local variations in depth to water*
2. Permeability	*Layer thickness, position of different layers*
3. Recharge potential	*Methods or assumptions utilized in analysis*
4. Topography	*Special landscape features (e.g., flood plains)*
5. Infiltration	*Macroporosity, degree of soil crusting, land-use, vegetation, precipitation patterns*
6. Attenuation	*Pesticide class, application rate*

5. Applying, Testing, Refining the System

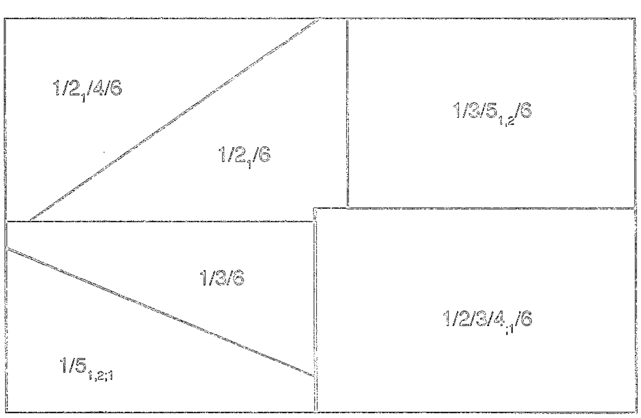
Once completed, the system should be applied to determine it's accuracy and ability to provide a sufficient range of scores for comparison of relative groundwater susceptibility between sites. After applying the system for a certain length of time (e.g., 1 year), refinements can be made. Adjustments to scales, weights, correction terms, and identifiers are expected. Factors may also be deleted, added, or modified at this time. Applying the system to areas known to be susceptible to contamination or testing the accuracy of the method (e.g., through the use of tracers) provide the best means of evaluating the system.

In applying a rating system, it is important that methodology be consistently employed. It's particularly important to be consistent in assigning the scale, weight, and correction values. To reduce inaccuracies of generalizations, particularly when data is poor or incomplete, development and application of hazard ratings should occur at a centralized location.

DEVELOPING SATURATED ZONE HAZARD RATINGS AND MODIFICATION OF EXISTING SYSTEMS

Several existing methods evaluate unsaturated and saturated zones in the same analysis. However, the fate of a contaminant in one particular zone may be independent of it's fate in another. Assessing both zones in one analysis tends to smooth differences and may misrepresent risks. It will be more appropriate to evaluate each zone separately and later present

Figure 11: Hypothetical factor distribution for sample factor analysis shown in Figure 10.



Indexes indicate factors, correction terms, and identifiers to be identified in each area. Equation 1b is required since factor 3 cannot be evaluated throughout the area. An index of $1/2/3/4/5_{1;1}/6$ indicates that all six factors can be evaluated in an area (see Figure 10 for factor descriptions). Factor 5 (Infiltration) is corrected and contains an identifier as indicated by the subscripts. An identifier must be added for each subarea to identify the aquifer of concern.

summarized evaluations in an overall site assessment.

This paper primarily focuses on evaluation of the unsaturated zone. Similar techniques can be applied to the saturated zone. As with the unsaturated zone, variables such as depth-to-water, recharge, infiltration/permeability, topography, and a variety of attenuation factors may be considered with the following exceptions:

- *flow is saturated;*
- *flow direction is horizontal rather than vertical;*
- *knowledge of recharge and discharge zones is critical and may be inferred from topography, well location and density, and position of surface water bodies;*
- *attenuation is often minor or insignificant;*
- *the rate of contaminant delivery to an aquifer may be more critical than the rate of water recharge.*

The analysis of flow in the saturated zone is eased by the application of various mathematical models and theories, particularly continuity of flow, and through the availability of good data on saturated hydraulic conductivities for classes of geologic materials. Difficulties commonly arise from data inadequacies on such variables as recharge rates, geologic characteristics of an aquifer (e.g., anisotropy, thickness, bedding angles), and the effects of recharge/discharge zones, particularly near flowing wells.

The primary effect on contaminant transport for horizontal, saturated flow is gradient or head difference. This is illustrated by increased variance in flow rates near recharge/ discharge sites, in areas of large topographic relief, and in aquifers with large variations in thickness and gradient. A rating system must properly evaluate the influence of hydraulic gradients.

Factors are similar for saturated and unsaturated systems but are measured differently. The need for correctors is reduced in the saturated zone because assumptions regarding flow are often relatively accurate. Comments regarding development of base systems, to which specific factors can be added or deleted, apply to analysis of both saturated and unsaturated systems.

Existing rating systems (e.g., DRASTIC) can easily be modified to use the methodology described here. Existing systems have the advantage of being already developed, although factors should be modified to form scales with these modifications and to consider local hydrogeologic conditions. Existing systems can be applied to equation *1a* or *1b*. This will allow the determination of hazard ratings for geographic areas having large variations in available data.

For assessment of existing contamination sites or structures, the LeGrand method (1983) appears to be the best currently available. It provides good site information, a wide range of contaminant considerations, and clear analysis of factors. It does not, however, provide a substitute for contaminant monitoring.

CONCLUSIONS

The Trojan-Perry Method of site analysis presented here provides a useful means of evaluating hazard or risk on a site. There are several advantages to this methodology:

1. Analysis is flexible. Any number of factors can be included in the analysis; at a minimum, a rating score at a site can be produced with only one factor. Factors can be added as data becomes available. Specific factors can be added to a base system to make specific evaluations, allowing a variety of potential assessments for different uses.

2. The system allows use of all available and/or potential data. Thus a site can be evaluated to the fullest and most accurate degree possible. In addition to factor analysis, correction terms and identifiers can be utilized to improve the accuracy of an assessment.

3. The user is provided with abundant site information. With computer application, a site index with factor scores, correctors, identifiers, and a site score can be displayed. Detailed information on, or interpretation of site characteristics may be provided if required.

4. System designers may refine the design to fit their needs. Existing systems may be modified to fit equations *1a* or *1b*. A base analysis may be done initially, and priority areas may be assessed first if detailed analysis is not desired. Specific evaluations, correction terms, and identifiers may be implemented in the future.

5. The final analysis requires coordination of existing data and identifies future data needs and priorities.

The reader should also recognize the existence of significant cautions in using this methodology.

1. The final system is only as accurate as the methodology, which in turn relies on data inputs. Of particular concern is the extrapolation of data from incomplete maps

and the reliability of existing maps. Maps may be outdated or may present data at a very broad scale (e.g., use of 100-foot contour intervals for water table elevations).

Factor scales and numerical values for correctors are intended to present "relative risk" based on highly variable natural processes. However, most methods of analysis are inexact. Support for development of factors and correctors is generally weak in the literature, particularly in the area of field application. It is likely that any susceptibility analysis will require refinement and iteration.

2. System designers must maintain a clear view of goals. If the initial goal is to design a screening tool such as DRASTIC, then correctors and identifiers should be temporarily omitted.

3. The Trojan-Perry Method is not site specific. Because of the generalized nature of the analysis, a rating system can never achieve site specificity. The investment in design and implementation of a system must therefore be balanced by expected benefits. If computer capabilities are to be utilized, a highly informative analysis can be provided for the cost of software and costs for digitizing maps.

4. A system must be consistently applied throughout an area, particularly with respect to scales, weights, and correction terms.

5. The utilization of different data bases for different sites may seem to invalidate or weaken resulting interpretations. Data limitations force such a practice in many situations. The Trojan-Perry Method provides the user with indices describing quality of the data utilized in a site evaluation. This insight may reduce the problem of differing data bases.

CHAPTER II: DEVELOPMENT OF FACTOR INDICES

The transport of contaminated water to groundwater is controlled by two processes which affect nearly all potential contaminants:

1. The amount and rate of recharge through the unsaturated zone and into an aquifer, and
2. Attenuation characteristics of the unsaturated medium.

Recharge and attenuation are complicated processes. Even over small distances, point estimates of these parameters are impractical due to variations in geology, climate, soils, and land-use. A logical alternative is to assess these processes through related characteristics which reflect more generic relationships. Taken together, evaluating factors and developing factor indices is the process of compressing point estimates to integrated variables.

This chapter discusses development of factor indices. The importance, methods of analysis, and natural variability of factors are presented. Correction terms and identifiers are introduced, developed, and when possible, presented in a format acceptable for incorporation into equations *1a* and *1b*. General relationships are shown. The discussion is not exhaustive and numerical values and scales are not developed. System design teams will need to expand on this discussion, including the development of necessary scales and values, particularly as they relate to local conditions.

RECHARGE

Recharge, as used in this discussion, is the amount of water passed through the unsaturated zone and into an aquifer during a specified period of time. The time period is generally one year. Recharge may be expressed as either of the following:

1. Annual recharge rates to aquifers, expressed as a depth across the evaluation area, or
2. The amount of water available for recharge. These latter methods consist of water budgeting approaches.

Either or both methods may be utilized in a susceptibility analysis. A summary of general methods for determining recharge is presented below, followed by a discussion of specific recharge potential analyses.

Methods for Determining Recharge to Aquifers

Method A

In regions where surface streams or rivers are hydrologically connected to groundwater, low-flow data from reliable gaging stations may be assumed to represent groundwater contribution to flow. These measurements are often derived from 30-day low flow records during periods of insignificant precipitation (Ackroyd *et al.*, 1967; Riggs, 1972).

However, recharge estimates derived from low-flow data are often significantly less than those calculated with other methods. Often this is due to aquifer-derived water being lost above the stream gaging station through evapotranspiration.

An additional error may result when recharge estimates during dry periods are less than those from wet periods, particularly in areas where precipitation-derived groundwater moves relatively quickly to the stream or river. In addition, there are inherent large variations in low-flow data, making extrapolation to adjacent gaging stations difficult.

Finally, there may be significant contributions to aquifers from surface water during times of low precipitation (Ackroyd *et al.*, 1967; Riggs, 1972 and 1973; Kanivetsky, Minnesota Geological Survey, personal communication). Kanivetsky (1979) has applied this low-flow technique to river basins throughout Minnesota and found good correlation with observation well data for about 30 percent of the basins.

Method B

Water levels in observation wells are often used to indicate vertical recharge rates. With this technique, horizontal flow is assumed to either be insignificant or quantifiable. Since an observation network must be adequate to determine recharge across large areas with diverse climate, geological, and soil characteristics, extrapolation between well sites is difficult.

Method C

Estimates of recharge may be based on climatic data (Ackroyd et al., 1967). In a study by Steenhuis *et al.* (1985) in New Jersey, recharge was found to equal 75-90 percent of the total precipitation between October 15 and May 15, and 0 percent for the remainder of the year. Rehm *et al.* (1982) found recharge equal to 2-9 percent of annual precipitation for three different methods of analysis in North Dakota.

In the Leningrad Province of the Soviet Union, Zaitsburg (1982) presented equations used to forecast recharge based on average air temperatures and water levels at certain times of the year (figure 12). Recharge in Saskatchewan prairies was found to equal 1-7.5 percent of mean annual precipitation (Stephens and Knowlton, 1986).

In a similar fashion, in arid regions of Saudi Arabia, Australia, and the High Plains of Colorado, well defined relationships have been found between recharge and storm intensity (Stephens and Knowlton, 1986). The method of determining recharge from climatic records appears to be accurate for areas with an adequate water level network, when climatological records are good, and when predictable recharge patterns exist.

Method D

Under certain circumstances, water chemistry and tracing studies have been used to estimate groundwater recharge. In limestone aquifers of south-central Texas, Campana and Mahin (1985) estimated recharge by tracing tritium through discrete cells and into groundwater. Recharge in their study area occurred through stream outflow and through precipitation infiltration. The results (figure 13) show that recharge values determined by the tritium method were somewhat lower than with an infiltration-streamflow model. The major source of difference is attributed to the inability to differentiate between recharge from primary and secondary sinkholes with the tritium method.

Despite the lower estimates of recharge, the tritium method was shown to yield consistent, relatively accurate results. Limiting the method is the assumption that groundwater losses due to evapotranspiration and outflow are zero or can be accounted for with the use of discrete cells. In deep desert aquifers unaffected by evapotranspiration and having low flow rates, these losses can be negligible.

Allison *et al.* (1985) determined recharge in karst

Figure 12: ***Regression equations and their multiple correlation coefficient used in predicting groundwater table elevation in the USSR (Zaitsburg, 1982)***

Well no.	Observation period, years	Forecasting equation	R ± SE
Leningrad and Leningrad Province (Northwest of the USSR)			
2	18	$y1=0.32+0.746x1-0.054x2$	0.93 ±0.03
11	22	$y1=0.68+0.573x1-0.046x2$	0.83 ±0.06
160	20	$y1=0.78+0.516x1-0.054x2$	0.78 ±0.09
251	16	$y1=0.53+0.887x1-0.080x2$	0.72 ±0.12
170	18	$y1=0.37+0.904x1-0.052x2$	0.76 ±0.10
Novgorod Province (Northwest of the USSR)			
1	15	$y1=1.32+0.320x1-0.040x3$	0.86 ±0.07
11A	11	$y1=0.78+0.373x1-0.026x3$	0.81 ±0.11
26	15	$y1=1.25+0.557x1-0.012x3$	0.82 ±0.08
27	14	$y1=1.38+0.377x1-0.028x3$	0.81 ±0.09
Average	17		0.81
Leningrad and Leningrad Province (Northwest of the USSR)			
1003	24	$y2=6.19+0.574x4-0.016x5$	0.80 ±0.08
1011	24	$y2=12.27+0.199x4-0.016x5$	0.69 ±0.11
1042	20	$y2=12.37+0.442x4-0.021x5$	0.88 ±0.05
106	16	$y2=0.59+0.205x4-0.003x6$	0.91 ±0.04
153	17	$y2=0.95+0.228x4-0.005x6$	0.86 ±0.06
160	19	$y2=1.61+0.257x4-0.006x6$	0.68 ±0.13
Average	20		0.81

R = multiple correlation coefficient
SE = standard error
y1 = pre-spring minimum ground-water table
y2 = spring maximum ground-water table
x1 = mean monthly ground-water table for December
x2 = average temperature for December-January
x3 = average temperature for December-February
x4 = mean monthly ground-water table for February
x5 = precipitation for December-March
x6 = precipitation for December-February

Figure 13: ***Estimated recharge, in inches, from two methods (Campana and Mahin, 1985)***

Year	DSC Model	Stream Loss + Precipitation
1953	0.1773	0.2060
1954	0.1817	0.1985
1955	0.2277	0.2369
1956	0.0486	0.0539
1957	1.1238	1.4101
1958	1.1150	2.1157
1959	0.7243	0.8517
1960	0.8407	1.0175
1961	0.7403	0.8847
1962	0.2777	0.3072
1963	0.1892	0.2106
1964	0.4427	0.5073
1965	0.6428	0.7692
1966	0.6415	0.7374
1967	0.4940	0.5757
1968	0.9055	1.0914
1969	0.6069	0.7117
1970	0.6922	0.8162
1971	0.9906	1.1350
Total	11.6635	13.8375

and dune elements in a semi-arid location in Australia by using chloride, deuterium and oxygen-18. Below the root zone, chloride concentration may be linearly related to water content with depth. If this relation holds and the water table is below the root zone, then measuring the chloride concentration at the water table provides a measure of the recharge rate.

Deuterium and oxygen-18 are natural isotopes; their concentration in rainfall is a function of rainfall intensity. A profile of the isotopic and water concentrations with depth to the top of the water table gives an indication of rainfall intensities needed to cause recharge, and of recharge amounts.

Allison *et al.* (1985) determined recharge rates of approximately 0.17 mm/year for the study area. They determined the following recharge rates for various landscapes: secondary sinkholes, 60 mm/year; primary sinkholes, 0.1 mm/year; undisturbed dunes, 0.06 mm/year; disturbed dunes, 13 mm/year. The authors ascertained that on disturbed sites where native vegetation was removed, due to increased recharge, groundwater was more susceptible to contamination by salts. This increased recharge led to increased salt concentrations in local rivers.

Gunn (1983) performed similar experiments in New Zealand karst areas. He correlated magnesium and calcium concentrations for different types of flow regimes. For example, seepage water in the vadose zone contained low concentrations of these ions, while water flowing through vadose zone joints and fissures contained high concentrations. By determining ion concentrations in groundwater, Gunn concluded that flow through joints and fissures produced more recharge than seepage.

Dreiss (1983) concluded that water chemistry and tracing studies are best for determining recharge areas in karst topography, while general methods fail to determine locations of rapid response.

Use of natural isotopes and water chemistry is suitable for areas where the necessary relationships can be established. These are generally localized areas where groundwater systems are somewhat simplified or are subject to rapid response processes.

Method E

Water balance methods attempt to budget water over the components in the following equation:

equation 2

$$R = P + I - RO - ET - dS$$

where
R = *recharge to groundwater*
P = *precipitation*
I = *irrigation* [1]
RO = *runoff*
ET = *evapotranspiration*
dS = *change in soil storage*

There are a number of simplifying assumptions used in applying this method, including:

1. No lateral water transport (unless it can be quantified);
2. Accurate estimation of ET and RO;
3. A well-defined, established vegetative cover;
4. All water available for soil storage is consumed as storage, meaning water moves as a wetting front through the unsaturated zone;
5. Winter precipitation is liquid or is considered for the month in which it melts.

Despite these restrictions, the method is frequently used. Leach (1982) studied the response of recharge to changes in water budget components for aquifers in Hong Kong. Runoff was insignificant and irrigation was zero, leaving soil moisture deficit, precipitation, and evapotranspiration as the major factors influencing recharge. He found that soil moisture deficits of up to 150 mm had little effect on recharge, while varying the evapotranspiration coefficient by 20 percent affected recharge by 20 percent. A strong and almost immediate response was found with precipitation. These results indicated that short duration, high intensity storms, where soil wetting is incomplete, are primarily responsible for groundwater recharge in Hong Kong.

Alley (1984) studied three water balance methods on 10 drainage basins (29-126 square miles in area) in New Jersey. Alley's methods and results are described in figure 14. The first method, developed by Thornthwaite and Mather, identifies the dS component as a function of precipitation and evapotranspiration. Runoff coefficients were estimated based on local cover and slope conditions. Excess soil moisture was assumed to result in groundwater recharge. The method is easy to use but soil moisture showed large variability.

The second method cited by Alley (1984) was developed by Palmer and is similar to

[1] Irrigation is a water input because the recharge component (i.e., the amount of water passing through the unsaturated zone) is increased, thus affecting contaminant transport.

Figure 14: Summary of Fitted Model Parameters for Estimating Recharge (Alley, 1984). Published by the American Geophysical Union

Parameter	Mean	Coefficient of Variation	Range
model T1			
I	1.4	1.3	0.003-5.4
r1	0.68	0.15	0.50 -0.77
model T2			
I	3.5	0.92	0.34 -10
r1	0.61	0.35	0.17 -0.84
r2	0.076	0.50	0.002-0.13
model T3			
I	7.3	1.3	0.003-25
r1	0.46	0.77	0.001-0.77
r3	0.46	0.93	0.014-0.99
model P			
I1	0.82	1.2	0.001-2.8
I2	0.44	1.4	0.001-1.6
r3	0.68	0.14	0.50 -0.78
model abcd			
a	0.992	0.007	0.975-0.999
b	30.	0.35	14-50
c	0.16	1.0	0.001-0.46
d	0.26	1.5	0.007-1.0

T = Thornthwaite's Model
P = Palmer's Model
abcd = Thomas' Model
I(1,2) = soil water capacity (layers 1 and 2)
r1,2,3 = runoff coefficients
a = saturated soil runoff coefficient
b = $(ET + dS)_{max}$ coefficient
c = groundwater coefficient
d = (groundwater residence time)$^{-1}$

Thornthwaite and Mather's but considered two layers in the unsaturated zone. It was assumed that water cannot enter the lower layer until the top layer reaches saturation. The assumption that excess soil moisture leads to groundwater recharge showed inaccuracies due to large variations in the ***dS*** component. These were particularly evident when the top layer exhibited a large degree of macroporosity, leading to a significant amount of unsaturated flow.

The third method presented in Alley (1984) was an ***abcd*** model developed by Thomas. This method modified equation ***2*** in developing coefficients for runoff, evapotranspiration, soil storage, and groundwater recharge. The groundwater component showed considerable variation, largely because it is affected by the other components in the equation.

Freeze and Banner (1970) studied recharge to Canadian prairies and found that recharge events were isolated in time and space. Their ***P***, ***ET***, and ***dS*** showed strong areal variations and could not be used to determine recharge over large areas. The problem of lateral soil flow was particularly troublesome.

A study of groundwater recharge on eastern Long Island, New York (Steenhuis *et al.*, 1985) showed similar results. Precipitation varied significantly in duration, amount, and intensity; soil storage was influenced strongly by variations in permeability and layering; and runoff and evapotranspiration could not be adequately approximated.

Water balance equations must be approached with caution. For systems in which runoff and evapotranspiration are insignificant or can be accurately determined or approximated, soils are relatively uniform, and precipitation events are well recorded, water balance methods appear to give reliable estimates of groundwater recharge. The method has been shown to be accurate for point estimates of recharge (Remson, 1986; Fenn *et al.*, 1975). However, in diffuse source areas or areas where the other conditions do not hold, water balance methods are not a good choice.

Method F

A theoretically correct method for determining recharge is estimating vertical water flux as a function of moisture content and hydraulic conductivity. Discussing the basis for this relationship is beyond the scope of this paper, but some applications will be presented. Stephens and Knowlton (1986) used two methods to estimate recharge. In one, moisture content with depth (using neutron probes) was correlated with the corresponding hydraulic conductivity. Hydraulic conductivity provides a measure of vertical water flux. This method requires accurate determination of hydraulic conductivity as a function of water content. This becomes difficult at low moisture contents, in non-uniform soils, or when lateral water flux is significant.

Their second method estimates recharge from the relationship between matric potential and hydraulic conductivity. A potential hysteretic effect leads to an overestimation of recharge. Estimates for recharge were 0.70-3.66 cm/year with the first method and 3.70 cm/year with the second.

Rehm et al. (1982) used a Darcy flux method to calculate recharge in upland prairie locations in central North Dakota. In the study area, water table wells and piezometers were nested between the water table and the first zone of strongly lateral groundwater flow. Darcy's law was then used to determine recharge. The model worked well in homogenous media, but showed considerable variation for anisotropic conditions. Despite this, estimates of average recharge rate were comparable to those obtained by hydrograph analysis and water budgeting for the same area.

Wikrameratna and Reeve (1984) used a mathematical model designed to fit a least-squares approach to recharge of various types. By establishing a grid system with points or recharge nodes and then type of recharge occurring at each node, and by applying the model, the authors reduced variation by 50-70 percent compared to implied recharge techniques. In deriving the node approach it was assumed that the major recharge areas and hydraulic processes affecting recharge at each node were known.

Other authors have derived mathematical models of aquifer recharge based on hydrologic properties of the unsaturated zone. These include studies by Morel-Seytoux (1984), Freeze (1969 and 1971), and Freeze and Banner (1970).

Mathematical models work well for one-dimensional, vertical flow problems. They provide good point estimates of recharge but are costly and limited in the extent of application.

Method G

Temperature gradients have been used to monitor water as it moves through the unsaturated zone and into groundwater. They can be used when well established relationships exist between temperature and water content over depth. That relationship is weak in areas with lateral flow. Sammis *et al.* (1982) used the method in Arizona, but the estimate of 9 cm/year is considerably below estimates achieved with Darcy's equation (18 cm/year) and tritium tracers (40 cm/year). Actual recharge is considered to be closest to the Darcy approximation.

Method H

Rehm *et al.* (1982) used a flow-partitioning approach. Establishing blocks with streamlines and equipotentials, water flux for the block was a function of the hydraulic conductivity, potentiometric gradient, and area. As with any mathematical model, use of this method required well established values for conductivity and a "steady-state" system. Results correlated well with Darcy flux and hydrograph analysis methods.

Summary

A variety of methods for determining recharge over large areas have been described. Several variations of each method exist. The choice of method should be a function of local hydrologic conditions, availability of data, and preference. Several states employ more than one method. Others employ none. Studies by Rehm *et al.* (1982), Steenhuis *et al.* (1985), and Sammis *et al.* (1982) indicate that the choice of method is more critical for areas with low or highly variable recharge.

WATER AVAILABLE FOR RECHARGE

Previously discussed methods determine general recharge over large areas. They may be difficult to use and may fail to account for large, localized variations. An alternative or supplement is to determine the amount of water available for recharge.

In water balance equations, precipitation and irrigation can be budgeted among the outputs of runoff, evapotranspiration, drainage and storage. Over large areas, runoff and lateral flow (a component of soil storage) can be ignored or generalized. Storage can be assessed by the thickness of the unsaturated zone and location of the water table. This leaves precipitation, irrigation, and evapotranspiration to describe the amount of water available for recharge. Precipitation can be well established with proper techniques and a sufficient network of recording stations (Harrold *et al.*, 1976). Irrigation is often well established but may be a critical concern where records are poor or non-existent.

Evapotranspiration is a more difficult computation. A number of methods are available, most of which give point estimates of evapotranspiration. Thornthwaite (1948) and Penman conceptualized potential evapotranspiration (PET), defining it as "the evaporation from an extended surface of short green crop which fully shades the ground, exerts little or negligible resistance to the flow of water, and is always well supplied with water. Potential evapotranspiration cannot exceed free water evaporation under the same weather conditions." This concept has application for areas where recharge occurs under saturated conditions (Rosenberg, 1974).

Qualifications should be made when using PET.

The concept is useful only when water flow through the root zone is saturated, fulfilling the requirement of a free water surface. These conditions generally occur following snowmelt, during rainy seasons, or following significant precipitation events.

For regions where significant unsaturated flow occurs through the root zone, the concept of PET is inadequate. These conditions are common in areas of intense storms or in arid regions, particularly if the soil is coarse textured or cracked. Under these circumstances, establishing potential recharge is difficult and it may be best to eliminate this term and attempt to make an evaluation of soil infiltration (Hillel, 1971).

Another qualification is the degree of cover, height, and type of vegetation. Leaf area index has been shown to correlate well with PET rates. Thus row crops, which do not fully shade the ground, show decreased PET compared to rates calculated under ideal situations. Taller crops have more effective exchange of energy with the ambient air and show increased rates of evaporation. Finally, broadleafs are shown to transpire more per unit of leaf area than grasses. Various coefficients can be used to adjust for these effects (Rosenberg, 1974; Fritschen, 1965; Janes, 1960; Brun *et al.*, 1972; Stanhill, 1965; El Nadi and Hudson, 1965; van Bavel and Ehrler, 1968).

Empirical methods can estimate PET. Thornthwaite's (1948) equation to determine PET is based on mean monthly air temperature. Blaney-Criddle (1950) correlate pan evaporation with mean monthly air temperature, relative humidity, and percentage of total yearly daylight hours for each individual month. Jensen and Haise (1963) determine PET from mean air temperature and solar radiation. These methods are summarized in figure 15.

Other methods include direct measurement of evaporation with potential evapotranspirometers, weighing lysimeters, floating lysimeters, pan evaporimeters, and atmometers. Evaporation determined with these instruments can be related to PET with various equations or coefficients which Rosenberg (1974) details.

When relating precipitation and PET to water available for recharge, several considerations must be met, including:

1. Use of reliable data;
2. Proper determination of precipitation, including contouring techniques and consideration of orographic effects;
3. Monthly computation of precipitation, PET, and the factors that affect these terms, including depth of the root zone, and crop height, type, and cover.

Figure 15: Summary of Some Methods for Estimating Potential Evapotranspiration (Rosenberg, 1974)

Thornthwaite Method: $ETP = Ct^a$

ETP = *potential evapotranspiration (in)*
a, C = *constants*
t = *mean monthly temperature (°C)*

Blaney-Criddle Method: $E1 = 0.0167T(114 - RH)$

$E1$ = *evaporation (in)*
T = *mean annual temperature (°F)*
RH = *mean annual relative humidity*

Jensen-Haise Method: $ETP = (0.014T - 0.37)Rs$

ETP = *potential evapotranspiration (in)*
T = *mean air temperature (°C)*
Rs = *evaporation equivalent of the solar radiation (in/day)*

The amount of water potentially available for recharge is not linearly related to the amount of contaminant that may be transported. During initial leaching, solute concentrations are high. As recharge continues, concentration decreases until at some point the concentration in drainage water is less than that in the aquifer. Drainage water then begins to dilute the aquifer.

The amount of recharge where hazard risk is maximized depends on contaminant toxicity, mobility, and quantity; moisture and attenuation characteristics of the unsaturated zone; flow velocity; and transport characteristics of the aquifer. Evaluating all these factors in a general rating system is impractical. When recharge rates are generally insufficient for aquifer dilution, it may be assumed that each additional, equal recharge increment leads to increased hazard risk. In equation *1a*, this increment exists in a 1:1 relationship with hazard risk and can be mathematically adjusted. In equation *1b*, it is necessary to establish a rating scale for available recharge.

Development of a factor index for recharge may proceed as follows:

Factor --- *Recharge (R)*
implementation --- Equation 1a*: determine (P + I -PET) and use this value for R;*
Equation 1b*: determine (P + I - PET) or recharge across an aquifer and*

use one of th'ese terms as a factor. Index score is determined from a rating scale.

Correctors --- If using PET, make adjustments for vegetation effects if data is available.

Identifiers --- Identify any assumptions made in the analysis. These may include methods used or recharge conditions considered.

A sample rating scale was shown in Figure 9.

PERMEABILITY OF THE VADOSE ZONE

Permeability influences the rate of water movement through porous media. This in turn affects the rate at which a contaminant is transported and attenuated. Most rating methods assume that the layer of lowest permeability between the land surface and the top of the aquifer controls the rate of vertical water flux, often regardless of its thickness or position with respect to depth or other layers.

Permeability is typically established as a function of the texture of the unsaturated zone media. Figure 16 lists permeabilities of various geologic materials. General textural classifications, degree of fracturing, and thickness are some parameters describing permeability. Differences in structure, texture, porosity, and moisture content cause significant variation in permeability over short distances. A number of potential correction terms can adjust for these effects.

Figure 16: Permeabilities of some geologic materials (Fetter,1980; Silka and Swearingen, 1978;)

Soils	*Permeability (m/sec)*
clays	10^{-9} - 10^{-8}
peat	10^{-9} - 10^{-7}
silt	10^{-8} - 10^{-7}
loams	10^{-7} - 10^{-5}
very fine sands	10^{-6} - 10^{-5}
fine sands	10^{-5} - 10^{-4}
coarse sands	10^{-4} - 10^{-3}
sands with gravel	10^{-3} - 10^{-2}
gravels	*greater than* 10^{-2}
Consolidated materials	
unfractured shale, igneous and metamorphic rock	10^{-9}
siltstone, massive limestone and dolomite	10^{-8}
compact sandstone, moderately fractured shale, igneous, and metamorphic rock	10^{-7} - 10^{-5}
poorly cemented sandstone, highly fractured shale, igneous, and metamorphic rock, permeable basalt and lavas, cavernous limestone and dolomite	*greater than* 10^{-5}

Hillel and Talpaz (1977) studied soil water dynamics in layered soils. Using combinations of a sand, clay, and silt with saturated hydraulic conductivities (*Ks*) of 2.5 X 10^{-5}, 0.2 X 10^{-5}, and 0.7 X 10^{-5} meters per second respectively, they determined cumulative drainage and moisture profile with depth. Soil columns were initially saturated and allowed to drain gravimetrically for ten days. Figure 17 illustrates their findings.

Sublayers tend to dominate drainage due to increased thickness. Sand sublayers drain quickly compared to clay and loam layers. As texture of the top layer becomes finer, drainage decreases in a sublayer class. Figure 18 illustrates the effects of evaporation. Cumulative drainage is lower for all combinations. Sand sublayers again have increased drainage compared to finer textures. The effect of top

Figure 17: Cumulative drainage from initially saturated two-layer profiles. S, L, and C designate sand, loam, and clay, respectively (Hillel and Talpaz, 1977)

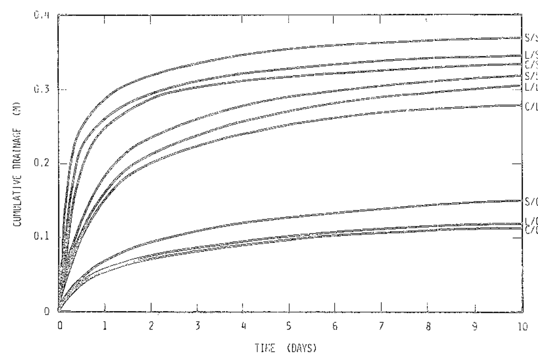


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Figure 18: Cumulative drainage from initially saturated layered profiles during simultaneous drainage and evaporation (Hillel and Talpaz, 1977)

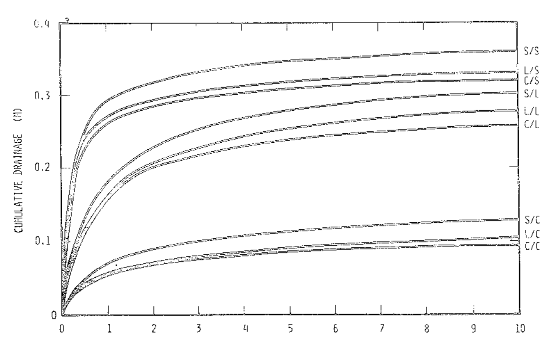

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Figure 19: Water balance of layered profiles for a 10-day simulation, including two rainstorms (first and third nights) totaling 144.0 mm, and two periods of evaporation (the first during the two days between rainstorms; the second during the eight days after the last rainstorm). Symbols S, L, C stand for sand, loam, and clay, respectively. S/C represents a two-layered profile of sand over clay, and other layered profiles are designated accordingly. Profiles with asterisk were subjected to the same rainfall with doubled intensity (Hillel and Talpaz, 1977).

Soil profile	Total infiltration(mm)	Total runoff(mm)	Total evaporation(mm)	Total drainage(mm)	Storage increment(mm)	Storage efficiency(mm)
Sand	144.0	0.0	66.3	1.2	76.9	53.3
Loam	144.0	0.0	78.7	0.0	65.7	45.5
Clay	144.0	0.0	87.3	0.0	57.1	39.5
S/L	144.0	0.0	51.2	0.5	92.7	64.2
S/C	144.0	0.0	49.8	0.02	94.6	65.5
L/S	144.0	0.0	93.4	0.0	51.0	35.3
L/C	144.0	0.0	70.7	0.02	73.7	51.0
C/S	144.0	0.0	97.4	0.02	47.0	32.5
C/L	144.0	0.0	94.4	0.01	50.0	34.6
S/L/C	144.0	0.0	47.5	0.03	96.9	67.1
C/L/S	144.0	0.0	104.7	0.0	39.7	27.5
C/S/C	144.0	0.0	89.6	0.2	54.8	38.0
S/C/S	144.0	0.0	60.9	0.0	83.4	57.8
C/S/L	144.0	0.0	90.3	0.04	54.1	37.5
L/S/C	144.0	0.0	85.3	0.02	59.1	40.9
C/S*	119.1	25.3	91.4	0.0	28.7	19.9
C/L/S*	122.1	22.3	92.7	0.0	29.4	20.4

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layer texture is uncertain, though evaporation increases with finer textures.

Figure 19 summarizes results from infiltration on soils allowed to drain gravimetrically for 10 days. Sand layers above fine textured layers showed much larger storage functions than fine over coarse profiles. Total drainage was low except on the pure sand profile. Water moved from fine textured layers into coarse layers prior to saturation of the top layer. This is illustrated by the occurrence of drainage in the soil columns.

An equation was developed by Selim et al., (1977) to describe the transport of chlorine and fluometuron (1,1-dimethyl 1-3-(a,a,a,-trifluoro-m-tolyl) urea) through an individual layer of a multilayered soil:

equation 3

$$dCi/dx = (Qi + pi(dSi/dt) + \theta i(dCi/dt))/ (\theta iDi(d/dx) - q)$$

where

Ci = concentration of solute in solution in ug/cm ^{3}, for ith layer
x = distance from soil surface
Q = sink for irreversible solute interaction (ug/cm ^{3}– hr)
p = bulk density (g/cm ^{3})
S = amount of solute adsorbed by soil (ug/g)
t = time (hr)
θ = moisture content (cm ^{3}/cm ^{3})
D = solute dispersion coefficient (cm ^{2}/hr)
q = Darcy soil-water flow velocity (cm/hr)

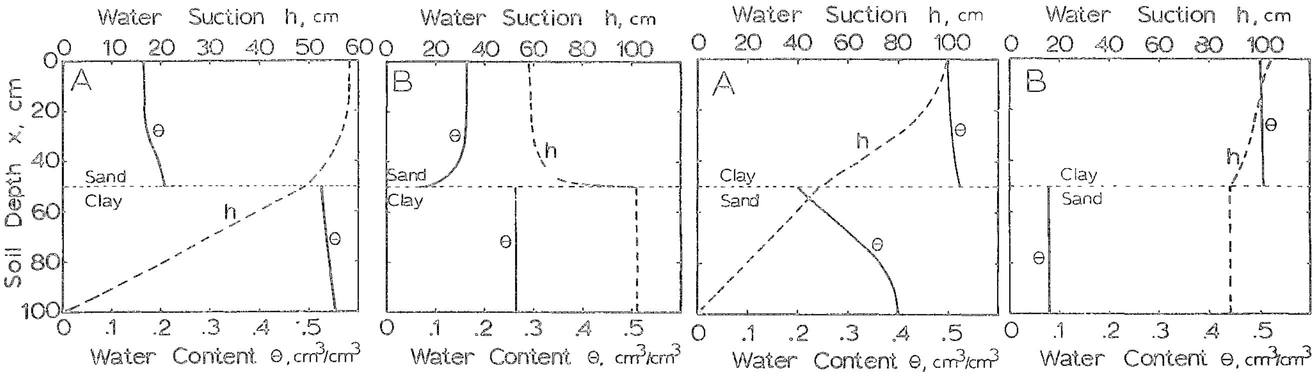
As (dCi/dx) increases, less solute is transported. Increasing Q, p, and S results in decreased solute transport. Transport rate increases as D, q, and O increase with depth.

Studies were performed on two soils, a fine sand and a loam. Combinations of layering were examined, varying in thickness and position. Experimental results indicated that layers did not act as independent units but as a single homogenous unit which could best be described as intermediate between the two layers. Principal factors affecting effluent volume and solute concentration were found to be thickness and retardation factor for each layer.

Moisture profiles were studied as a function of layering and water table position. Results (figure 20) indicate that the fine textured layer dominates the moisture profile, but that a water table near the surface is more likely to induce unsaturated flow and that this flow into a sand layer occurs prior to the saturation of a clay layer.

Hillel and Gardner (1970) studied the effect of a surface impeding crust on vertical water flux. They developed the following equation to fit this situation:

Figure 20: Soil water content (θ) and water suction (h) versus depth in sand-clay and clay-sand profiles having a water table at (A) x=100-cm depth and (B) great depth (x approaches infinity) (Selim et al., 1977)



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equation 4

$$I = (dO)\mathrm{sqrt}(Ku^2Rc^2 + 2Ku(dH/dO)t) - KuRc$$

where

I = infiltration (cm)
O = water content increment of wetted zone in infiltrating soil
Ku = unsaturated hydraulic conductivity (cm/hr)
Rc = hydraulic resistance of the crust (hr)
H = effective head drop between surface and wetting front (cm)
t = time (hr)

Figure 21: I vs. $\sqrt{t}$ for uniform and crusted profiles. The I data are corrected for the initial quantity of water sorbed by the crusts (Hillel and Gardner, 1970)

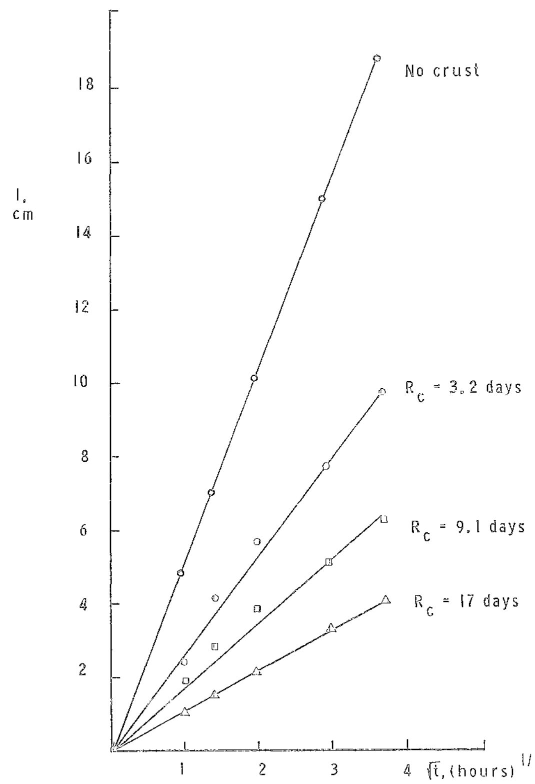

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Figure 21 summarizes Hillel and Gardner's results. If K is held constant for the crust, increasing the crust resistance decreases infiltration at a decreasing rate.

Another effect of layering was shown to be the depth at which different layers are encountered. Miller and Gardner (1962) varied the depth to a sand layer (0.1-0.3 mm diameter) below a silt loam topsoil. A sand layer near the surface greatly decreased infiltration rate, but the impact of this decreased with depth (figure 22). It is likely that the infiltration rate of the topsoil was highly variable, resulting in an unstable wetting front. This variation decreased with depth. The main effect on downward movement of a wetting front was the difference in interlayer pore diameter (figure 23).

Palmquist and Johnson (1962), studying the movement of nuclear wastes in Colorado, found that when a coarse textured layer (diameter = 0.47 mm) overlaid a fine textured layer (diameter = 0.036 mm), thin layers of wetting zones emerged in times less than those predicted by flow equations. With fine over coarse sequences there was large lateral spread of the wastes over time, but individual threads of infiltration into the coarse material were observed. The difference in area of wetting fronts was 100:1 for coarse-over-fine and fine-over-coarse sequences, respectively. This study illustrated a potential effect of layering on lateral

Figure 22: Effect of distance from the water source on time between wetting front reaching and crossing soil-sand interface and on rate of decrease of infiltration rate due to the interface (Miller and Gardner, 1962)

Distance from water source source to sand layer*	Time required for water to cross soil-sand interface		Average rate of decrease of infiltration rate after wetting front reached sand#	
	downward wetting	upward wetting	downward wetting	upward wetting
(cm)	(sec)	(sec)	ml/sec/sec X 10^{-5}	
4.0	105	195	13.7	14.1
8.0	180	550	2.1	1.3
12.0	390	1860	0.9	0.3

* 0.1 to 0.3 mm sand.
# Based on time interval from when wetting front reached sand until observance of the minimum infiltration rate due to the sand

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Figure 23: Summary of pore-size distribution in quartz materials used for layers* and their effect upon water movement into the layer (Miller and Gardner, 1962)

Diameter of separate	Diameter range of most pores	Proportion of pores in indicated diameter range	Interval between times water reached and entered sand	
			downward wetting	upward wetting
(mm)	(mm)	(%)	(sec)	(sec)
<0.05	<0.04	93	-	-
0.05-0.1	<0.04	97	-	-
0.1-0.3	0.06-0.12	82	180	550
0.3-0.5	0.08-0.20	79	260	4120
0.5-1.0	0.20-0.60	77	303	(did not enter sand#)

* Sand-soil interface was 8 cm. from soil-water contact
# Time period of 15 hours

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flow of contaminants. Layers which cause lateral water transport increase contaminant and recharge capabilities of adjacent sites.

It is apparent that the assumption of a fine-textured layer dominating the flow regime of an unsaturated profile holds true, but with qualifications. An accurate assessment of the true permeability should consider the thickness, sequence, and position of the various layers, pore-size distribution between layers, and the effects of horizontal flow.

It may be desirable to separate the analysis of unsaturated zone permeability into infiltration through the root zone and permeability-of-the-vadose-zone. The root zone is eliminated from this factor analysis and incorporated into a separate analytical step (discussed later as Infiltration).

Development of an index for permeability-of-the-vadose-zone may proceed as follows:

Factor --- *Permeability-of-the-vadose-zone (P)*

Implementation --- *Factor scores are determined from rating scales which consider average permeability rates of geologic materials. The layer of lowest permeability is evaluated unless overriding conditions, such as the occurrence of karst aquifers, exist. Permeability may be determined from the land surface or some desired depth (ex. bottom of root zone) to the top of confined aquifers or the seasonally high water table.*

Correctors --- *1. Thickness:as the assessed layer becomes thicker, it reduces permeability.*

2. Pore Size Distribution:as the average pore-size distribution between the impeding layer and adjacent layers increases, permeability is reduced.

3. Number of Layers:permeability is reduced as layering increases.

4. Position of Impeding Layer:impeding layers near the surface greatly restrict total infiltration, which reduces the amount of water available for recharge. If infiltration is a factor being analyzed, this corrector may be deleted.

Identifiers --- *1. Presence of impermeable layers restrict vertical flow and may lead to perched conditions or extensive lateral flow.*

2. Numerous discontinuous clay lenses across the evaluation area cause large localized variations in vertical and lateral flux.

3. Discontinuous confining layer or occurrence of karst conditions across the evaluation area may lead to extensive, localized variations in vertical flux.

A factor scale for permeability was illustrated in figure 3. Correction terms are developed in figure 24. Examples of indices are presented in figure 25.

Figure 24: Correction terms for permeability

Correction term	Effect on permeability	Comments
1. Thickness of of layers	As the thickness of an individual layer in a layered profile increases, it exerts a greater effect on the overall overall permeability of the profile. This effect is more pronounced for layers of low permeability.	Literature indicates that two layers of differing texture act as a single layer, with permeability related to the thickness of the respective layers.
2. Pore-size distribution	As the difference in pore-size between adjacent layers increases, permeability is reduced.	The reduction in permeability is related to soil pressure difference between wet and dry layers. The effect is difficult to quantify but is illustrated by the difference in permeabilities for poorly sorted and well sorted glacial outwash.
3. Number of layers	As layering increases, permeability is reduced.	This relates directly to pore-size distribution. Vertical wetting is controlled by pressure distribution with depth. This distribution becomes more variable as layering increases.
4. Position of an impeding layer	Impeding layers near the land surface greatly restrict vertical flow.	This effect increases as unsaturated flow increases in the soil water zone. This is likely on soils with macroporosity.

Figure 25: Example indices for permeability

Index*	Description**
$5.5_{1,4;2}$	Site score of 5.5 indicates a permeability of approximately $10^{-5.7}$ meters per second. The geologic material may be a poorly-cemented sandstone or fractured shale, igneous, or metamorphic rock. The infiltration rate has been mathematically adjusted to consider the effects of layer thickness and position. The site is located in an area with numerous clay lenses.
$9.8_{;3}$	Site score of 9.8 indicates a permeability of approximately 10^{-2} meters per second. The geologic materials may include basalt or fractured and cavernous limestone. The site is located in an area with discontinuous confining layers or karst aquifers.
1.7_{3}	Site score of 1.7 indicates a permeability of approximately $10^{-8.2}$ meters per second. The geologic materials may include siltstone or unfractured shale, igneous, or metamorphic rock. The site is located in an area with numerous layers in the vertical profile and has been mathematically adjusted for this effect.

* Index consists of a site score for permeability with correction terms and identifiers. Correctors appear before the semi-colon and identifiers after it in the index. This format is arbitrary.

** Site scores are applied to a rating scale developed in this paper. The mathematical values of correctors vary locally and should be derived accordingly. These values are not discussed here, although the general effect of a correction term is discussed in the paper. Correction term and identifier numbers correspond with those discussed in this section.

DEPTH-TO-WATER

Thickness of the unsaturated zone or depth-to-water is the distance from the land surface or the bottom of a contaminant layer to the top of a confined aquifer or seasonally high water table. It affects the amount of potential recharge water that can be stored and the moisture profile (due to redistribution processes) above the water table. Most attenuation processes occur in the soil zone, but thickness may have an influence below this as a result of specific chemical or physical environments, such as pH or clay type and content.

The influence of thickness on saturated flow is described relatively easily if depth-to-water and relative porosities of geologic media with depth are known. These variations influence the quantity of infiltrating water that may be stored. Regions with shallow depth-to-water are often well described. Deeper unsaturated profiles are often poorly described, but the effect of thickness may diminish rapidly beyond the soil zone or below confining layers.

Unsaturated flow is difficult to describe mathematically but may be important to overall recharge, particularly in arid regions, areas with intense storms, and regions with deep water table aquifers. The effect of distance to water on unsaturated flow is described by Selim et al. (1977). Establishment of a moisture profile is necessary to predict accurately the amount of recharge as a function of depth-to-water, although the amount of unsaturated flow decreases rapidly as depth increases.

Development of an index for depth-to-water may proceed as follows:

Factor — Thickness or Depth-to-Water (D)

Implementation— Scores are determined from scales which evaluate distance from the *land surface or some specified depth to the top of the seasonally high water table or the top of confined aquifers.*

Correctors --- *Depth-to-Bedrock: If the uppermost bedrock deposit strongly influences recharge processes, it may be used as an adjustment or alternative factor for depth-to-water.*

Identifiers --- *1. Karst: evaluation area has undergone extensive karstification, which may lead to large variations in thickness over relatively small areas due to the presence of primary or secondary sinkholes. These are not found in available maps.*

2. Discontinuous confining layers: numerous discontinuities in confining layers occur over artesian aquifers and may lead to large, localized variations in depth-to-water. These discontinuities are not found in available maps.

3. Buried Drift Aquifers: numerous buried drift aquifers occur in the evaluation area, leading to large, localized variations in depth-to-water. These aquifers are not described in available maps.

4. Clay Lenses: numerous clay lenses are present in the evaluation area, leading to localized perched conditions. Depth-to-water may therefore be depth to a perched aquifer.

A rating scale for depth-to-water was illustrated in figure 8. Figure 26 illustrates example indices for depth-to-water.

Figure 26: Example indices for depth to water

Index	Description*
$7.0_{:2,4}$	*Site score of 7.0 indicates a depth-to-water of 15.3 feet. The site is located in an area with discontinuous confining layers and numerous clay lenses. These increase the variability in depth-to-water and may lead to perched conditions.*
10.0	*Water table or aquifer is at or near the surface.*
$3.4_{1;1}$	*Site score of 3.4 indicates a depth to water of approximately 47 feet. This has been corrected to include the effect of depth-to-bedrock, which implies that the uppermost bedrock layer influences flow processes. The site is located in areas of karst aquifers, indicating the potential for rapid response systems.*

** see comments at bottom of Figure 25*

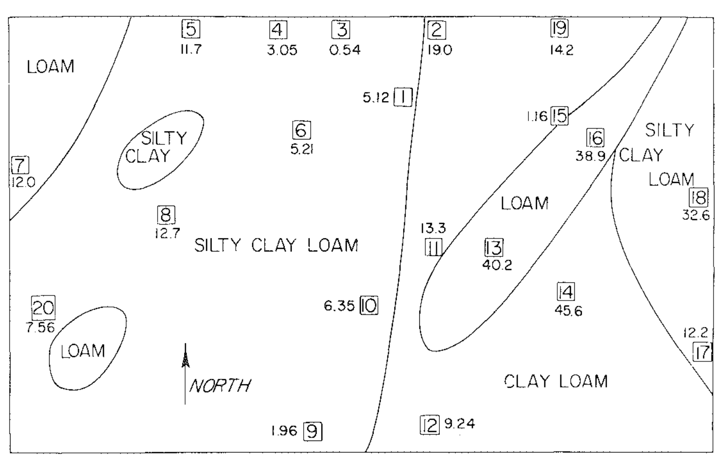
INFILTRATION

Infiltration is generally evaluated as the permeability of the root zone, a depth ranging from 2-4 feet for most locations. The evaluation is based on average permeabilities of soil materials, as shown earlier (figure 16). Infiltration influences the amount and rate of water flow through the root zone, into underlying material. Once water reaches this point, it is below the zone of maximum potential for evapotranspiration. It is then largely influenced by redistribution processes in the unsaturated zone. This leads to aquifer recharge at some point in time and space, provided the water is not chemically altered. Thus, as infiltration increases, recharge at some point is likely to increase. Infiltration rate also affects attenuation of contaminants, but this may best be evaluated as a correction term for an attenuation factor.

Data availability aside, infiltration is the most difficult parameter to assess in a groundwater sensitivity analysis. Infiltration is a function of soil texture and structure, soil wetness, land slope, meteorological conditions, vegetation, degree of soil surface crusting, and degree of macroporosity. Some of these parameters are in turn affected by other processes. Because of this complexity, there is a tremendous spatial variability in hydrologic properties of field soils.

Nielson et al. (1973) studied steady-state infiltration rate at 20 locations in a 150 hectare field. The locations, with infiltration rates, are shown in figure 27. There was considerable

Figure 27: Diagram of the field site showing locations of the plots (1 through 20). The number by each plot indicates the measured value of the steady-state infiltration rate (cm/day). Textures at the soil surface of the Panoche soil are also indicated (Nielson et al., 1973)



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variation among sites on the same soil and within short distances of each other. Infiltration rate showed a log normal distribution and was poorly correlated with texture (figure 28). Correlation coefficients improved with depth in the upper 60-90 cm of soil, then showed no further relationship with depth.

Figure 28: Correlation of steady-state infiltration and hydraulic conductivity with soil properties (Nielson et al., 1973)

Soil depth (cm)	n	r^2: % clay	% sand	θ vs. Io	θ vs. lnIo
30.5	20	-0.447	0.468	-0.2431	-0.1887
61.0	20	-0.416	0.408	0.1099	0.2357
91.4	20	-0.501	0.470	0.1553	0.4067
121.0	20	-0.524	0.492	0.0997	0.4094
152.4	20	-0.366	0.233	0.1282	0.3723
182.9	20	-0.367	0.172	0.0153	0.2479

r^2 = simple correlation coefficient
θ = soil moisture content (cm^3/cm^3)
Io = initial soil infiltration capacity

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The poor correlation near the surface was probably due to soil macroporosity, which led to variations in infiltration rate other than those predicted by pore-size alone. The low correlation between water content and infiltration rate at time zero support this assumption, although laboratory experiments with homogenous, isotropic soils show a strong negative correlation between the two. The authors claim that three hydrologic units exist, based on infiltration properties. These are:

1 - plots 5,7,8,20 : moderate to high infiltration rates, moderate hydraulic conductivity (K) at high soil water content (-θ), and low K values at low values of θ.

2 - plots 1,3,4,6,9,10,11,12,15 : low infiltration rates, low K values at high values of θ, and variable K values at low values of -θ.

3 - plots 13,14,16,17,18 : high infiltration rates, high K values at low and high values of -θ.

The units correlate poorly with soil type.

Luxmoore et al. (1981) is a study of infiltration rates into weathered shale at a site adjacent to low-level radioactive waste. Their results for 48 plots on a 250 square meter field are shown in figure 29. They indicate both a wide range and large variability in infiltration rate, with no significant distributional pattern. Up to one-half of the field variation occurs in any square meter block. The soil, an ultisol, was considered uniform in textural distribution.

Sisson and Wierenga (1981) obtained similar results (figure 30). No significant distribution pattern of steady-state infiltration rate or variance was found for 25 plots in a 1000 square meter field. A large amount of water infiltrated in a relatively small area. The soil was an entisol with 70 cm of silty clay loam over fine to medium sands. No correlations of profile characteristics with infiltration rate were made.

Sharma et al. (1980) studied infiltration in a 9.6 hectare watershed in Oklahoma. Three silt loam soils were identified in upland, flat, and alluvial locations. Average slope was approximately 3 percent with native grass cover. They found no pattern of infiltration with soil type or position in the watershed. An arithmetic mean[1] of infiltration rate was established for the watershed, with a relatively small standard error

Figure 29: Steady-state infiltration rates for 48 plots. Total area is 250 m^2 on a Typic Hapludult soil (Luxmoore et al., 1981)

x\y	A	B	C	D	E	F
1	2.4	0.6	0.7	0.3	1.1	2.3
2	7.9	0.4	6.8	3.1	0.7	3.7
3	1.9	9.8	2.2	1.8	1.8	4.1
4	9.2	0.9	1.4	4.6	3.1	0.6
5	0.9	4.1	1.5	0.3	0.7	0.8
6	4.7	1.3	1.4	2.6	3.2	1.8
7	3.5	17.1	1.8	0.4	3.9	3.0
8	3.2	6.0	5.3	2.1	1.9	1.1

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Figure 30: Distribution and variance of steady-state infiltration rate for 25 plots on a Typic Torrifluvent soil (Sisson and Wierenga, 1981)

Row or column number	Row u	Row s^2	Column u	Column s^2
1	8.48	1.76	0.95	5.60
2	8.45	1.85	7.69	2.57
3	8.25	5.17	7.50	0.20
4	8.96	6.34	8.79	1.18
5	7.76	--	8.91	--
1°	0.92	0.0045	0.95	0.013
2°	0.92	0.0010	0.88	0.0094
3°	0.91	0.012	0.87	0.00068
4°	0.95	0.0028	0.94	0.0028
5°	0.89	0.025	0.95	0.015

u = average steady-state infiltration rate in cm/hr
s^2 = variance
° = log transformation

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[1] Several authors have found that use of geometric or harmonic means are more accurate field hydraulic property indicators.

Figure 31: Arithmetic means of measured I (infiltration in cm), ul, and standard errors, SE, at selected times for the whole watershed and also separately for the three soils of the R-5 watershed (Sharma et al., 1980)

t	Whole watershed		Soil 1		Soil 2		Soil 3	
(min)	ul(cm)	2SE	ul(cm)	2SE	ul(cm)	2SE	ul(cm)	2SE
2	0.849	0.134	0.776	0.168	1.055	0.216	0.572	0.305
5	1.477	0.258	1.353	0.332	1.856	0.412	0.920	0.534
10	2.215	0.410	2.033	0.534	2.792	0.648	1.335	0.858
15	2.818	0.534	2.600	0.708	3.535	0.828	1.684	1.136
30	4.271	0.840	4.054	1.216	5.176	1.196	2.562	1.828
60	6.624	1.377	6.514	2.110	7.654	1.832	4.050	3.152

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(figure 31). Despite this, the three soils were often significantly different in average cumulative infiltration versus time, and showed large individual standard errors. The major factor influencing infiltration rate was pore-size distribution both with depth and laterally.

Several authors use scaling factors to correlate infiltration rates for soils with similar porosities (Peck et al., 1977; Warrick et al., 1977; Wilkinson and Klute, 1959; Klute and Wilkinson, 1958). The concept suggests that two soils have similar conductive properties if multiplying certain characteristic microscopic parameters for one soil by a constant gives the same values for the other soil. For vertical infiltration this characteristic parameter is pore length. The scaling factor is used to statistically relate and compare soils and is established by the relationship:

equation 5

$$a = l / l_{ave}$$

where a = scaling factor
l = pore length (cm)
l_{ave} = average pore length (cm)

Pore length can be related to hydraulic conductivity by the following equation:

equation 6

$$l_r^2 / l^2 = Kr/K$$

where K = hydraulic conductivity (cm/sec)
r = reference soil

Use of scaling factors has given excellent results in comparing infiltration between similar media. The method is not practical for use in a broad analysis of groundwater susceptibility, but establishes the importance of pore-size distribution on infiltration in the unsaturated profile. Macroporosity or crusting of the soil surface, layering in the vertical profile, and textural distribution are major factors to consider in addition to average permeabilities associated with textural classifications.

Dixon and Peterson (1971) studied the effect of pore-size on volume of flow in the surface layer and established the relationship:

equation 7

$$V = C(P)^4$$

where V = flow volume (cm ^{3})
C = a constant (1/cm)
P = pore-size (cm)

The results of their study (figure 32) indicate that a highly functional channel system, connected at the surface, significantly increases infiltration rate. The effects become more pronounced as channel size and number increase, the surface becomes rougher, and crusting is reduced. The results of a study by Edwards et al. (1979) support these findings (figure 33), indicating that infiltration rate and cumulative infiltration are higher on soils with macropores.

Ehlers (1975) studied the effect of earthworm channels on tilled and untilled loess soils. The volume of channels was approximately two times higher on no-till sites, with cumulative water intake proportional to pore-diameter to the 3.8 power. Ritchie et al. (1972) found hydraulic conductivity to be three times higher on undisturbed versus disturbed cores for Houston black clay, an extensively swelling soil.

Bouma and Dekker (1978) studied the effects of rate and amount of water applied to swelling soils. Their results (figure 34) illustrated the increase in wetting due to macropores as a function of application rate. The effect of amount was not as dramatic as for rate, though infiltrated water tends to move vertically as a wetting front.

Bouma et al. (1978) compared two soils for the effect of structure on infiltration and outflow rates through 100 cm of soil at various water application rates. Their results (figure 35) indicated that soil II, with very strong structure, had increased infiltration and outflow rates, a

Figure 32: Infiltration rates and total infiltration under imposed conditions Ae, Ac, A, or F and naturally occurring conditions D and E (Dixon and Peterson, 1971)

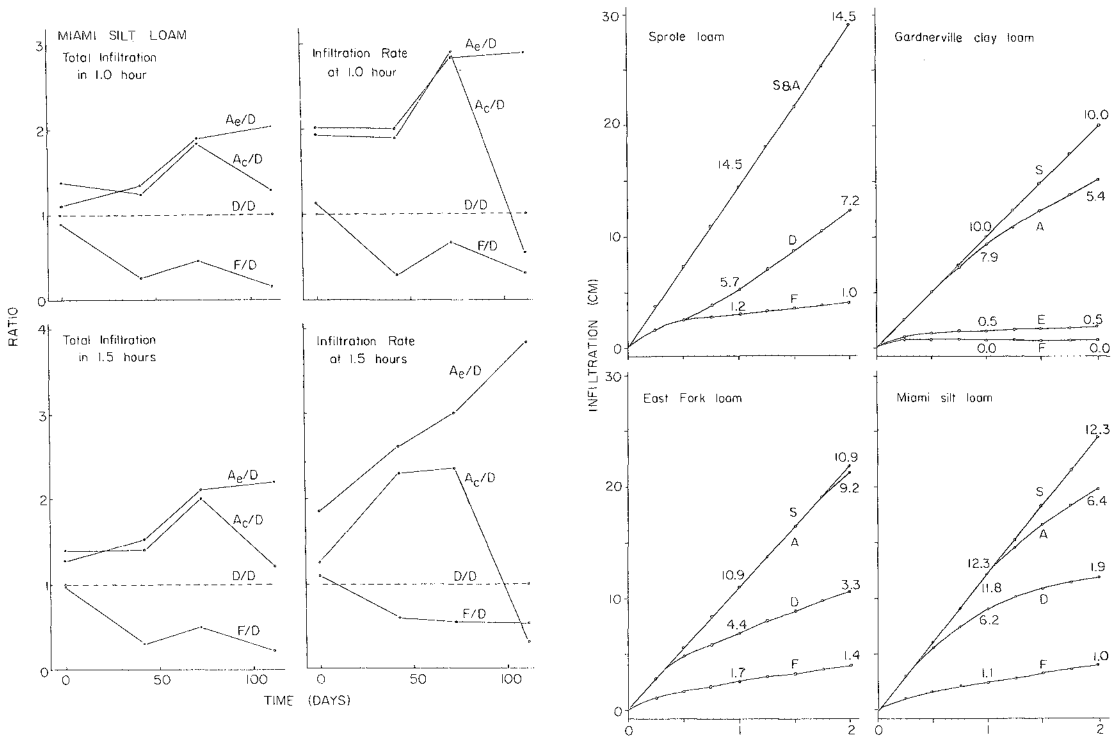

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Figure 33: Cumulative infiltration for a) 50-cm diameter soil with no central hole, b) 50-cm diameter soil with a central hole 60-cm deep and 0.5- or 1.0-cm in diameter, c) 50-cm diameter soil with a central hole 1.0-cm in diameter and 30- or 60-cm deep, d)30- and 50-cm diameter soil with a central hole 0.5-cm in diameter and 60-cm deep (Edwards et al., 1979)

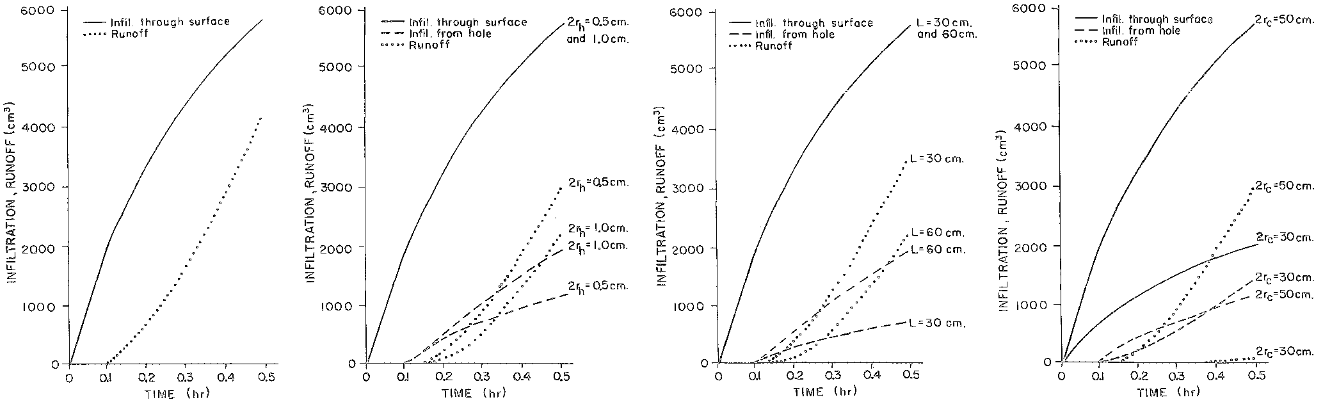

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Figure 34: Application rate effect on depth of water penetration (Bouma and Dekker, 1978)

Application rate (mm/hr)	Application amount (mm)	number of bands at 30-70 cm
40	5	55
40	10	55
500	10	220
16	10	15
8	10	5

shorter time to outflow, and decreased sorption compared to soil I, which had a more moderate structure. After approximately three hours, infiltration became higher in soil I as a result of a wetting front moving through the profile.

Hillel and Gardner (1969) conducted an extensive study of infiltration through crust-topped profiles (figures 36 and 37). A dramatic

Figure 35: Physical data, calculated to document the effects of different flow regimes through large undisturbed cores from upper 20cm of two dry clay soils with contrasting macrostructure (Bouma et al., 1978)

Soil core	Initial moisture condition (Vol %)	Inflow rate (mm/hr)	Period to first outflow (min)	Steady adsorption rate (mm/hr)	Steady outflow rate (mm/hr)
dry soil, low application rate (8mm/hr)					
Ia	41	8.0	45	0.9	7.1
Ib	41	8.6	70	1.1	7.5
IIa	36	9.7	45	1.1	8.6
IIb	38	8.0	28	1.1	6.9
dry soil, high application rate (28mm/hr)					
Ic	38	26.0	14	9.8	16.2
Ib	38	26.0	14	10.7	15.3
IIa	36	28.5	18	1.0	27.5
IIb	39	29.5	10	2.4	27.1
moist soil, high application rate (28mm/hr)					
Ic	52	27.5	5	0.5	27.0
IIa	46	34.0	15	0.9	33.1
IIb	48	32.0	5	1.5	30.5

Soil I - moderate to strong structure throughout profile
Soil II- very strong structure throughout profile

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Figure 36: Effect of surface condition on soil hydraulic properties (Hillel and Gardner, 1969)

Soil characteristics	stable aggregates	slaked crust	puddled crust	cemented crust
	----------Surface condition----------			
Toplayer				
thickness (cm)	1.0	1.0	1.0	1.0
porosity (%)	49.0	45.0	38.0	30.0
hydraulic conductivity (cm/day)	22.1	0.7	0.3	0.1
resistance (days)	0.5	1.4	3.2	9.1
Sublayer				
porosity (%)	48.0	48.0	48.0	48.0
a value	4900	4900	4900	4900
n value	2.0	2.0	2.0	2.0
saturated hydraulic conductivity (cm/day)	16.0	16.0	16.0	16.0
Results				
water flux (cm/day)				
predicted	-	13.0	8.0	4.0
measured	16.8	13.4	9.5	5.0
suction head of subcrust (cm)				
predicted	-	19.0	25.0	36.0
measured	5.0	64.0	88.0	102.0

a,n = soil constants

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decrease in steady-state infiltration was observed with decreased porosity in the upper 1 cm of soil. This effect became more pronounced as the difference in pore-size between the crust and sublayer increased.

Figure 37: Effect of aggregate size on soil hydraulic properties (Hillel and Gardner, 1969)

Crust properties			
Toplayer			
thickness (cm)	1.0	1.0	1.0
porosity(%)	29.0	29.0	28.3
hydraulic conductivity (cm/day)	0.081	0.079	0.078
resistance (days)	12.4	12.6	12.7
Sublayer			
aggregate size (mm)	0–0.2	0.25–0.5	2–5
porosity (%)	49.8	61.7	65.0
a value	4900	63000	2.5×10^9
n value	2.0	4.4	8.0
saturated hydraulic conductivity(cm/day)	16.0	22.0	30.0
Results			
water flux (cm/day)			
predicted	3.2	1.5	1.2
measured	6.1	2.1	2.0
suction head of subcrust (cm)			
predicted	39.0	19.0	15.0
measured	50.0	50.0	26.0

a,n = soil constants

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Figure 38: Effect of freeze-thaw cycles on drainage (Benoit and Bornstein, 1970)

Aggregate size (mm)	saturation −18 °C	saturation −2 °C	0.5 bar pressure −18 °C	0.5 bar pressure −2 °C	average
	relative change (initial/final)				
bulk density (g/cm ^{3})					
0.00-2.00	0.86	0.86	0.93	0.94	0.90
2.00-4.80	0.75	0.77	0.83	0.85	0.80
4.80-19.1	0.73	0.77	0.79	0.80	0.77
water content (cm ^{3}/cm ^{3})					
0.00-2.00	0.90	0.87	0.85	0.86	---
2.00-4.80	0.67	0.67	0.81	0.93	0.88
4.80-19.1	0.59	0.60	0.84	0.86	0.73
hydraulic conductivity (mm/h)					
0.00-2.00	8.14	9.40	1.11	0.87	0.76
2.00-4.80	289.87	170.91	4.93	15.56	120.30
4.80-19.1	177.83	21.50	6.62	16.51	55.62

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Another structural effect is the influence of freeze and thaw cycles. Figure 38 summarizes the findings of a study by Benoit and Bornstein (1970) in which freeze-thaw cycles led to a destruction of soil aggregate structure, reduced conductivities, and decreased drainage. A similar study by Benoit (1973) shows no effect of freeze-thaw cycles for small aggregates.

Wood (1977) studied the hydrologic differences between selected forested and agricultural soils in Hawaii (figure 39). Infiltration rates on forested soils were significantly higher than on agricultural soils, even for the same soil order. Although pore-size and porosity were higher on the forested soils, there was low correlation of these factors with infiltration rate. The author did not explain the large differences in infiltration rate, but greater uniformity of pore-size with depth in forested soils is a possible explanation.

Figure 39: Comparison of infiltration rates for similar soils with differing land-use (Wood, 1977)

Soil	Infiltration rate (cm/hr) forested	pasture	sugarcane	pineapple
Histosols				
soil 1	35.0	1.8	—	—
soil 2	4.0	0.1	—	—
Oxisols				
soil 1	29.3	1.8	—	—
soil 2	30.4	—	1.7	—
soil 3	2.8	—	2.2	—
soil 4	7.1	2.9	1.2	—
Ultisols				
soil 1	39.3	—	—	9.1
soil 2	23.9	4.7	—	1.5
Inceptisols				
soil 1	7.4	0.05	0.08	—
soil 2	17.1	0.5	1.1	—
soil 3	0.3	—	0.03	—
soil 4	2.5	—	0.01	—
soil 5	14.7	—	0.03	—
soil 6	20.2	4.8	—	—
soil 7	1.1	3.7	—	—

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Site-specific factors are difficult to incorporate into a general rating method. There are many complexities and potential adjustments for determining infiltration. Estimates of infiltration based solely on textural classification provide a limited analysis and do not identify potential variation. Areas considered to have low susceptibility to contamination as a result of favorable textural profiles would be more accurately described as having the potential for sites suitable for some waste disposal purpose. Further analysis may then be done addressing the points in this discussion. Qualifiers may be used to describe the types of conditions which are desirable for low-susceptibility in these areas. These conditions might include fine-textured soils with low macroporosity and unstable aggregates, tilled agricultural land, uniform profiles with depth, and regions of low-intensity, short-duration storms.

Generalized assumptions can account for natural variation in infiltration rates, if there is adequate data to support the assumptions.

- *Loam soils form large stable aggregates and are influenced significantly by tillage, freeze-thaw cycles, and local meteorological conditions.*
- *Soils high in 2:1 clays, particularly montmorillonite, are subject to extensive shrinking and swelling, leading to wide variations in infiltration rate.*
- *Tilled agricultural soils generally have lower infiltration rates than untilled soils having macroporosity, unless tillage reduces the effects of soil crusting.*
- *Forested soils generally have more uniform pore-size distribution with depth than agricultural soils of the same soil order, due primarily to land management practices and biological effects.*
- *Localized high intensity, long-duration precipitation events can lead to significant deep infiltration on cracked soils but decreased infiltration on crusted or smooth soils.*
- *Soils of poor structure with high silt content tend to form surface crusts, particularly as precipitation intensity increases.*

Development of an index score for infiltration may proceed as follows:

Factor --- *Infiltration (I)*

Implementation --- *Determined as a rate, in length per unit time, based on average permea bilities of soils and geologic materials. Factor scores are determined from rating scales.*

Correctors --- *1. Degree of macroporosity: as macroporosity increases, infiltration rate increases. This effect may be assessed generally or as several smaller effects, including the influence of shrink-swell, freeze-thaw, soil aggregation, earthworms, tillage, land-use, and vegetation.*

2. Pore-size distribution: as the range in pore-size increases, infiltration decreases (macroporosity effects excluded) and becomes more variable.

3. Crusting: soil surface crusting leads to significant reduction in infiltration rate.

4. Precipitation patterns: high intensity precipitation events lead to rapid infiltration deep into or below the root zone, particularly on soils with macroporosity.

5. Total infiltration: knowledge of the thickness of the root zone and the topographic slope gives an estimate of the amount of water storage potential in the root zone. Sites can be adjusted to consider this effect.

Identifiers --- *Any of the above correction terms may be used as an identifier if they cannot be quantified but are judged to have a significant impact on infiltration or variability of infiltration.*

Figure 40 shows the development of correction terms and identifiers for infiltration. Figure 41 illustrates example indices. A rating scale for infiltration was shown in figure 9.

Figure 40: Correction terms for infiltration

Correction terms	Effect	Comments
1. Degree of macroporosity	As macroporosity increases, infiltration increases	Macroporosity includes effects of soil shrinking, root channels, and biological activity (e.g. earthworm activity). The effect is larger as a soil initially wets, as a pore increases in diameter, and as a pore increases in depth of penetration.
2. Pore-size distribution	As the range in pore size increases, infiltration decreases.	As a soil becomes more poorly-sorted, the pressure differences between adjacent pores increase and lead to reduced water flux. Examples include glacial till, which is generally poorly sorted and has low infiltration rates, and outwash plains, which are often well sorted and permeable.
3. Soil surface crusting	Surface crusting reduces infiltration.	Crusting creates an impermeable layer at the soil surface and reduces infiltration by up to two orders of magnitude.
4. Precipitation patterns	High intensity, long duration precipitation increases infiltration rate initially and total infiltration.	A difficult parameter to adjust for quantitatively unless data exists on the infiltration rates of soils as they relate to precipitation intensity.
5. Total infiltration	———————	This may apply more to recharge but may be used as a separate factor or in a matrix with infiltration rate.

Figure 41: Example indices for infiltration

Index	Description*
$3.9_{2;4}$	Site score of 3.9 indicates an infiltration rate of $10^{-6.5}$ meters per second. It is likely to be a silt-loam unless the correction for pore-size distribution significantly alters the initial infiltration rate. Some pattern of precipitation is described by use of an identifier.
1.1_{3}	Site score of 1.1 indicates an infiltration rate of 10^{-9} meters per second. Soil is crusted as shown by the use of a correction term.
$9.1_{1;1}$	Site score of 9.1 indicates an infiltration rate of 10^{-4} meters per second. The soil exhibits macroporosity, which in turn may be related to texture. The identifier describes some other potential influence of macroporosity which can not be quantified (ex. variability of spatial distribution of macroporosity).

* See comments at bottom of figure 25. Correction term and identifier numbers correspond with those shown in Figure 40.

TOPOGRAPHY

Upland sites are generally at lower risk than lowland sites, other factors being equal. As slope increases, this effect becomes more pronounced.

Topography affects surface runoff and subsurface flow. It may have a further effect on the position of unconfined water tables. These are difficult parameters to assess. Combined with the site specific nature and large variability of topographic relief, it is difficult to establish a definite hydrologic risk due to topography. Use of identifiers which indicate the topographic position of sites or coding (e.g., use of color codes) to indicate slope position may be more useful, practical, and provide more information to the system user.

An example of coding using colored maps is shown in figure 42 (in Color Appendix). Once a rating evaluation has been completed for an area, coding establishes topographic position. Inferences may then be made regarding runoff, lateral flow, and position of the water table. Coding may be done at any scale if adequate topographic maps exist, allowing inferences for the influence of natural features such as river courses.

A general sequence of increasing risk based on topography would be:

1. Upland sites with strong relief,
2. Upland sites with low to moderate relief,
3. Lowland sites with strong relief,
4. Lowland sites with low to moderate relief.

Figure 43 shows this sequence schematically.

Topography may also be used as a correction term for other factors, particularly if runoff coefficients are well established. Rating scales can be developed as necessary.

Development of an index for topography may proceed as follows:

Factor --- *Topography (T)*

Implementation --- *Evaluation is based on percent slope as it affects runoff and subsurface flow. Factor analysis is not recommended, but a rating scale may be used to determine factor scores if desired.*

Correctors --- *1. Slope, as percent:as slope increases, less water is retained on-site and risk decreases.*

2. Slope position:evaluation sites located at downslope positions are at greater hydrologic risk than upland sites due to increased quantities of water delivered from upslope sites.

These correctors may be used in another factor analysis, such as infiltration.

Identifiers --- *Depending on scale of coding, the following identifiers may be used:*

1. Slope position

2. Percent slope

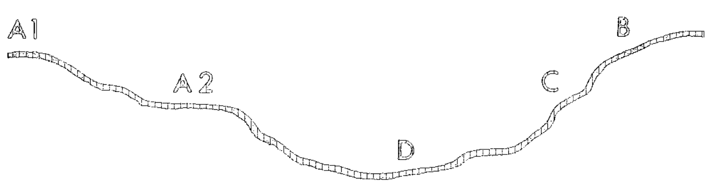
3. Presence of natural features such as floodplains, surface catchments, etc.

Figure 44 illustrates development of correction terms for topography. A rating scale is shown in figure 9. Figure 45 illustrates example indices for topography.

Figure 42: Example for color coding for typography

(See Color Appendix)

Figure 43: Example of hydrogeologic risk as a function of topographic position



A1 - flat uplands, low risk
B - moderately-sloping uplands, low risk
A2 - flat uplands, moderate risk
C - moderately-sloping to steep uplands, low to moderate risk
D - flat bottomlands, high risk

Figure 44: Correction terms for topography

Correction term	*Effect*	*Comments*
1. Percent slope	*As slope increases, risk decreases*	*Slope relates to the amount of water that runs off or is transported to an area. Coefficients exist to describe runoff as a function of percent slope and cover type.*
2. Slope position	*Upland sites are at lower risk than lowland sites, other factors being equal.*	*Greater quantities of water are delivered to sites located downslope, thus increasing water available for recharge and increasing risk to ground-water. Runoff coefficients can be used to calculate the amount of water delivered to or leaving a site with a known slope percent and position. At topographic lows, all water is assumed to infiltrate.*

Figure 45: Example indices for topography

*Index**	*Description**
$4.9_{;3}$	*Site score of 4.9 indicates a slope of 16%. The identifier describes some special feature such as a floodplain.*
$10_{1,2;3}$	*Site score of 10.0 indicates a slope of near 0%. The correctors indicate water inputs from upland sites. The identifier may describe the site as a floodplain or alluvial valley.*
1.0_{2}	*Site score of 1.0 indicates a slope of 50%. The slope has been corrected to consider the effects of water lost due to runoff.*

** See comments at bottom of figure 25. Correction term and identifier numbers correspond to those shown in figure 44 or are arbitrarily described here.*

ATTENUATION

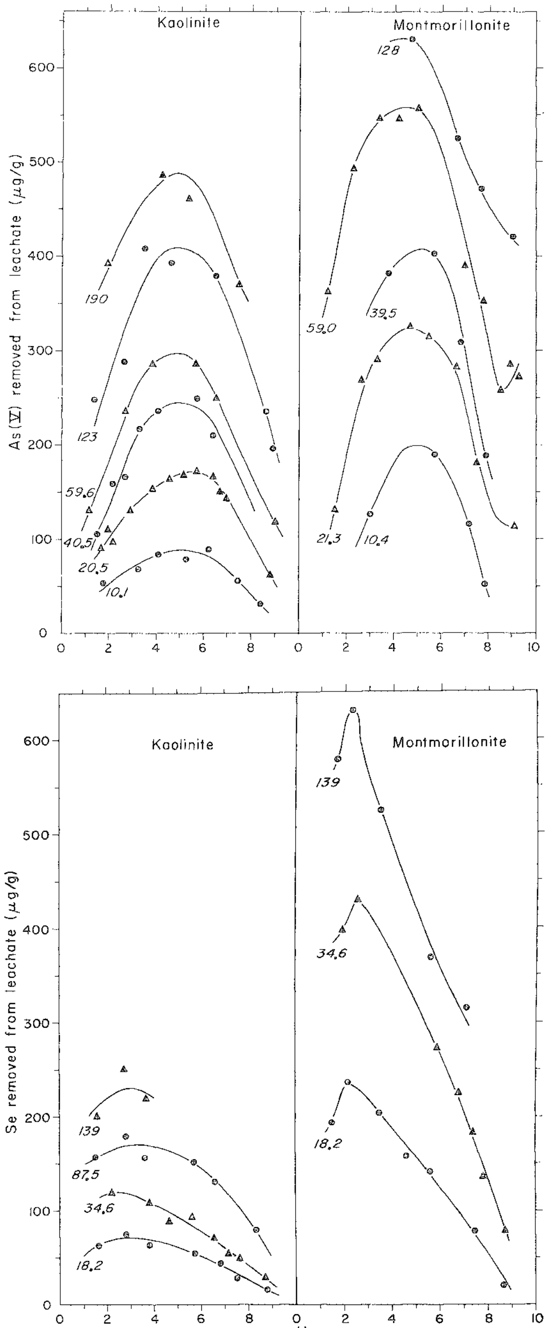
It is not possible to adequately describe hydrogeologic sensitivity to general or specific contaminants without evaluating the potential for contaminant attenuation in the unsaturated zone. There are several broad classes of potential contaminants, each affected by different attenuation processes. Within each class there may be further classifications, unless a worst case situation is always assumed. This assumption is unnecessarily limiting.

Major zones of contaminant attenuation are the upper soil horizons. Unless stated otherwise, the following discussion considers these zones.

Metals

Frost and Griffin (1977) studied arsenic and selenium adsorption by clay minerals at landfill sites. Their results (figure 46), indicated that montmorillonite adsorbs approximately twice the arsenic and more than twice the selenium that kaolinite does. Surface area and cation exchange capacity (CEC) were 2.5 and 5.3 times higher for the montmorillonite. Adsorption was strongly influenced by pH, with maximum sorption at pH 5 for arsenic and pH 2-3 for selenium. They indicated that montmorillonite had a high degree of interlayer metal sorption, highly resistant to desorption. Total quantity of metal ions did not significantly affect the ratio of montmorillonite to kaolinite metal sorption, but sorption decreased as a percent of total metal.

Figure 46: The amount of As(V) or Se(IV) removed from DuPage leachate solutions by kaolinite and montmorillonite at 25°C plotted as a function of pH. Initial solution concentrations are given in ppm. Total solution volume was 52.5 ml (Frost and Griffin, 1977).



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McBride et al. (1981) correlated cadmium sorption and plant uptake with soil properties for nine soils from five soil orders. Their results are shown in figure 47. Total bases, percent exchangeable calcium, CEC, and pH were significantly ($p<0.001$) related to cadmium retention capacity. Clay and organic matter were significant at the 0.05 level; plant uptake was significant at the 0.001 level for percent exchangeable calcium, CEC, and cadmium retention. Clay and organic matter had a weak effect on cadmium uptake, primarily because they relate more to strength of sorption.

Harter (1979) studied lead and copper adsorption in Ap and B2 horizons of 15

Figure 47: Correlation of cadmium retention with soil properties. Nine soils from five soil orders are examined (McBride et al., 1981)

Cadmium retention capacity versus	Correlation coefficient
sum of bases	0.91***
sum of bases, Ap horizon	0.94***
sum of bases, B horizon	0.93***
exchangeable calcium	0.94***
exchangeable calcium, Ap horizon	0.95***
exchangeable calcium, B horizon	0.96***
exchangeable magnesium	0.72**
exchangeable sodium	0.71**
exchangeable potassium	0.45
cation exchange capacity (buffered)	0.82***
cation exchange capacity, Ap horizon	0.82***
cation exchange capacity, B horizon	0.82**
% clay	0.49*
% clay, B horizon	0.67*
% organic matter	0.53*
% organic matter, B horizon	0.66*
pH	0.79***
pH, Ap horizon	0.82**
pH, B horizon	0.83**

Ap = plow layer
* = significant at the 0.05 level
** = significant at the 0.01 level
*** = significant at the 0.001 level

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northeastern U.S. soils. Five soil orders were represented. All soil factors were significant at the 0.05 level, with most factors significant at the 0.01 level (figure 48). Percent sand was negatively correlated with metal retention. Total exchangeable bases and exchangeable calcium were the most highly correlated soil properties. Vermiculite, an expanding 2:1 clay, adsorbed more metal than chlorite, a non-expanding clay. The influence of soil horizon was not significant. Harter found that addition of 0.01M $CaCl_2$ reduced metal sorption by 50-70 percent due to calcium competition for exchange sites.

Street et al. (1977) studied cadmium adsorption in 4 soils amended with sewage sludge (figures 49 and 50). As CEC increased, metal sorption increased. Soil D, with a higher pH and similar CEC compared to soil A, showed increased sorption. Hahne and Kroontje (1973) illustrated the effect of pH on mercury, lead, calcium, and zinc sorption. Solubility decreased as pH increased for zinc, lead, and calcium, with minimum solubilities at a pH of approximately 8.

Bitell and Miller (1974) report on lead, cadmium, and calcium selectivity coefficients for 10-90 percent metal saturation. Their results (figure 51) indicate that no preference is evident for the Ca:Cd system, lead is preferred 2-3:1 over calcium, and cadmium is preferred 2-3:1 over lead. Lead and cadmium show stronger selection relative to calcium on kaolinites when

Figure 48: Correlation coefficients of lead and copper adsorption with soil properties (Harter 1979)

Independent Variable	correlation coefficient: -Pb adsorption capacity- All[2]	A Hor.	B Hor.	-Cu adsorption capacity- All[2]	A Hor.	B Hor.
Cu adsorption capacity	0.894	0.884	0.981	----	----	----
pH	0.670	0.660*	0.823	0.740	0.752	0.767
CEC[1]	0.758	0.649*	0.898	0.818	0.695	0.911
calcium[1]	0.858	0.827	0.967	0.917	0.926	0.916
magnesium[1]	0.687	0.738	0.732	0.652	0.916	0.670*
sum of bases[1]	0.873	0.833	0.961	0.917	0.943	0.911
% sand	−0.556	−0.553	−0.606	−0.621	−0.728	----
% silt	0.516	0.624*	----	0.449*	0.726	----
% clay	----	----	0.721	0.639	0.582*	0.756
% illite	----	----	0.751	0.575	----	0.751
% vermiculite	0.531	----	0.920	0.689	----	0.886
% chlorite	0.534	----	0.622*	0.703	0.742	0.685*
% SiO_2	−0.630	−0.561	0.738	−0.698	−0.723	−0.703
% Al_2O_3	0.498*	----	0.740	0.651	0.630*	0.722
% MgO	0.640	0.589*	0.853	0.767	0.749	0.822
% K_2O	0.420*	----	0.809	0.623	----	0.785
% P_2O_5	0.667	0.775	----	0.503*	0.704	----
% loss on ignition	0.579	0.677*	----	0.447*	0.651	----

1 = concentrations in meq/100g
2 = entire soil profile
*All relationships are significant at the 0.01 level except *, which is significant at the 0.05 level.*

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Figure 49: Cadmium adsorption in soils as a function of pH (Street et al., 1977)

Soil	pH	% $CaCO_3$	% OM	CEC (meq/100g)	ug/g in solution, time (hours): 0.0	1.0	8.0	24	48	96
A	5.71	0.0	3.7	23.9	50	1.4	0.2	0.1	0.1	0.1
B	7.82	0.4	0.7	5.8	50	0.7	0.7	0.6	0.6	0.6
C	7.43	0.3	0.6	3.6	50	1.8	1.9	1.9	1.8	1.8
D	8.25	6.0	1.4	23.0	50	0.1	0.1	0.1	0.1	0.1

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Figure 50: Freundlich adsorption isotherms for cadmium in four experimental soils (Street et al.,1977)

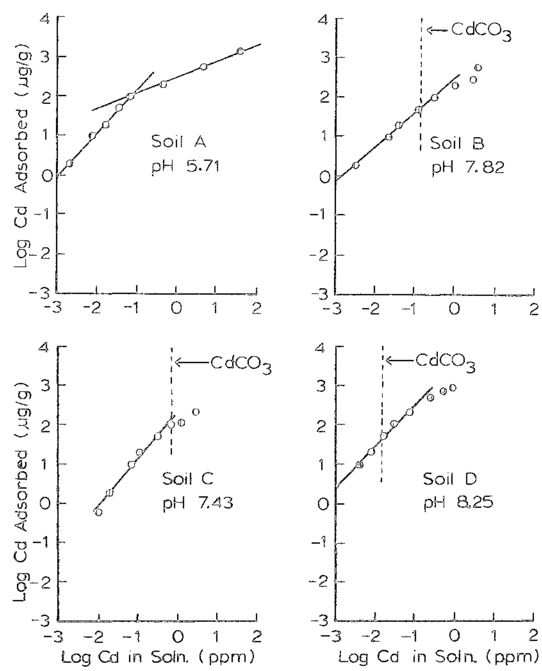

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Figure 51: Selectivity coefficients for lead, cadmium, and calcium sorption on montmorillonite, illite, and kaolinite (Bittell and Miller, 1974)

System	Ks*: montmorillonite	illite	kaolinite
Ca:Pb	0.60	0.44	0.34
Ca:Cd	1.04	1.01	0.89
Pb:Cd	0.58	0.56	0.31

**Ks = adsorption coefficient. Numbers represent ratios of metal sorption to one another*

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compared to the 2:1 clays. However, total sorption is higher on the 2:1 clays. The results are averages of 10 values for 10-90 percent metal saturation. Increasing saturation has little effect on selectivity coefficients.

Mehta et al. (1984) studied zinc adsorption in calcium and sodium saturated soils in India. There was no significant difference in zinc sorption for the two systems across a range of metal concentration. Zinc sorption was approximately equal to the soil CEC.

The pH dependence of zinc and cobalt sorption to hydrous manganese oxides is illustrated in figure 52, based on a study by Loganathan et al. (1977). Sorption increases with pH, particularly above pH 6.5.

Soil factors influence the fate of metals. Conclusions may be drawn with regard to developing a rating system designed to evaluate the potential for contamination from wastes containing high concentrations of heavy metals (e.g., sewage sludge).

1. If CEC data is available, it gives an indication of the potential sorption of heavy metals in soils. Percent of total metal sorbed decreases as total metal concentration increases. Maximum potential metal concentrations in waste should be used to determine the amount, as a percent of total, that may be adsorbed by soil as a function of CEC. Scales may then be developed relating CEC to hazard risk scores.

2. Calcareous soils may retard metal sorption up to 50 percent as a result of calcium competition with metals.

3. Soil pH strongly influences metal sorption, but the effect varies for each metal. Mercury and selenium are two metals with maximum sorption at low pH, while most metals exhibit maximum sorption above pH 6. The use of pH as an indicator of potential contamination by metals is a waste-specific analysis, best used as an indicator. If it is used in determining a factor score, a worst-case situation is produced.

4. Clay type and content can be used to give an indication of metal sorption when CEC is unknown. Textural classification gives a minimum clay content. Clay type often exhibits strong regional distribution and can be approximated by correlating this distribution with soil data and input from experienced field personnel. Figure 53

Figure 52: Electrophoretic mobility of a-MnO_2 as a function of pH at different equilibrium concentrations of Zn^{+2}, Co^{+2}, and 1×10^{-3} M $NaNO_3$ (Loganathan et al., 1977)

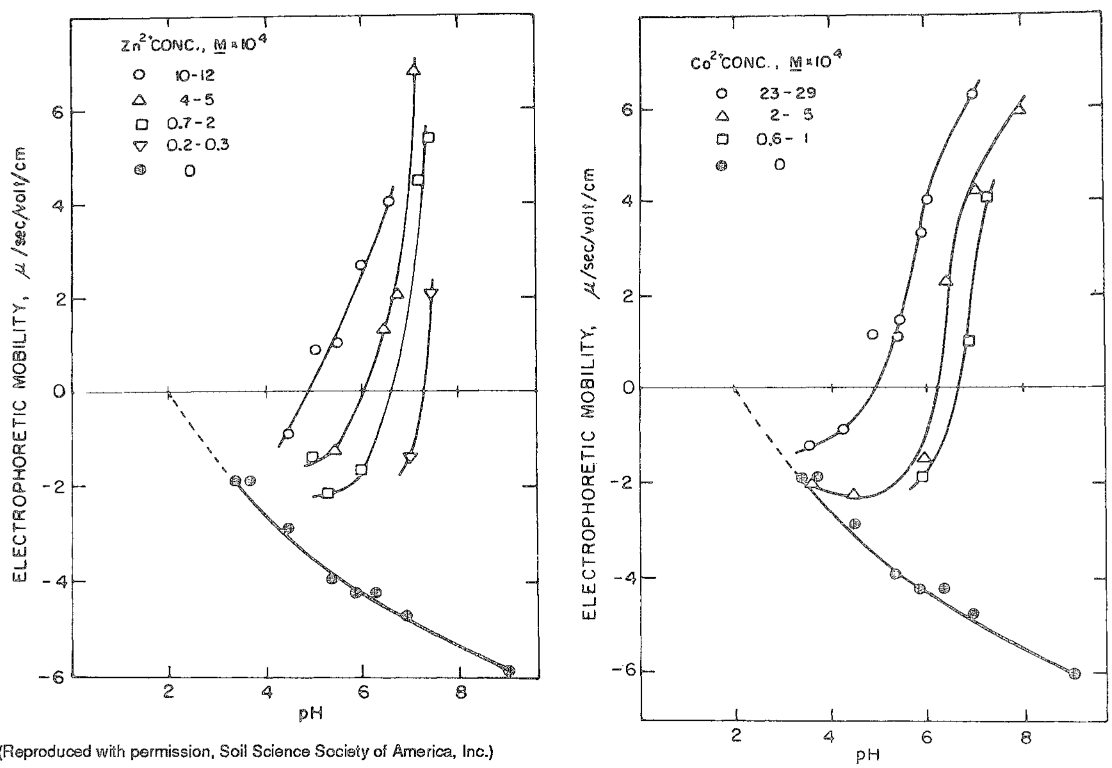

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Figure 53: Summary of selected properties of solid phase components (Bohn et al., 1979)

Componnent	mineral type	layer charge	CEC meq/100g	surface area (m^2/g)	Expan-sible	Colloidal Activity*
kaolinite	1:1	near 0	1-10	10-20	no	low
montmorillonite	2:1	0.25-0.6	80-120	600-800	yes	very-high
vermiculite	2:1	0.6-0.9	120-150	600-800	yes	high
mica	2:1	1.0	20-40	70-120	no	medium
chlorite	2:1:1	1.0	20-40	70-150	no	medium
allophane	-	-	10-150	70-300	-	medium
organic matter	-	-	100-300	800-900	-	medium

*pH dependency of charge is approximately inversely related to colloidal activity

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Figure 54: Fate of radioactive strontium in soil versus time (Lagerwerff and Kemper, 1975)

Application rate (ml Sr/cm^2/hr)	Depth (cm)	—% of original activity left— 26 hours	94 hours	168 hours	322 hours
0.32	1	4.7	-	1.6	-
2	4.2	-	1.6	-	-
0.16	1	-	21.1	-	2.5
2	-	21.5	-	1.3	-
3	-	-	-	3.4	-
4	-	-	-	1.5	-

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describes selected properties of solid phase components which affect the amount of metal sorbed.

In addition to soil factors, rapid infiltration rates and large quantities of water may reduce the amount of metal adsorbed. (See section on Infiltration) Metals, however, are strongly and rapidly sorbed by soil components. Only soils with extensive macroporosity can attain infiltration rates high enough to reduce metal sorption in the soil zone.

Metal sorption is generally reversible only when another metal is preferred at an exchange site. Metal sorption may be reduced in initially saturated or wet soils due to steric hindrance by soil adsorbed water molecules.

Radioactive Wastes

Radioactive waste covers a broad spectrum of contaminants. Plutonium and strontium have been intensively studied and are representative of the environmental fate of many radioactive wastes.

Francis (1973) conducted a literature review of plutonium mobility in soils and uptake by plants. Plutonium was considered immobile in soils, with ratios of concentrations in plant tissue to soil equal to 10^{-4} to 10^{-6}. Plutonium in a storage crib was predicted to move less than 10 cm after 24,000 years, with a maximum of 0.1 percent leached by infiltrating water. Nine meters of water were required to leach this amount. Plutonium at detonation sites moved little except at very high concentrations. Soil particles over 44 microns diameter were strongly preferred for adsorption.

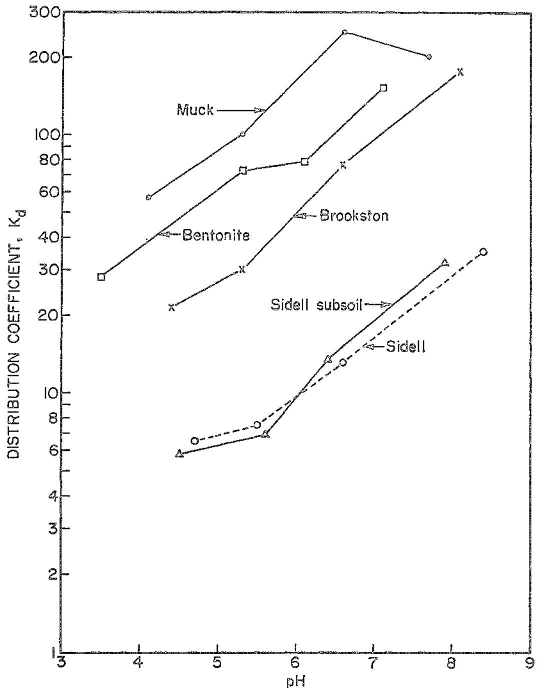
Plutonium sorption is pH dependent, with sorption near 100 percent at pH 2 and 80 percent at pH 8-12. Plutonium from fallout in Denver is located in the top 20 cm except in city parks, where liming and fertilizing enhanced movement. Below pH 2, anion-complexation dominates the fate of plutonium. Above pH 2, plutonium is adsorbed as positively charged polymers. The presence of organic salts thus enhances plutonium movement. Organic salt to plutonium ratios of 1:1 decreases sorption from 96.5 percent to 58.9 percent compared to a salt-free system. Organic-plutonium complexes have been found at depths of 8.5 cm, primarily along soil macropores. Soils subject to extensive freeze-thaw cycles may also show vertical movement due to physical transport.

Other radionucleides were found to be more mobile than plutonium, with greater concentrations in plant tissue. Strontium 89 and Americanum 241 had uptake rates over 50 times higher than plutonium.

Lagerwerff and Kemper (1975) did extensive studies on radioactive strontium sorption in three soils. Nearly all added strontium was removed from solution and immobilized in soil (figure 54). Rapid infiltration led to less leaching than continuous infiltration. Even with large infiltration amounts, leaching was small, with 1.5 percent of the original strontium found at a depth of 4 cm.

Juo and Barber (1970) studied the effect of organic matter, pH, and saturating cations on strontium movement for four soils (figure 55). Sorption increased with organic matter, CEC, and pH; effects of CEC and organic matter were not separated. The percent of non-exchangeable strontium was small (figure 56) indicating that CEC played an important role in total sorption, with organic matter forming stable complexes. A further indication of this was the high sorption of strontium by bentonite (organic matter = 0), though all the strontium was removed with NH_4Cl. The effect of exchangeable cation followed the pH-dependent reaction, with little exchange of hydrogen (low

Figure 55: The sorption of strontium by soils and bentonite from 3 X 10^{-3} M $SrCl_2$ solution as a function of pH (Juo and Barber, 1970)



Soil	% clay	% organic matter	CEC (meq/100g)
muck	-	49.8	70.0
Brookston	29.1	7.1	39.4
Sidell	20.8	2.9	19.2
Sidell subsoil	18.2	1.5	18.7
bentonite	-	0.0	-

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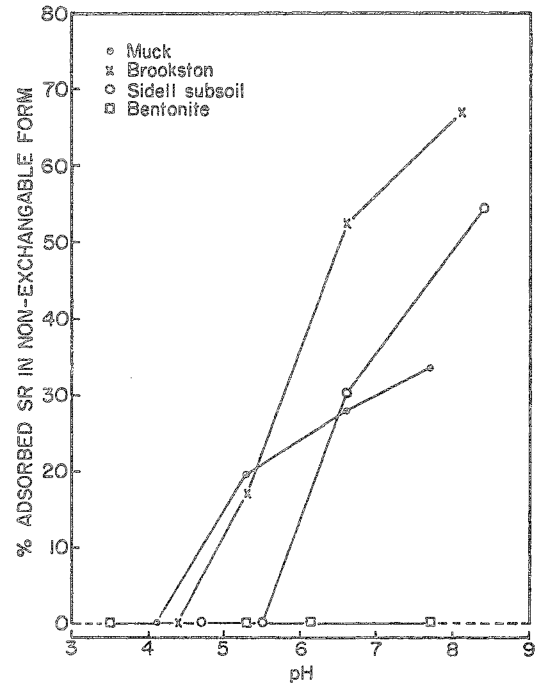
Figure 56: The percentage of sorbed strontium remaining in non-exchangeable form in soils as a function of pH (Juo and Barber, 1970)



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pH) and large exchange of sodium (high pH).

As with metals, radionucleides show a strong contaminant specificity. Radioactive wastes are easier to assess for content and their fate in the environment is easier to describe because they are generally strongly adsorbed. The most important factor in assessing radionucleide fate appears to be soil pH, though CEC and particle size may affect movement.

Nitrates, Phosphorus

In a study of unsaturated flow in England, Wellings and Bell (1980) found that the rate of downward nitrate movement was approximately 0.7 and 0.6 times that of water for clay-loams and sands respectively. These rates were a function of the soil-water pressure profile and vegetation. Kissel et al. (1977) found that carbon:nitrogen ratios of 20 or higher increased nitrogen immobilization as a result of nitrate being converted to organic nitrogen.

Liming increased nitrogen mobility by raising pH and decreasing the number of positive adsorption sites. Plysier and Juo (1981) found no affect of liming, as nitrate concentrations in drainage water exceeded 90 percent of added nitrogen for limed and unlimed plots. Sidle and Kardos (1979) found that nitrate leached in greater concentration and to greater depth than predicted by modeling water flux. Nitrogen was apparently weakly held and easily removed from adsorption sites, and nitrate tended to travel with water molecules through macropores. Kissel et al. (1976) found maximum nitrate concentrations of 2.4 ppm in spring drainage water for a Houston black clay. Concentrations were 0 ppm on soils with actively growing crops.

Nitrates would be expected to be a contamination problem on soils with a large water flux, rapid infiltration below the root zone, where fertilizer applications have been heavy or poorly timed, where specific contamination sources exist (e.g., feedlots, septic tanks), or where mineralization rates of organic nitrogen are high because of cultivation.

Other inorganic ions leach in proportion to their concentration in soil, and with water flux. The following order is expected for decreasing risk of leaching: Cl-SO_4-Na-Mg/Ca-K-PO_4. Phosphorus is strongly associated with iron, aluminum, and calcium in soil. It is rarely leached more than 10 cm at concentrations exceeding 5 ug–P/g–soil, except when application rates exceed 120 kg-P/hectare or on heavily leached soils such as Oxisols (Logan and McLean, 1973).

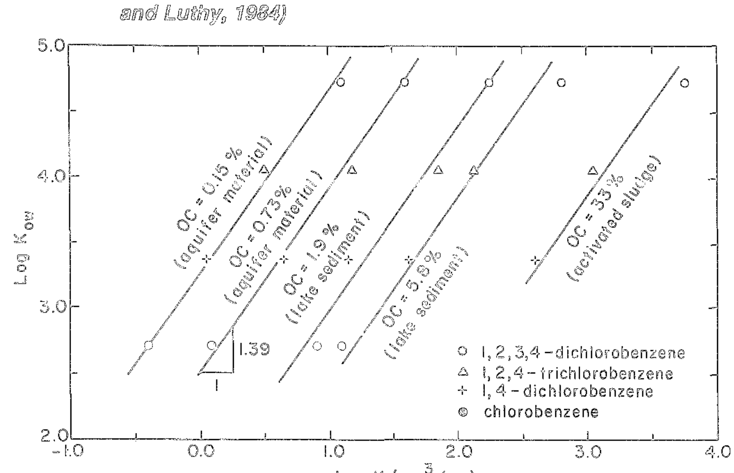
Organics

Organic contaminants vary widely in mobility due to chemical structure. Sax (1979) has established relative ratings for mobility of a large number of organic substances. Most organics tend to form neutral, negatively, or positively charged species in soil. For broad application, most organics can be separated into one of these three classes.

Organics which behave as acids (that form anionic species) are weakly sorbed in soils. Examples include Dicamba (2-Methoxy-3,6-Dichlorobenzoic acid), Fenac ((2,3,6-Trichlorophenyl) acetic acid), Picloram (4-Amino-3,5,6-Trichloropicolinic acid), and 2,4-D ((2,4-Dichlorophenoxy) acetic acid). Neutral organics (do not form charged species) are affected strongly by soil organic matter. The bulk of organic compounds fall into this category. Positively charged species are strongly retained in the soil. An example is the pesticide diquat (1,1'-Ethylene-2,2'Bipyridylium ion).

In applying a rating system, classes of organic contaminants can be evaluated. Knowledge of pesticides used in an area, composition of landfill waste, or composition of hazardous waste then allows contaminant classifications. For each

Figure 57: Correlation of log K_{ow} with log K for chlorobenzenes and sorbing materials of various organic contents (Dzomback and Luthy, 1984)



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class evaluation, a system designer should assume a worst case analysis unless specific contaminant data is known.

Dzomback and Luthy (1984) develop a model for adsorption of polyaromatic hydrocarbons (PAH). These are neutral, non-polar organics. They compete effectively with water molecules for hydrophobic surfaces such as waxes, fats, resins, aliphatic side chains, humic and fulvic acids, and lignins. The model is described by the following equation:

equation 8

$$K = 0.482(K_{ow})(OC)$$

where

K = adsorption coefficient
K_{ow} = octanol-water separation coefficient
OC = soil organic carbon content (g/g)

The model was applied to geologic materials of differing organic carbon content. Results are shown in figure 57. They indicate a high degree of correlation (r=0.95) for the variables used in the model.

Helling (1971) did an extensive study on the influence of soil properties on pesticide mobility (figure 58). Thirteen pesticides and 14 soils were examined. Mobility was correlated with several properties. Nonionic or weak-acid pesticide mobility was proportional to water flux and inversely proportional to organic matter content, clay content, and the amount of other pesticides adsorbed. Anionic pesticide mobility was positively correlated with pH and water flux. Cationic pesticide mobility was positively correlated with water flux. Relative mobilities were 0.72-0.95, 0.17-0.54, and 0.00-0.03 for anionic, neutral, and positive species, respectively.

Water flux and field capacity, inversely related to each other, had the most consistent effects on pesticide mobility, particularly for nonionic species. Clay type was shown to have a significant effect, with sorption enhanced by kaolinite and vermiculite for anionic and nonionic species and reduced for cationic species.

Helling's results are shown in figure 59. Multiple regressions are developed for each pesticide, as shown in figure 60. Correlation coefficients are higher ($\geq$ 0.85) for nonionic species than for charged species ($\leq$ 0.75).

Wilson and Cheng (1978) observed that 60-70 percent of applied 2,4-D was adsorbed within three weeks in a silt-loam soil. Less than 20 percent of 2,4-D adsorbed after 48 hours could

Figure 58: Simple correlation coefficients between relative mobility of twelve pesticides and soil properties for fourteen soils (Helling, 1971)

Pesticide	O.M.	clay	F.C.	pH	CEC	water flux	picloram adsorbed	diuron adsorbed
Dicamba	0.15	-0.01	-0.19	0.62	0.30	0.50	-0.21	-0.22
Picloram	-0.25	-0.23	-0.47	0.48	0.02	0.65	-0.41	-0.19
Fenac	-0.08	0.08	-0.25	0.76	0.30	0.60	-0.56	-0.07
2,4-D	-0.26	-0.11	-0.44	0.62	0.15	0.68	-0.48	-0.13
Monuron	-0.73	-0.71	-0.92	-0.12	-0.70	0.67	-0.24	-0.73
Atrazine	-0.66	-0.61	-0.88	0.11	-0.56	0.75	-0.31	-0.62
Diphenamid	-0.49	-0.57	-0.77	-0.08	-0.65	0.72	-0.29	-0.50
Simazine	-0.54	-0.53	-0.85	0.15	-0.52	0.74	-0.32	-0.51
Diuron	-0.72	-0.72	-0.93	-0.15	-0.68	0.66	-0.21	-0.72
Chlorpropham	-0.61	-0.65	-0.91	-0.01	-0.52	0.80	-0.34	-0.57
Azinphosmethyl	-0.56	-0.62	-0.85	-0.20	-0.61	0.46	-0.11	-0.57
Diquat	-0.17	-0.34	-0.51	0.12	-0.34	0.81	-0.48	-0.16

Trifluralin is omitted due to immobility

O.M. = organic matter
clay = clay as a weight percent in soil
F.C. = field capacity moisture content
CEC = cation exchange capacity in meq/100g
r^2 greater than $\pm$ 0.67 is significant at the 0.01 level
r^2 less than $\pm$ 0.67 but greater than $\pm$ 0.52 is significant at the 0.05 level

Dicamba --- 2-Methoxy-3,6-dichlorobenzoic acid
Picloram --- 4-Amino-3,5,6-trichloropicolinic acid
Fenac --- (2,3,6-Trichlorophenyl) acetic acid
2,4-D --- (2,4-Dichlorophenoxy) acetic acid
Monuron --- 3-(p-Chlorophenyl)-1,1-dimethylurea
Atrazine --- 2-Chloro-4-ethylamino-6-isopropylamino-S-triazine
Diphenamid --- N,N-Dimethyl-2,2-diphenylacetamide
Simazine --- 2-Chloro-4,6-bis(ethylamino)-s-triazine
Diuron --- 3-(3,4-Dichlorophenyl)-1,1-dimethylurea
Chlorpropham --- Isopropyl-m-chlorocarbanilate
Azinphosmethyl --- O,O-Dimethyl S-((4-oxo-1,2,3-benzotriazin-3(4H)-yl)methyl) phosphorodithioate
Diquat --- 1,1'-Ethylene-2,2'-bipyridylium ion
Trifluralin --- a,a,a-Trifluoro-2,6-dinitro-N,N-dipropyl-p-toluidine

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Figure 59: Correlation coefficients for twelve pesticides on fourteen soils as a function of clay type effect on mobility. The first set of twelve pesticides is for montmorillonite soils, the second set for non-montmorillonite soils (Helling, 1971)

Pesticide	O.M.	clay amount	field capacity	pH	CEC	water flux	picloram adsorbed	diuron adsorbed
Dicamba	-0.07	0.72	-0.48	0.95	0.46	0.27	0.40	-0.01
Picloram	-0.76	0.39	-0.80	0.57	-0.10	0.48	0.92	-0.72
Fenac	-0.61	0.52	-0.75	0.71	0.06	0.47	0.83	-0.57
2,4-D	-0.84	0.24	-0.81	0.48	-0.24	0.49	0.94	-0.81
Monuron	-0.80	-0.09	-0.80	0.53	-0.47	0.38	-0.76	-0.77
Atrazine	-0.63	0.06	-0.77	0.74	-0.28	0.34	-0.66	-0.57
Diphenamid	-0.24	-0.32	-0.17	0.52	-0.28	0.46	-0.05	-0.21
Simazine	-0.31	-0.02	-0.52	0.84	-0.14	0.29	-0.31	-0.25
Diuron	-0.87	-0.12	-0.78	0.43	-0.52	0.42	-0.80	-0.84
Clorpropham	-0.93	0.11	-0.87	0.31	-0.41	0.34	-0.98	-0.91
Azinphosmethyl	-0.52	-0.02	-0.52	0.70	-0.23	0.55	-0.48	-0.48
Diquat	-0.28	-0.86	-0.42	-0.42	-0.47	-0.36	0.65	0.26
Dicamba	0.19	-0.12	-0.27	0.45	-0.13	0.57	-0.21	0.23
Picloram	-0.12	-0.36	-0.55	0.39	0.14	0.72	-0.28	-0.09
Fenac	-0.01	0.01	-0.35	0.78	0.18	0.66	-0.62	-0.14
2,4-D	-0.18	-0.20	-0.58	0.62	0.02	0.77	-0.46	-0.06
Monuron	-0.73	-0.83	-0.94	-0.14	-0.68	0.78	-0.02	-0.70
Atrazine	-0.66	-0.71	-0.89	0.06	-0.56	0.86	-0.16	-0.60
Diphenamid	-0.59	-0.68	-0.86	0.11	-0.54	0.94	-0.33	-0.53
Simazine	-0.61	-0.65	-0.90	0.17	-0.49	0.88	-0.30	-0.54
Diuron	-0.69	-0.82	-0.95	-0.17	-0.65	0.74	0.01	-0.67
Chlorpropham	-0.58	-0.70	-0.92	0.06	-0.51	0.86	-0.22	-0.52
Azinphosmethyl	-0.62	-0.68	-0.88	-0.17	-0.55	0.50	0.01	-0.57
Diquat	-0.19	-0.31	-0.53	0.45	-0.17	0.93	-0.65	-0.11

correlation coefficients greater than $\pm$ 0.82 are significant at the 0.01 level
correlation coefficients less than $\pm$ 0.82 but greater than $\pm$ 0.66 are significant at the 0.05 level

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be recovered by dilution with distilled water, indicating that organic matter was the primary adsorption medium. Correlation of sorption with organic matter was found to be 0.988.

Figure 60: Multiple regressions and correlation coefficients for mobility of twelve pesticides on fourteen soils (Helling, 1971)

Pesticide	equation	r^2
Dicamba	Y=0.80 + 0.035pH -0.0021C	0.525
Picloram	Y=0.80 + 0.0073F	0.416
Fenac	Y=0.07 + 0.1pH + 0.0113F	0.768
2,4-D	Y=0.21 + 0.009F + 0.099pH + 0.007S	0.864
Monuron	Y=0.88 - 0.012FC -0.004S + 0.005F	0.920
Atrazine	Y=0.70 - 0.007FC - 0.006S + 0.005F	0.935
Diphenamid	Y=0.10 - 0.0094FC + 0.02P - 0.004S	0.988
Simazine	Y=0.65 - 0.0107S + 0.012F	0.918
Diuron	Y=0.72 - 0.0183FC	0.865
Chlorpropham	Y=0.046 - 0.0134FC + 0.0082F	0.938
Azinphosmethyl	Y=0.43 - 0.0118FC	0.724
Diquat	Y=-0.02 + 0.00575F	0.654

Y = pesticide mobility
F = water flux
S = Simazine adsorbed
r^2 = correlation coefficient
C = clay amount
FC = field capacity moisture content
P = Picloram adsorbed

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Figure 61: Values of soil active fractions (O) as a function of soil organic matter (Lambert, 1968).

Soil	State	Organic matter(%)	O
sandy loam	standard	1.0	1.0
clay	California	2.4	2.3
clay loam	California	3.1	2.6
loam	California	10.8	5.4
silt loam	Oregon	0.69	0.73
clay loam	California	2.2	1.9
clay loam	Minnesota	3.8	2.5
clay loam	Nebraska	2.9	2.6
clay loam	California	1.2	1.4
clay loam	California	2.1	1.9
loam	California	5.3	6.1
loamy sand	California	0.52	0.51
silt loam	Washington	1.0	1.1
Armix A	composite soil	15.5	12.9
Armix B	composite soil	23.9	17.2
Armix C	composite soil	30.8	23.3
#1	North Carolina	2.1	0.66
#2	North Carolina	0.69	0.31
#3	North Carolina	3.1	2.3
#4	North Carolina	5.5	5.7
#5	North Carolina	10.0	6.6
#6	North Carolina	38.8	29.8
#7	North Carolina	0.52	0.45

O = CeKX(100)

where Ce = equilibrium concentration of chemical in solution
K = distribution coefficient
X = amount of chemical sorbed to soil

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Koskinen et al. (1979), studying 2,4,5-T, observed values of 6.5 and 0.91 for the Freundlich constant K (an indicator of adsorption by soil particles) in silt-loam and clay-loam soils, respectively. Organic matter was 3 percent for the silt-loam and 0.8 percent for the clay-loam. There was considerable hysteresis in the silt-loam, indicating a high degree of organic matter-herbicide complexation.

Rodosevich and Winterlin (1977) found 45-60 percent of applied 2,4-D and 2,4,5-T (2,4,5-Trichlorophenoxy-acetic acid) in litter at chaparral sites. These percentages remained consistent for the entire study year.

Fuhremann et al. (1976) observed that bound parathion (O,O-diethyl-O-4-nitrophenyl phosphorothioate) was 3-7 times higher for a loam soil with 4.7 percent organic matter than for a sandy bog with an organic matter content of 1.3 percent. This indicated stronger adsorption associated with organic matter surfaces.

Lambert (1968) studied soil adsorption of three nonionic pesticides: Planavin (4-Methylsulphonyl-2,p-dinitro-N,N-dipropylanaline), Landrin (3,4,5-and 2,3,5- isomers of trimethylphenyl methylcarbamate), and Supona (2-chloro-1-(2,4-dichlorophenyl)vinyl diethyl phosphate). It was found that the distribution of a chemical between soil organic matter and water was proportional to soil organic matter content regardless of soil type and other soil properties. The concept of an "active fraction" of the soil was developed and expressed as:

equation 9

$$K = X/(O/Co)$$

where
K = distribution coefficient (ug/ml)
X = chemical sorbed to soil (ug/g)
O = active fraction of the soil
Co = equilibrium concentration of chemical in solution (ug/ml)

Figure 61 indicates that determining soil active fraction as a function of organic matter content works well for nonionic species. The model is not extended to polar or charged species.

Rhodes et al. (1970) is a study of the movement of five agrichemicals for four soils of varying organic matter content. The results (figure 62), show a strong correlation between organic matter content and chemical adsorption.

Karickhoff et al. (1979) computed correlation coefficients of 0.85 or higher for adsorption of hydrophobic pesticides versus organic carbon content of natural sediments. Means et al. (1982) computed correlation coefficients of 0.9 and higher for substituted PAH sorption on soils with organic carbon content of up to 2.38 percent. Correlation coefficients decreased slightly as solubility increased, though the significance of solubility was not tested.

A secondary effect of organic content is on biological degradation of organic contaminants. Studies by Gerstl et al. (1979), Abdelmaqid and Tabatabai (1982), Boyd and King (1984), Holstun Jr. and Loomis (1956), Konrad and Chesters (1969), and Lavy et al. (1973) illustrate the importance of microorganisms in attenuating and degrading organic contaminants. Organisms must have a food source prior to introducing a contaminant. Consequently, many studies indicate that there is no threshold value for mineral soil at which increasing organic carbon content results in no further attenuation of a contaminant.

Yaron (1975) refutes the findings of no threshold for organic content in a study illustrated in figure 63. Biodegradation is reduced as a result of pesticide immobilization by soil organic matter. Attenuation does appear to occur at a much lower rate above organic carbon contents of 4-5 percent, although total attenuation is not reduced.

Figure 62: Mobility and adsorption of five herbicides in soil (Rhodes et al., 1970).

Soil	*Bromacil*	*Terbacil*	*Monuron*	*Diuron*	*Cloroneb*
Rf					
muck	*0.15*	*0.15*	*0.13*	*0.00*	*0.00*
Muscatine	*0.60*	*0.55*	*0.28*	*0.16*	*0.00*
Keyport	*0.61*	*0.58*	*0.43*	*0.24*	*0.00*
Cecil	*0.89*	*0.85*	*0.68*	*0.42*	*0.00*
Rb					
muck	*0.00*	*0.00*	*0.00*	*0.00*	*0.00*
Muscatine	*0.25*	*0.08*	*0.00*	*0.00*	*0.00*
Keyport	*0.40*	*0.24*	*0.00*	*0.00*	*0.00*
Cecil	*0.42*	*0.34*	*0.00*	*0.00*	*0.00*

Rf = 1/(K + 0.15) where K is a sorption coefficient
Rb = distance moved by labeled herbicide/distance traveled by water

muck : pH=6.7; O.M.=83.5%; average particle size=8.7u
Muscatine : pH=6.4; O.M.= 6.0%; average particle size=8.6u
Keyport : pH=5.4; O.M.= 2.1%; average particle size=5.6u
Cecil : pH=5.8; O.M.= 0.7%; average particle size=10.5u

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Many organic materials act as weak acids and are pH dependent. This is shown in the study by Helling (1971). Saltzman and Yariv (1975) discuss the pH dependence of phenol adsorption. At low pH, phenols act as proton donors and can be adsorbed into clay interlayers or on deprotonated sites of organic matter and clays. Maximum sorption would be expected near the pK value for an organic material, but this must be balanced by the effect of pH on other soil parameters such as bioactivity. A pH of 5-6 might be expected to maximize adsorption and degradation for organics that behave as weak acids.

Clay type, as shown by Helling (1971), is important in contaminant attenuation. Mingelgrin (1977) found that organophosphate pesticide adsorption and degradation were enhanced by kaolinite when compared to montmorillonite. Degradation was several orders of magnitude greater for the kaolinite, with illite and gibbsite intermediate. The effect of saturating cations on sorption was Fe > Ca > Mg > K > Al > Na. Similar results were reported by Yaron (1975).

Figure 63: Effect of soil organic matter on parathion immobilization (Yaron, 1975)

Soil type	*dominant clay*	*clay (%)*	*O.M. (%)*	*pH*	*% parathion remaining dry*	*% parathion remaining wet*
sandy regosol	*montmorillonite*	*2.9*	*0.45*	*8.2*	*96.8*	*79.0*
loessial light brown	*kaolinite*	*16.9*	*0.66*	*8.2*	*88.0*	*84.0*
grumusolic brown	*montmorillonite*	*40.0*	*0.08*	*7.9*	*83.0*	*90.7*
red terra rossa	*kaolinite*	*75.5*	*1.07*	*6.8*	*77.0*	*87.0*
red-brown grumusol	*montmorillonite*	*57.6*	*1.50*	*7.6*	*82.9*	*90.8*
red terra rossa	*kaolinite*	*76.6*	*1.89*	*6.9*	*81.1*	*99.0*
basaltic brown	*kaolinite*	*22.4*	*2.47*	*6.4*	*92.0*	*97.5*
hamma	*kaolinite*	*9.7*	*2.75*	*6.8*	*92.0*	*97.5*
basaltic brown	*kaolinite*	*32.0*	*3.61*	*6.5*	*84.0*	*97.4*
calcareous brown	*montmorillonite*	*42.9*	*3.94*	*7.5*	*91.7*	*98.8*
red-brown terra rossa	*montmorillonite*	*65.3*	*4.10*	*7.7*	*96.5*	*98.0*
red-brown terra rossa	*montmorillonite*	*71.1*	*4.94*	*7.5*	*96.0*	*99.7*
red terra rossa	*kaolinite*	*68.5*	*4.98*	*7.5*	*80.7*	*97.8*
brown rendzina	*montmorillonite*	*46.5*	*12.10*	*7.3*	*95.7*	*99.8*

O.M. = organic matter *wet = 50% moisture content by weight*
dry = air-dried soils

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Figure 64: Relative retention capacity of soils contaminated with dense, non-aqueous phase liquids (Villaume, 1985)

Medium	*R value*
stone, coarse gravel	*5*
gravel, coarse sand	*8*
coarse-medium sand	*15*
medium-fine sand	*25*
fine sand, silt	*40*

R = relative retention capacity

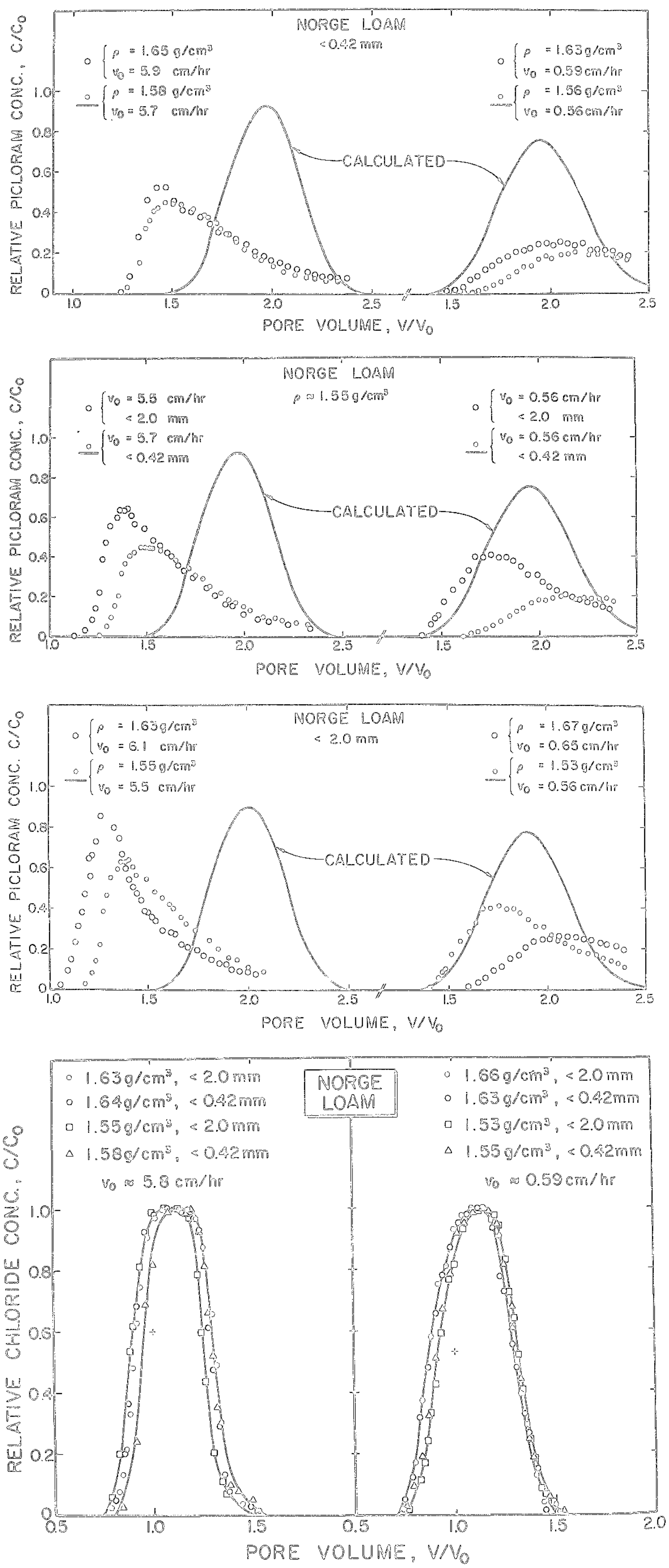

Figure 65: Experimental and calculated picloram concentration distributions from Norge loam (1.55g/cm³ bulk density, less than 0.42 mm aggregates) for two average pore-water velocities (Davidson and Chang, 1972)

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Soil grain size may influence the fate of organics. van Genutchen et al. (1977) indicate that bulk density and aggregate size do not affect the relative location of adsorption sites, though aggregate size may affect the number of sites by influencing surface area. This is illustrated by Davidson and Chang (1972), who report sorption 30 percent higher for aggregate size less than 0.42 mm versus a soil with an average aggregate size less than 2 mm.

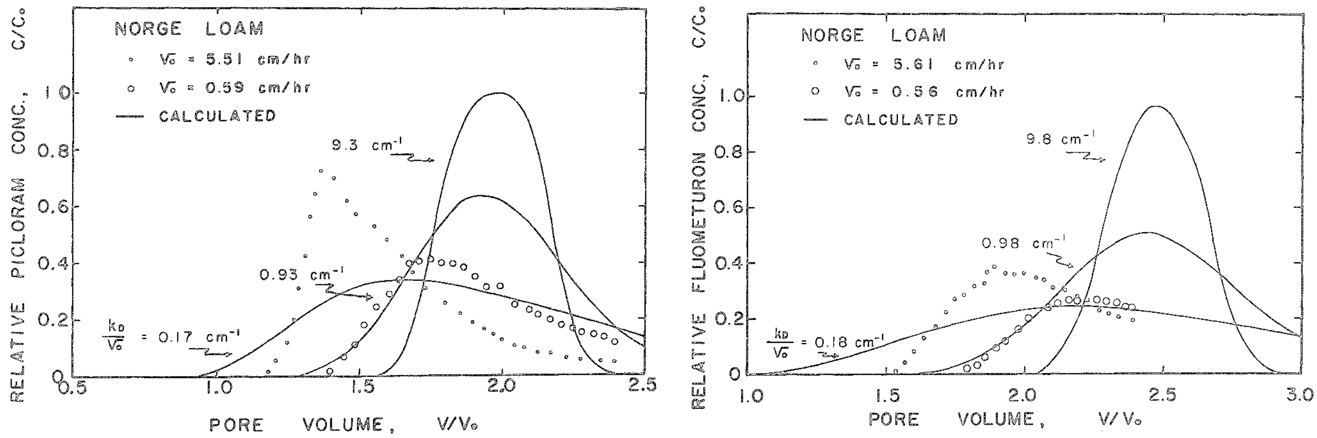
Villaume (1985) studied the movement of dense, nonaqueous phase liquids as a function of soil texture. Figure 64 illustrates relative retention capacities of different soil media for oils used in the study. Grain size was an important factor determining contaminant mobility.

The Villaume study suggests that hydrologic factors also influence organic contaminant mobility. Helling (1971) shows that flux was significantly correlated with pesticide mobility for 10 of 12 pesticides studied. Figure 65 illustrates the results of a study by Davidson and Chang (1972) correlating pore-water velocity with picloram mobility. Figure 66 illustrates a study by Davidson and McDougal (1973) showing the effect of pore-water velocity on parathion mobility. Increasing velocity increases mobility, although the indicated velocities are high. These velocities may be encountered with intense storms on unsaturated soils.

If an organic contaminant has had sufficient time to react with soil and organic matter, the effect of velocity is significantly reduced due to strong contaminant adsorption. On nearly saturated soils, pore-water velocity is reduced, although water adsorbed to exchange sites would repel organic contaminants and reduce attenuation (Yaron and Saltzman, 1972). This effect is shown in figure 67, where saturated soils have virtually no parathion adsorption.

A final variable affecting attenuation is temperature. Yaron and Saltzman (1972) show the adsorption of parathion in three soils as a function of temperature (figure 68). Their results indicate that increasing temperature leads to water desorption from soil particles and a consequent increase in pesticide adsorption.

Figure 66: Experimental and calculated relative a) picloram and b) fluometuron effluent distributions from Norge loam soil (Davidson and McDougal, 1973)



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Lavy et al. (1973) finds that increasing temperature increases degradation and therefore increases contaminant attenuation up to 25-30°C. The composite effect of temperature and soil moisture on parathion adsorption is illustrated in figure 69. Adsorption is maximized in warm, dry soils. Biological degradation is likely to be low in such systems.

When incorporating an attenuation factor for organics into a rating analysis, the following conclusions may be drawn:

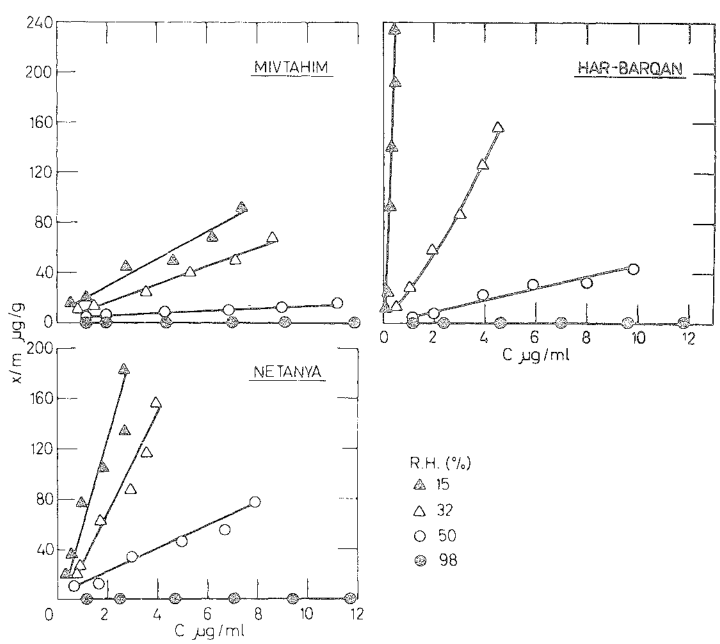
1. As soil organic matter increases, to about 5 percent, attenuation increases. The effect varies with different organic classes. Organic content generally decreases across the following textural sequence: loams > silt loams > clay loams > sandy loams > sands. Forests generally have thick organic litters but low organic content in the soil horizons; deciduous forests generally have greater organic matter concentrations than coniferous. Agricultural soils have moderate to high organic contents; soil type, location (geographic and topographic), fertilization practices, crops grown, and tillage practices affect concentrations. Native prairie soils have thick organic layers;

2. A soil pH of 5-6 maximizes attenuation for most organics, particularly those degraded biologically;

3. Soil temperatures of 25-30°C maximize attenuation for most organics;

4. The effect of clay type (e.g. kaolinite montmorillonite, etc.) on attenuation varies with contaminants;

Figure 67: Adsorption isotherms of parathion from hexane on partially hydrated soils (30°C) (Yaron and Saltzman, 1972)



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5. Infiltration rate may have an effect on organic adsorption depending on pesticide concentration, initial soil moisture, and degree of pesticide sorption prior to infiltration;

6. Attenuation should be considered only over the thickness of the attenuating layer. For organic matter, this is generally the upper soil horizons. The pH level is important throughout the unsaturated zone, but should only be considered when significant variations occur with depth. The influence of clays,

Figure 68: Normal adsorption isotherms of parathion from water at 10°, 30°, and 50°C (Yaron and Saltzman, 1972)

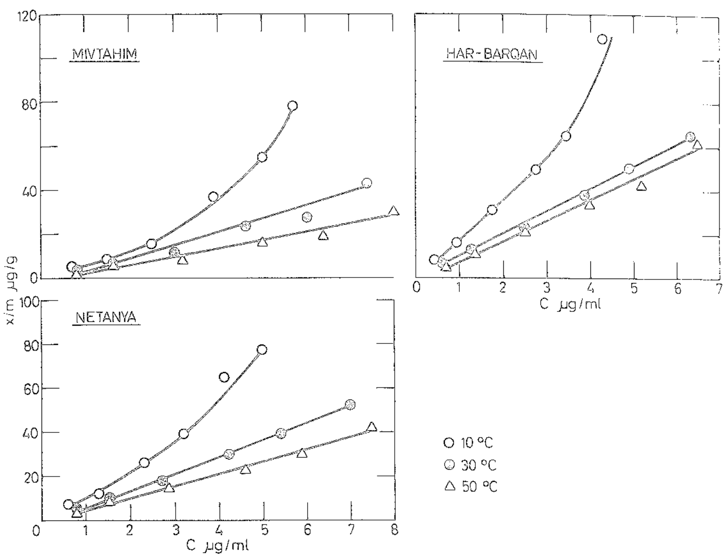

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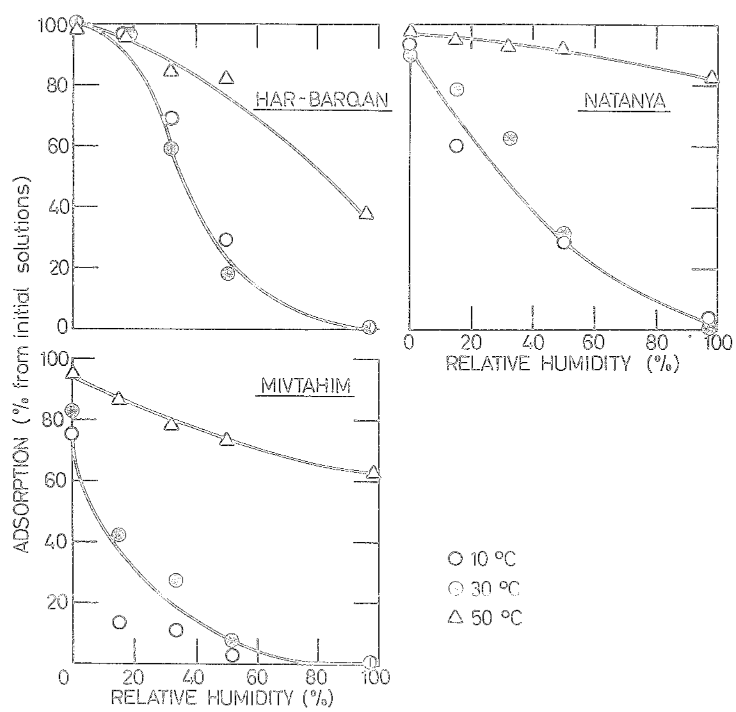
Figure 69: Composite effect of water and temperature on soil parathion adsorption from hexane solutions (Yaron and Saltzman, 1972)



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particularly kaolinite, may be important below the upper soil horizons where organic matter content is highest. Temperature is important in the upper soil horizons where it influences bioactivity and rates of chemical reactions;

7. The composition of organic matter affects contaminant attenuation. Separating the various fractions (humic and fulvic acids, etc.) is difficult however, and is too specific to be very useful in a rating analysis.

Incorporating attenuation into a factor analysis provides increased practicality and accuracy. Because of the tremendous variation in potential contaminants, it is a difficult factor to assess. This discussion, despite the limited scope of contaminants considered, implies that broad contaminant classes exist. These classes can be utilized to simplify the analysis. Attenuation factors were not placed in usable form for equation *1a* or *1b* due to the limited availability of background literature and the importance of local characteristics which influence attenuation. Figure 70 illustrates the potential utilization of an attenuation analysis.

SUMMARY

Figure 71 illustrates example site indices. The format is arbitrary. Hazard index is the first value given in the site index, followed by factor indices. Factor indices are separated by slashes, though semi-colons or other separation marks may be substituted. Correction terms and identifiers are included in a factor index, with numbers corresponding to those presented in the preceding discussion. Actual numbers are not included but can be envisioned. Site description would be given as a computer printout explaining the terms in the site index. These descriptions are derived from computer files. A site location would be included for an actual situation. An identifier must be used to identify the aquifer of concern.

Nielson et al. (1973) states that the low correlation between particle-size distribution and soil-water parameters implies that traditional soil survey data are inadequate for describing soil-water movement and retention. Establishment of hydraulic conductivity/soil moisture content relationships and steady state infiltration rates are necessary to adequately describe water flux in the unsaturated zone. Freeze and Banner (1970) offer the same conclusion, stating that the areal variation in meteorological factors, moisture profiles, and soil hydrologic properties can only be described with knowledge of conductivity/moisture relationships and storage functions.

Biggar and Nielson (1976) reinforce their earlier conclusion with a study of the leaching characteristics of a 150-hectare field soil (figure 72). Variations for diffusivity and pore-velocity

Figure 70: Development of indices for attenuation factors

Metals

Analysis - rating scale of risk versus soil cation exchange capacity; concentration of metal is included if data exists; hydrogeologic risk is based on an analysis of the most likely contaminant to reach groundwater.

Correction terms - 1. pH--for the majority of metals a pH of 6-8 maximizes adsorption;

2. clay type--montmorillonites adsorb more metal than kaolinites, although much of this effect is due to the higher cation exchange capacity of montmorillonite;

3. calcium content of soils--calcareous soils may have reduced metal adsorption due to competition with calcium, other factors being equal.

Radioactive Wastes

Analysis - contaminant specific but the content of most radioactive wastes should be well described, allowing specific factor analysis. Organic matter concentrations, soil pH, and clay content (type and amount) are possible factors.

Correction terms - 1. volume of infiltration water or water available for recharge;

2. velocity of infiltrating water.

Solutes

Analysis - Quantity and volume of infiltrating water have the largest effect on solutes such as nitrate and chloride. Others such as phosphorus require specific analysis which can be justified only when these contaminants are a special concern.

Organics

Analysis - There are classes of organic compounds, each affected by different soil properties and processes. Anionic species are influenced strongly by soil moisture content and pore-water velocity; organic matter has a moderate effect as does clay type, with kaolinite leading to increased adsorption over other clay types. A rating matrix of soil moisture and organic matter/clay contents may provide a suitable method for determining score for a factor. Neutral species are strongly influenced by soil organic matter. Cationic species are influenced by cation exchange capacity and organic matter content. For each class of organic contaminant, assess the most mobile compound unless specific data on composition exists.

Correction terms - 1. pH--pH is most important for anionic species;

2. moisture content--moisture content becomes important for all organic compounds above field capacity, particularly reducing adsorption of polar contaminants;

3. temperature--temperatures of 25-30°C appear to maximize attenuation of organic materials due to a combined effect of increased bioactivity and chemical reaction;

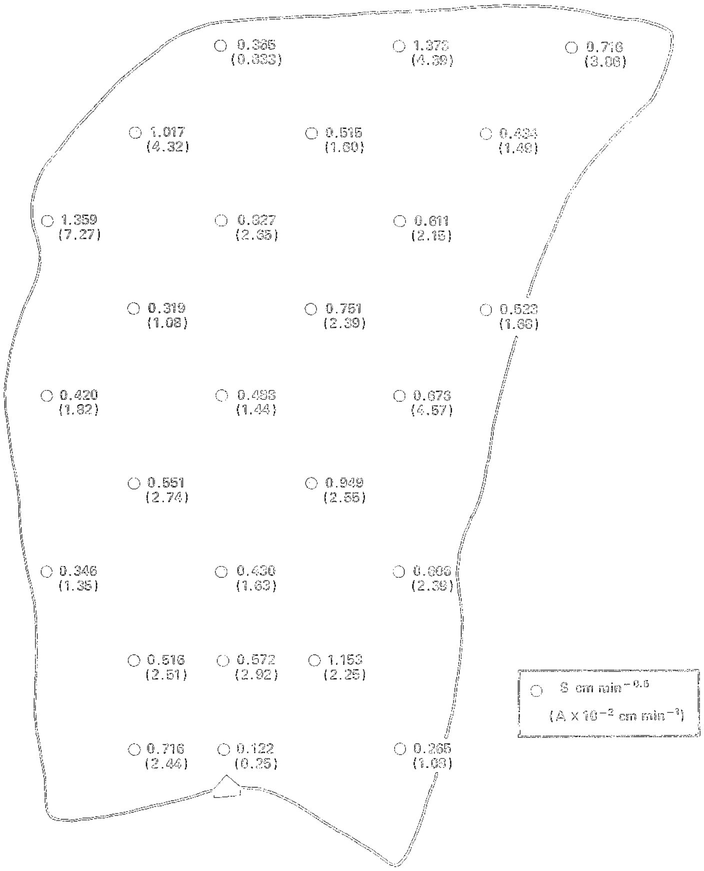
4. flow velocity--organics are often slowly adsorbed in the soil and are thus subject to vertical movement at high pore-water velocities. These velocities are most common in soils with macroporosity, but the effect of macroporosity on movement of organic materials is uncertain.

Figure 71: Example indices for contamination susceptibility of sites contaminated by neutral pesticide species

Index*	Description
41.78 / 7.49 / $4.19_{1;3}$/ $7.4_{;2}$/ 6.3 / 3.8_{2}/ 6.2	Site score is 41.78. Recharge potential is 7.49 inches. Permeability score is 4.19 and has been corrected for layer thickness, with a final permeability of $10^{-6.4}$ meters per second. Discontinuous layers in the confining beds may lead to localized areas of high permeability. Depth-to-water score is 7.4 and corresponds to a depth of 13 feet. Discontinuous layers in confining beds may lead to varied depth-to-water. Infiltration score is 6.3, corresponding to an infiltration rate of $10^{-5.3}$ meters per second and a sandy-loam soil. Topography score is 3.8 and has been corrected for runoff as a function of slope position. Attenuation score is 6.2, corresponding to a soil organic matter content of 1.5%. Six factors out of six possible factors have been analyzed.
70.29 / 9.9 / – / – / 7.9 / $8.7_{;3}$/ 4.7	Site score is 70.29. Recharge potential is 9.9 inches. Insufficient data exists to evaluate permeability. Insufficient data exists to evaluate depth-to-water. Infiltration score is 7.9, corresponding to an infiltration rate of $10^{-4.5}$ meters per second and a fine sand soil. Topography score is 8.7 corresponding to a slope of 2.7%. The site is located on a 100-year floodplain. Attenuation score is 4.7, corresponding to a soil organic matter content of 3.1%. Four factors out of six possible factors have been analyzed.

* Factors are prompted from computer files in the following order: Recharge potential, permeability, depth-to-water, infiltration, topography, and attenuation of neutral pesticide species. An identifier must be included in each index identifying the aquifer of concern. Values expressed here are derived from previously displayed figures or are arbitrarily set.

Figure 72: Spatial distribution of the infiltration parameters S and A across a 150-hectare watershed (Sharma et al., 1980)



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are so large that detailed studies would be necessary to predict leaching losses.

The question of concern to the developer of a rating system is: "Can the variation in soil-hydrologic properties be evaluated to a degree where broad generalizations about recharge processes accurately describe groundwater susceptibility?" The preceding discussion presents a far-reaching method to accomplish this, incorporating localized conditions into an informative, practical, and accessible methodology. Application of this methodology presents the best chance to answer this question in the affirmative.

CHAPTER III: COMPUTER APPLICATION

With a properly designed computer program and available digitized maps, the methodology described here can be quickly utilized, leading to a flexible, informative, and practical analysis of groundwater sensitivity across an area. Such a program must be user friendly to allow incorporation of local features that influence groundwater quality. The program would require an extensive number of files including scales for factors and correction terms; lists of factors, correction terms, and identifiers; and explanations of these features.

The concept of expert systems has significant application in groundwater sensitivity analysis. Figure 73 provides exemplary literature sources for expert systems. An algorithm for a computer program designed to apply the described methodology is illustrated in figure 74.

Figure 73: Literature sources for Expert Systems

Davis, J.R., 1986. Possibilities for expert systems in agriculture, Acta Horticulturae, no. 175, March, 1986.

Davis, J.R., P.M. Nanninga, K.D. Cocks, 1985. The usefulness of computer aids that capture expert knowledge about land management, CSIRO, Division of Water and Land Resources, P.O. Box 1666, Canberra, Australia.

Davis, J.R., and P.M. Nanninga, 1985. GEOMYCIN: Towards a geographic expert system for resource management, Journal of Environmental Management, Vol. 21, p. 377-390.

Denning, P.J., 1986. The science of computing: Expert systems, American Scientist, Jan.-Feb. 1986.

Fersko-Weiss, Henry, 1985. EXPERT SYSTEMS: Decision-making power, Personal Computing, November 1985, p. 97.

Giboney, Vance, 1986. Conventional Programming and expert systems, Computer Language, Vol. 3, no. 8, August 1986, p. 53.

Huggins, Larry F., John R. Barrett, and Don D. Jones, 1986. Expert systems: Concepts and opportunities, Agricultural Engineering, Vol. 67, no. 1, Jan.-Feb. 1986, p. 21-23.

Lecot, Koenraad, and D. Stott Parker, 1986. Control over inexact reasoning, AI Expert, Premier Issue, 1986, p. 32-43.

Figure 74: Algorithm for computer program designed to assess groundwater sensitivity to contamination

Function	File/Inputs	Example
Choose objective	File with list of objectives displayed to user.	Sensitivity of groundwater to contamination by pesticides.
Choose factors	File with list of necessary and potential factors is displayed to user. User may be prompted for contaminant class.	Base factors include recharge, topography, infiltration, permeability-of-the-vadose-zone, and depth-to-water. Specific attenuation factor is organic matter content of soil. Contaminant class is anionic pesticides (e.g. 2,4-D).
Can the objectives be accom-plished?	Computer program requires inclusion of at least one factor and all necessary factors at each site.	Each site must have data on organic matter content. Lack of data results in an 'insufficient data for evaluation' message.
IF YES - continue		
Choose equation	Equations 1a and 1b are displayed with term descriptions.	If recharge can be evaluated throughout the area, equation 1a is used.
Enter site coordinates	Enter x and y coordinates. Computer translates these to proper positions on final map.	x=1 y=1
Enter factor data	Original factor values are entered. Help commands display factor scale explanations. Prompts ask for specific forms for input.	Enter recharge potential in inches per year. Help command displays the equation used in the calculation (P + I - ET). Scale from Figure 9 may be displayed. User enters a value of 7.0.

(Table continues on next page)

Figure 74 continued

Function	*File/Inputs*	*Example*
Enter correctors	*Prompt for correction terms with corresponding values is given. Correction terms are numbered and exist as scales or mathematical values in files. These files can be retrieved and displayed.*	*For recharge the following correctors may be displayed:* *1.cover type;* *2.cover height.* *User responds with 1, indicating type of vegetative cover will affect the factor value. Another prompt displays possible cover types and asks for the appropriate type and percent cover. User enters deciduous forest with 100% cover. Correction term is determined from scales or numerical relationships on file and is given as 0.9.*
Enter Identifiers	*Potential identifiers are displayed for the factor being considered. The procedure is similar to that for correction terms.*	*List of identifiers may include:* *1. No summer recharge is assumed;* *2. No winter recharge is assumed.* *User answers the prompt with 2.*
Data is processed	————	*Factor value (7.0 inches) is multiplied by the correction term to give a new factor value. Index for factor is compiled and stored until site is completed.*
Next factor	————	————
Site results	*See Figures 25, 26, 41, 45, and 70 for example factor indices, and Figure 71 for example site indices.*	*The index for recharge is given as $6.3_{1;2}$. 6.3 is the factor value, 1 is the correction term used, and 2 the identifier. The user may ask for a site description, which is displayed.*

NEW SITE OR END EVALUATION?

CHAPTER IV: APPLICATION EXAMPLE : GROUNDWATER SENSITIVITY ANALYSIS FOR WINONA COUNTY, MINNESOTA

The methodology of Chapter I was applied to a susceptibility analysis for Winona County in southeast Minnesota, shown in figure 75. The county is bordered on the east by the Mississippi River, which, along with numerous small streams, has dissected deeply into the geologic profile over the eastern one-third of the county. Aquifers are found in bedrock deposits, which consist of layers of sandstone, shale, and limestone or dolomite deposited from Late Cambrian (525 million years ago) to Middle Ordovician time. The geologic sequence is described in figure 76.

The Prairie du Chien Group and Ironton and Galesville Sandstones are the two primary aquifers in the county. Figure 75 indicates location of the important aquifers. The Prairie du Chien Group consists of dolomites which have undergone significant dissolution and karstification. The lower aquifers become important near the Mississippi River where steep valleys have cut into the upland formations. These aquifers consist primarily of sandstones which are separated from the upper aquifers by confining layers. Near the Mississippi River these sandstones have been dewatered.

Figure 75: Location of Winona County, Minnesota. Locations of major aquifers are included.

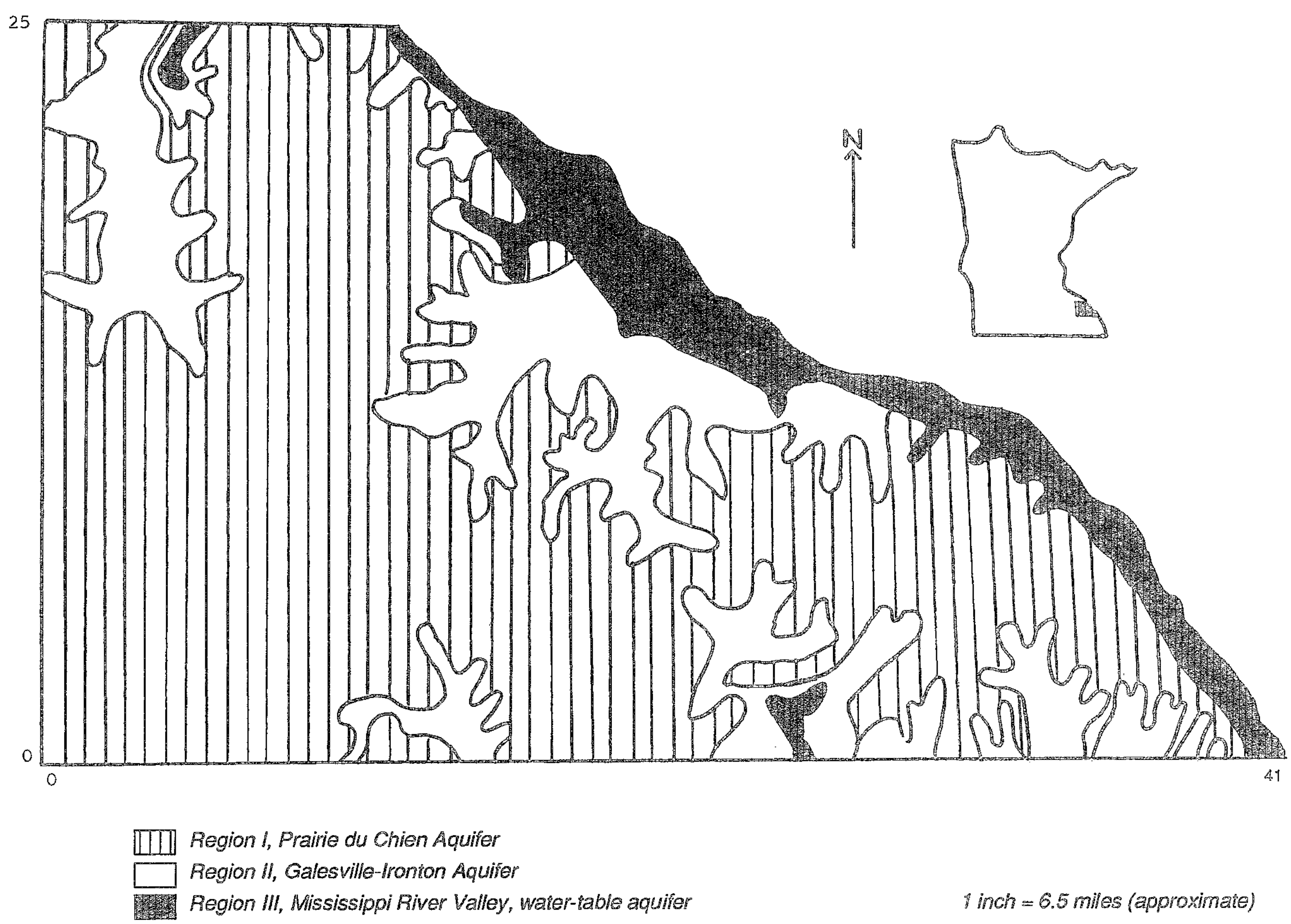
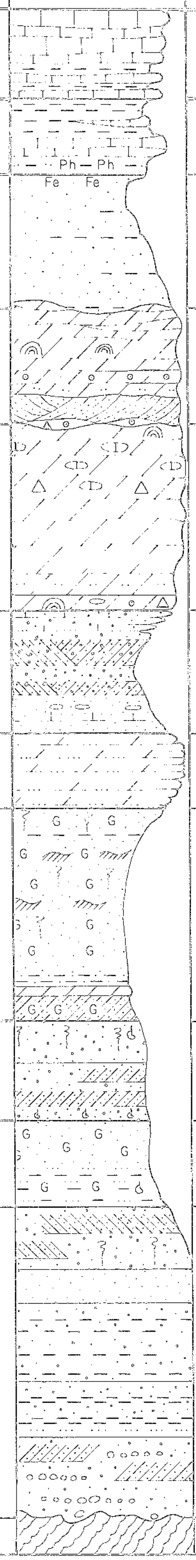


Figure 76: Geologic Sequence for Winona County, Minnesota (Minnesota Geological Survey, 1984)



SYSTEM SERIES	GROUP OR FORMATION NAME	SYMBOL	THICKNESS (feet)	DESCRIPTION
MIDDLE ORDOVICIAN	GALENA FORMATION	Og	60	Fine-grained fossiliferous limestone. Many shale partings in basal 15-20 feet
MIDDLE ORDOVICIAN	DECORAH SHALE	Od	45	Shale and thin interbeds of limestone. Commonly fossiliferous
MIDDLE ORDOVICIAN	PLATTEVILLE Fm	Od	20	Fine-grained fossiliferous limestone
MIDDLE ORDOVICIAN	GLENWOOD Fm	Od	4	Sandy shale
MIDDLE ORDOVICIAN	ST. PETER SANDSTONE	Os	90 to 100	Fine- to medium-grained, poorly cemented, quartzose sandstone; basal contact minor erosional surface. Upper surface commonly iron crusted. Generally massive and unbedded
LOWER ORDOVICIAN	PRAIRIE DU CHIEN GROUP: SHAKOPEE FORMATION	Ops	90 to 115	Thin-bedded and medium-bedded dolomite with thin sandstone and shale beds. Basal 20 to 30 feet is fine-grained quartzose sandstone. Local red iron staining. Basal contact minor erosional surface
LOWER ORDOVICIAN	PRAIRIE DU CHIEN GROUP: ONEOTA DOLOMITE	Opo	160 to 180	Thick-bedded to massive dolomite. Some sandy dolomite in basal 10 to 20 feet. Vugs filled with coarse calcite in upper part. Minor chert nodules. Upper part near contact with Shakopee commonly brecciated
UPPER CAMBRIAN	JORDAN SANDSTONE	Єj	100 to 120	Sandstone. Top 30 feet is thin bedded and well cemented by calcite. Middle part is medium- to coarse-grained quartzose sandstone; generally uncemented and iron stained in outcrop. Basal 35 to 40 feet is very fine to fine-grained sandstone
UPPER CAMBRIAN	ST. LAWRENCE[1] FORMATION	Єs	50 to 75	Thin-bedded dolomitic siltstone. Minor shale partings
UPPER CAMBRIAN	FRANCONIA[1] FORMATION	Єf	140 to 180	Thin-bedded, dolomite-cemented glauconitic sandstone. Very fine to fine grained. Contains minor dolomite beds near base and shale partings throughout
UPPER CAMBRIAN	IRONTON & GALESVILLE SANDSTONES	Єig	90 to 120	Ironton: Poorly sorted, silty, fine- to medium-grained quartzose sandstone with minor glauconite Galesville: Fine- to medium-grained, well-sorted quartzose sandstone
UPPER CAMBRIAN	EAU CLAIRE[2] FORMATION	Єe	90 to 125	Very fine to fine-grained sandstone and siltstone. Some is glauconitic. Interbedded shale
UPPER CAMBRIAN	MT. SIMON[2] SANDSTONE	Єm	290 to 350	Fine- to very coarse grained, poorly cemented sandstone. Contains pebbles in basal 20 to 40 feet. Sandstone generally moderately to well sorted. Greenish-gray shale mottled with grayish-red in basal third of formation. Basal contact major erosional surface
PRECAMBRIAN[3]		pЄ		Biotitic granite gneiss in eastern part. Poorly known in west

Legend:

- LIMESTONE
- DOLOMITE
  - sandy
- SANDSTONE
  - fine to very fine
  - medium to coarse
  - shaly
- SILTSTONE
- SHALE
- GNEISS

- Vugs (filled with coarse calcite)
- Chert
- Oolites
- G Glauconite
- Fe Iron stain
- Ph Phosphate pellets
- Algal stromatolites
- Fossiliferous
- Worm bored
- Pebbles
- Flat-pebble conglomerate
- Cross-bedded
- Ripple cross-laminations
- Dolomitic
- Calcareous

[1] St. Lawrence and Franconia Formations undivided on map. Symbol: Єsf
[2] Eau Claire Formation and Mt. Simon Sandstone undivided on map. Symbol: Єem
[3] Precambrian shown only on sections

The topography ranges from gently rolling upland sites in the west to steep stream valleys and flat alluvial plains in the east. Precipitation is approximately 30 inches annually throughout the county.

Bedrock deposits are overlain by silt deposits less than ten feet thick in the west. Toward the east the thickness of these deposits increases, with increasing amounts of till, colluvium, and older loess deposits, eventually reaching a thickness of over 150 feet. Along the Mississippi River, deposits of alluvium are over 100 feet thick. Soils follow similar patterns of thickness and are generally silt-loam in texture. There are numerous bedrock exposures and sinkholes throughout the upland areas.

Agriculture is the primary land-use in the county except on steep valley hills, which have been left forested (Minnesota Geological Survey, 1984).

The goal of the analysis was to develop a base system designed to assess relative recharge potential throughout the county. Specific areas could then be further evaluated for sensitivity of specific aquifers to pesticide contamination.

The county was divided into three areas as shown in figure 75. Upland areas (Region I) were evaluated for recharge sensitivity to the Prairie du Chien Group. Along stream valleys and hills, the Galesville-Ironton Formations were evaluated (Region II). Along the Mississippi River a water-table aquifer underlying the surface deposits was evaluated (Region III). This division accounted for the primary water sources and allowed a more complete analysis of the county. Factors included depth-to-water, depth-to-bedrock, infiltration, and permeability-of-the-vadose-zone for Region I; depth-to-water, infiltration, and permeability for Region II; and infiltration and permeability for Region III. Equation 1a was utilized and water available for recharge was assessed throughout the county. Topography was color coded and used as a corrector for infiltration. Identifiers included: karst aquifer (K); sinkholes common (S); floodplain site (F); presence of sand terraces (T); and bedrock at or near the surface (B).

Depth-to-water was determined as the distance, in feet, from the land surface to the water table. Depth-to-bedrock was considered in Region I because the bedrock is highly dissolved and has the potential to rapidly transmit water. This created a check on depth-to-water. Sources of data included topographic and geologic maps (Minnesota Geological Survey; United States Geological Survey, 1972).

Potential recharge was computed as precipitation minus evapotranspiration (P-ET). ET was computed using the Blaney-Criddle method. Data was derived from climatological records of eight weather stations in or near Winona County (National Oceanic and Atmospheric Administration, 1987). Recharge was assumed to occur under saturated conditions, primarily in the spring following snowmelt. The layer of lowest conductivity in the vadose zone was used to determine the permeability score. Corrections were made if the bottom of this layer was located within 15 feet of the land surface. This depth was assumed as the maximum depth of saturated soil in the spring. Data was derived from geologic maps. Infiltration was based on permeability of soil materials. A root zone of 3 feet was assumed. Soils with deeper root zones were corrected based on increased storage capacity for evapotranspiration, while thin soils had the opposite effect. Geologic maps and soil atlases provided infiltration data (Minnesota Agricultural Experiment Station, 1973).

The slope classes for color coding topography included:

1. Flat bottomland with slope less than 2 percent,
2. Bottomland with slopes of 2-12 percent,
3. Upland sites with slopes less than 6 percent,
4. Upland sites with slopes of 6-12 percent, and
5. Steep slopes over 12 percent.

A portion of the county, color-coded for topography, was shown in figure 42. Identifiers were based on geologic, soil, and climatic data. Figure 77 summarized the analysis for the county. Rating scales used for this analysis were developed earlier (figures 8 and 9).

Correctors and identifiers, described in figure 77, were primarily derived from soil texts and literature presented in Chapter II of this paper.

The final analysis was made by overlaying a 41 x 25 grid on a 1:100,000 county map (see figure 75 for grid coordinates). Each grid intersection corresponded to a section corner and an analysis point. There were a total of 691 sites, each located one mile from adjacent sites.

A sensitivity map derived from this analysis is shown in figure 78 (in Color Appendix), with selected indices shown in figure 79. Areas of extreme sensitivity are located along the Mississippi River, where high inputs of runoff water exist, and on upland sites with little surficial cover overlying the karst aquifer. Areas of low sensitivity are primarily the result of confining layers. The indicated sensitivity

Figure 77: Factor analysis for Winona County, Minnesota

Objective : *Develop a base system designed to assess relative recharge potential throughout Winona County (located in southeastern Minnesota).*

Factors : *Recharge potential, depth-to-water, depth-to-bedrock, permeability-of-the-vadose-zone, and infiltration for Region I; recharge potential, depth-to-water, permeability-of-the-vadose-zone, and infiltration for Region II; and recharge potential, permeability-of-the-vadose-zone, and infiltration for Region III.*

Methodology : *Recharge potential in inches, as precipitation minus evapotranspiration. Precipitation was determined at eight weather stations in or near the county. Evapotranspiration was computed using the Blaney-Criddle method. Depth-to-water and depth-to-bedrock, in feet, were computed as the difference in elevation between the land surface and the top of bedrock or the underlying aquifer. Sources of data were Minnesota Geological Survey maps and USGS topographic maps. Permeabilities, in meters per second, were determined from tabled values containing average permeabilities of geologic materials. Sources of data included geologic maps and texts. Infiltration was computed in a manner similar to permeability but was applied to the root zone (2-4 feet below the land surface). Sources of data were soil atlases. Topography was color coded into slope classes using topographic maps. Classes included red (flat bottomland with slopes less than 2%), blue (lowlands with slopes of 2-12%), green (slopes of 12% and greater), orange (uplands with slopes of 6-12%), and brown (uplands with slopes of 0-6%).*

Equation : *Equation 1a was utilized.*

Scales : *Scales developed in Figures 3 and 8 were utilized.*

Correctors : *Infiltration was corrected for topographic position. Infiltration values were multiplied by the following values for each particular color class:*

Red — 1.61
Blue — 1.11
Green — 0.65
Orange — 0.92
Brown — 1.00

These were adapted from runoff coefficients for silt-loam soils (Beasley, 1972), which are the predominant soil textures found in the county.

Identifiers : *K - karst aquifer*
S - sinkholes common
F - floodplain site
T - presence of sand terraces
B - bedrock at or near the surface
1 - Prairie du Chien Aquifer
2 - Ironton-Galesville Aquifer
3 - Alluvial water-table aquifer

Final index : *Site coordinates / hazard index / recharge potential / depth-to- water / depth-to-bedrock / permeability-of-the-vadose-zone / infiltration / topographic code / identifiers. Correction terms are displayed as subscripts. The format used here is different than that used in figure 71 (example indices), although the format of figure 71 could be adapted for this evaluation.*

Figure 78: Hydrogeologic sensitivity in Winona County, Minnesota

(See Color Appendix)

classes (e.g. extreme, high, etc.) are relative and have not been tested for accuracy.

With a computer program, a site index and description can be readily displayed. Examples are shown in figure 80 for three sites located within three miles of each other. Using the sensitivity classes from figure 78, these three sites are at extreme, moderate, and low risk. The descriptions displayed in figure 80 indicate that the site of low risk is located over the Ironton-Galesville aquifer. The other two sites, located over the Prairie du Chien aquifer, differ in the degree and type of surficial coverage. Site descriptions such as those in figure 80 provide a resource planner with a great deal of information for use in land-use decisions or for general information.

An updated soil survey of the county is currently being completed; maps are unavailable but soil descriptions exist. Resource managers wishing to evaluate the relative sensitivity of groundwater to pesticide contamination must incorporate an attenuation factor into their analysis.

Considering 2,4-D, a widely used herbicide in Winona County, attenuation scales were developed and are shown in figure 81. Descriptions for some selected soils in the county are shown in figure 82 (United States Soil Conservation Service, 1987). In figure 83, attenuation scores are shown for these soils based on organic matter content. From figure 82, clay content, pH, and infiltration rate may be used as correction terms or identifiers if the necessary relationships can be established. Once soil survey maps are digitized, values from figure 83 can be incorporated into the existing sensitivity analysis to give an assessment of relative groundwater susceptibility to 2,4-D contamination. This is accomplished by including the attenuation score into equation *1a* and developing the proper factor index.

The analysis described here is not yet a useful management tool due to inaccuracies in the methodology. These include the small number of data points (a grid of 81 x 49 would provide a more useful analysis); minimal support for correction terms, factor scales, and identifiers;

Figure 79: Selected indices from a sensitivity analysis for Winona County, Minnesota

coordinates x	y	A	R	D	DB	P	I*	T	HI	ID
15	0	2	10.48	0.0	—	3.4	4.0	O	25.85	K
15	1	2	10.53	0.0	—	3.4	3.4	G	23.87	K
15	2	2	10.57	0.0	—	3.4	4.0	BR	26.07	
15	3	2	10.59	0.0	—	3.4	3.4	G	24.00	K
15	4	1	10.64	0.3	6.20	8.65	4.0	BR	50.94	S
15	5	2	10.75	0.0	—	3.4	3.1	O	23.29	S,B
15	6	1	10.84	0.4	9.30	9.5	4.0	BR	62.87	S
15	7	1	10.92	0.5	10.00	10.0	10.0	BR	83.27	S
15	8	1	10.99	0.3	6.00	9.0	3.1	O	50.55	S
15	9	1	11.04	0.1	6.60	9.3	3.1	O	52.44	K
15	10	2	10.60	0.0	—	3.4	3.4	G	25.07	K,B
15	11	2	10.80	0.0	—	3.4	4.0	BR	27.33	K
15	12	2	11.10	5.0	—	3.4	3.4	G	25.16	
15	13	2	11.12	0.0	—	3.4	3.4	G	25.21	B
15	14	2	11.13	0.0	—	3.4	10.0	G	49.71	K,B
15	15	2	11.13	0.0	—	3.4	3.1	O	24.12	K,B
15	16	2	11.10	0.0	—	3.4	3.1	O	24.05	K,B
15	17	3	11.02	—	—	6.0	6.0	R	66.12	F

A = aquifer number corresponding to regions of analysis
R = recharge potential in inches
D = depth-to-water score
DB = depth-to-bedrock score
P = permeability score
I = infiltration score
T = topographic color code
HI = hazard index
ID = identifiers

* infiltration was corrected for topographic position as it relates to water retention (see figure 77)

Figure 80: Sample descriptions of hydrogeologic sensitivity for selected sites in Winona County, Minnesota

Site	Index	Description
15,5	15,5/23.29/ 10.75/0.0/ –/3.4/3.1_1/ 0/S/B/2	site coordinates x=15, y=5; hazard index is 23.29, low risk; recharge potential is 10.75 inches; depth-to-water is over 260 feet; depth-to-bedrock is deleted; permeability is $10^{-6.8}$ meters per second; infiltration is $10^{-6.9}$ meters per second and has been corrected for topographic position; topographic code is orange, an upland site with 6-12% slope; sinkholes are common in the area; bedrock may be at or near the surface; site considers the Ironton-Galesville Aquifer;
15,7	15,7/83.27/ 10.92/0.5/ 10.0/10.0/ 10.0_1/BR/ S/1	site coordinates x=15, y=7; hazard index is 83.27, extreme risk; recharge potential is 10.92 inches; depth-to-water is 180 feet; depth-to-bedrock is 0 feet; permeability exceeds 10^{-2} meters per second; infiltration exceeds 10^{-2} meters per second and has been corrected for topographic position; topographic code is brown, an upland site with slopes less than 6%; sinkholes are common in the area; site considers the Prairie du Chien Aquifer;
15,8	15,8/50.55/ 10.99/0.3/ 6.0/9.0/ 3.1_1/O/S/1	site coordinates x=15, y=8; hazard index is 50.55, moderate risk; recharge potential is 10.99 inches; depth-to-water is 220 feet; depth-to-bedrock is 22.5 feet; permeability is 10^{-4} meters per second; infiltration is 10^{-7} meters per second and has been corrected for topographic position; topographic code is orange, an upland site with slopes of 6-12%; sinkholes are common in the area; site considers the Prairie du Chien Aquifer;

Figure 81: Attenuation scale used for analysis of groundwater sensitivity to 2,4-D contamination

Score	0	1	2	3	4	5	6	7	8	9	10
Organic matter content (%)	7.0	6.7	6.2	5.6	4.9	3.7	3.1	2.6	2.0	1.0	0

Figure 82: Characteristics of selected soils from Winona County, Minnesota (United States Soil Conservation Service, 1987)

Soil name and map symbol	Depth	Clay	Moist bulk density	Permeability	Available water capacity	Soil reaction	Shrink-swell potential	Erosion factors K	Erosion factors T	Wind erodibility group	Organic matter
	In	Pct	G/cc	In/hr	In/in	pH					Pct
198C, 198D-------- Rollingstone	0-5	12-25	1.30-1.45	0.6-2.0	0.22-0.24	5.6-7.3	Low----------	0.43	3	6	1-3
	5-10	15-27	1.35-1.50	0.6-2.0	0.22-0.24	5.1-6.5	Low----------	0.43			
	10-60	60-80	1.45-1.65	0.06-0.2	0.09-0.14	4.5-5.5	Moderate------	0.28			
215B, 215C, 215D- Southridge	0-9	8-17	1.40-1.50	0.6-2.0	0.22-0.24	5.6-7.3	Low----------	0.43	3	5	2-3
	9-29	10-30	1.45-1.55	0.6-2.0	0.20-0.22	4.5-6.5	Low----------	0.43			
	29-60	55-80	1.50-1.65	0.06-0.2	0.09-0.13	4.5-6.0	Moderate------	0.28			
262B------------- Medary	0-9	15-27	1.35-1.60	0.6-2.0	0.22-0.24	5.1-6.5	Low----------	0.37	3	5	1-2
	9-13	25-40	1.55-1.65	0.2-0.6	0.18-0.22	4.5-6.0	Moderate------	0.37			
	13-56	35-60	1.55-1.70	0.06-0.2	0.11-0.20	4.5-6.0	High----------	0.37			
	56-60	40-60	1.80-1.90	0.06-0.2	0.09-0.13	5.1-7.3	High----------	0.37			
271-------------- Minneiska	0-8	5-18	1.35-1.50	2.0-6.0	0.15-0.18	7.4-8.4	Low----------	0.20	5	3	2-5
	8-46	5-18	1.40-1.60	2.0-6.0	0.13-0.18	7.4-8.4	Low----------	0.28			
	46-60	2-5	1.50-1.65	6.0-20	0.05-0.08	7.4-8.4	Low----------	0.10			
283B, 283C, 283D, 283E------------ Plainfield	0-8	2-5	1.50-1.65	6.0-20	0.04-0.09	5.1-7.3	Low----------	0.15	5	1	.5-2
	8-31	0-4	1.50-1.65	6.0-20	0.04-0.07	4.5-6.5	Low----------	0.15			
	31-60	0-4	1.50-1.70	6.0-20	0.03-0.07	4.5-6.5	Low----------	0.15			
285A, 285B, 285C- Port Byron	0-16	18-27	1.10-1.20	0.6-2.0	0.22-0.24	5.1-8.4	Low----------	0.32	5-4	6	2-4
	16-42	18-27	1.15-1.30	0.6-2.0	0.20-0.22	5.6-7.3	Low----------	0.43			
	42-60	18-27	1.20-1.40	0.6-2.0	0.20-0.22	5.6-8.4	Low----------	0.43			
299B------------- Rockton	0-9	18-28	1.30-1.40	0.6-2.0	0.20-0.22	5.1-6.5	Low----------	0.28	4	6	2-6
	9-22	25-35	1.40-1.55	0.6-2.0	0.17-0.19	5.1-6.5	Moderate------	0.28			
	22-27	35-60	1.35-1.45	0.6-2.0	0.10-0.14	5.6-7.3	High----------	0.28			
	27	---	---	---	---	---	-------------	----			
301A, 301C, 301D- Lindstrom	0-9	10-20	1.20-1.30	0.6-2.0	0.20-0.22	5.6-7.3	Low----------	0.28	5	5	3-5
	9-36	18-24	1.20-1.30	0.6-2.0	0.22-0.26	5.6-7.3	Low----------	0.28			
	36-60	18-24	1.30-1.40	0.6-2.0	0.20-0.22	5.6-7.3	Low----------	0.43			
322C2, 322D2, 322E2, 322F----- Timula	0-32	10-18	1.30-1.60	0.6-2.0	0.20-0.24	6.1-7.8	Low----------	0.37	5-4	5	1-2
	32-60	10-18	1.40-1.60	0.6-2.0	0.18-0.20	7.4-8.4	Low----------	0.37			
331-------------- Tripoli	0-16	28-32	1.40-1.45	0.6-2.0	0.19-0.21	6.1-7.3	Moderate------	0.24	5	6	6-7
	16-26	22-28	1.45-1.70	0.6-2.0	0.17-0.19	6.6-7.8	Low----------	0.32			
	26-60	20-28	1.70-1.80	0.6-2.0	0.17-0.19	7.4-8.4	Low----------	0.32			
369B, 369C------- Waubeek	0-9	19-24	1.25-1.30	0.6-2.0	0.21-0.23	5.6-7.3	Moderate------	0.32	5-4	6	2-3
	9-26	25-34	1.25-1.35	0.6-2.0	0.18-0.20	5.1-6.0	Moderate------	0.43			
	26-60	20-28	1.65-1.80	0.6-2.0	0.17-0.19	5.1-7.3	Low----------	0.43			
388C, 388D, 388E- Seaton	0-8	10-22	1.10-1.45	0.6-2.0	0.22-0.24	5.6-7.3	Low----------	0.37	5	6	1-3
	8-46	18-27	1.20-1.60	0.6-2.0	0.20-0.22	5.1-7.3	Low----------	0.37			
	46-60	10-25	1.20-1.50	0.6-2.0	0.20-0.22	5.6-8.4	Low----------	0.37			
401B, 401C, 401D- Mt. Carroll	0-8	15-22	1.10-1.20	0.6-2.0	0.22-0.24	5.6-7.3	Low----------	0.32	5-4	6	2-3
	8-40	18-27	1.15-1.30	0.6-2.0	0.20-0.22	5.1-7.3	Low----------	0.43			
	40-60	16-24	1.30-1.40	0.6-2.0	0.20-0.22	5.6-8.4	Low----------	0.43			
455A, 455B------- Festina	0-11	18-24	1.30-1.35	0.6-2.0	0.22-0.24	5.6-7.3	Low----------	0.32	5	6	2-3
	11-53	24-29	1.35-1.40	0.6-2.0	0.20-0.22	5.1-6.0	Moderate------	0.43			
	53-60	22-26	1.40-1.45	0.6-2.0	0.20-0.22	5.1-6.5	Moderate------	0.43			
457E------------- Lacrescent	0-8	18-33	1.25-1.40	0.6-2.0	0.15-0.22	6.6-7.3	Low----------	0.24	2	8	3-5
	8-15	8-23	1.30-1.50	0.6-6.0	0.06-0.09	6.6-7.3	Low----------	0.24			
	15-48	8-20	1.30-1.50	2.0-6.0	0.05-0.08	7.4-7.8	Low----------	0.24			

the preparation of maps by hand from base maps of differing scales; and contouring and extrapolation of incomplete data. Despite these inaccuracies, the final map provides a general description of recharge sensitivity in Winona County.

Figure 83: Ranges of attenuation scores for selected soils from Winona County, Minnesota (see Figure 82 for attenuation scale)

Soil name	*Organic matter (%)*	*Attenuation score*
Rollingstone	*1-3*	*5.8-9.0*
Southridge	*2-3*	*5.8-8.0*
Medary	*1-2*	*8.0-9.0*
Minneiska	*2-5*	*3.8-8.0*
Plainfield	*0.5-2*	*8.0-9.6*
Port Byron	*2-4*	*4.8-8.0*
Rockton	*2-6*	*2.3-8.0*
Lindstrom	*3-5*	*3.8-5.8*
Timula	*1-2*	*8.0-9.0*
Tripoli	*6-7*	*0.0-2.3*
Waubeek	*2-3*	*5.8-8.0*
Seaton	*1-3*	*5.8-9.0*
Mt. Carroll	*2-3*	*5.8-8.0*
Festina	*2-3*	*5.8-8.0*
Lacrescent	*3-5*	*3.8-5.8*

SUMMARY

Standardized rating systems have been developed and implemented as a means of assessing relative groundwater susceptibility to contamination across large areas. Relative site indices or scores are calculated, usually as the sum of factor scores. These scores are used to develop sensitivity contours across an area. Factors are considered important natural parameters influencing contaminant migration. Examples include depth-to-water, recharge, permeability, infiltration, and topography.

Existing rating systems are often inflexible due to data restrictions or failure to account for highly diverse, localized hydrogeologic environments. They rarely utilize all existing data and cannot be expanded or altered for a variety of analyses.

A methodology has been presented to reduce these limitations. The methodology is applicable for any system designer wishing to implement a factor analysis. Utilization of modern computer techniques add tremendous flexibility to the rating analysis.

The fundamental equation used in developing a rating system is either:

$$HI = R((WaAab...n + WbBab...n + ...\ WnNab...n)/(Wa + Wb + ...\ Wn))$$

or:

$$HI = ((WrRab...n + WaAab...n + ...\ WnNab...n)/(Wr + Wa + ...\ Wn))$$

These equations allow the system designer to evaluate any number of available factors at a site and obtain a final site score which can be compared to other sites utilizing different data inputs. The concept of correction terms and identifiers increase the accuracy and information given in a site index.

In developing a rating system, a five-step approach is applied, using the preceding equation. These steps include defining specific objectives; determining factors to be analyzed; identifying data available for use in the analysis; developing scales, correction terms, and identifiers; and applying, testing and refining the system. This methodology may be applied to the unsaturated zone, saturated zone, or to existing contamination sites. Development of a base system which evaluates recharge potential across an area provides flexibility for incorporating specific factors to assess different contaminant susceptibilities.

The advantages of this methodology include tremendous flexibility in system design, factor evaluation, data utilization, and future application; excellent site information and evaluation; and improved accuracy over traditional system evaluations.

As an exemplary application, the methodology was applied to a recharge sensitivity analysis for Winona County, Minnesota.

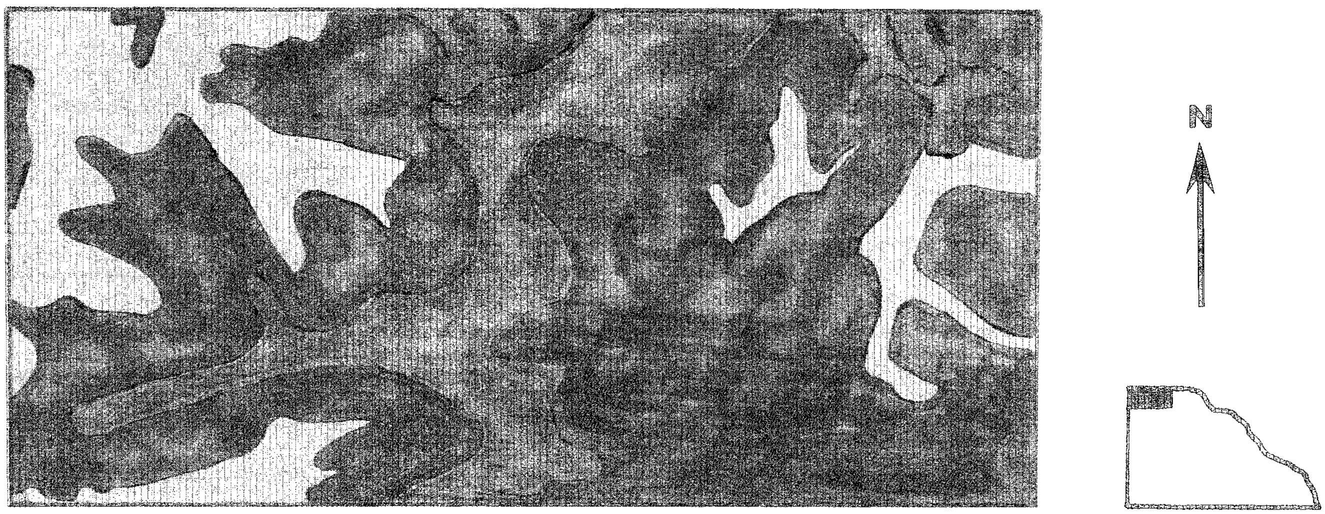
CONCLUSIONS

The methodology described in this paper is intended to provide a tool which system designers can utilize in developing a local susceptibility analysis that is accurate and informative. The discussion provides insight into available evaluation options and processes that influence groundwater recharge and contamination. No system based on factor analysis can replace site-specific evaluations, but the described methodology allows a means to maximize existing and future data in a practical analysis of diverse, extensive geographical areas.

Development of a groundwater susceptibility system requires fiscal expenditures and is intended to accomplish an intended objective. This objective should be the creation of a tool useful in land-use decisions and as a source of general information. Using modern computer techniques, a tremendously informative, accurate, and flexible analysis can be made at relatively little cost. Most important, this analysis can be continually modified to meet new objectives, correct inaccuracies, and incorporate new data, eventually leading to an analysis capable of aiding site-specific evaluations.

COLOR APPENDIX

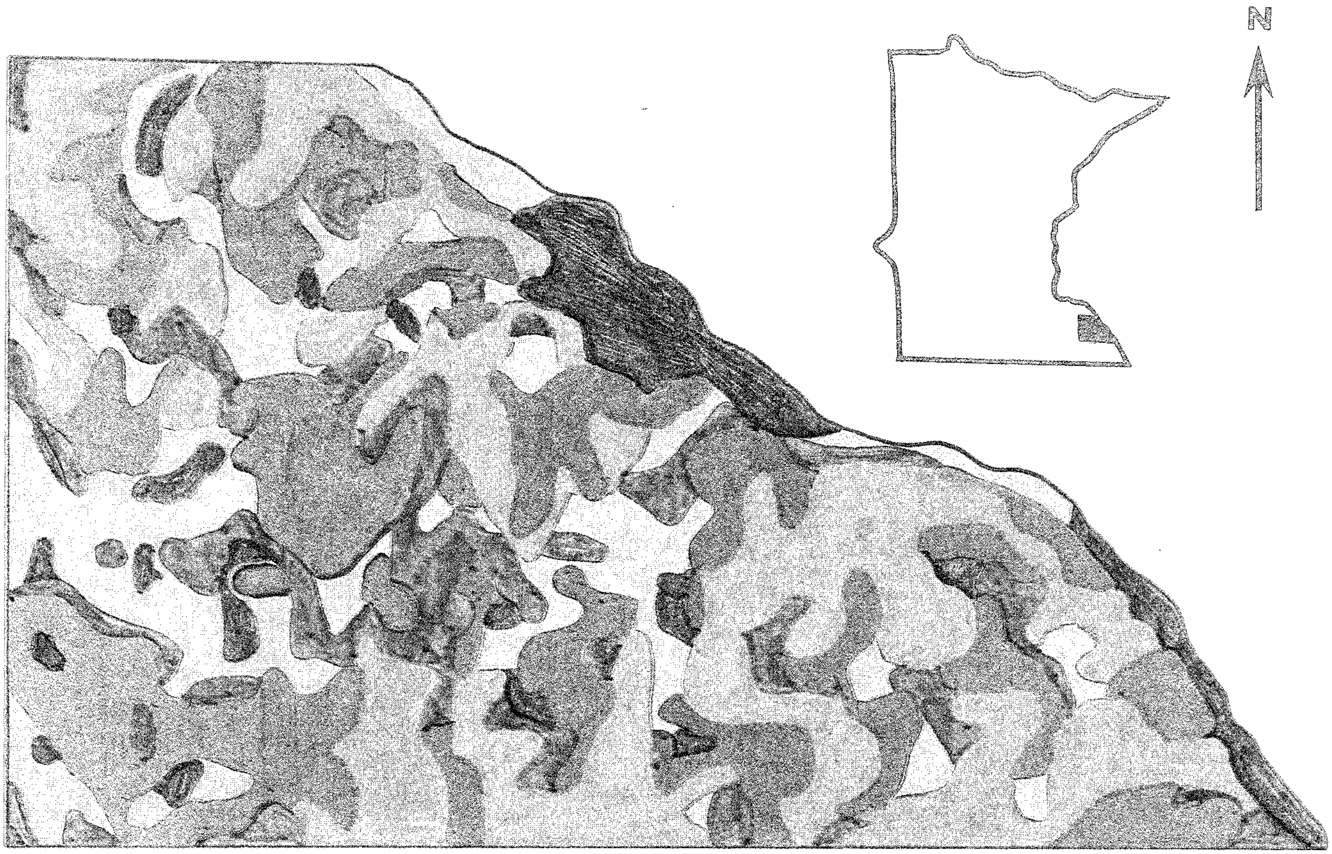
Figure 42: Example of color coding for topography. This area is located in Winona County, Minnesota (location in county indicated, see figure 75 for location of Winona County)



- *bottomland slopes less than 2%*
- *steep slopes of greater than 12%*
- *lowlands with slopes of 2-12%*
- *uplands with slopes of 6-12%*

1 inch = 1.75 miles (approximate)

Figure 78: Susceptibility analysis for Winona County, Minnesota



RISK (site scores)

- *– low (less than 30)*
- *– low-moderate (30–50)*
- *– moderate-high (50–60)*
- *– high (60–80)*
- *– extreme (greater than 80)*

1 inch = 6.25 miles (approximate)

GLOSSARY

Adsorption — The attraction and adhesion of a layer of ions from an aqueous solution to the solid mineral surfaces with which it is in contact.

Anisotropy — The condition under which one or more of the hydraulic properties of an aquifer vary according to the direction of flow.

Confining Bed — A body of material of low hydraulic conductivity that is stratigraphically adjacent to one or more aquifers. It may lie above or below the aquifer.

Correction term — A mathematical value which, when multiplied by a factor value, gives an adjusted factor value.

DRASTIC — An acronym for a hydrogeologic sensitivity analysis designed by the U.S. Environmental Protection Agency. The acronym consists of the following terms:

- D - depth-to-water
- R - recharge
- A - aquifer media
- S - soil media
- T - topography
- I - influence of the vadose zone
- C - conductivity of the aquifer

Factor — A parameter used in assessing groundwater sensitivity to contamination or recharge.

Hydraulic Conductivity — A coefficient of proportionality describing the rate at which water can move through a permeable medium. The density and kinematic viscosity of the water must be considered in determining hydraulic conductivity.

Identifier — An unquantifiable parameter used in a groundwater sensitivity index to acknowledge the presence of processes or features that may affect a site's sensitivity to contamination or recharge.

Infiltration — The flow of water downward from the land surface into and through the upper soil layers.

Isotropy — The condition in which hydraulic properties of an aquifer are equal in all directions.

Perched Groundwater — The water in an isolated, saturated zone located in the zone of aeration. It is the result of the presence of a layer of material of low hydraulic conductivity, called a perching bed. Perched groundwater will have a perched water table.

Root Zone — The zone from the land surface to the depth penetrated by plant roots. It may also be considered as the zone of maximum evapotranspiration, usually taken to be a depth of 2–4 feet.

Vadose Zone — Water in the zone of aeration.

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