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Recommended Protocols & Implicit Assumptions

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GROUND-WATER MONITORING IN KARST TERRANES: RECOMMENDED PROTOCOLS AND IMPLICIT ASSUMPTIONS

by

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FOREWORD

This document was written for four major reasons: 1) The hydrology of karst terranes is significantly different from that of terranes characterized by granular and fractured rocks--flow velocities in karst may be several orders of magnitude higher than in other ground-water settings; Darcy's Law describing flow is rarely applicable; 2) For monitoring to be relevant and reliable in karst terranes, monitoring procedures must be radically different from those in non-karst terranes; 3) There is a need for a practical guide that tells engineers, geologists, hydrologists, and regulators what the monitoring problems are in karst terranes and how to solve them; and 4) Create awareness of the state-of-the-art in monitoring in karst terranes--and provoke thought and discussion about the subject and its implications for ground-water protection strategy. Some of the conclusions may be controversial, but I believe it is better to be aware of problems and to try to solve them than to ignore them.

Approximately 20 percent of the United States (and 40 percent of the country east of the Mississippi River) is underlain by various types of karst aquifers. The vulnerability of these aquifers to contamination--and the consequent threats to public health and safety, as well as to the environment--make it imperative that monitoring of these aquifers be reliable. This document tells how to achieve such needed reliability.

This is a synthesis of the results of research and practical experience by the author and others. My research has been sponsored by the National Park Service, the U.S. Environmental Protection Agency, and the Kentucky Water Resources Research Institute. Much of my research has been done in collaboration with Dr. Ralph Ewers of Eastern Kentucky University.

Reliability, however, is not obtained easily or automatically. Karst is complex. Although general principles are applicable almost universally, reduction of them to absolute rules is difficult. Therefore, rules to guide monitoring should be based on goals or performance, not procedure. Stated generally, there are two goals of monitoring:

1. Acquire and correctly interpret data relevant to the hydrology of a site.

2. Logically monitor water-quality parameters that are indicative of contaminant-related changes in the hydrology.

There are two problems with the attainment of these goals: It is easy to assume that one has achieved them, and the design of a monitoring system is a research project, not the automatic installation of a few wells downgradient and upgradient from a site followed by collection of water samples on a regular schedule. This document discusses reliable procedures for attaining these goals. Following internal EPA review of this document, the report will be upgraded to a Project Report, for general release and use buy three groups of people:

1. Administrators who must evaluate existing or proposed networks for monitoring water quality in karst terranes, but have minimal experience in karst hydrogeology;

2. Consultants and others who must design monitoring networks but may or may not have extensive experience in karst terranes; and

3. The well-experienced tracer who is already familiar with the hydrology and geomorphology of karst terranes but has minimal familiarity with monitoring problems.

Accordingly, some sections of this document may be of greater or lesser interest to one group than to another.

This document is a highly revised, nearly threefold expansion of a review by Quinlan (1989a). It includes a summary of six other papers published in the proceeding of various conferences (Quinlan, 1988a; Quinlan and Alexander (1987); Quinlan, Aley and Schindel, 1988; Quinlan and Ewers (1984, 1985); and Quinlan, Ewers, and Field, 1988), but most of this document is in no other publication or report. It is a unique synthesis and summary of the experience of the writer and his peers who are gratefully acknowledged.

GROUNDWATER MONITORING IN KARST TERRANES: RECOMMENDED PROTOCOLS AND IMPLICIT ASSUMPTIONS

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ABSTRACT

Reliable monitoring of ground-water quality in any terrane is difficult: There are many ways in which violation of sound principles of monitoring-network design and good sampling protocol make it easy to acquire data that are not representative of the water or pollutants within an aquifer. In karst terranes it is especially likely that irrelevant data, inadvertently misrepresenting aquifer conditions, will be obtained.

The special problems of monitoring ground water in most karst terranes can be grouped into four major categories of problems that are rarely as significant in other terranes. These categories are:

1. Where to monitor for pollutants: At springs, cave streams, and wells shown by tracing to include drainage from a facility to be monitored--rather than at wells to which traces have not been run but which were selected because of convenient downgradient location. Wells on fracture traces and fracture-trace intersections and wells located randomly can be successfully used for monitoring, but only if traced positively from the facility to them. Often, the monitoring can only be done several kilometers away from the facility.

2. Where to monitor for background water quality: At springs, cave streams, and wells in which the waters are geochemically similar to those to be monitored for pollutants but which are shown by tracing not to include drainage from the facility--rather than at wells selected because of convenient location upgradient from it. This, too, may have to be done several-kilometers away from the facility.

3. When to monitor: Before, during, and after storms or

meltwater events and also at known base-flow conditions--rather than regularly with weekly, monthly, quarterly, semi-annual, or annual frequency.

4. How to reliably and economically determine the answers to problems 1, 2, and 3: Reliable monitoring of ground water in karst terranes can be done, but it is not cheap or easy.

These problems exist because many of the assumptions made for monitoring ground-water flow in granular media are not valid in karst terranes. Many axiomatic, implicit assumptions are made about flow-systems by those who use various tracers and who may or may not use the traced-spring, -cave-stream, and -well monitoring strategy discussed herein. Many of these assumptions are valid only about 95 percent of the time. Ten of them are reviewed and exceptions to eight are discussed. Among the ten reviewed, the most insidious is that the tracing tests have been well designed, properly executed, and correctly interpreted--and that they are capable of yielding an unambiguous interpretation. Additionally, there are at least thirteen major ways to inadvertently obtain falsely negative results in tracer tests. These too are cited and discussed.

The recommended monitoring strategy is widely applicable in most U.S. karst terranes. Most of them are characterized by local recharge that is discharged at springs. The strategy is not applicable in karst terranes characterized only by recharge, by diffuse discharge into sediment, or by sublacustrine or submarine discharge.

Tracing agents are fundamental and necessary tools for studies, model confirmation and calibration, and predictions of movement of ground water and pollutants. Their legitimate use needs legal recognition as being benign and harmless in the concentrations generally employed--rather than regulation as contaminating substances.

Many of the passionate conflicts between designers of monitoring systems and interpreters of their data are philosophical. Most of these conflicts are between individuals with different perceptions of reality.

A checklist can be used to guide the general sequence of operations necessary in the design of a monitoring system for ground water in karst terranes. The checklist given, however, is no substitute for acquisition of an understanding of the concepts espoused in this document and application of them.

Although some state agencies now require ground-water monitoring at springs, official Federal regulatory recognition of this necessity is needed.

FIGURES

ACKNOWLEDGEMENTS

Ideas expressed here have been critically reviewed by E. Calvin Alexander, Jr., Thomas J. Aley, John V. Brahana, Ralph O. Ewers, Malcolm S. Field, Eric N. Koglin, Arthur L. Lange, Donald S. Mull, Joseph A. Ray, Geary A. Schindel, Christopher C. Smart, and A. Richard Smith. Each has critically reviewed drafts of the manuscript and made numerous valuable suggestions and improvements that significantly enhanced the clarity and content. The reductional talents of A. Richard Smith and Eric N. Koglin repeatedly improved organization and coherence. I am greatly indebted to each of them. Much of the field work in Kentucky that initially stimulated many of my conclusions was meticulously done by Joseph A. Ray and more than 30 other highly motivated National Park Service research assistants, many of whom were near-fanatical in their dedication and heroic in their efforts. I am greatly indebted to them also. Some of my conclusions are a result of work done as a consultant for various clients in several states and Puerto Rico.

INTRODUCTION

OBJECTIVES OF THIS DOCUMENT

The hydrology of karst terranes is significantly different from that of non-karst terranes. Accordingly, the monitoring techniques necessary for reliable, accurate assessment of their ground-water quality are significantly different. This document discusses these techniques, recommends ways to solve the special problems posed by monitoring in karst terranes, calls attention to possible regulatory problems, discusses the assumptions made when monitoring, and makes recommendations throughout the text.

Recent comprehensive reviews of the hydrology and geomorphology of karst terranes are the well-illustrated texts by White (1988) and Ford and Williams (1989). These are the most readable introductions to the subject. A comprehensive analysis of the physics, chemistry, and geology of karst aquifers has been written by Dreybrodt (1988). None of these books is specifically concerned with environmental problems, but they are useful, experience-based syntheses of current knowledge. Present and future trends in karst hydrologic research have been briefly reviewed by Atkinson (1985). A useful guide to the nature and distribution of North American karst aquifers is given by Brahana *et al.* (1988).

To many hydrologists, geologists, and engineers, the flow of ground water in karst terranes is mysterious, capricious, and unpredictable. Many others recognize the term, karst, but do not understand its significance. Few publications adequately discuss predictive aspects of environmental hydrologic problems of karst terranes or offer practical, experience-based insights for solving them. Notable exceptions are Alexander *et al.* (1988), Aley (1977, 1988), Aley & Thompson (1984), Bonacci (1987), Crawford (1984), Milanović (1979), Palmer (1986), Quinlan (1986b, 1988a, 1988b), Quinlan & Alexander (1987), Quinlan and Ewers (1985, 1986), Quinlan & Ray (1981, 1989), Smart & Hobbs (1995), Smoot *et al.* (1987), and White (1988). Both the actual problems and the policy problems of flow prediction in karst terranes are real. Yet they are totally ignored in nearly all of the environmental monitoring literature and by most current U.S. Environmental Protection Agency (EPA) and state ground-water monitoring regulations. These problems can, nevertheless, be solved. The National Water Well Association regularly offers practical course in karst hydrogeology to professionals who to learn about these problems and attempt to solve them.

Although some EPA publications take note of the problems of monitoring in karst terranes and endorse the traced-spring, -cave-stream, and -well monitoring concepts first espoused by Quinlan and Ewers (1984, 1985) and analyzed in this document (Office of Ground-Water Protection, EPA, 1987, 1988; Mull et al., 1988), most EPA documents, and specifically the Technical Enforcement Guidance Document concerning ground-water monitoring at sites governed by the Resource Conservation and Recovery Act (RCRA), ignore these concepts. The significance of the proposed strategy for monitoring in karst terranes was strongly endorsed in letters to EPA from numerous authorities who urged, albeit unsuccessfully, that they be included in the revision of the Draft TEGD on monitoring for RCRA sites (Office of Waste Programs Enforcement and Office of Solid Waste and Emergency Response, EPA, 1986).

Most current regulations concerning ground-water monitoring disregard the manifold problems of doing it reliably in karst terranes. In part, this is because they are performance-based, and rightly so. The true goal of ground-water monitoring should be to detect the nature and magnitude of changes, if any, in ground-water quality, as a result of natural processes or human activity--rather than just to comply with the letter of the law and regulations. Monitoring can be reliable, and it can achieve performance goals, but sometimes it needs the aid of technical guidance in order to achieve its goals.

Proposed EPA regulations concerning municipal solid waste landfills are some of the first to specifically address problems of karst terranes and to prohibit siting them in such areas (U.S. Environmental Protection Agency, 1988, p. 33333-35). This prohibition could be waived if an owner or operator can demonstrate structural stability of the facility. The regulation (40 CFR Section 258.15) is an important official recognition of a significant problem in karst terranes, sinkhole development, but it ignores the most widespread problems--those associated with reliable monitoring of ground-water quality as affected by leakage from a facility located in a karst.

This document discusses where and when to take relevant water samples and how to get the data essential for making the where and when decisions; it also discusses implicit assumptions made when monitoring with the strategy recommended herein. No attempt is made to discuss the design and construction of monitoring wells, protocol for sample custody, or quality assurance/quality control (QA/QC) for dye-tests. All but the last of these topics have been adequately addressed by others; many topics are being reviewed and codified by the American Society for Testing and Materials (ASTM) Subcommittee D-18.21 on Ground-Water and Vadose Zone Investigations. Section .09 of this ASTM subcommittee, chaired by the writer, is developing standards for dye-tracing and ground-water monitoring in karst terranes.

Two major types of ground-water flow occur in karst aquifers-- conduit flow and diffuse flow, each of which is an end-member of a continuum. Springs and cave streams in conduit-flow systems are “flashy”, as expressed by high ratios between their maximum discharge and base-flow discharge, typically 10:1 to 1000:1. Discharge responds rapidly to rainfall. Flow is generally turbulent. The waters possess low but highly variable hardness; turbidity, discharge, and sometimes temperature also very widely. Where a karst aquifer is less developed and is characterized primarily by diffuse flow, its behavior is less flashy; the ratio between maximum discharge and base-flow discharge of major springs is low (4:1 or less) , and the response of their discharge and water quality to rainfall is slower than in conduit-flow springs. Flow is generally laminar. Hardiness is higher than in conduit-flow springs, but hardness, turbidity, discharge, and temperature have low variability (Quinlan and Ewers, 1985). The variations in and relations among these properties and their variability as a function of aquifer flow, storage, and recharge have been described in a significant paper by Smart and Hobbs (1986).

Two important and seemingly contradictory points need to be made about diffuse flow:

1. Movement of water through most parts of a diffuse-flow aquifer is similar to movement of water through granular aquifers. Darcy’s law is operative (Hickey, 1984; Wailer & Howie, 1988).

2. Although water from a diffuse-flow spring may be discharged from an obvious conduit, perhaps 3 m (10 ft) in diameter, the geometry and configuration of the “plumbing system” that feeds it near the orifice is trivial. Wilson and Skiles (1988) and Stone (1989) have published maps of different cave systems with more than 11 km (6.5 mi) of braided passage that feeds diffuse flow springs.

   The most significant controls on flow-type are the types of recharge and storage, as discussed by Smart & Hobbs (1986). These most influence the degree of variability of water chemistry and the magnitude, timing, and duration of response of springs and wells to storms. For very large ground-water basins there is additional dampening of response to storms as a consequence of their sheer size and the greater time necessary to transmit the storm input to their spring (White, 1988, p. 186-187). Also, individual storms will tend to overlap and seasonal trends will comprise the most obvious part of the annual record.

A quick, inexpensive way to distinguish between a conduit-flow spring and a diffuse-flow spring is to observe its turbidity and to measure its specific conductivity before, during, and after several large storms. If the spring is characterized by conduit flow, the water will be turbid; the coefficient of variation (standard deviation ÷ mean x 100) of its specific conductivity will be 10 to more than 25 percent. If the spring is characterized by diffuse flow, its water will always be clear to slightly turbid; the coefficient of variation of its specific conductivity will be less than 5 percent (Quinlan and Ewers, 1985). Remember, however, that conduit flow and diffuse flow are end-members of a continuum. Many springs are fed by a mixture of both types of flow (White, 1988, p. 183-187).

Ground-water flow in conduits and fractures of karst aquifers differs radically from flow in other aquifers. Most commonly, it is to springs by way of caves. Such flow is generally faster than in other aquifers; extreme velocities of 2300 m/hr (7500 ft/hr) have been observed, while a range of 10 to 500 m/hr (30 to 1500 ft/hr) is typical for many conduits (Quinlan & Ewers, 1985, p. 202). [The latter two flow velocities are equivalent to approximately 90 and 4400 km/year (55 and 2700 mi/year).] Thus the effects of leakage or a spill of hazardous material on water quality in a karst aquifer can be sensed at great distances in less than a day. Conceptual and porosity relations between conduit-flow aquifers, fractured aquifers, granular aquifers, and diffuse-flow aquifers are shown in Figure 1.

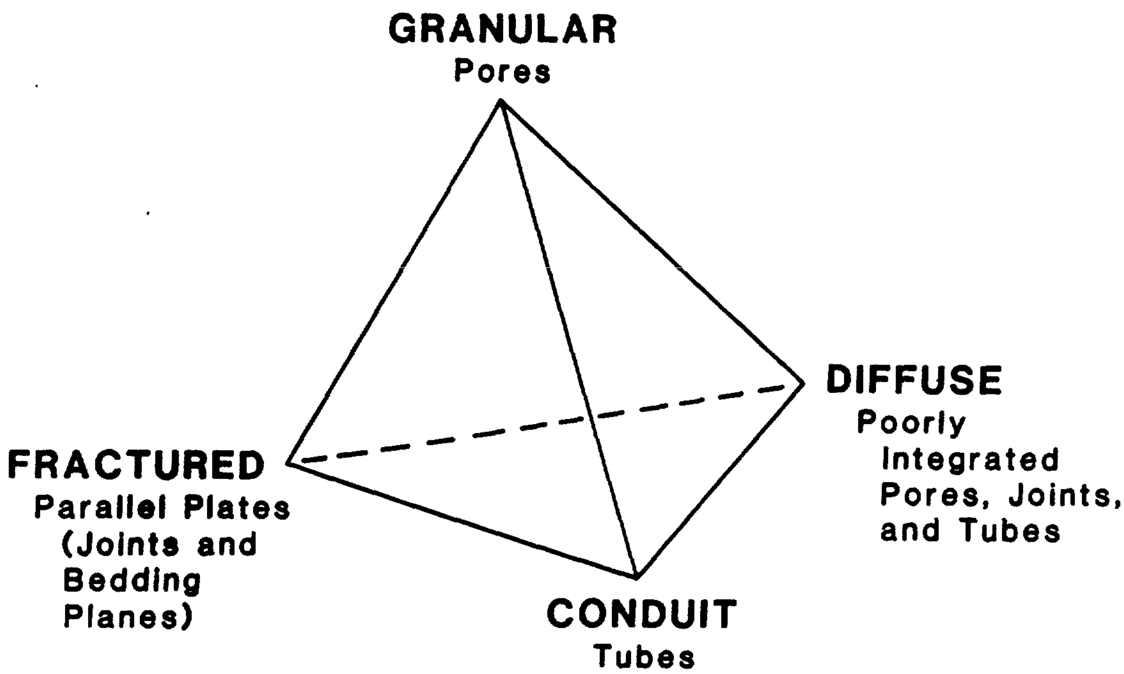

Figure 1. Relation between four major types of aquifers and the dominant porosity geometry in each. This is a tetrahedral continuum. (after Quinlan & Ewers, 1985)

Waste-disposal facilities should not be located within a karst terrane unless one is willing to risk sacrificing the use of at least part of the subjacent karst aquifer as a source of potable water. This is a high risk, almost a certainty. Nevertheless, many facilities already exist within a karst environment, and hazardous materials are disposed of without supervision by a karst-aware hydrologist. Also, hydrologists are rarely consulted in the selection of a proper site for an accidental spill of hazardous material.

Immediately beneath the soil and overlying a domain of horizontal flow in diffuse-flow and conduit-flow systems, is the epikarst (also known as the subcutaneous zone). It comprises the strongly karstified water-bearing rock in the vadose zone beneath the soil and above the phreatic zone, but it may be separated from the phreatic zone by tens to hundreds of feet of dry, inactive, waterless bedrock that is locally breached by vadose percolation. Commonly, the epikarst is 3 to 10 m thick (10 to 30 feet). Williams (1983) estimates that 50 to 80 percent of the dissolution done by recharging waters takes place in this interval. Thus its storativity and transmissivity are significantly greater than in the subjacent, less karstified, relatively dry rock in the vadose zone. The peculiar hydrology of the epikarst, and how it differs from that of the diffuse-flow and conduit-flow part of karst aquifers, has been discussed by Williams (1983), Friedrich (1981), Gouisset (1981), Headworth et al. (1982) and Walsh and Puri (1985).

The complex relations between hydrology, landforms, and stratigraphy in karst developed in gently-dipping rocks in a low-relief terrane are shown schematically in Figure 2. The cave stream is recharged through the soil, by sinking streams, seepage through bedrock, and underground tributaries. Flow is integrated at a trunk stream that ultimately discharges at a spring. Many analogies can be made between the hydrology of cave rivers and surface rivers. The terrane shown is hypothetical, but representative of karst terranes in much of the Midwestern United States. Ten different types of ground-water flow are shown; most of them are discussed by Gunn (1986) and White (1988). A comprehensive discussion of this figure is beyond the scope of this document.

Additional concepts of karst hydrology will be introduced in the discussion of where to monitor for pollutants and background. Case histories will be cited to illustrate these concepts.

DESIGN OF MONITORING SYSTEMS IN KARST TERRANES

WHERE TO MONITOR FOR POLLUTANTS

The ground-water monitoring regulations specified by RCRA for

treatment, storage, and disposal facilities (TSDF's) (40 CFR Part 264, Subpart F; and Part 265, Subpart F) prescribe a minimum of four wells to be installed at a facility to be monitored--one well upgradient and three wells downgradient, commonly near the facility boundary and almost always on the facility property. Sampling is to be performed annually, semi-annually, and perhaps quarterly, but rarely as often as monthly. Monitoring wells installed and sampled in this manner in most karst terranes will generate great amounts of carefully collected, expensive data, most of which are useless because the wells usually fail to intercept the contaminants they are intended to detect. They fail to do so because the wells do not encounter the flow lines (cave streams) draining from a site; if they <u>do</u> intercept them, an insufficient number of samples is taken at inappropriate intervals and at the wrong time (Quinlan & Ewers, 1985; Quinlan and Alexander, 1987).

The easiest and most reliable sites at which to monitor ground-water quality in a karst terrane are springs and directly accessible cave streams shown by dye-tracing to drain from the facility being evaluated (Quinlan & Ewers, 1985). It is, however, naive, erroneous, and dangerous to assume that all or most of the springs that must be regularly sampled during tracer tests and ground-water monitoring are indicated on U.S. Geological Survey (USGS) 7.5-minute topographic quadrangles. Experience in numerous karst areas has shown that only about 5 percent of the springs discharging non-isolated <u>local</u> flow are shown on maps. Inclusion of a spring on a topographic map is not necessarily an indicator of the significance of its discharge. Many trivial springs are included on a map because of their cultural associations. Field work to discover springs is mandatory; there is no substitute!

The preferred alternative to the use of springs as monitoring sites is a suite of wells that intercept cave streams shown by tracing to flow from the facility. Cave streams may be difficult or impossible to find with traditional geophysical techniques. New techniques such as the use of streaming potential (measurement of the electrical potential-gradient caused by displacement of ions from fissures and rock grains as water moves from a recharge area to a discharge area; Lange, 1988; Lange & Quinlan, 1988) and acoustic detection (measurement of sound waves caused by the knocking of pebbles against one another during saltation, by cavitating water, or by cascading, riffling, and dripping water in partly air-filled cavities) have been tested with encouraging results by Lange (1988). Stokowski (1987) has also tested streaming potential as a guide to discovery of caves, but his results are not supported by his data. Other non-traditional geophysical techniques also have promise (Lange, 1988).

A second alternative for monitoring sites in karst terranes is a suite of wells located on fracture traces or on fracture-

trace intersections. These wells are usable only if tracer tests show a connection with the facility under base-flow as well as flood-flow conditions. Although some cave passages are coincident with various types of fracture traces and lineaments, not all fracture-related features are vertical and therefore directly above cave passages (Werner, 1988). Many cave streams are developed along bedding planes and are unaffected by vertical fractures. This fact lessens the probability that a well drilled on a fracture trace, a lineament, or the intersection of such linear features will intercept a cave stream. This fact does not challenge the well-known correlation of such linear features with increased water yields (Parizek, 1976).

As a third alternative, randomly located wells could also be used, but only if tracing has first proven a connection from the facility to each of the monitoring wells under various flow conditions. Domestic, agricultural, industrial, and monitoring wells are generally sited where convenient, but for a specific purpose. Even though monitoring-well locations are designed for detection of contaminants or to intercept contaminant plumes oriented parallel to the hydraulic gradient or to be upgradient from them, I consider most such wells to be randomly located--unless they were deliberately sited along fracture traces or fracture-trace intersections. Throughout this document I use the phrase randomly located wells in this sense. Most randomly located and nonrandomly located wells were not intended for aquifer-testing or dye-tracing, but they can be so used. Such wells can not be used as monitoring wells for a facility in a karst terrane unless they have been shown to be positive for tracer released at or very near the facility to be monitored.

At most locations in karst terranes, proper and reliable monitoring can only be done at sites outside the boundary of a facility. Frequently this monitoring can only be done at places up to several kilometers away from the facility.

WHERE TO MONITOR FOR BACKGROUND WATER QUALITY

Springs, cave streams, and wells in settings geochemically and culturally similar to the traced monitoring sites are the only suitable places to monitor background water quality. This is true, however, only when these places have been shown by carefully designed, repeated dye-traces done under conditions ranging from base flow and flood flow not to drain from the facility. There may be no obvious place that can be monitored for background water quality. This is especially true for a facility located on a hill that is also a potentiometric high and characterized by radial flow of ground water. If randomly located, non-traced wells are irrelevant for monitoring possible contaminants in karst ground water, they are equally irrelevant for monitoring background for contaminants.

One must be exceedingly cautious in interpreting negative results of a tracer test. Only some springs and randomly located wells that are negative for dye in a judiciously designed, properly executed trace are in settings geochemically and culturally similar to those of the monitoring wells. Use of such springs and wells for monitoring background <u>when no other monitoring sites are available</u> is rational, but not specifically addressed by current regulations.

Background data at a new facility can also be obtained from wells that are monitored before, during, and after storms. Ideally, this will be for at least a year before operations begin. This duration would probably detect storm-related and seasonal trends in water quality. Analysis of continuous records of stage in these wells can be used for selection of storm-related sampling frequencies and for possible differentiation of wells into several categories that are based on hydrographic and chemical responses to storms. Traces should be run during that year-long period in order to demonstrate the presence or absence of connections between the various wells and the facility to be monitored. If a well does not test positive for dye after allowance is made for probable flow velocity, it can not be considered to be an effective part of a monitoring system. If none of the wells, or an insufficient number of wells to which traces are attempted, tests positive for dye, an <u>effective</u> monitoring system does not exist; one probably does not understand the hydrogeology of the facility. New hypotheses about flow must be devised and tested in order to verify the correctness of the understanding of its hydrogeology.

There are problems associated with the validity of statistical comparison of data from background wells with purportedly relevant data from monitoring wells. No matter what sophisticated statistical tests "prove", if common sense and field observations demonstrate that samples from monitoring sites and background sites are taken from the equivalent of two different populations, the statistical analyses are invalid. EPA, in recognition of problems of comparability, has been seeking ways to make objective, valid comparisons. It has proposed the concept of “ground-water trigger-levels” for determining if a facility is in compliance with regulations. State agencies are to set trigger-levels for compounds and metals after critical review of data on their toxic effects on health of people and aquatic life. If the trigger-level of a compound or metal is exceeded in a monitoring well for a facility, an investigation of possible sources must be made (U.S. Environmental Protection Agency, 1988, p. 33370-71). No matter what concept of background is used, and no matter what the concept of background evolves to, there is still the problem, addressed herein, of validly comparing two sets of water-quality data which may not be related in any way.

DISCUSSION OF WHERE TO MONITOR FOR POLLUTANTS AND BACKGROUND

All six types of monitoring sites and background sites--springs, directly accessible cave streams, wells drilled to cave streams, wells drilled on fracture traces, wells drilled on fracture-trace intersections, and wells drilled randomly--must be tested by tracing, not only during moderate flow but also during flood flow and base flow, in order to prove the usefulness of these sites for monitoring. This must be done during the extremes of expected flow conditions because flow routing in karst terranes commonly varies as stage changes. During flood conditions the water level in conduits will rise. Some of the water may temporarily move through conduits that are dry during low-flow conditions and be switched (decanted) into adjacent ground-water basins, thus being temporarily diverted by them. An example of such hydraulic switching is depicted in Figure 3. During moderate and flood flow, water draining from Park City and from the west boundary of Cave City in the Turnhole Spring ground-water basin may flow via intermediate-level and high-level crossover routes (shown by the north-trending dashed lines with arrows) to as many as three other basins (#4, Sand Cave; #6, Echo River; and #7, Pike Spring, as indicated by the short, north-trending arrow that crosses the boundary of the ground-water basin at the south end of Roppel Cave). These three subsurface diversion routes are also shown more schematically in the western third of Figure 4. They are part of the distributary with 12 springs. Other more complex examples of hydraulic switching are known (Smart & Ford, 1986; Smart, 1988a, 1988b).*

Another peculiarity of water movement in many karst aquifers is distributary flow. An <u>underground distributary</u> is a dispersal route analogous to the distributary at the mouth of a major river that empties into a sea. However, its origin is different, as discussed by Quinlan & Ewers (1985, p. 205, 207) in their description of the similarities between surface rivers and large underground rivers. Figure 4 shows numerous distributaries in the Mammoth Cave area. The positions and geometry of the underground branching shown are schematic, but their existence has been confirmed by cave-mapping and by dye-tracing. Some of these underground flow paths have been mapped. Knowledge of the occurrence and functioning of distributaries is important in the de-

*It would be interesting to make an analysis of the similarities between hydraulic switching in caves and the principles of <u>fluidics</u> (fluid logic circuits that can be used for sensing, logic, memory, timing, and interfacing; Esposito, 1980, p. 338-367). The whimsical idea of a cave as a giant fluidic computer is reminiscent of Douglas Adams's proposal that the Earth is a giant organic computer designed to calculate <u>the</u> Question to <u>the</u> Ultimate Answer (Adams, 1980, p. 181-183).

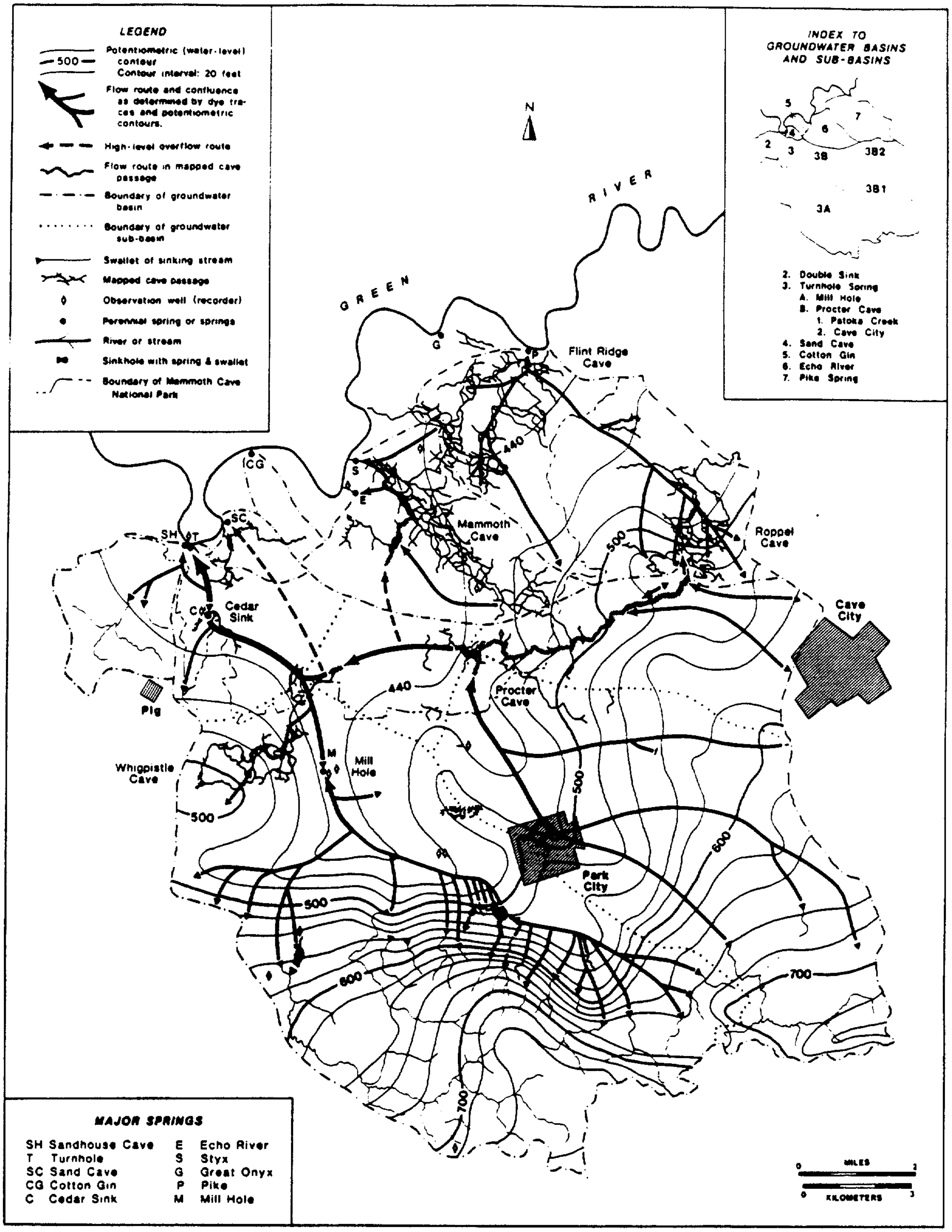

Figure 3. Hydrology of the Turnhole Spring ground-water basin, the major basin draining into Mammoth Cave National Park, Kentucky. Beds dip north at about 11 to 40 m/km (60 to 210 ft/mi). (after Quinlan & Ewers, 1985)

sign of a monitoring system in a karst terrane because pollutants from a point-source in the headwaters or mid-reaches of a ground-water basin or subbasin may flow to all springs in its distributary system or subsystem during periods of high stage. For example, pollutants reaching ground water beneath a point-source in the east-central area of Figure 4 south of Green River would, depending upon flow conditions, disperse to as many as 52 springs in 19 isolated segments along a 19 km (12 mi) reach of Green River. Part of this distributary is shown also in Figure 5. Not all springs between the extremities of a distributary are necessarily a part of the distributary.

The extent to which ground-water basins and their subsurface flow routings can be deciphered by tracing, mapping of the potentiometric surface (water table), and mapping of caves is shown in Figures 3 and 5. The east half of Figure 5 summarizes some of the results of a study of the dispersal of heavy metals from a

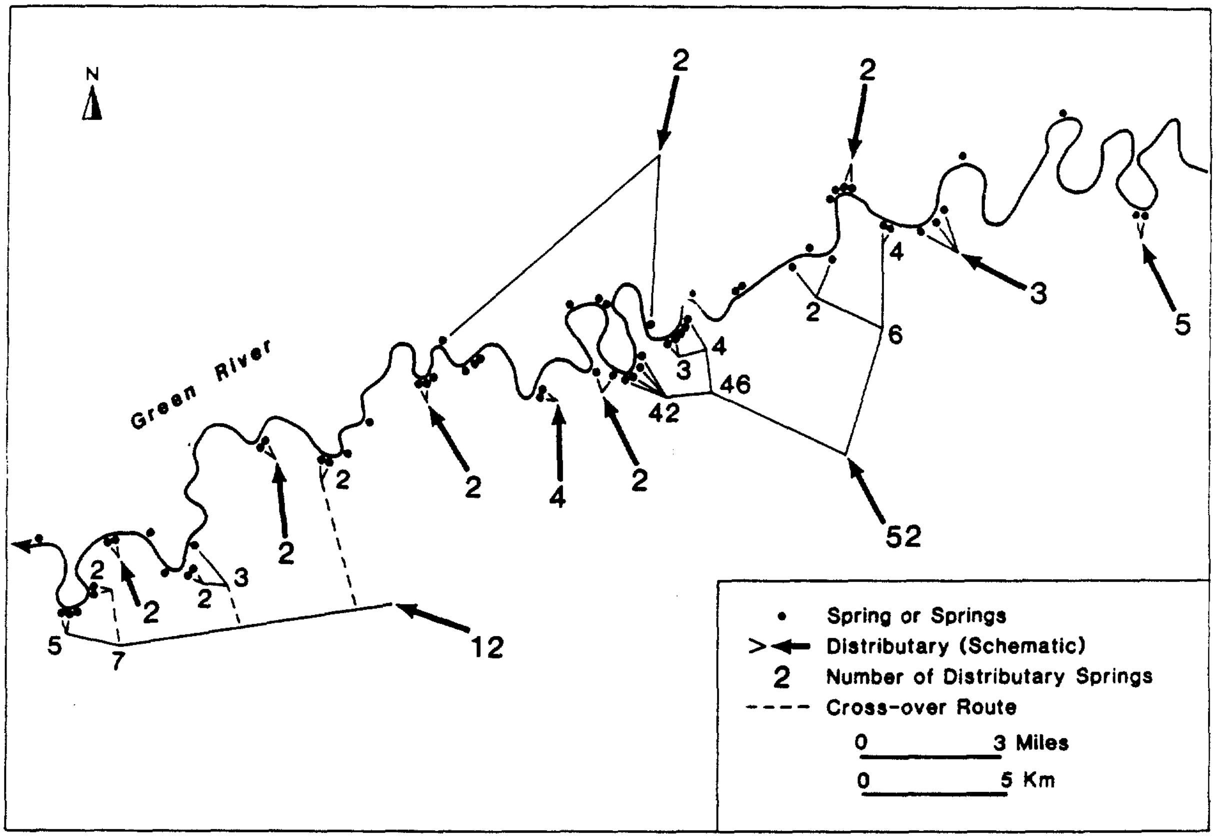

Figure 4. Distributary springs along Green River in the Mammoth Cave area, Kentucky. The numbers indicate the total number of springs in a given distributary system or subsystem. The westernmost large distributary, with 12 springs, is a schematic representation of part of the Turnhole Spring ground-water basin shown in Figure 3. (after Quinlan & Ewers, 1985)

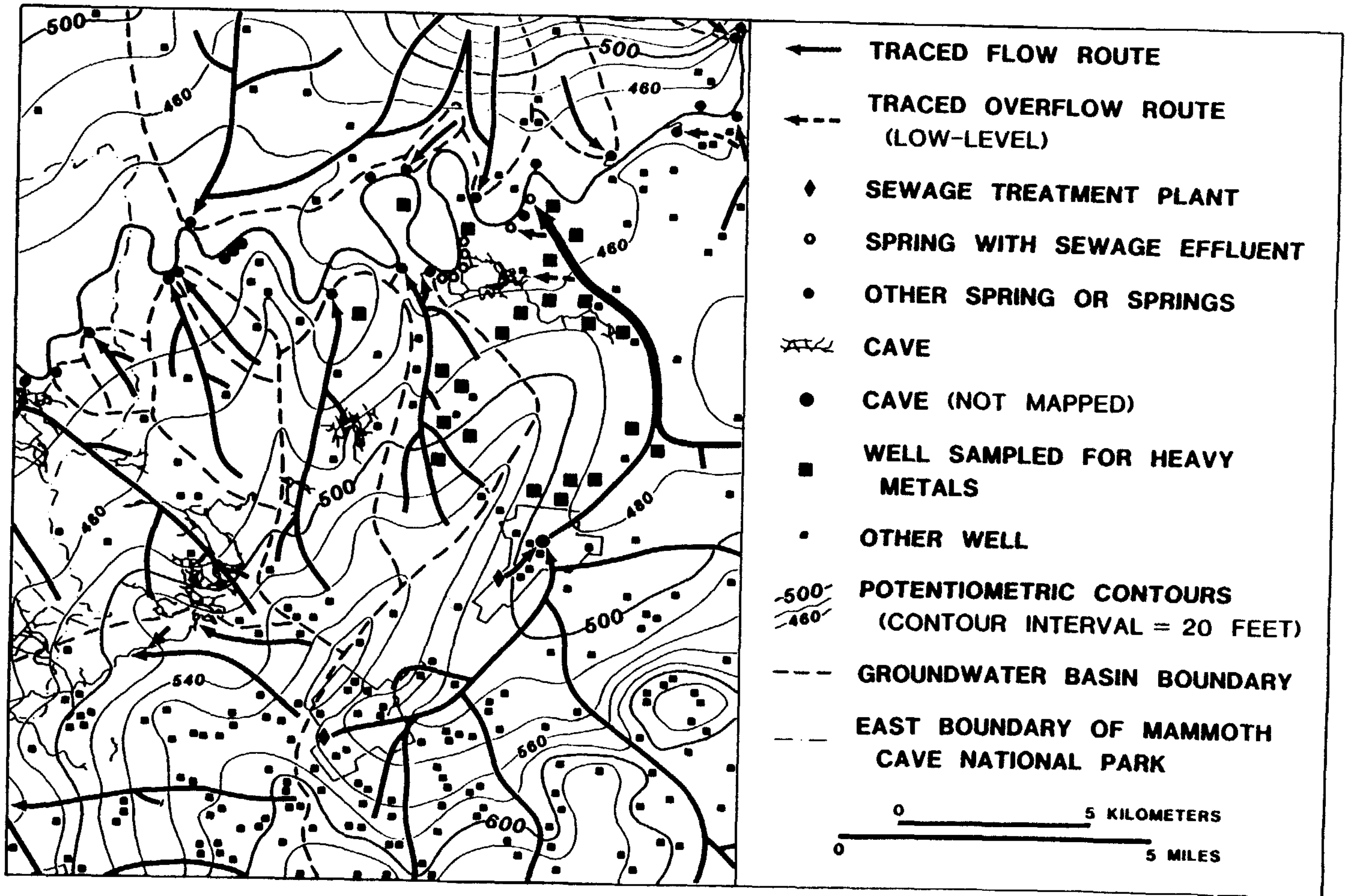

Figure 5. Ground-water flow routes, potentiometric surface, and pollutant dispersal in the vicinity of the towns of Horse Cave (in the east-central part of the map) and Cave City (in the south-central part of the map), Kentucky. Wells are shown as an indicator of the amount of data used to map the potentiometric surface. Part of the east half of Figure 3 overlaps the west half of this map. The outline of Cave City is in both figures. (modified from Quinlan and Ewers, 1985)

metal-plating plant; the metals had been discharged into the ground (in concentrations of more than 10 mg/l via a municipal sewage treatment plant at the town of Horse Cave. [The outline of the town is just east of the center of the map.] A study of the chemistry of water from springs, wells, cave streams, and the west-flowing Green River showed that effluent from the sewage treatment plant flows underground 1.6 km (1 mi) northeast to the unmapped cave beneath the town and then, depending upon stage, to a total of as many as 46 springs in 16 segments along an 8 km (5 mi) reach of the river, about 7 km (4 mi) to the north (Quinlan & Rowe, 1976). None of the 23 wells monitored for heavy metals during base flow showed concentrations higher than background levels, but they should have been sampled also during flood-flow conditions, when water movement might have been reversed from the trunk conduit to some of the wells (Reeker et al., 1988).

Figure 6 shows complex, radial flow of ground water in faulted, flat-lying rocks in the Ozarks of Arkansas. The divergent results of six dye-tests, four of which are within a mile of a proposed landfill, are summarized in this figure. Although the flow routing in Figure 6 is more complex than that shown in Figures 3 and 5, the routing illustrates the advocated maxim, "Monitor the springs!" Monitoring wells that could be drilled on fracture traces at the proposed landfill site might detect seepage of leachate from it, but there is no means--other than by tracing--to identify correctly and conclusively the places to which leachate would (and probably would not) flow. Stated another way, no matter how superbly efficiently the hypothetical monitoring wells on the landfill property were able to detect leakage from the landfill--assuming, for sake of discussion, that they would function so reliably--there is no way, other than by dye-tracing or by monitoring of numerous springs and wells off the property, that one could discover the many consequences of leakage from the proposed landfill!

All four wells at the town of Pindall (Figure 6), which were pumped continuously during the dye-trace from the east boundary of the landfill site (the first trace run), tested positive for dye. By inference, if other wells in the town had been pumped continuously, many of them, perhaps all, would have been positive. Note that none of the four wells immediately east and southeast of the first dye-injection site and neither of the two wells immediately west of the western dye-injection site were positive for dye during the first test. [None of these six wells was sampled for dye during the five subsequent tests.] During the second and third dye-tests, in which dye was injected 1.6 km (1 mi) east and west of the proposed landfill site, respectively, only two wells at Pindall were pumped continuously. The southernmost well there was positive for dye in both tests, as shown; the easternmost well was ambiguous for dye from the east and negative for dye from the west. The high-yield well on the fracture trace (at F) was positive for dye from the second,

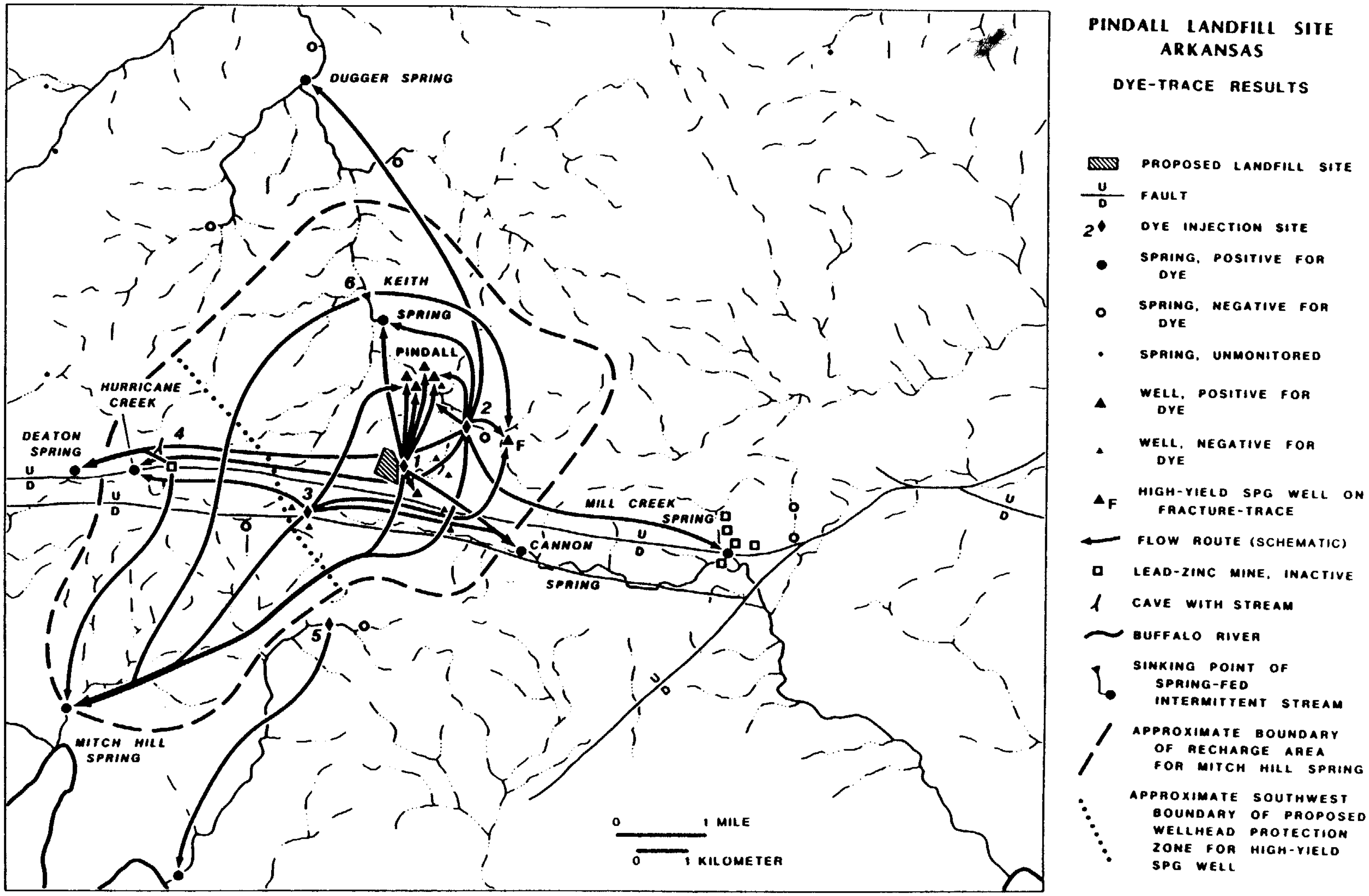

Figure 6. Complex radial flow of ground water near a proposed landfill in flat-lying, residuum-mantled limestone in the Ozarks of Arkansas. An upper and lower karst aquifer, faulting, fracture traces, lineaments, spill-over levels, differences in sampling and analytical protocols, and the intrinsic complexity of flow within the karst all influenced the divergent results of the traces. (after Aley, 1988)

third, and sixth dye-tests, but negative for dye from the first, fourth, and fifth. Dye recovery from this well during the first test was probably hampered by the effects of chlorine added to the well water; chlorine may react with tracer dyes and can destroy low concentrations of any that are present (Smart & Laidlaw, 1977). The difference in tracer results at the high-yield well may also have been a consequence of greater efficiency in sampling and analysis during these latter tests and of the use of the writer's pumped-well dye-sampling device (Aley et al., 1989). The results of the fourth and fifth dye-tests are consistent with the hypothesis that Mitch Hill Spring is fed by deep, complex circulation of ground water which is recharged near faults, lineaments, and fracture traces (Aley, 1988). The implications of these dye-test results for wellhead and springhead protection are discussed by Aley (1988) and Quinlan, Aley, and Schindel (1988). [Dye-trace results from the first test were instrumental in the 1987 defeat of a proposal to use this site as a landfill.]

Dye for the six traces near Pindall was injected near the headwaters of several surface streams and at low- to moderate-flow conditions. The results might be different if the dye-tests were run during base-flow and flood-flow conditions, but flow would still be radial.

Radial flow occurs in many karst terranes and has also been documented in them at waste disposal facilities (Quinlan & Ewers, 1985, p. 214-219; Aley, 1988); it tends to be associated with locations on topographic highs. When evaluating a facility by dye-tracing, one must keep an open mind and place dye-detectors at not only the logical, obvious places, but also at the illogical, the unlikely, and the "No, it couldn't ever go there" places. The cliche "Expect the unexpected" applies, no matter how experienced one is in tracing.

It is tempting and all too easy to take what I think is false comfort in the interpretation of negative tracing results to a well to mean that a waste disposal facility and its liner (if present) are functioning as designed. More specifically, it is easy to say that a facility is either not leaking or is adequately attenuating everything put into it, but I think it is naive to say so unless one is extremely and justifiably confident in the validity of the tracer-test results (Quinlan, Ewers, & Field, 1988). Alternatively, the negative tracing results could just as logically be a consequence of monitoring for dye in wells that do not intersect that part of the aquifer in which ground water actively circulates. In most karst terranes, the latter explanation is more likely to be correct. If water is not standing on the ground in pools or ponds, infiltration must be flowing someplace. If infiltration is flowing someplace, an uncapped facility is probably leaking to somewhere.

Sometimes there is resistance on the part of owners/operators to the performance of a dye-test from their facility or its immediate vicinity. This resistance is based on their fear that environmental administrators and the public would judge that the rate of flow of dye from an adjacent sinkhole, for example, to a spring would be interpreted to be the same as the rate of leakage from the facility. The rates are not necessarily the same. But it is essential to perform the trace so that the consequences of possible leakage become known before leakage actually occurs. The sinkhole is used as an injection-site because flow velocities from it will probably be more rapid than through a liner and because of the economic constraints of investigation time. In such a situation, the most important monitoring question about leakage is <u>where</u> <u>to,</u> not <u>when</u> or <u>how</u> <u>fast.</u>

In some epikarsts and soil zones characterized by diffuse flow, it may be possible to determine the optimal location for reliable monitoring wells by applying standard, topography-based principles of well-siting developed for non-karst terranes. Springs may be unreliable monitoring sites because of the long residence time of pollutants "hung-up" in the epikarst. Not enough research has been done on monitoring of epikarsts to formulate specific protocols for optimally doing so.

RELEVANCE OF OFF-FACILITY MONITORING TO RCRA REGULATIONS

Although no EPA regulations specifically sanction the monitoring protocols recommended herein for karst terranes, these protocols are not inconsistent with any EPA regulations. The following paragraphs discuss how the existing regulations can be interpreted as relevant to these protocols and to off-facility monitoring.

Conventional monitoring practice and EPA regulations for RCRA, 40 CFR Section 264.95, require that ground-water monitoring be done at the compliance boundary of a treatment, storage, and disposal facility (TSDF). Whenever hazardous constituents from a regulated unit exceed concentration limits in ground water (set under Section 264.94) in wells located between the compliance point (under Section 264.95) and the property boundary downgradient from a facility, the owner or operator must institute a corrective action program (under Section 264.100). The concept of off-facility monitoring is not mentioned or implied, but the RCRA TEGD for ground-water monitoring recommends that "Geologic environments with discrete solution channels such as Karst formations must have detection monitoring wells located in those solution channels likely to serve as conduits for contamination migration." (Office of Waste Programs Enforcement and Office of Solid Waste and Emergency Response, EPA, 1986, p. 47).

It has been shown that analysis of off-facility water samples from traced-springs, -cave-streams, and -wells may be the only way to reliably detect and monitor whether the facility is leaking (Quinlan & Ewers, 1985; Quinlan, 1988a; Aley, 1988). Achievement of the recommendation of the TEGD is most practically and most economically achieved by following the strategy these authors advocate. The non-traditional geophysical techniques described by Lange (1988) and Lange and Quinlan (1988) may expedite finding the conduit to be monitored, but these techniques are still in a developmental stage and there are limitations on their applicability. Also, even if a large water-filled conduit is found by geophysical techniques, it must be tested by tracing.

I do not believe that lack of specific mention in the RCRA regulations of monitoring at springs, cave streams, and traced wells prevent their use. Why? Section 264.97(a) of RCRA regulations requires that a ground-water monitoring system consist of a sufficient number of wells, installed at appropriate locations and depths to yield ground-water samples that represent background water quality not affected by the regulated unit represent the quality of water passing the point of compliance. I contend that unless downgradient and upgradient wells intercept a cave stream (not necessarily the same cave stream) they are incapable of doing what they were intended to do; there is an insufficient number of wells unless one or more of them intercepts a conduit that is traced from the facility to be monitored.

I believe that dye-traced springs and cave streams must be recognized as valid, essential components of a ground-water monitoring system in a karst terrane. Section 264.97(d) requires that the ground-water monitoring program include consistent sampling and and analysis procedures designed to ensure monitoring results that provide a reliable indication of ground-water quality below the waste-management area. In karst terranes such reliable indications are most easily and best obtained from springs, cave streams, and wells drilled to intercept them. The relevance of such interceptions must be proven, however, by tracing.

Additional support for monitoring at traced-springs and -cave-streams, rather than at randomly located wells, is implictly but strongly given by other RCRA regulations such as 40 CFR Section 270.14(c), which is concerned primarily with site characterization. Subpart (c)(2) of this regulation requires identification of the uppermost aquifer and aquifers hydraulically interconnected beneath a facility property, determination of ground-water flow direction and flow rate, and statement of the basis for the conclusions of hydrogeologic investigations of the facility. In most karst terranes, an approximation of the required flow direction can be obtained from potentiometric data but it should be confirmed by tracing. Flow-rate data can be obtained only by tracing.

Section 270.14(c) (4) requires description of any plume of contamination that has entered the ground water from a regulated unit and delineation of it on a topographic map. A contaminated cave stream is a plume, albeit a confined plume; the only way to delineate it is by potentiometric and/or geophysical mapping confirmed by drilling (as discussed herein) and by dye-tracing. Although Section 270.14(b) (19) states that the required topographic map should show the area around a facility to a distance of 1000 ft (305 m) from its boundary, it would be reprehensibly legalistic and short-sighted to map a plume to that boundary and stop. Such a map would imply that the plume ceased to exist at the boundary or ceased to be the responsibility of the owner/operator. The plume should be mapped farther, to its end at a spring, when it is technically feasible to do so. The polluted spring will contaminate surface waters.

Section 270.14(c) (5) requires detailed plans and an engineering report describing the proposed ground-water monitoring program to be implemented to meet the requirements of Sections 264.97(a)(2) and 264.98(b). These two Sections require that the monitoring system represent the quality of ground water passing the point of compliance. Since the water flowing from the facility is in a conduit or in a zone of diffuse flow that may include a conduit, wells must be drilled to intercept and monitor it. Substitution of monitoring of a spring for monitoring of a well drilled perhaps 30 m (100 ft) from its point of discharge would be logical, cost-efficient, and not inconsistent with existing regulations.

WHEN TO MONITOR FOR POLLUTANTS AND BACKGROUND

Current conventional monitoring protocol generally requires well sampling annually, semi-annually, occasionally quarterly, and rarely as often as monthly. This is reasonable in most non-karst terranes. In karst terranes, however, even at springs and cave streams judiciously and correctly selected as monitoring sites by the dye-tracing procedures recommended herein, by Quinlan & Ewers (1985), and by Aley <u>et al</u>. (1989), the analytical results of such regularly collected data can be inadvertently misleading. The net result is a waste of time and money. Why? If a karst aquifer is characterized by conduit flow, the chemical quality of water at a spring to which it drains can be greatly affected by the effects of storms and meltwater events (Quinlan & Alexander, 1987; Libra <u>et al</u>., 1986; Hallberg <u>et al.,</u> 1985). In contrast, the water quality of diffuse-flow systems is generally only slightly affected (Quinlan & Ewers, 1985). Sampling must also be conducted during base-flow conditions and analyses should be compared with water quality during and after storms and meltwater events. Water quality is dependent upon the type of pollutant source (point-source vs. nonpoint-source), its volume, and its concentration.

In order to reliably characterize the natural, storm-related variability in water-quality of a spring in a conduit-flow system, sampling must be done much more frequently than was customary in the past. Characterization of water quality, when using typical semi-annual sampling for study of a conduit-flow system is analogous to estimating annual rainfall of an area solely on the basis of rainfall data collected on the same two days of each year.

The effect of sampling frequency on the accuracy of characterizing and depicting storm-related variability in water quality affected by non-point agricultural pollutants is illustrated in Figure 7. Data are from the Big Spring basin, Iowa. Figure 7A is a composite of three figures published by Libra et al . (1986). Discharge was recorded continuously. Pesticides were sampled up to 6 times per day, nitrate was sampled up to 20 times per day, and suspended sediment was sampled up to 17 times per day, depending upon stage (Bernard E. Hoyer, Iowa Geological Survey, Iowa City, oral communication, 1987) . For the sake of discussion, however, assume that these parameters were monitored continuously for the n-day interval shown. Assume also that apparent variations in water quality as a result of its natural variability, the statistics of sampling and analysis, and analytical error are trivial--even though they possibly are not. The most important thing to be seen in Figure 7A is that pesticides and suspended sediment have a maximum that approximately coincides with that of the discharge. Nitrate, however, has its minimum when the others are at a maximum, and reaches its own maximum several days later when the others are approaching their low pre-storm values. The reasons for this peculiar lack of synchrony are discussed elsewhere (Libra et al ., 1986; Hallberg et al., 1985; Quinlan & Alexander, 1987). Note also that pesticide concentrations increased. by more than an order of magnitude during and after the storm. Such storm-related variation in nonpoint-source pollutants is common, but longer term precipitation-related variation in what can falsely be called "background" for pesticide concentration may range over more than three orders of magnitude at some sites in just a few months (Quinlan & Alexander, 1987).

Figures 7B through 7F are based on the data represented in Figure 7A, but they assume "sampling" at intervals of 12, 24, 48, and 96 hours. For the storm event that occurred, sampling at 48-hour and 96-hour intervals (Figures 7E and 7F) is totally incapable of suggesting any significant change in water quality. The 24-hour sampling frequency (Figures 7C and 7D) is better, but the midnight samples happen to miss the decrease in nitrate and most of the increase in pesticides. Even a 12-hour sampling interval (Figure 7B) is only a crude approximation of the continuous sampling represented by Figure 7A.

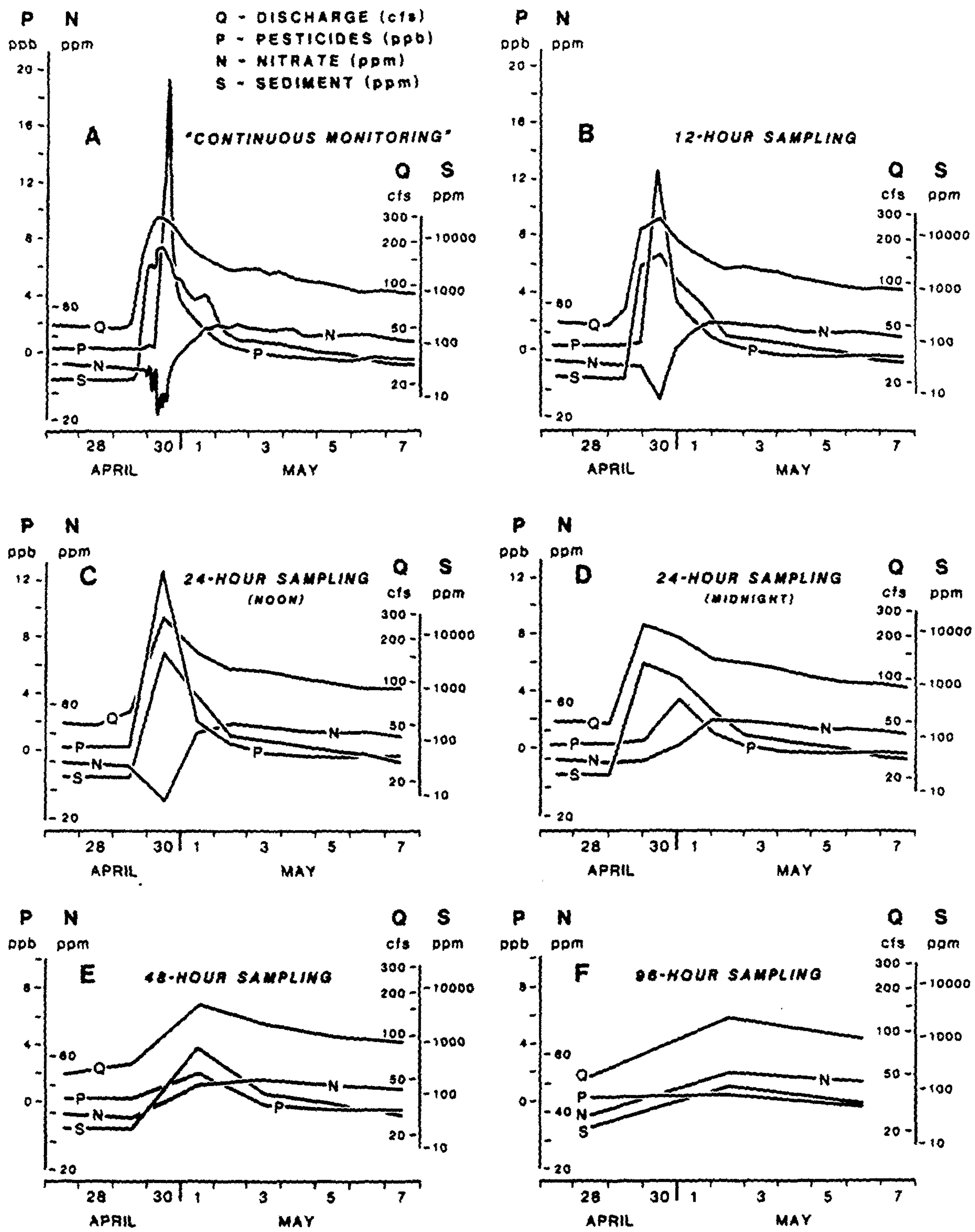

Figure 7. Water quality and discharge during a 1984 storm in a karst aquifer characterized by conduit flow, Big Spring groundwater basin, Iowa. The data shown in A are assumed to be accurate and continuous. Figures B through F show the same data which are "sampled" at the indicated intervals and are derived from A which is composite from 3 figures in Libra et al. (1986).

In September 1987 the EPA announced proposed standards for the concentration of various pesticides in ground water. For atrazine, a herbicide used to control weeds in corn, sorghum, sugarcane, pineapple, and citrus groves, the maximum allowable concentration is 3 parts per billion (ppb) (Office of Drinking Water, EPA, 1987). About 90 percent of the pesticides found in Big Spring basin are atrazine. Therefore, the peak pesticide concentrations shown in Figures 7B, 7C, and 7A are about 4 to 6 times higher than this 3 ppb limit. Midnight sampling barely detected violation of the 3 ppb maximum (Figure 7D) ; the 48-hour and 96-hour sampling (Figures 7E and 7F) totally failed to do so.

It is to be stressed that the “optimal sampling frequency", however defined and determined, will vary with the event, pollutants to be monitored, discharge, flow dynamics, and flow type (conduit flow versus diffuse flow) of the karst aquifer studied, as well as whether one is trying to sample for point-source or nonpoint-source pollutants. For example, the proper sampling frequency necessary to accurately characterize changes in the chemical composition of spring discharge affected by nonpoint agricultural pollution in the Big Spring ground-water basin (and in many other conduit-flow basins in the Mammoth Cave area and elsewhere) will, for similar storms and similar antecedent moisture conditions, be far more than is necessary to give the same accuracy of characterization at the diffuse-flow springs draining the Edwards aquifer in Texas, the Floridan aquifer, and much of the Ozarks of Missouri and Arkansas. Another example: For a given spring and a given set of antecedent precipitation conditions, the optimal sampling frequency for detection of leakage from a landfill would be affected by the thickness and integrity of its cover and the distance of the landfill from the spring. Therefore, one would be wise to allow for a possible lag of pollutants behind maximum discharge in response to rain storms. The optimal sampling frequency at such a landfill could be different from that for point-source contamination such as a spill of hazardous materials along a highway.

Perhaps the most economical solution to water quality problems caused by storm-related or meltwater-related pulses of water that exceed proscribed limits or guidelines for one or more pollutants is to temporarily divert the potentially polluted water out of the water-supply circuit (Quinlan & Alexander, 1987). Study would first be needed to identify which pollutants are present in such waters, when (relative to the hydrography peak) they reach their maximum concentration, whether it is practical to continuously monitor for them or a surrogate, or whether it is best instead to divert stormwater and meltwater when stage and other flow-related parameters reach certain critical values.

Statistical procedures for designing and evaluating sampling strategies are available (Sanders et al ., 1983; Gilbert, 1987;

Makridakis et al ., 1983; Chatfield, 1984; Bendat & Piersol, 1986; Gibbons, 1987; Montgomery et al ., 1987; Rouhani & Hall, 1988), but they are complex.

Until an acceptable, economically realistic, reliable procedure for sampling ground water in karst terranes is developed and tested, probably the best protocol is that proposed and discussed by Quinlan & Alexander (1987, p. 281). In brief, sampling, especially for nonpoint-source pollutants, should start at base flow, before the beginning of a storm or meltwater event, and continue until 4 to 30 times the time from the start of hydrography rise to the time of its crest, depending upon the extent to which an aquifer is characterized by conduit flow or diffuse flow. Sampling may have to be done as often as at 1- to 6-hour intervals in the early part of a precipitation event and at 4- to 24-hour intervals in the recession limb of its hydrography. Appropriate sampling frequency can be determined by analysis of either continuous records of stage and specific conductivity or hourly readings for them.

After an event, the decision about which samples to analyze, if any, should be based on a careful evaluation of the significance of the event, interpretation of data from previous events, and an estimation of the data needed to characterize the monitoring site. Many samples, sometimes all, can be rightly discarded. Hard, judicious decisions must be made. These analytical data must be compared with those for samples taken several times per year during base flow, storms, and meltwater events. Only then can one possibly make a reliable assessment or characterization of the true quality of water draining from a facility. After several years of data have been accumulated and the aquifer behavior is understood, sampling frequency may be decreased.

On the basis of conceptual models and limited data, I believe that monitoring in karst terranes characterized by diffuse flow is easier and cheaper than in those characterized by conduit flow because fewer samples are required. Also, the more a karst aquifer is characterized by diffuse flow, the higher the probability that fracture-trace-sited wells and randomly located monitoring wells can be used reliably. All wells proposed for monitoring use, however, must still be tested positively by tracing. This tracing adds time and cost to the design of a monitoring system, but in the long run, it is cheaper to design it properly.

Much remains to be learned about when to sample ground water in karst terranes. The likely possibility for either deliberate or inadvertent acquisition of falsely negative or falsely positive data from them makes it imperative that people in charge of-sampling and officials in charge of evaluation of sample data have an understanding of karst problems. They should carefully scrutinize all analytical results from such terranes.

APPLICABILITY OF RECOMMENDED TRACED-SPRING, -CAVE-STREAM, AND -WELL MONITORING STRATEGY

The applicability of the traced-spring, -cave-stream, and -well monitoring strategy and the conventional (randomly located well) monitoring strategy in various types of aquifers is shown in Figure 8. Both monitoring strategies may be applicable in some fractured aquifers, some diffuse aquifers, and some karst aquifers. It is to be stressed that there are some diffuse-flow karst aquifers in which the traditional randomly located well strategy works efficiently and is better (Beck, 1986; Wailer & Howie, 1988; Benson et al ., 1988).

The recommended monitoring strategy is not applicable universally. But it is applicable in all diffuse-flow and conduit-flow aquifers that drain to springs which discharge on land or along the shores of streams, rivers, lakes, or seas. Accordingly, the strategy is applicable in most karsts of the following 16 states that have significant amounts of karst: New York, Pennsylvania, Maryland, West Virginia, Virginia, Tennessee, Georgia (Appalachians), Alabama, Kentucky, Indiana, Arkansas, Missouri, Iowa Minnesota, Texas, and Oklahoma. Their karst terranes are characterized by local recharge and discharge. Many of these states include some of the more densely industrialized and populated areas of the U.S. Karst comprises approximately 25 to 30 percent the total area of these states.

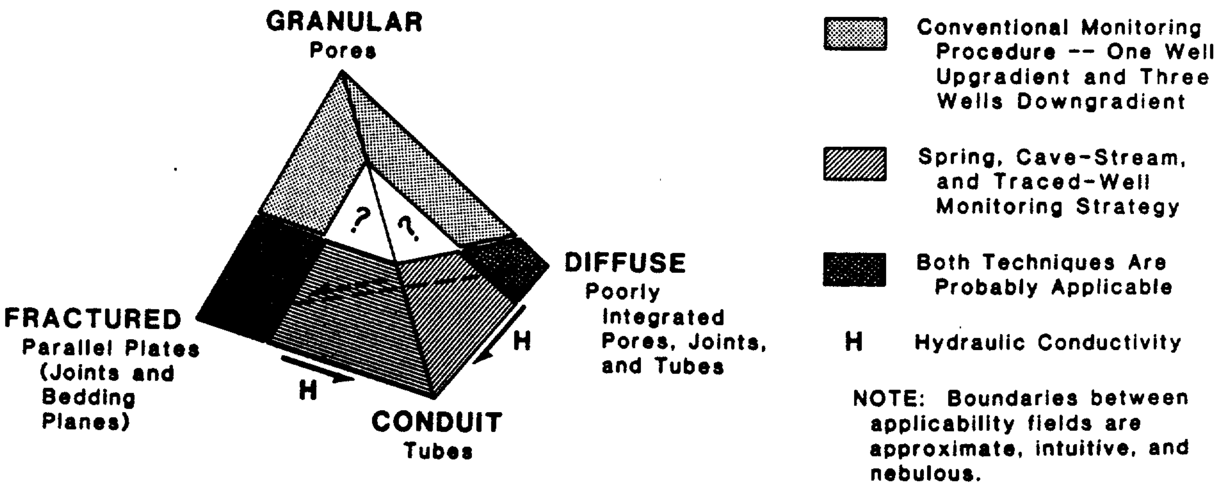

Figure 8. Tetrahedral continuum between four major types of aquifers, dominant pore geometry in each type of aquifer, applicability of the traced-spring, -cave-stream, and -well monitoring strategy in each type of aquifer, and applicability of the traditional monitoring technique (using randomly located wells) in each type of aquifer. The direction of increasing hydraulic conductivity of each aquifer is shown by the arrows. (modified from Quinlan & Ewers, 1985)

The recommended monitoring strategy is applicable only locally in parts of the Floridan aquifer of Florida and in Puerto Rico. In both areas there is significant discharge at springs. The strategy is not applicable in karst terranes that are merely recharge areas of regional aquifers such as the Upper Floridan aquifer of Florida, Georgia, and South Carolina. It is also minimally applicable in karsts mantled by glacial sediments and in which discharge is diffuse, into sediment and/or bodies of water, rather than at discrete springs. I estimate that these karsts comprise about 30 percent of the total area of these states.

The strategy would be applicable in most of the Edwards aquifer of Texas, much of the Upper Floridan aquifer, and part of the North Coast limestone aquifer of Puerto Rico where flow is to springs at the surface, and where most springs are diffuse-flow rather than conduit-flow. Although much of each of these aquifers is characterized by sponge-like permeability, many of their springs are fed by conduits that are commonly braided (anastomosed) (Veni, 1988; Beck, 1986, p. 240; Wilson & Skiles, 1988; Stone, 1989). Accordingly, one might use geophysical techniques for trying to find the main conduit and drill to intercept it, but probably it would be missed. Nevertheless, monitoring could be successfully accomplished at springs. Wells in these sponge-like aquifers could be used as monitoring sites only if there were a positive trace to them from the vicinity of a facility or from the facility itself during low-, moderate-, and high-flow conditions.

There are numerous small areas of karst in the western U.S., but nearly all of them are in isolated, non-industrialized, unpopulated terranes.

Research is needed on the distribution of and criteria for applicability of the recommended strategy in karsts of the states cited in the four preceding paragraphs.

REGULATORY ASPECTS OF DYE-INJECTION INTO WELLS

As discussed under the next major heading, dye-tracing strategy might include proposals for injection of tracers into wells. Strictly speaking, injection of dye or other non-toxic tracers into a well, no matter how noble the reason for doing so, makes it possible to construe the well to be a Class V injection well (a well that is not included in Classes I through IV and that generally injects non-hazardous fluid into or above an underground source of drinking water; 40 CFR Section 144.6(e)) and is thus subject to State and Federal regulations governing its use. Also, some state agencies have interpreted tracing agents used in ground-water investigations to be pollutants or contaminants. If the tracer is toxic, the well would be a Class IV injection well (a prohibited type of well which disposes of hazardous or radioactive wastes into or above a formation within 1/4 mile (0.4 km) of an underground source of drinking water; 40 CFR Section 144.6(d)).

Well-meaning as the above interpretation of regulations for injection wells may be, it is not justifiable in terms of potential benefits for environmental protection, intent of the law-makers, or risk of exposure to pollutants. Like boats put into a lake, tracing agents are used in the water for a good and definite purpose, not put in it for disposal. And like boats, dyes generally used for tracing ground water are benign and harmless in the concentrations commonly employed (Smart, 1984).

Tracing agents are fundamental tools for discovery and prediction of the velocity and dispersal-path of pollutants in ground water and surface water. Interpretation of data from tracer studies makes it possible to protect water quality, public health, and aquatic life. Such data are crucial to the development of wellhead and springhead protection strategies and can be essential for the calibration of computer models of water flow and pollutant movement. Tracing is cost-efficient and is often the only way to get critically needed data.

A further analogy describing the use of tracing agents can be made. Doctors use vaccines and a wide range of diagnostic techniques to prevent and treat illnesses. Some of these vaccines and techniques have definite risks associated with their use. These risks are assumed by an informed patient because the consequences of not preventing or not diagnosing an illness far outweigh the slight risk from use of the vaccine or diagnostic technique.

If and when state officials establish regulations governing the use of dyes or any other ground-water tracer, they should require their use by knowledgeable, experienced professionals.

Additional discussion of regulatory problems concerning the use of dyes for tracing ground water, and a recommended solution to these problems, is given by Quinlan and Field (1989).

Many Federal and State agencies have sanctioned the use of dye-tracing studies in the study of ground-water pollution and time-of-travel of pollutants in rivers. Guidance manuals for tracing techniques exist and have been sponsored by EPA (Davis et al., 1985) and by the Société Géologique Suisse (Parriaux et al., 1988). Updated manuals on ground-water tracing have been written by the U.S. Geological Survey (under contract to the EPA: Mull et al., 1988) and are in preparation for the National Water Well Association (Aley et al., 1989). Several manuals on the use of dyes for measurement of discharge, time of travel, and dispersion in surface streams have been written by the U.S. Geological Survey (Wilson et al., 1986; Kilpatrick & Cobb, 1985; Hubbard et al ., 1982).

The following section on dye-tracing and the design of monitoring systems in karst terranes is written on the assumption that it is legally permissible to inject dye into wells. Qualifying statements that would be necessary if tracer-injection into wells were illegal are omitted for clarity.

HOW TO DETERMINE WHERE, HOW, AND WHEN TO MONITOR GROUND WATER IN KARST TERRANES RELIABLY AND ECONOMICALLY

INTRODUCTION

Where to monitor is typically best determined with the aid of three types of field investigation: dye-tracing, surveying of cave rivers that are shown by tracing to drain from a facility, and detailed mapping of the potentiometric surface.

DYE-TRACING

Selection of Dye-Injection Sites

Under ideal circumstances, one can run the dye-tests necessary for the design of a monitoring system from a perennial sinking stream on the facility. Often no stream is available for dye-injection. In that situation, one can use tank-trucks of water and inject dye at (in decreasing order of desirability) a:

1. Sinkhole with a hole at its bottom.

2. Sinkhole without a hole at its bottom; excavation may reveal a hole that can be used.

3. Losing-stream reach with intermittent flow.

4. Class V stormwater drainage well.

5. Well drilled on a fracture trace or a fracture-trace intersection.

6. Abandoned domestic, agricultural, or industrial well.

7. Well randomly drilled for dye-injection.

Alternatively, it is sometimes practical (or necessary) to trace by injection through the drain field of a septic tank. This is a difficult trace to perform: it should be done with great care and by experienced personnel.

Before going to the trouble and expense of dye-tests from injection-site types 2 through 7, they should first be given either a percolation test or a slug test by injection of potable water or a cylinder of known volume in order to determine if the well is open to the aquifer and to see how rapidly they drain. Alternatively, a pumping test can be run. An electric tape or a pressure transducer can be used to determine the rate of water-level decline in a well during a slug test; plots of such data can be used to select the site that probably has the most direct and open connection to the conduits that are the aquifer drains. If the percolation test of an injection site shows little or no drainage, do not inject dye into it. The procedures for dye-tracing with trucked water are discussed elsewhere (Quinlan & Ewers, 1985, p. 222; Aley et al ., 1989). Wells drilled for dye-injection should probably extend about 8 m (25 ft) below the potentiometric surface or upper bed that may confine a karst aquifer, but this is a site-specific determination.

It is almost economically impossible to design a dye-trace that simulates the conditions beneath a landfill in a terrane characterized, for example, by 20 m (60 ft) of residual soil. Why? Residuum is anisotropic: The distribution of macropores within it is unpredictable. Their permeability may be several orders of magnitude higher than that of the bulk of the residuum (Quinlan & Aley, 1987). The larger macropores, through which fluids move most rapidly, may be several centimeters to several meters apart. The problem of intercepting them with a drill hole intended for dye-injection is analogous to the problem of searching for orebodies with a drill hole; only the size of the targets and therefore the necessary spacing between holes for the same probability of target interception is different. In addition, it is likely that the act of drilling or coring through clayey residuum, no matter how carefully it is done, will smear many of the macropores shut, thus masking their presence. Also, a hole drilled for dye-injection is likely to miss the subjacent conduit system. A randomly-drilled hole tests only a tiny percentage of a facility’s surface and subsurface.

The relative suitability of different drill holes in residuum or bedrock can be evaluated by slug tests, as discussed above. If the holes are very close to one another it would be prudent to have an electric tape or a pressure transducer in the adjacent holes--just to be sure that the rapid fall of the water level in the tested hole is not a consequence of leakage into one or more of the adjacent holes.

Instead of using drill holes for dye-injection, one might excavate a 10 m x 10 m (30 ft x 30 ft) pit to the depth of the bottom of a proposed landfill in the example terrane, say 5 m (15 ft), and carefully construct either a simulated compacted or lined bottom of a cell--or try to make an “undisturbed” bottom. Even then, one could not be sure of having simulated or tested the long-term permeability conditions at the bottom of the land-

fill. A similar carefully excavated and constructed pit, if dug above a solutionally enlarged joint in the subjacent limestone, might leak during the test or several years later. Its failure could be hastened by leakage accompanied by synergistic subsidence or collapse of soil that bridged the joint.

Whether using a sinkhole, a drill hole, or a pit for dye-injection, when designing a dye-trace to evaluate a proposed landfill site, it is necessary to:

1. **Assume that the soil or liner has or will have differential permeability (leakage) that can not be remediated by economically justifiable construction methods. This assumption is supported by an extensive literature, as discussed by Quinlan & Ewers (1985, p. 199). The long-term permeability of any kind of liner may be affected by chemical changes induced by leachate (Quinlan & Ewers, 1985, p. 199; Hettiaratchi & Hrudey, 1987), aging and degradation of polymers used in liners (Segrestin & Jailloux, 1988), and flaws in construction and installation (Rogowski, 1985; Jaywickrama *et al.,* 1988). The urgent questions about leakage are *when, to where* and *how fast.* [This is consistent with the EPA policy of assuming a worst possible case scenario.]**

2. **Test for the consequences of the leakage that is certain to occur. The argument that a dye-trace is irrelevant to evaluation of a particular landfill because the dye was not injected at the bottom of the actual landfill and precisely at its location is specious.**

Leakage may reasonably assumed to be a certainty. Therefore, a dye-test should be designed to maximize the probability of getting the dye through the soil or residuum as rapidly as possible. Few consulting firms or their clients can afford to wait a year or more for test results that, until the dye is recovered, remain negative.

The monitoring system for a facility and the consequences of leakage from it should be tested by tracing from the facility itself. This may not be possible. If it is not, traces should then be run from sites adjacent to the facility, preferably from opposite sides of it, and at points lying on a line approximately perpendicular to the suspected flow direction. This increases the probability of discovering if a facility is near the boundary of a ground-water basin (or well away from a boundary) and whether the facility consistently drains to the same spring (or springs). Such tracer results are relevant to objective evaluation of the facility.

Tracing from sites adjacent to a facility is justified by the principle of hydrologic juxtaposition. This principle is easily

explained. If the geology of a dye-injection site is similar to that of a site immediately adjacent, it is highly probable that tracer results from the two sites will be to the same spring. Obviously, this may not be so in the immediate vicinity of the boundary between two ground-water basins, but it is the reason why a second test is recommended for the opposite side of a facility. In fact, a higher degree of confidence would be achieved for predictions concerning ground-water flow and contaminant transport if they were based on the results of a tracer test run from each side of a facility (on or off it) rather than a single tracer test run from the middle of it.

Selection of Dye-Monitoring Sites

All springs within a radius of perhaps 8 to 25 or more kilometers (5 to 15 or more miles) from a facility, especially those within ±90 degrees of the likely vector of the hydraulic gradient from it, should be found and monitored during dye-traces. At the beginning of an investigation, a prudent designer of a dye-trace will generally assume the possibility of radial flow and will have 360 degrees of coverage with dye-detectors--if only to defend the test design from criticism of inadequacy. The radius of spring search is determined by evaluating stratigraphy, structure, and physiography, and by proposing various tentative hypotheses about possible flow routes and resurgences. These working hypotheses must be tested for each area.

Dye-traces to springs at the bottom of sinkholes are especially important for recognizing segments of the plumbing system of a karst aquifer. These segments between a facility and the spring to which it drains can be used for monitoring. They offer two advantages over the use of springs: less dilution of pollutants or surrogate compounds, and earlier detection of them.

Imagine how extremely different the tracing results in Figure 6 would appear if the designer of the dye-tests shown had followed a hunch (or perhaps the dip of the beds beneath the facility) and monitored only the springs in one particular direction from the landfill rather than in all directions! At the administrative hearing on whether or not to grant a permit for construction of the proposed landfill, held after dye-test #1 (from a point adjacent to the site) but before the other five dye-tests were run, the state's witnesses alleged that it was impossible for dye (or pollutants) to flow in opposite directions. They vigorously but erroneously impugned the validity of the dye-test. Each of the five subsequent tests resolved all questions about the alleged impossibility of radial flow.

Prediction of flow within a ground-water basin characterized by local flow is usually very much like prediction of flow within a surface-water basin. It moves "downhill" (downgradient) to the trunk drain at local base level. If the boundaries of either basin are known, it can be confidently stated that although small-scale local flow from an area (or facility) may be in almost any direction, macro-scale local flow will be to tributaries and ultimately to the trunk that drains the basin. For example, in Figure 3, any dye (or pollutant) injected south of Mill Hole and west of both Park City and the subbasin boundary shown as a dotted line can confidently be predicted to flow to Mill Hole. Similar predictions can be reliably made for potential facilities elsewhere in the Turnhole Spring ground-water basin and for anywhere in Figure 5.

Dye-Tracing Methods

Ground-water tracing can be done with many different tracers, but in general the cheapest, most efficacious ones are fluorescent dyes such as fluorescein* (CI Acid Yellow 73), Rhodamine WT (CI Acid Red 388), CI Direct Yellow 96, and optical brighteners such as CI Fluorescent Whitening Agents 22 and 28. Brief summaries of practical techniques for dye-tracing have been published (Quinlan & Ewers, 1985; Davis *et al.,* 1985; Quinlan, 1981, 1982, 1986a, 1987; Mull *et al* ., 1988). [A comprehensive, plain-English guide to the use of dyes as tracers is planned for publication during 1989 (Aley *et al* ., 1989). It will include practical hints and suggestions that will enhance the rigor of test-design with any tracer and will increase the reliability of test results. It will also include discussion of appropriate QA/QC procedures.] A useful review of the chemistry of dyes has been published by Zollinger (1987).

Three types of dye-tracing can be used for evaluating the suitability of springs, cave streams, and wells for ground-water monitoring. They are:

1. Qualitative tracing, using either of the following:

   A. Visual observation of the dye-plume. Generally this is wasteful of dye and may cause aesthetic and public relations problems. Also, there is great risk of missing the dye-pulse when it arrives at the monitoring site, especially at night.

*Fluorescein is the name generally (but erroneously) used in America for sodium fluorescein ($C_{20}H_{10}O_5Na_2$) which is highly soluble in water and sometimes confused with fluorescein ($C_{20}H_{12}O_5$) which is insoluble in water. Both dyes are CI Acid Yellow 73. This water-soluble dye is known in Europe as uranine. In order to minimize confusion, I follow the American convention of using *fluorescein* when I actually mean *sodium fluorescein.*

B. Passive detection (with passive detectors consisting of activated charcoal or cotton, depending upon the tracer used) plus either visual observation of dye eluted from charcoal or ultraviolet observation of cotton (Aley _et_ al., 1989; Quinlan, 1981, 1982, 1987).

Qualitative tracing is sufficient for most dye-tests; when done with passive detectors, it is generally the most cost-efficient tracing technique.

2. Semi-quantitative tracing, using passive detectors and instrumental analysis of dye with a filter fluorometer or a scanning spectrofluorophotometer of cotton or elutant from activated charcoal (Aley _et al_ ., 1989: Duley, 1986; Thrailkill _et al.,_ 1983; Behrens, 1982, 1987, 1988; Vo-Dinh, 1981). Although instrumental analysis can identify dye-concentrations several orders of magnitude smaller than those detectable visually, the many variables associated with changes in spring or stream discharge, with reaction kinetics of sorption of dye onto passive detectors, and with elution of dye from charcoal all make it impossible to precisely quantify the varying concentrations of dye that passed any specific monitoring site during a given period of time.

3. Quantitative tracing, using instrumental analysis of dye in water samples (either grab-samples or those taken with an automatic sampler) or of water continuously flowing through a filter fluorometer, preferably (for either option) with continuous measurement of discharge. Instrumental analysis enables more precise determination of flow velocity, the breakthrough tune characteristic of a tracer's arrival and retardation, and aquifer dispersivity. It also allows calculation of dye-recovery (the mass balance relation between the amount of dye injected and recovered). Many inter-well traces are done with this type of quantification; some evaluations of wells as potential sites for monitoring can only be done with inter-well traces (Molz _el al_ ., 1986).

For a given trace, quantitative tracing is the most expensive procedure, but it can give answers not available by any other technique (Smart & Ford, 1986; Mull _et al_ ., 1988; Smart, 1988a, 1988b; Aley _et al_ ., 1989.

If quantitative tracing results are needed in the design of a monitoring system--generally, they are not--it is commonly far more cost-efficient-to first do a qualitative or semi-quantitative study. This eliminates substantial costs of sampling and analysis of numerous sites to which no dye travels.

There is a need to publish the results of tracer tests in which quantitative field data, rather than laboratory data, are compared with semi-quantitative field data that is obtained concurrently.

Each of these three types of dye-tracing is sufficient and satisfactory for establishing a hydrologic connection between two points. Semi-quantitative and quantitative tracing techniques are more sensitive to detection of small concentrations of dye; for litigation, they are more convincing. Quantitative tracing techniques are most sensitive to detection of small, temporary changes in dye concentration, and they are sensitive indicators of accidental contamination. If quantitative tracing indicates that nearly all dye is recovered, one can be confident that the monitoring system will operate effectively.

No matter which of the three types of dye-tracing is used for an investigation, it is important to avoid two of the more common mistakes of neophytes: not sampling at enough sites at which dye could possibly be recovered (generally using passive detectors) and not sampling long enough. If not enough sites are sampled or if sampling is stopped too soon after the first positive results, one would fail to detect dye at the other places to which it also goes (if, indeed, it goes elsewhere) at either the same velocity or a different velocity. Also, one would fail to discover if some of the dye were stored in the epikarst (as discussed subsequently under EXCEPTIONS TO ASSUMPTION #3) and released over a long period of time. Any of these consequences of inadequate sampling for dye will prevent discovery of aquifer properties that adversely influence the adequacy of the design for a monitoring system. Other procedural errors that can result in falsely negative results are discussed on pages 50-58 and by Quinlan, Aley, and Schindel (1988).

A "reliable?" tracer test can be defined as one that is a product of careful, prudent design and execution that leaves no significant doubt about its validity. Commonly there will be at least one background sample taken at all sites monitored for dye. Such a test unambiguously demonstrates that there is a hydrologic connection between A and B. There are degrees of reliability. The ultimate standard in a reliable test would produce a smooth, well defined breakthrough curve that accounts for 100 percent of the injected tracer at concentrations that are greatly above well defined background levels, and tracer identification would be done with a scanning spectrofluorophotometer. For most investigations, however, such an ultimate standard is not only unnecessary, but also impossible to attain. There is a need, however, to establish practical standards to ensure the reliability of tracer tests.

The need for tracing during low-, moderate-, and flood-flow conditions for many studies, but not all, is stressed repeatedly in this document. Moderate flow is the ideal time for tracing because generally it can be done efficiently and one can rapidly obtain a tentative understanding of the hydrology. Low flow is difficult because of the extreme duration that may be required and because many flow routes may cease to function. But some of the more difficult traces to perform are those attempted during flood flow. There are three reasons why this is so: Tracer dilution is extreme, and access to monitoring sites may range from difficult to impossible (because of flooding). Also, one must have an intimate familiarity with the monitoring site during non-flood conditions so that, during flood conditions, dye-detectors can be placed in a spring orifice or channel where the amount of dye contacting the detector is maximal, and where the detector will not be left high and dry when the water stage goes down. This can be very difficult to do when a site is under 3 to 8 m (10 to 25 ft) of floodwater and almost unrecognizable.

Dye-Test Analysis

Ideal tracers are conservative. They do not react with soil, bedrock, or ground water, and they do not undergo microbial decay. However, most tracers, including dyes, are slightly reactive and may undergo adsorption-desorption and cation-exchange reactions. . Organic pollutants may undergo similar reactions which affect their rate of migration. This rate is predictable and correlated with their octanol-water partition coefficients (Fetter, 1988, p. 397-405; Winters & Lee, 1987); organic pollutants may also undergo microbial decay. Depending upon their mobility, pollutants may travel faster or slower than a dye. Although tracer velocities can be used as a reliable guide for prediction of flow velocities of pollutants under similar antecedent moisture conditions, especially in conduit-flow aquifers, tracer velocities in diffuse-flow aquifers (as well as in granular aquifers) will be significantly higher than those of most pollutants. In conduits, where reactions of both tracers and pollutants with the rock matrix will be minimal, there will be less difference in their velocities.

Interpretation of dye-recovery curves from quantitative tracer tests can yield much information about the nature of ground-water flow in a karst aquifer and the structure of its conduit system, as discussed by Brown and Ford (1971), Maloszewski and Zuber (1985), Zuber (1986), Lepiller and Mondain (1986), Gaspar (1987), Mull et al. (1988), Quinlan (1988b), and Smart (1988a, 1988b). Similarly, much can be learned from the interpretation of discharge hydrography of springs, as shown by Wilcock (1968), Brown (1972), Sara (1977), Podobnik (1987), Meiman et al . (1988), White (1988, p. 183-186), and others.

Mull et al . (1988) have described an empirical technique for studying the travel time of surface streams to the study of cave streams and spring discharge. The application is more relevant to spill-response rather than to monitoring of a facility, but it is very useful. It makes possible a good approximation of the time of travel, peak concentration, and flow duration of contaminants accidently spilled into a karst aquifer and flowing to a spring or well; it does so for various discharge conditions. The technique is a powerful predictive tool for the protection of water supplies, but it is most judiciously employed <u>after</u> most of the boundary of a ground-water basin or wellhead protection area has been determined by dye-tracing. It is not applicable to monitoring of continuous leakage from a facility. The implicit assumptions of their practical application of dye-recovery analysis are discussed by Quinlan (1988b).

If dye-tracing is needed in order to respond to a spill (to determine, for example, that it drains to a given spring that must be monitored for assessment of the long-term effects of the spill) , then laboratory investigation is needed to determine the effects of the spilled material on the tracer to be used. This should be done before the tracer test. For example, the spilled material could react with the tracer dye, causing it to form a non-fluorescent compound or causing significant quenching of the dye's fluorescence. Either reaction could cause trace results that could be falsely negative. Limited data have been published on such effects, except for chloride ions and chlorine (e.g., Smart & Laidlaw, 1977). A filter fluorometer would be inadequate for such an investigation; synchronous scanning with a spectrofluorophotometer might identify traces of the dye or detect exotic organic compounds that might be formed. [This instrument is also known as a scanning spectrofluorometer, a scanning spectrophotometer, and a fluorescence spectrometer.]

Although it has no bearing on proving if a site will *or* will not leak, rigorous tracing protocol requires that the design of a test include determination of tracer background at all tracer-recovery sites. This determination will influence selection of the tracer to be used and its quantity. Acquisition of background data is good protocol in any scientific investigation, but it is also highly desirable if there is any potential for litigation involving the site.

It is sometimes possible to detect with a fluorometer or a spectrofluorophotometer trace quantities of what appears to be a green dye in background samples. The background can be derived from dyes in various foods, household products, antifreeze, crack-detection penetrant, etc., but such background is extremely rare, except in urban areas and near leaking landfills. It is easy to identify the compound(s) contributing to such background. A synchronous scan of the samples makes it possible to differentiate between various dyes. Coloration of foods and various products is imparted by mere trace concentrations of dyes, quantities that are usually four or more orders of magnitude smaller than commonly used in tracing ground water.

Agricultural runoff may also contribute to background for compounds with green emission. Jones (1989), using a filter fluorometer, has discovered green emission in a water extract of horse manure. This green fluorescence, when sensed with a fluorometer, could be mistaken for fluorescein. I have used a spectrofluorophotometer and found similar fluorescence in cow manure, but these background values are trivial and equivalent to only a few parts per trillion (ppt) of fluorescein.

It should be noted that the green fluorescence peak of cow manure in water (518 nanometers [rim]) does not correspond to that of a water extract of crushed grass (509 nm) or that of fluorescein in water (513 rim). An alcohol extract of crushed grass (saturated solution of KOH in a 70% solution of isopropyl alcohol, a solution commonly used for eluting dye from activated charcoal) also fluoresces weakly in green, but strongly in red-violet. This is because the chlorophyll, which fluoresces red-violet, is highly soluble in alcohol and relatively insoluble in water. Manure and grass is mentioned here in order to establish that their probable contribution to background in dye-tests is detectable, but trivial. Nevertheless, this is a topic worth investigating in moderate detail.

Qualitative evaluation of background cannot be relied on for distinguishing between various green dyes. Also, investigators with little experience in qualitative identification of dyes can be easily fooled by green algae and exotic hydrocarbons.

Fluorometric analysis will distinguish between background samples which include only a green dye such as pyranine (CI Solvent Green 7; D&C [Drug and Cosmetic] Green No. 8; fluorescent) and those samples that consist of a common mixture of a blue dye such as Brilliant Blue FCF (CI Acid Blue 9; FD&C [Food, Drug, and Cosmetic] Blue No. 1; non-fluorescent) and a yellow dye such as tartrazine (CI No. 19140; FD&C Yellow No. 5; non-fluorescent). Dye nomenclature and other dyes are discussed by Quinlan and Smart (1977), Quinlan (1989b), Aley _et al_ . (1989), Marmion (1984), Zuckerman and Senackerib (1979), and Zollinger (1987). A distinction between fluorescein and various other fluorescent green dyes cannot be made reliably with a filter fluorometer because their emission spectra overlap one another; a scanning spectrofluorophotometer must be used (Duley 1986, Aley _et al.,_ 1989).

Success in tracing to randomly located wells that are to be regularly sampled during a dye-test can be maximized if they are pumped continually to discharge at a low rate, say 4 l/rein (1 gpm), through a passive dye-detector (such as cotton or activated charcoal) which is regularly changed once or twice a week for weeks or months. In most settings this small amount of water can be wasted onto the ground at a reasonable distance from any building or structure with no adverse effect. Pumpage at high rates, say 400 l/rein (100 gpm) or more, may distort the flow field near a well, which is acceptable if recognized. A high pumpage rate for weeks to months is expensive and wasteful; furthermore, disposal of the pumped water can be a problem, especially if it is contaminated. Selection of the rate of pumping must be based on knowledge of what constitutes a low rate in the aquifer being investigated.

Although a dye-test is like the birth of a baby--no matter how many men and women are put on the job, it will take however long is necessary to complete the task--it is also quite different. The birth of a baby can confidently be predicted to probably occur about 9 months after conception. A dye-test, however, may be completed within a few hours or days after injection, but it could as easily be weeks, months, or even years. One must be patient while waiting for tracing results, or risk malpractice and loss of valuable data. It might take just a few weeks to do the dye-traces necessary to design and test the monitoring system for a facility, but it is likely to require 6 to 9 months of intermittent, careful tracing. Warning: it could take even longer.

I believe that one well-designed tracer test, properly done and correctly interpreted, is worth 1000 expert opinions . . . or 100 computer simulations of ground-water flow. The only disagreement that colleagues have expressed with this statement is to jocularly suggest that the two numbers should be reversed.

In dye-tracing, wisdom is knowing what essential questions need to be asked, and asking them; experience is knowing the most expedient, most prudent way to get the answers to these essential questions.

The dye-tracing strategies discussed in this document are also applicable to delineation of wellhead and springhead protection areas (Quinlan, Aley, and schindel, 1988; Office of Drinking Water, 1987, EPA, 1988). They should be used.

SURVEYING OF CAVE RIVERS

The surface location of a drill hole sited to intercept a known cave stream can be determined by any of many conventional surveying techniques using a transit, compass, tape, or electronic distance-measurement equipment. Such surveys should include closed loops. It is easier and more accurate, however, to use low-frequency electromagnetic induction equipment, a so-called “cave radio”, which transmits a signal to the surface from an accessible cave passage with a stream (Reid, 1984; Cole, 1988). Such equipment has been used successfully in the Mammoth Cave area to drill to cave streams at depths as great as 143 m (470 ft). The horizonal error in one well drilled to 41 m (135 ft) was only 18 cm (7 in); it was attributed to deviation of the drill hole rather than to error in the survey.

If cave rivers are not accessible from the surface, geophysical studies such as mapping of streaming potential of descending waters and mapping of acoustic emissions from cave rivers (Lange, 1988; Lange & Quinlan, 1988) may follow mapping of troughs on the potentiometric surface. Such geophysical studies should precede drilling of wells planned to intercept cave rivers which are inferred from the potentiometric troughs and which are to be tested by tracing for a connection to a facility.

MAPPING OF POTENTIOMETRIC SURFACE

The potentiometric surface of a karst aquifer should be mapped with as many control points (water levels) as possible. For basin analysis, a minimum of 1 well per square kilometer (2.5 wells per square mile) is recommended for most aquifers. A facility analysis could require more than 40 times this well density. A carefully contoured potentiometric map, if based on valid measurements of an adequate number of wells, can be used to:

1. Predict flow routes of dye (or pollutants);

2. Judiciously select dye-injection sites;

3. Minimize the number of dye-traces necessary for definition of boundaries of a ground-water basin or evaluation of a facility;

4. Interpolate flow routes in the areas between dye-traces; and

5. Detect the possible influence of shale beds and other low permeable rocks on perching and confinement of water within or below a karst aquifer.

Water levels for mapping the potentiometric surface can be measured with an electric tape or an acoustic well probe (Quinlan & Ewers, 1985; Quinlan, 1981, 1982), but such measurement must include a QA/QC program using a vertically hanging steel tape as a standard before, after, and preferably during periods of use, especially if water levels more than 30 m (100 ft) below the ground surface are being measured to the nearest tenth of a foot (0.03 m). Permanent tape-stretch of as much as 2 percent occurs in electric tapes employing copper twin-lead wire similar to those used for TV antenna lead-in. Whenever possible, all water-level measurements should be made with the same electric tape--or each tape should be calibrated against a standard. Measurements should be made of water levels unaffected by pumping, and during low flow or base flow, rather than during the rainy season or after storms when water levels can locally be significantly higher and the potentiometric surface can have a different configuration.

It is logical, correct, and conventional to interpret the flow direction of ground water to be perpendicular to the potentiometric contours and downgradient. Sometimes, however, flow lines appear to be parallel to the contours rather than perpendicular to them, as has been demonstrated in the Edwards aquifer in Texas (Maclay & Small, 1984; Waterreus & Hammond, 1989). The flow lines are actually perpendicular, but a lack of sufficient well data in areas characterized by extreme heterogeneity in aquifer properties precludes demonstration of such orientation.

Mapping of the potentiometric surface does not eliminate the need for dye-tracing nor does it not replace tracing. The two techniques are complementary, but data from a well-designed, properly executed, and correctly interpreted dye-trace or series of traces are less ambiguous than a potentiometric map that is based on water-level data.

IMPLICIT ASSUMPTIONS OF RECOMMENDED TRACED-SPRING, -CAVE-STREAM, AND -WELL MONITORING STRATEGY, WITH EXAMPLES OF EXCEPTIONS

Ground water has been recognized to circulate in three different types of flow systems: local, intermediate, and regional (Tóth, 1962; Freeze & Witherspoon, 1967), as reviewed by Fetter (1988, p. 217-258). Most monitored flow is in local-flow systems. Indeed, this is the flow system most practical to monitor and most needful of monitoring.

Many implicit flow-system assumptions will be made by those who may use the monitoring strategy advocated here and by Quinlan. and Ewers (1985), but they should realize what these assumptions are. The accuracy of the first eight of the following ten assumptions is affected by the hydrogeology of the karst beneath and adjacent to a facility. Assumption #9 involves logic, and #10 is influenced by the rigor of the design and execution of tracing tests--as well as by one’s understanding of the first eight assumptions and one’s ability to recognize the exceptions to them. The major implicit assumptions, stated axiomatically, are:

1. Ground-water discharge is concentrated at a point (a spring or group of springs) rather than diffused over a broad area or concentrated along a line (such as a stream).

2. Most flow systems to be monitored in karst terranes are characterized by local flow, in the sense of Tóth (1962) and Freeze & Witherspoon (1967).

3. Ground-water flow velocities in karst terranes characterized by local flow are the high values already cited herein. Flow velocities in karst terranes characterized by intermediate-flow systems and by regional-flow systems tend to be several orders of magnitude slower than in local-flow systems.

4. A ground-water basin is a discrete entity having a specific, well-defined boundary.

5. Ground-water basins are contiguous.

6. All the discharge of a spring is from the same ground-water basin.

7. Storm-related diversion of ground water out of a basin, if it occurs, is via intermediate- and high-level overflow routes (conduits) leading to adjacent ground-water basins.

8. Temporary, storm-related diversion of surface water within a karst ground-water basin is not a significant problem because all water will remain within the ground-water basin.

9. A positive trace from a facility proves that it is a source or the source of pollutants discovered at the dye-recovery site.

10. The tracing tests done for investigation of a facility were properly designed and executed and correctly interpreted.

Each of the first eight of these implicit assumptions is correct about 95 percent of the time--often enough to be fairly assumed until or unless data imply otherwise, but not so often as to be a certainty. One must always ask, “What are my conceptual models of the flow system and their assumptions?" and critically review the validity of each. Systematic analysis and review of examples of probable exceptions to the above assumptions could be the subject of another document. Only one or two exceptions to each of the first eight is cited and briefly discussed below, along with the relevance of these exceptions to a monitoring program.

The ninth assumption is patently false. it is a flaw in logic. A positive trace proves that the tested facility <u>may</u> be a source or the source of some or all of the pollutants but not that it is a source or the source. All the pollutants could have come from one or more other facilities in the same ground-water basin. Proving that pollutants could be derived from a facility is not the same as proving that they are derived from it. Nevertheless, if there is no other plausible source for the pollutants in a ground-water basin, the tested facility may be--and probably is--the source of them. For example, gasoline found in a spring could have leaked from any of 50 service stations within a basin, each of which could be shown by tracing to be the possible source. Tank tests, product audits, soil-gas analysis, suction lysimeters, and contaminant fingerprinting could be used to find the probable source (or sources) of gasoline. In contrast, in the average ground-water basin there are relatively fewer possible sources of an exotic organic compound. Conversely, if well-designed, properly executed, and correctly interpreted set of tracer tests conducted during both low-flow and high-flow conditions showed that a facility drains to a basin other than the one drained by a contaminated spring, there is sufficient proof that the facility is not the source of the contaminants. Again, this is simple logic, not hydrogeologic subtlety.

The tenth assumption, like the others, is made honorably, but it is the most insidious. It will be discussed after the following exceptions to the first eight assumptions are cited and described.

EXCEPTIONS TO ASSUMPTION #1: NONPOINT DISCHARGE

Although some discharge in the karsted dolomite of the Door Peninsula, Wisconsin, is to springs (Wiersma et al., 1984) that can be considered as point discharge, most of the discharge is through sediment and over a broad area beneath Green Bay and Lake Michigan (Bradbury, 1982; Cherkauer et al ., 1987). Discharge from a karst aquifer through sediment over a broad area occurs in the Caño Tiburones area north and west of Barceloneta, Puerto Rico; it occurs along a line in the valley of the Rio Grande de Manati, south of Barceloneta. These Puerto Rican terranes are in, Tertiary limestones of the alluvium mantled and paludal sediment mantled shallow aquifer on the north coast; the aquifer seems to be characterized by diffuse flow, but springs and conduit flow are locally important.

Monitoring in terranes characterized by areal discharge can best be done with randomly located wells, perhaps along fracture traces, but they might not intercept the relevant flow lines. Monitoring in terranes characterized by seepage along a line is best done by sampling at intervals along the line and upgradient from it. Although ground-water flow velocities may be very low, tracing must nonetheless be done in each hydrologic setting if confidence in a monitoring effort is desired. If a well proposed for monitoring a facility in a karst terrane does not have a positive trace to it (or to its site), it is not a monitoring well for ground water draining from the facility.

Discharge along the length of a stream is an example of non-point discharge. Smart (1988b) has described this in the Maligne karst aquifer in the Rocky Mountains of Canada. Another example of discharge along a line occurs in the eastern Snake River Plains aquifer in Idaho (Lindholm, 1986; Wood and Low, 1988). Admittedly, this basalt aquifer is not a karst aquifer, but it was for the hydrology of such rocks that the term pseudokarst was first proposed more than 80 years ago by von Knebel (1906, p. 182-183). Many of the monitoring principles advocated herein may be applicable to monitoring in this basalt aquifer and in other highly fractured rocks.

EXCEPTIONS TO ASSUMPTION #2: NONLOCAL FLOW

There are many examples of non-local flow (intermediate flow and regional flow) that must be monitored in karst aquifers. Some of the better known examples of such karsts are the Edwards aquifer (Maclay & Small, 1984; Campana & Mahin, 1985) the Great Basin carbonate aquifer (Mifflin & Quade, 1988; Fetter, 1988, p. 233-237); and the Floridan aquifer (Fetter, 1988, p. 237-243, 359-361; Miller, 1984; Beck, 1986). Ground-water flow velocities in them are likely to be much slower than in most conduit-flow aquifers and more like the low velocities characteristic of most diffuse-flow aquifers. Flow-paths are likely to be braided (anastomosed) and dispersive rather than convergent, but most are still to springs.

EXCEPTIONS TO ASSUMPTION #3: SLOW MOVEMENT IN LOCAL-FLOW SYSTEM

The epikarst, briefly described in the introduction, is a zone of karstification below the soil profile and above the phreatic zone; horizontal flow is dominant and storage is significant (Quinlan & Ewers, 1985: Smart & Hobbs, 1986: Bonacci, 1987, p. 28-35; Williams, 1985; Friederich & Smart, 1981, 1982; Smart & Friederich, 1986; and Ford & Williams, 1989). Commonly, it is 3 to 10 m (10 to 30 ft) thick. Although tracer studies in a British epikarst have shown that vertical flow velocities (to caves below) locally exceeded 100 m/hr (330 ft/hr), dye was still detectable in the epikarst 13 months later (Friederich & Smart, 1981, 1982; Smart & Friederich, 1986). Similar results were found by Even et al . (1986) in an Israeli epikarst. This flow dichotomy is actually a continuum, but it suggests that the common adage that ground-water pollution in karst areas is not a long-term problem because the aquifer is rapidly self-cleaned, is wrong--or at least unreliable. The duration of retention of a pollutant in a karst terrane is a function of soil thickness and sorptive capacity, the efficiency of connection between the ground surface and the epikarstic bedrock, and the efficiency of flow between the epikarst and the phreatic zone. Indeed, if one is “lucky” and the volume of a spill on a thick soil is minimal, all or most of the spilled substance may be retained and/or sorbed by the soil. Aquifer remediation, therefore, may be limited to excavation of the soil.

Most dyes are injected into sinking streams, sinkholes, or wells which bypass the soil and epikarstic zone. In contrast, contaminants are rarely injected purposely and directly into a karst aquifer. Pollutants in the epikarst may be monitored using traditionally sited, randomly located monitoring wells at the point source to be monitored and they may obey Darcian flow laws, but such a monitoring system may not intercept them. Nevertheless, contaminants may actually be detected and recovered with traditional or innovative techniques before they enter the conduits of a flow system. Once a contaminant has entered the subjacent conduits and solutionally enlarged joints, Darcian flow conditions usually do not apply; detection and recovery are difficult and the protocols recommended in this document are applicable.

EXCEPTIONS TO ASSUMPTION #4: FUZZY AND OVERLAPPING BASIN BOUNDARIES

It is quite logical to assume that a ground-water basin has a specific, well-defined boundary. Sometimes, however, the boundary temporarily migrates or ceases to exist during response to moderate- to flood-flow conditions; some water is diverted to adjacent ground-water basins, as already discussed. Nevertheless, this assumption is reasonable and generally correct; it can be extended to allow for temporary, slight shifts in the boundary between two basins, also in response to storm-related changes in flow conditions, In contrast, some diffuse-flow aquifers show significant exceptions to this implicit assumption; the basin boundaries may be nebulous and gradational during all flow conditions. For example, all dye injected near the center of a ground-water basin in the Great Oolite Limestone (in the Bath district of southwestern England) flows to only one spring. Successive tests at increasingly greater distances from the central axis of the basin show that less dye goes to its spring; the balance of the dye goes to the spring draining the adjacent ground-water basin. [As the dye-injection point gets successively farther from the central axis, more and more dye goes to the major spring in the adjacent basin.] Such fuzzy boundaries are characteristic of basins in this aquifer. Smart (1977) has recommended that the gradational “boundary” between any two such ground-water basins be chosen to coincide with tracer injection points from which the dye divides evenly between the adjacent springs. Similar results occur elsewhere in England (Atkinson & Smart, 1981) and probably in other places where diffuse flow predominates. Nebulous, gradational boundaries between ground-water basins have recently been recognized in Missouri (Thomas Aley, Ozark Underground Laboratory, Protein, Missouri, oral communication, 1987) and undoubtedly will be recognized in other karst terranes of the United States.

Although Smart's 50 percent boundary criterion is reasonable for hydrologic budgeting (Smart, 1977), for the delineation of a wellhead or springhead protection area it is necessary to know the entire area that contributes recharge. Therefore, for some hydrologic studies, his valid criterion must be ignored.

Gradational basin boundaries are suggested by the results of dye-injection into a swallet (a sinkhole into which a stream empties) at the west boundary of the Turnhole Spring ground-water basin (Figure 3, on the 520-ft contour). This swallet drains to two different ground-water basins; it flows both to the northeast (within the Turnhole Spring basin) and to the west, to a second major drainage basin (Quinlan, Ewers, et al ., 1983, p. 48-49). This particular boundary of the ground-water basins has not been studied, but I believe that the extent of overlap is less than half a kilometer.

One should be prepared to encounter fuzzy and overlapping boundaries between ground-water basins. The possible existence of such boundaries makes it necessary to be extremely thorough in the design of dye-tracing investigations and confirms the already recognized need for monitoring for dye at springs in ground-water basins adjacent to a proposed facility (Aley et al ., 1989).

Overlap of ground-water basins is convincingly illustrated by the Bear Wallow basin in Kentucky (Quinlan & Ray, 1981). It occupies 500 km^2 (190 mi^2) and its three subbasins, Hidden River, Uno, and Three Springs, resemble a Venn diagram (a diagram employed in symbolic logic; it uses circles and their relative position to represent sets and their relationships (Gardner, 1982; Miller, 1986; Edwards, 1989). The Hidden River and Uno subbasins occupy 80 percent, and 35 percent, respectively, of the total basin. The paradox of these two subbasins totaling more than 100 percent of the area of the Bear Wallow basin is explained by the fact that the Three Springs subbasin comprises 15 percent of the entire Bear Wallow basin and is part of the headwaters of each the other two, subbasins; it is common to each of them. This overlap is significant because the 97 km^2 (37 mi^2) of the Uno basin is the size of the area alluded to earlier in this document as the terrane in Figure 4 from which pollutants could flow to a total of 52 springs in 19 isolated segments along a 19 km (12 mi) reach of Green River. Probable consequences of leakage from a facility would be significantly fewer if it were located someplace other than in a ground-water basin in which the overlap of the headwaters is analogous to a Venn diagram and discharge is via distributary flow. This overlap is also significant because adequate monitoring of a facility in the Uno sub-basin would require more sites (and more expense) than monitoring the same facility in a basin that was not part of two other basins.

Similar overlap of recharge areas of springs occurs also in the Mendip Hills of southwestern England (Drew, 1968, 1975; Atkinson, 1977).

EXCEPTIONS TO ASSUMPTION #5: NONCONTIGUOUS GROUND-WATER BASINS

Thrailkill and his students have shown that there are two physically distinct spring types in karst of the Inner Bluegrass of Kentucky: local high-level springs discharging from shallow flow paths and major low-level springs discharging from a deep, integrated conduit system (Scanlon & Thrailkill, 1987; Thrailkill, 1984, 1985). The major low-level springs are characterized by larger catchment areas (>10 km^2; 4 mi^2) and higher discharges (10-2700 l/see; 2.6-700 gal/see); the local high-level springs are characterized by smaller catchment areas (<2 km^2; 0.8 mi^2) and lower discharges (0.1-0.8 l/see; 0.025-0.2 gal/see). The lack of integration of the local high-level spring catchments into the major low-level spring catchments can be explained by the impermeability of numerous interbedded shales and the lack of fractures passing through them. The catchment of each major low-level ground-water basin is interpreted to be near-elliptical, but there is minimal data to support this shape. Each of the catchments is unrelated to surface drainage, isolated from the nearby major basins, and commonly separated from its nearest similar neighbor by 1 to 4 km (0.6 to 2.5 mi) , thus making them noncontiguous. The reasons for the noncontiguity of these major basins are not yet understood but may be related to the nature of the epikarst between conduits and the inhibition of hydraulic integration by clay and shale in the bedrock (Ralph O. Ewers, Department of Geology, Eastern Kentucky University, oral communication, 1988).

Noncontiguous basins within a karst terrane in which the beds dip uniformly at a low angle also occur in the Nashville Basin, Tennessee (Geary M. Schindel, ATEC Environmental Consultants, Nashville, Term., oral communication, 1989); they probably exist elsewhere. The possibility of their occurrence in various settings should be anticipated and can be detected when traces are run from each side of a suspected boundary of a ground-water basin.

It is possible that traditional, randomly located wells may provide effective monitoring in the interbasin areas between noncontiguous basins. To use these wells reliably in such a karst, however, the tracing procedures advocated herein must first be employed and competently shown to yield negative results at springs and positive results at the wells.

EXCEPTIONS TO ASSUMPTION #6: SPRINGS THAT DERIVE SOME OF THEIR FLOW FROM MORE THAN ONE BASIN

If one or more tributaries to the trunk conduit of a ground-water basin join it at a distance upstream from the orifice that is equal to or greater than the length of the mixing zone for a given velocity and difference in water quality (Fischer et al., 1979, p. 105-147; Hubbard et al., 1982; Kilpatrick & Cobb, 1985; Mull et al., 1988, p. 43-45), monitoring of water quality at its spring during base flow will not detect the existence of the subbasin, even if there was a significant difference between the chemical and physical properties of the two basins or subbasins. But if the confluence of a basin or subbasin is at less than the length of the mixing zone for a given velocity and difference in water quality--or if the confluence is at the surface, possibly because erosion and slope-retreat progressed upstream--a series of measurements of a physical or chemical property such as specific conductivity may show variation across the width of the spring's stream channel. Many examples of this dual-basin phenomena are known. They include:

1. Seven Springs, in Fillmore County, Minnesota. Two distinct ground-water basins discharge from the same conduit orifice. Dye from one basin discharges from the left side of the spring, but not the right; dye from the other basin is discharged on the right side, but not the left (Mohring and Alexander, 1988). At Seven Springs, however, the discharge of water from the second basin occurs only during flood flow, not during base flow. Seven Springs is part of a distributary.

   If water quality of the two basins or subbasins is similar, the only sure way one could discriminate between them--or discover the existence of the second is to measure dye concentrations on both sides of a channel and in its middle. Prospecting for variations in water quality or dye concentration as an indicator of the nearby confluence of a subbasin must be done under low-, moderate-, and flood-flow conditions. It must be shown that these differences in dye-concentration vary with position, not with time. [Meiman et al. (1988) have shown that continous monitoring at the same point during flood flow detects significant short-term variations in the quality of waters that have not been mixed. This is interpreted to be a consequence of piston-flow from tributaries.]

2. St. Dunstan’s Well, in the Mendip Hills of southwestern England. The flow of a spring-fed stream is from two

adjacent orifices, nly 2 m (6 ft) apart. Water quality of one of the springs is significantly different from that of the other. One is usually supersaturated with respect to $CaCO_3$; the other is usually undersaturated. Tracing has shown that each spring has a separate recharge area but a partial overlap of these areas is common to both springs (Drew, 1968, 1970), like the Venn diagrams mentioned previously.

3. Aquas Frias Spring, on the west bank of the Rio Grande de Manati, Puerto Rico. A dye-trace has shown that a major tributary joins the inclined throat of the spring that rises from a depth of 27 m (90 ft). This tributary joins the throat that is the trunk passage at a depth of 9 m (30 ft) below the spring's surface. Most of the discharge is from the trunk passage that is part of the Rio Encantado Cave System, more than 19 km (12 mi) long (Wes C. Skiles, Karst Environmental Services, High Springs, Fla., oral communication, 1989).

The occurrence of dual-basin flow has been attributed above to be a consequence of distributary flow. As such, it may be the reoccupation of high-level flow routes that ceased to function perennially as base level was lowered. Dual-basin flow can also be a consequence of a spring location being base level for a large area that was never integrated into a single ground-water basin.

Of what relevance is this seemingly nit-picking discussion to the monitoring of ground water? If prospecting a channel for position-related and time-related variations in specific conductivity or dye-concentration during constant-rate injection shows that the variations are position-related, one can select the best place in the channel to monitor for dye and/or pollutants. Such data may also enhance the probability of more accurate delineation of the recharge area for each subbasin.

There is another continuum here. If the distinct orifice of two or more spring outlets feeding a stream can be seen at or from the ground surface, I would call the recharge area of each a separate basin, but with a common discharge point. If the distinct orifices can not be seen at the surface and if the existence and position of underground confluences must be inferred from measurements of water quality or position-related dye-concentration, or by diving, the recharge area of each is a subbasin. St. Dunstan's Well represents on end of this continuum; Seven Springs is near the same end of it. Aguas Frias Spring is farther from that end. The opposite end of the continuum is exemplified in Figure 3 by the confluence between the Mill Hole subbasin (3A in the index map) and the Procter Cave subbasin (3B, 3B1, and 3B2). The full spectrum of this dual- or multi-basin continuum is represented by various karst springs in the Hot Springs area of South Dakota (Alexander et al ., 1988).

EXCEPTION TO ASSUMPTION #7: DIVERSION OF GROUND WATER TO THE SURFACE

The Poorhouse Spring ground-water basin in Kentucky nicely illustrates the diversion of ground-water flow to a surface water basin (Quinlan & Ray, 1981, 1989). The basin drains 70 km^2 (27 mi^2) and most flow is to the southeast, through the 3 x 6 m (10 x 20 ft) trunk stream passage that is Steele's Cave. After very heavy rains (approximately once a year), some of the water in this trunk rises about 21 m (70 ft) above its normal level, out of the sinkhole entrance, and overflows onto the surface. From there it flows in the opposite direction to the northwest and west where it augments the flow of a surface stream which drains an area outside of the Poorhouse Spring ground-water basin. Such flow, if contaminated, could give spurious values at surface sites used to monitor pollutants or background in adjacent ground-water and surface-water basins.

EXCEPTION TO ASSUMPTION #8: DIVERSION OF SURFACE WATER TO A DIFFERENT GROUND-WATER BASIN

Sometimes surface waters overflow during storms from one ground-water basin to another. This is illustrated by the behavior of Cayton Branch in Kentucky (Quinlan & Ray, 1981, 1989; Quinlan, Ewers, et al., 1983, p. 22). Little Sinking Creek (the southwesternmost surface stream in Figure 3) drains north to the Green River and contributes to the discharge from the Turnhole Spring ground-water basin. At a point about 2 km (1.2 mi) south of the creek's northernmost swallet, where the south fork of the stream bends north at the 600-ft potentiometric contour, the creek goes out of its banks during floods and diverts some of its discharge westward. The diverted surface water flows about 600 m (2000 ft) west to a swallet that comprises part of a larger ground-water basin that drains westward to the Barren River, and in which the headwaters have been captured by the Turnhole Spring ground-water basin.

This diversion of surface water from the Turnhole Spring basin to another ground-water basin could be relevant to a monitoring effort in the second basin if there were significant quantities of pollutants in the surface waters of Little Sinking Creek and if they were "exported" to the second basin where they could adversely affect the reliability of data from cave streams used for monitoring of background.

The above exceptions to the eight implicit assumptions stated as hydrologic axioms are uncommon but important. Their existence justifies a thoroughness in the design of tracer tests and in the interpretation of tracer results that, to some people, might seem paranoid. Their existence, however, emphasizes the need for facility-related field work as a prerequisite for the design of a reliable monitoring system in a karst terrane.

ASSUMPTION #10, THE MOST INSIDIOUS

Introduction

Tracing for evaluation of a facility is easy to do: All one does is inject tracer such as a dye at one point and recover it at another. so, too, is well drilling: All one does is set up the rig, start the motor, and count the money. Both techniques yield good, reliable results when performed by those trained in their art. But there are so many things that can go wrong when trying to do either that it is more prudent and cost-efficient to have each done by an experienced professional.

It is easy to obtain falsely positive results in tracing--chiefly as a result of contamination that can be prevented by a proper QA/QC protocol. [The same can be said for errors in chemical analysis or almost any other analytical technique.] It is even easier to obtain falsely negative results. This is why I have called the naive and sometimes erroneous belief that tracer results and their interpretation are valid “the most insidious assumption”.

Procedural Causes of Falsely Negative Results of Tracer Test Results

In addition to occurrence of negative results as a result of any of the eight hydrogeologic assumptions discussed above, there are many procedural ways to inadvertently get a false negative in a tracer test and thus “prove” that leakage of harmful materials from a facility will not (or does not) occur. Thirteen of the more common ways are:

1. Inadequate field survey to locate springs or wells to be monitored for tracer.

2. Injection of dye in a well or at another site that has minimal hydraulic connection to the subjacent aquifer.

3. Sampling in only one or two directions from an injection site--rather than in all directions when radial or multi-directional flow is possible, as described in the discussion of Figure 6.

4. Sampling at only a few sites--rather than at all sites possible for recovery of tracer.

5. Not sampling often enough to detect the tracer in either grab samples or on activated charcoal that becomes loaded with other organic compounds before the dye arrives.

6. Premature cessation of the tracer test--before there is enough time for the tracer to reach any monitoring site or those sites that would be reached after recovery of tracer at the first site. (Premature cessation is most common during the dry season, when flow velocities are slowest.)

7. Sampling only at randomly-located drill holes--rather than at springs, cave streams, or wells that become turbid after heavy rains and wells drilled on photolineaments.

8. Use of an inadequate amount of tracer--an amount so small that it is likely to be diluted or sorbed to concentrations below the limit of detection.

9. Use of a tracer inappropriate for the system under study, one that is likely to be totally or greatly sorbed by sediment or rock through which it passes, one that is reactive with a dissolved constituent in water, or one in which the fluorescence is quenched or enhanced by a dissolved constituent or by pH.

10. Use of too high a value for the acceptable threshold of detectability of dye.

11. Use of organizations and individuals inexperienced in the design, execution, and interpretation of tracer tests.

12. Sampling for dye at only one side (the wrong side) of a spring with dual-basin flow.

13. Placement of a dye-detector where contact with spring water or stream water is not continuous for the time between placement and recovery.

In more detail, the procedural errors likely to yield falsely negative tracing results are described below. Some of these errors have already been been mentioned briefly in this document.

1. Inadequate field survey to locate springs or wells to be used for monitoring the presence of tracer. Published USGS topographic maps can not be relied upon as the sole source for data on spring locations. Field work is essential because generally fewer than 5 percent of the base-level springs are shown on topographic maps. An interpretation of regional hydrology based on dye-traces made only to springs shown on the USGS 7.5-minute topographic maps is likely to be a distorted, incomplete caricature of reality. Obtaining this caricature will

cost more in terms of time lost while waiting for dye to be recovered, than doing the tracing investigation correctly from the beginning.

Although most springs occur along the banks of a stream or river, some occur in channels. Therefore it is wise to also set dye-detectors in streams, rivers, and tributaries --just to sense discharge from unknown springs that might be in channels, from other springs that may not have been found, and from reaches characterized by diffuse seepage.

Field work will usually show that there are many domestic wells, both in service and abandoned, that can be used to monitor for dye during a test. For many states most of the wells will not be included in official State or Federal files; inclusion of only 10 percent is common. Probably no dye will be recovered in any of these wells, but it is always worth trying to recover dye in them, especially if their water gets turbid after heavy rains.

2. Injection of dye in a well or at another site that has minimal hydraulic connection to the subjacent aquifer. Dye injected into a well that does not intercept solutionally enlarged fissures, bedding planes, or conduits may remain there for a long time and not reach the drainage system of an aquifer except via intergranular flow. The same can be said of dye put into a trench excavated in soil or on a quarry floor. This error can be easily prevented by first running slug or percolation tests, as already discussed.

3. Sampling in only one or two directions from a dye-injection site--rather than in all directions when radial flow or multi-directional flow is possible. Radial flow, although not common, frequently occurs near topographic divides. Two excellent examples of radial flow at waste disposal sites are illustrated in Figure 6 and by Quinlan and Ewers (1985, p. 214-219).

   If one has ignored the possibility of radial flow in the design of a dye-test or a series of dye-tests, and then gets positive results in those tests which are run, one can be easily lulled into a false sense of security. For example, if a hydrologist's best professional judgement suggested that ground-water flow in the vicinity of dye-injection point #1 in Figure 6 was to the north or east and if he set dye-detectors (bugs) only in those directions and not at springs and wells in the other directions, he might be rather impressed by his perspicacity. He would also be professionally embarrassed and legally vulnerable when leachate was subsequently detected at springs or wells in the south and west.

4. Sampling at only a few sites--rather than at all sites possible for the recovery of tracer. One can not afford the “economy” of minimal sampling. Aside from the fact that one does not get an understanding of regional or basin hydrology without monitoring the sites to which dye is carried by ground water, months of valuable time can be lost while waiting for dye to be recovered at sites to which it will not go. Also, until and unless the dye from a test is recovered somewhere or it is determined where it went, that dye can not be used a second time in the basin. The reason is obvious: Dye recovered after a second dye-injection could be interpreted to be from the first.

5. Not sampling often enough to detect the tracer. For example, if a pulse duration is 24 hours and sampling is weekly, it is highly probable that grab samples will not detect the pulse. This is not a problem if activated charcoal is used for dye-detection, but a better understanding of the flow dynamics is achievable if one detects tracer at a site several times during a test (after at least one time when the detectors and/or water samples are negative for dye) and if one can recognize the approximate time of maximum concentration and monitor the decay of the tracer-concentration curve. Problems of sampling frequency in karst aquifers characterized by conduit flow and nonpoint pollutants are discussed by Quinlan and Alexander (1987).

   As a second example, if one is using charcoal detectors for dye-detection and is doing weekly sampling in waters highly polluted by organic waste and if the adsorption sites on the charcoal are thus totally occupied by organic compounds after 24 hours, elution of detectors changed weekly is incapable of detecting dye or a representative sample of dye--unless one is extremely lucky and happens to set a detector at a time when the dye-cloud is passing the monitoring site. Even such luck is not enough. Organic compounds (and possibly associated bacterial reactions) in streams and ground-water can also remove dye from charcoal detectors. In a well-designed test in which a cloud of Rhodamine WT was visually seen to be flowing by several adjacent detectors in a stream laden with organic pollutants, the amount of dye recovered on the detectors was inversely proportional to the duration of their exposure to polluted stream water; dye-recovery from detectors with the longest exposure was miniscule (Thomas Aley, Ozark Underground Laboratory, Protein, Me., oral communication, 1988).

6. Premature cessation of the tracer test--before there is enough time for the tracer to reach any monitoring site or those sites that would be reached after recovery of tracer

at the first site. An excellent example of the wisdom of continuation of detector recovery is shown in Figure 6. Dye injected next to the landfill site, at #1, arrived in 3 to 5 days at Cannon Spring, about 3.5 km (2.2 mi) away, having traveled at least 700 to 1200 m/day (2300 to 3800 ft/day): dye was detected in 26 to 33 days at Keith Spring, a shorter straight-line distance, having traveled at least 80 to 100 m/day (260 to 330 ft/day). This latter velocity range is consistent with the straight-line flow-velocity of about 120 m/day (390 ft/day) for dye that traveled 7.3 km (4.5 mi) southwest to Mitch Hill Spring in 61 days (Aley, 1988).

An extreme example of premature cessation known to the writer occurred when fluorescein put into an open sinkhole at a hazardous-waste disposal site during a drought took approximately 30 days to travel 3.0 km (1.8 mi) to a spring at a mean maximum velocity of 4 m/hr (13 ft/hr). Subsequently dye flowed from this spring for at least 30 days. The consultant doing the tracer test terminated it before dye ceased flowing from the spring. He did not allow enough time for dye to also reach more distant sites where recovery was possible, and he refused to have the detectors that had been changed weekly analyzed for dye. After he was persuaded to have them analyzed, it was learned that flow from the facility was radial; dye had traveled approximately twice as fast in almost the opposite direction to a domestic well!

Many examples of distributary flow (Quinlan & Ewers, 1985, p. 205, 207-208) would not have been detected if sampling for dye had not continued well beyond the time of first recovery of dye. During the dry season, when ground-water flow velocities are slowest, it is easy to err by premature cessation of a dye-trace. This is one of several reasons why the most efficient times for initial tracing *are* during moderate flow conditions and the recession of Storm flow.

Flow times significantly longer than anticipated (or even negative results) can also be a consequence of injection of dye at poor sites in which flow is significantly slower than in the subjacent drainage system. This is another reason to continue sampling longer than may appear to be necessary for dye-recovery.

7. Sampling only at randomly located drill holes rather than at springs, cave streams, and wells that become, turbid after heavy rains and wells drilled on photolineaments. The rationale for this statement is given and illustrated by Quinlan (1989a) and Quinlan and Ewers (1985). In brief, the probability of randomly located wells intercepting a conduit conveying waste from a site in a karst terrane is about equal to that of a dart thrown at a wall map of the U.S. hitting the Mississippi River. Both events are possible, but the probability of each is extremely low. One can not afford to prospect blindly for cave streams by random drilling.

Most randomly located wells in karst terranes are not suitable for monitoring the quality of ground water draining from a given site (Quinlan and Ewers, 1985). Some wells can be used as monitoring points, but <u>only</u> if dye-tests at high stage and low stage have shown that they are recharged from the site to be monitored. Each well that is to be dye-tested for suitability as a monitoring site should be pumped during the test. at a rate that adequately senses flow in an aquifer but doesn't distort the flow field. Pumping of domestic wells to yield a continuous discharge of 4 to 8 liters (1 to 2 gallons) per minute has been found to be quite satisfactory. A device for maximizing the efficiency of dye-recovery from pumped wells has been developed by the writer (Aley <u>et</u> <u>al</u> ., 1989) .

8. Use of an inadequate amount of tracer--an amount so small that it is likely to be diluted or sorbed to concentrations far below the limit of detection. I know of situations where either corporate parties or a regulatory agency--for reasons ranging from fear of potentially adverse public reaction to problems of alleged toxicity to strong desire not to discover the truth--tried to prevent proposed dye-traces from having the slightest chance of success by deliberately limiting the amount of dye that could be used. The probable dilution of the dye proposed for use would have been to a concentration far lower than the limit of detection of the instrument employed to analyze for it. Investigators in other situations have, through ignorance, used too small an amount of dye. One cannot routinely expect a few ounces of dye to be unequivocally detectable 16 km (10 mi) away.

   When starting a tracing investigation in an area, One should always, if there is a choice, start with the simplest, most obvious trace, the one in which the results are most easily anticipated. This "calibration" procedure enables a better estimate of the amount of dye needed for that trace and other traces in the adjacent area.

9. Use of a tracer inappropriate for the system under study, one that is likely to be totally or greatly sorbed by sediment or rock through which it passes, one that is reactive with a dissolved constituent in water, or one in which the fluorescence is quenched or enhanced by a

dissolved constituent or by pH. Until the "ideal" dye is synthesized and economically available, we must live with problems of sorption of dyes. As a generalization, the least sorbed dye commonly used for tracing is fluorescein; it is superior to Rhodamine WT in most settings where photodecomposition is not a problem (Smart and Laidlaw, 1977). Traces through coal mines are more likely to be successful if CI Acid Red 52 is used; other conventionally used dyes have a nigher affinity for sorption by ferric hydroxide (Aldous and Smart, 1988).

Quenching of dye fluorescence by chloride ions and decomposition of dyes by chlorine have been briefly described on p. 36. Graphs showing the quenching and enhancement of dye fluorescence by pH have been published by Smart and Laidlaw (1977) and Behrens (1982, 1987, 1988). Indeed, use of pH-control for selective suppression or enhancement of excitation and/or emission spectra can be used to minimize the interference effects of one dye upon another when a mixture of two or more of them are used simultaneously and recovered at the same site (Behrens, 1982, 1987, 1988).

Many fluorescent dyes are suitable for tracing groundwater. Before beginning a dye-test, the characteristics of the site and the recovery areas must be evaluated and properties of various possibly suitable dyes must be compared (Quinlan, 1989b).

10. Use of too high a value for the acceptable threshold of detectability of dye. A practical value for the detection limit of a fluorometer or a spectrofluorophotometer is the concentration of analyte which gives rise to an analytical signal equal to twice the background noise (Rendell, 1987, p. 136). Some investigators informally define the acceptable threshold of detectability, however, as many times this limit of detection. Such a practice is foolish and can prevent the recognition and acknowledgement of radial flow, distributary flow, and flow to a sole recovery site or multiple sites when a less than optimal amount of dye is used or recovered. One should always be skeptical and paranoid about anomalous results, especially those near the limit of detection, but they should be critically evaluated for their consistency with results that are more certain.

    Definition of background fluorescence and its variation as an aid to reliable determination of limit of detection is yet another reason why pre-trace background determinations are important. An interesting study of daily natural variations in background fluorescence has been made by Jones (1989). He found no variation for orange (Rhodamine WT), moderate variation for green (fluorescein) [possibly related to leaching of horse manure], and extreme variation in blue (optical brightener) background. This is not to say that any of these dyes were present as background. Rather, there were traces of unidentified substances in the water which a filter fluorometer detected as though they were dyes.

11. Use of organizations and individuals inexperienced in the design, operation, and interpretation of tracer tests. Dye-tracing, like neurosurgery, can be done by anyone. But when either is needed, it is judicious and most cost-efficient to have it done by experienced professionals, those who have already made the numerous mistakes associated with learning or those who have trained under the tutelage of an expert and learned to avoid numerous procedural errors that could have economically and physically fatal consequences.

12. Sampling for dye at only one side (the wrong side) of a spring with dual-basin flow. This procedural error has been explained in the discussion of EXCEPTIONS TO ASSUMPTION #6.

13. Placement of a dye-detector where contact with spring water or stream water is not continuous for the time between placement and recovery. The most common way for this to occur is to set the detector too high in the spring or stream channel rather than as low as possible. When stage falls, the detector is left high and dry, but it rises before the detector is recovered. A pulse of dye could come by after the detector was out of the water and be gone by the time it was again submerged. Another way of having the same effect is to have a curious passerby remove the detector and return it to the water a few days later. [This has actually happened!]

Discussion

These procedural errors are listed and discussed not as a "knave's guide to duplicity" but as an aid for recognizing inadequate investigations.

Quinlan, Ewers, and Field (1988) admitted having mixed feelings about telling how to get spurious results from tracer studies. Nevertheless, they believed that administrators and others who evaluate hydrogeologic studies of facilities must be able to differentiate between skilled, thorough, rigorous work and shoddy or inadequate work. If review of the evaluation report for a facility shows that one or more of the thirteen procedural deficiencies described above are present or that one or more of the exceptions to the first eight routinely made assumptions may occur, the validity of the report is questionable. The ground-water traces are probably incomplete. Interpretations based upon them are unreliable.

It is easy to conduct poorly-designed tracer studies that yield indeterminate results. For example, when the tracer is not recovered, what do the results, more specifically, the lack of positive results, mean? Both investigators and report evaluators who are inexperienced with tracer-test design are not likely to recognize that poor recovery of tracer may be a result of either poor design of the test or inept execution of it--or both. More commonly, both parties erroneously tend to accept a lack of tracer recovery as an indicator of diffuse flow, non-radial flow, or the alleged unreliability of tracers for characterizing the hydrology of a site. Investigators and report evaluators may then develop a false sense of security, believing that leakage has never occurred, or that ground-water flow velocities are very slow and like those in granular aquifers, or that a facility can be monitored by randomly drilled wells.

There is another reason for listing the above causes of falsely negative tracer tests. The consultant or agency employee who knows well what constitutes bad tracer-test design and protocol knows better what constitutes good design and protocol. Such knowledge can improve the quality of dye-test performance and evaluation.

As repeatedly implied and stated throughout this document, short-cutting on rigorous design and protocol of water-tracing is a false economy. The environmental consequences, the ethical consequences, and the legal consequences of malpractice may be far too high to be ignored.

Tracer tests are not intrinsically unreliable. They are reliable. It is tracer tests done by the inexperienced who are unaware of the assumptions they have made and unaware of how one can inadvertently get falsely negative and falsely positive results that are unreliable.

One can learn much about dye-tracing by reading; one should. But before attempting complex traces that can affect the expenditure of hundreds of thousands or millions of dollars, it is judicious to learn also by experience on simple traces and by working with someone who has extensive dye-tracing experience. With dye-tracing, as with love-making, there is only so much one can learn from a book. In the quest for success, there is no substitute for experience.

SISYPHEAN LAMENT

People concerned with monitoring are usually afflicted with one of two syndromes: "nothing leaks" or "everything leaks".

The "nothing-leaks syndrome" is dominated by a rationale that "We designed it not to leak, therefore, it won't." In the early stages this syndrome is characterized by a conviction that there is no problem with a facility unless an anomalous value for a parameter is detected by its monitoring system. In the advanced stages it is characterized by a conviction that there is no problem unless the anomalous parameter exceeds the proscribed limits for drinking water. In the terminal stages this syndrome is characterized by a conviction that there is no problem unless the anomalous parameter repeatedly exceeds drinking water standards. By then it may be too late to prevent the consequences of pollution.

The "everything-leaks syndrome" is dominated by the rationale that few systems work as designed and Murphy's Law is always operative. In the early stages this syndrome is characterized by a conviction that pollutants detected by a well-designed monitoring system are proof that the facility leaks. A deadly Catch-22 logic rapidly brings the syndrome to a terminal stage in which those afflicted fanatically proclaim that if pollutants are not detected, the monitoring system is defective and needs to be improved.

A symptom common to zealots afflicted with the pernicous terminal stage of each syndrome is intolerance for the irrationality of those afflicted by the terminal stage of the other.

When people with each of these syndromes, especially in their terminal stages, must work together on a project, each side tends to keep the other honest--but often at the expense of a harmonious working relationship. Religion and politics are safer, less controversial topics of conversation.

Would that reality were perceivable objectively and the same for all. This topic is eloquently discussed by Alexander (1989).

CHECKLIST FOR DESIGN OF MONITORING SYSTEMS IN KARST TERRANES

The following checklist can be used as a guide to the general sequence of operations necessary for the design of a monitoring system for ground water in most karst terranes, especially those characterized by conduit flow. The checklist is useless, however, without an understanding and application of the concepts espoused in this document. One should do the following:

1. Review the geologic and hydrologic literature.

2. Study and interpret topographic and geologic maps.

3. Make a spring survey.

4. Review this document.

5. Map the potentiometric surface, if possible.

6. Design the dye-tracing study.

7. Do the first trace, preferably during moderate flow conditions.

8. Evaluate results of the first trace and modify the design of the tracing study, if necessary.

9. Determine whether springs to which a facility drains are characterized by conduit flow or diffuse flow.

10. Run other dye-traces during moderate flow, always modifying the tracing plan, as necessary, in the light of the results of each trace. For many facilities it is necessary to perform only two traces during moderate flow.

11. Repeat selected traces during base flow and flood flow.

12. Synthesize tracing results, available potentiometric data, and conductivity and turbidity data used to discriminate between conduit flow and diffuse flow into a monitoring plan.

13. Have results of field studies and the proposed monitoring plan peer-reveiwed.

SUMMARY

In order to be relevant to detection monitoring for pollutants from waste-disposal facilities, water-quality data from karst terranes must be from springs, cave streams, and wells shown by tracing to drain from the facility to be monitored. Tracing should typically be done at least three times: when first convenient (during moderate-flow conditions, to quickly give a preliminary, tentative understanding of local movement of ground water) and later during both base flow and flood flow. The general prudence of tracing during all three types of flow conditions can not be overemphasized.

Sites for monitoring background should be selected on the basis of:

1. Negative results of these tracing tests but from settings in which the rocks and waters are geochemically similar to those of the locations where the tracer tests were positive.

2. Cultural similarity.

A map of the potentiometric surface, if it is based on enough data from an aquifer not complicated by aquicludes and aquitards, will greatly enhance one's ability to efficiently design the necessary tracing tests, interpret them with greater confidence, and design an effective, reliable monitoring network.

Sampling for water quality must be frequent and done before, during, and after storm and meltwater events. Base flow should be sampled between such events.

Ground-water monitoring in karst terranes can be done reliably, but the analytical costs are likely to be significantly higher than those for other terranes. It could be far less expensive to locate a proposed facility in a non-karst terrane.

There are many ways to design the placement and sampling frequency of a ground-water monitoring network in a karst terrane so that it inadvertently yields falsely negative results for the chemical compound(s) being sought. Accordingly, environmental consultants and regulators must be ever vigilant to be sure that negative results are not falsely negative and that positive results are not falsely positive--either accidentally or intentionally.

There is need for Federal and State regulations to officially recognize the utility and wisdom of ground-water monitoring at springs, cave streams, and traced wells.

Numerous plausible, axiomatic rules (assumptions) can be stated about ground-water flow in karst terranes, but they are not absolutes. Exceptions are known for all' but two of the ten rules cited; more exceptions will be discovered.

It is easy to inadvertently get falsely negative results with any tracer, thus "proving" that a facility will not (or does not) leak and will not affect "x". If an investigator knows what these pitfalls are, he can avoid them. If a report evaluator knows them, he can recognize flawed investigations, reject them, and have them redone properly.

Conflicts between designers, evaluators, and regulators over whether a monitoring system works or doesn't work and how to interpret its data are often a result of irreconcilable philosophical convictions. Some people sincerely believe that a facility designed not to leak will not leak. Opponents are equally convinced of the impossibility of long-term prevention of leakage and place great importance on judicious siting and skeptical interpretation of monitoring data.

The monitoring strategies advocated here and discussed in more detail by Quinlan and Ewers (1985) work in most karst terranes characterized by recharge and by discharge at springs. They are also applicable to the delineation of wellhead and springhead protection areas. They are a significant advance over traditional monitoring strategies and have been recognized as such (Beck *et al.,* 1987), but this document is far from the last word on the subject. Much remains to be learned and described. There is a need for publication of more case studies.

EPILEGOMENON

Dye-tracing is essential for the design of a monitoring system in a karst terrane. But why is it that professional geologists and engineers who would not think of venturing into the design of a building foundation, a landfill, or a well field without first having obtained some initial experience with such matters, and without a review of its design by a competent peer, all too often assume that anyone can do professional-quality dye-tracing on the first attempt? If one's tracing experience is limited or nonexistent, the most astute, most ethical, and least expensive way to minimize the risks of costly litigation for tracing-related malpractice are to choose one of the following:

1. Gain experience with one of the fewer than 10 individuals in North America who are adept at tracing with various types of dyes in complex hydrologic settings.

2. Arrange for one of these individuals to do the tracer investigation--with the understanding that he will give some rudimentary training.

3. Hire one of them to design the tracer investigation.

4. At an absolute minimum, use one of them to review the design of the proposed tracer investigation.

A thoroughly-experienced, adept tracing-consultant potentially saves his client many times his fee and greatly enhances the reliability of both tracer results and a proposed monitoring system.

Some readers may choose to interpret the preceding part of this epilegomenon as a self-serving solicitation offering consultation services. That would be unfortunate. It is not. Rather, it is an objective assessment of facts as viewed in the light of extensive experience.

Karst hydrology is foreign to the training and experience of most American geologists, hydrologists, and engineers. Indeed, only one of the commonly used hydrology texts gives more than lip-service recognition to the significant differences between flow in karst aquifers and flow in other aquifers (Fetter, 1988, p. 285-295, 233-246).

Only three of the dozens of hocks, monographs, and manuals published on ground-water hydrology and/or monitoring during the past eight years include either a discussion or implication of the recommended strategy of monitoring springs, cave streams, and traced wells (Office of Ground-Water Protection, EPA, 1987, 1988; Mull et al., 1988), but the authors of at least one hydrology text in preparation strongly endorse it. Two recent textbooks on karst geomorphology and hydrology also endorse it (White, 1988; Ford & Williams, 1989). No journal papers even hint at the strategy, but there are numerous recent papers, chiefly by karst specialists, that do so in conference proceedings (Alexander et al., 1988; Aley, 1988; Beck, 1986; Field, 1988, 1989; Jennings, 1988; Quinlan, 1986b, 1988a, 1989a; Quinlan & Alexander, 1987; Quinlan, Aley, & Schindel, 1988; Quinlan, Ewers, & Field, 1988). The review by Field (1988) summarizes evolving EPA thought on monitoring in karst terranes, but it is not an official statement or a draft of proposed agency policy. In brief, the scientific community has just begun to recognize the effectiveness of the traced-spring, -cave-stream, and -well monitoring strategy.

It would seem that many pollution control agencies are about to recognize, at last, the “peculiarities” of karst and to promulgate their regulations accordingly.

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